THE BLUE CITY: URBAN METABOLISM AND THE ENERGY-WATER NEXUS

BY

CHRISTOPHER CHINI

DISSERTATION

Submitted in partial fulfillment of the requirements
for the degree of Doctor of Philosophy in Civil Engineering
in the Graduate College of the
University of Illinois at Urbana-Champaign, 2018

Urbana, Illinois

Doctoral Committee:

Assistant Professor Ashlynn Stillwell, Chair
Professor Ximing Cai
Professor Julie Cidell
Assistant Professor Megan Konar

ABSTRACT

Urban environments sit at an intersection of technology, resource consumption, population, culture, and economics. With an increasingly urbanized population across the globe comes an increased demand for resources including water, food, and energy. The study of resource consumption in cities and urban environments, therefore, offers potential for conservation and efficiency increases. Water resources are integral to the necessary functioning of the city and its future sustainability. Not only are water resources directly procured and utilized within the city, water is also consumed in the production of other resources, including food and energy. However, these direct and indirect water resources of cities are understudied, with data that are scattered and inaccessible, if they exist at all.

This dissertation utilized the principles of the food-energy-water nexus, urban water, material flow analysis/urban metabolism, and urban water governance to discuss the magnitude and importance of water resources in the urban environment. Open data and data availability played an integral role throughout the analysis. First, the availability of direct water volume (drinking water and wastewater) and its embedded energy data were discussed. The lack of existing data prompted the use of open records requests to build a database of urban water and energy data for cities across the United States. The collected data were then evaluated to quantify the state of the urban energy-water nexus. Additionally, the information from this database was compared to other material flows and their water footprints to characterize the extent of direct and indirect water resources in cities. Following the quantification of water resources and their impact in cities from a civil engineering framework, statistical modeling was completed to identify indicators of urban water use considering socioeconomic and environmental factors. Finally, through these studies, the underlying theme of open data in water resources was discussed in its relationship to governance regimes. The role of open data in sustainable urban water governance revealed a path forward for policy and future data publication to promote sustainable water systems and the concept of the *blue city*.

In summary, this dissertation both quantifies the magnitude of water resources in cities throughout the United States and promotes the need for further open data. The resultant databases of water and wastewater utilities represents a service population of over 80 million people. The annual embedded energy within national water and wastewater resources was estimated to be 1% of total electricity produced in the United States. Additionally, non-revenue water, estimated at 16% of total treated drinking water, contributed to a significant amount of both water and energy loss. Data for indirect water resources were even more scarce and require a shift in urban water governance to create opportunities for greater data collection and synthesis. The overall results promote greater understanding of the urban water cycle through data collection of direct and indirect water resources, inclusion of embedded energy at the urban scale, and the need for a social sciences perspective when studying the drivers and governance structures of urban water resources.

To my wife.

ACKNOWLEDGMENTS

When thinking back on the last five-and-a-half years, it is pretty amazing to think of all the individuals that have helped me to achieve this goal. Part of me is a little concerned that I am going to forget someone, but I ask for forgiveness in advance. It has been a long and winding path to get to 2018 and a dissertation defense, with challenges and opportunities I could never imagine. So without further ado, I would like to start by acknowledging my great friend, first college roommate, and co-best man at my wedding, Brett Bradshaw. We were walking back from Texas A&M's campus one night and had a deep discussion on what our futures might be. It was then that Brett suggested I consider getting my doctorate. So here I am.

Next, to all my friends I met at A&M: it was difficult to leave you and the State of Texas. Thank you for your support and continued friendship, even though we get to see each other way less than we would like. Thank you, especially, to Wesley Gilmore and our monthly calls over the past few years. Thank you to Andy Ortegon for always supporting me. Thank you to Sterling Debner and Paul Banschbach for being great roommates and great friends. Thank you to Texas A&M University and the opportunities given to me there, that inspired me to be where I am today. I will always be the proudest member of the Fightin' Texas Aggie Class of 2011, Gig 'Em.

Upon arriving at the University of Illinois, I met a great group of friends that continues to support me to this day. Thank you to Jeff Wallace for showing me the ropes of research and helping me complete my Master's Degree. Thank you to Professors Cassandra Rutherford and Joshua Peschel for your mentorship and advising. Thank you to Andrew Rehn for being a great roommate and friend for three years. Thank you to Matt Czapiga for being a great friend and confidant. Thank you to Zachary Barnaal for being just an all-around good dude.

And to all my other friends and colleagues that helped and encouraged me at the University of Illinois (including: Saki Handa, Joseph Harmon, Ryan Sisk, Rich Lo, Nathan Liggett, Elizabeth Depwe) thank you for helping make graduate school just that much better.

Thank you to the Stillwell Research Group. Thank you to Lauren H. Logan (yes I remembered the 'H') and Reshmina William for being great research partners and always having outstanding insights. Thank you Zachary Barker for being a great friend, research partner, and roommate. Thank you to Lucas Djehdian, Jimmy Canning, and Rohini Gupta for always being able to have a good laugh and good ideas. Thank you to all of the undergraduates that helped me become a better researcher and mentor. I would especially like to thank William Lubega for being one of the most genuine people I have ever met and for having some of the greatest insights and opinions ever. Thank you Dr. Ashlynn Stillwell for giving me such an amazing opportunity to learn and grow as a researcher.

Thank you to Diana Byrne for being a writing buddy, sounding board, and research helper. I can honestly say that I would not be defending my dissertation without your friendship.

Thank you to St. John's Catholic Newman Center, for the growth I have experienced there and the friendships I have gained, especially Ross Liederbach. Most especially, thank you to the Graduate Rosary group at St. John's where I met my loving and wonderful wife, Corryn. Thank you to Father Chase, Father Lampitt, and Father Luke.

Finally, thank you to my family: Mom and Dad, Grams and Opa, Grandma Joanne, Zach, Courtney, and all my wonderful Aunts, Uncles, and Cousins. Also, thank you to my wonderful in-laws, Keith and Colleen, Rachel, and the Callahans. Thank you for loving, supporting, and cheering me on even from hundreds of miles away. Thank you to my wife, Corryn, for giving me a wonderful life and son, Oliver Chase. This work is for you.

TABLE OF CONTENTS

LIST OF ABBREVIATIONS

AWWA	American Water Works Association
CAFR	Comprehensive Annual Financial Report
CDF	Cumulative Distribution Function
CFS	Commodity Flow Survey
CUAHSI	Consortium of Universities for the Advancement of Hydrologic Science, Inc.
EIA	Energy Information Administration
FAF	Freight Analysis Framework
FOIA	Freedom of Information Act
GHG	Greenhouse Gas
MFA	Material Flow Analysis
MSA	Metropolitan Statistical Area
MWRA	Metropolitan Water Resources Authority (Boston)
NPDES	National Pollutant Discharge Elimination System
NRW	Non-revenue Water
ORR	Open Records Request
SCTG	Standard Classification of Transported Goods
TRACI	Tool for Reduction and Assessment of Chemicals and Other Environmental Impacts
U.S.	United States of America
USCB	United States Census Bureau
USDA	United States Department of Agriculture

USDoT	United States Department of Transportation
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
VWC	Virtual Water Content
WRCB	Water Resources Control Board (California)

LIST OF SYMBOLS

\$	U.S. Dollars
billion	10^9
gal	U.S. gallon
GJ	giga-Joule (10^9 Joule)
GWh	giga-Watt hour (10^6 kWh)
kg	kilogram
kg CO_{2e}	kilogram equivalent carbon dioxide
km	kilometers
kt	1000 metric tonnes (10^6 kilograms)
kWh	kilo-Watt hours
lpcd	liters per capita per day
m^3	cubic meter
Mgal	million gallons (10^6)
MMBtu	million British thermal units
MW	Megawatt (10^6 Watt)
ton	2000 pounds (907.19 kilograms)
tonne	metric tonne (1000 kilograms)

CHAPTER 1

INTRODUCTION

> The very sustainability of cities and the practices of everyday life that constitute the urban are predicated upon and conditioned by the supply, circulation, and elimination of water.
>
> –Erik Swyngedouw (2004) [1]

Cities, or urban environments, sit at an intersection of technology, resource consumption, population, culture, and economics. Within this intersection, there lies opportunity for action in generating a sustainable environment. With most of the world's population now residing in urban environments [2], the impact of the human species on the planet's ecology now centers on population-dense cities [3]. Cities are open systems relying on the surrounding ecosystem, or hinterlands, to provide resources and dispose of waste [4], and they are central nodes in global flows of these resources and wastes [5]. Studying a city as an individual entity with respect to its resource consumption, as opposed to a national or regional scale, reflects the population at a greater resolution [6]. The relatively small geographical footprint of cities is juxtaposed against a large ecological and economic footprint. Cities are critical nodes in flows of material, handle a majority of the global gross domestic product, and are responsible for approximately 75% of global final energy and greenhouse gas emissions [7, 8]. Water resources are integral to the sustainability of an urban system [9, 10], with understanding human modifications to the water cycle important for determining water scarcity [11].

Cities, especially those in the United States, have been thrust to the forefront of the fight against climate change with the U.S. federal government's reluctance to acknowledge human impact on the environment and a withdrawal from the Paris Climate Accords [12]. This political decision is contrary to a predominant trend in academic literature that considers a new ecological paradigm rejecting the notion that humans are exempt from ecological constraints [13]. The paradigm derives from sociological studies on environmental pollution, resource scarcity, and their impacts on society at large [14–17]. The United Nations' *Local Agenda 21* furthers the discussion of climate action and sustainability on a local scale, encouraging a rescaling of policy from the national or international level [18–20]. This rescaling of policy in sustainability aligns with the predominantly local management of water resources through municipal water utilities. Therefore, urban environments provide the ideal laboratory for the study of water resources within the context of sustainability by promoting urban water conservation and efficiency for the rise of *blue cities*. The *blue city* moniker, devised here, requires a comprehensive look at the urban water cycle from both direct and indirect resources:

> A *blue city* promotes policy, public awareness, and economic efficiencies of water resources to provide equitable, secure, and sustainable water systems. These water systems consist of both direct water resources in the form of drinking water, wastewater, and stormwater (and their embedded material requirements, such as energy), while also considering water as an embedded resource in other material fluxes (food, energy, and other materials). The *blue city* considers the greater impacts of its water resource demands and how its citizens are provided these resources, by way of energy and emissions, through an openness of data and sharing of information.

The *blue city* is derived from discussions on the green city, which can refer to the literal greening of the city in terms of trees or green spaces, but more commonly refers to a sustainable and resilient city that embraces renewable energy, efficient public transit, innovative waste treatment, affordable housing, walkability, etc. [21–23]. Along this vein, a *blue city* provides a sustainable environment through management of direct and indirect water resources and promotes open data on water resources. Just as the color green is the combination of two primary colors (blue and yellow), the *blue city* is a component of the overall green city initiative with a focus on water resources conservation and efficiency. How can research foster a sense of competition towards the sustainability trend already evident across the world [24–26], with more focus on the integral operations and conservation of direct and indirect water resources?

Sustainability metrics are often unitless and relative, depending on the variables and methodology; therefore, it is difficult to compare results of assessments across metrics. Currently, there is a lack of a unifying framework for comparing cities [10]. These difficulties necessitate the use of comparative sustainability studies to define a relative metric based on comparisons between a large breadth of case studies. Noiva et al. [27] conducted one such study for international water consumption values, concluding that there is a need for more comparative sustainability studies. The confluence of water, urban environments, and the growing desire for sustainable operations motivates research to study multiple cities to compare and contrast the different urban water cycles across the United States. Understanding and quantifying of urban water resources begins with data availability; therefore, the *blue city* framework and its emphasis on open data is essential to further comparative analyses of cities. This dissertation utilized a holistic approach in studying the urban water cycle with urban metabolism and embedded resource accounting. The study of water and embedded resources by themselves is not novel; however, the application of these concepts at the city scale with a large number of represented cities is necessary, unique, and fills a knowledge gap in the literature.

This dissertation builds on four major research areas: (i) urban water data, (ii) the food-energy-water nexus (specifically, the energy-water nexus), (iii) urban metabolism, and (iv) urban water governance. Chapter 2 discusses each of these focus areas to provide the necessary background for the research. These areas of study provide the basis for answering the following five research questions, answered in Chapters 3 through 7.

1. **What data are required to describe the urban water cycle and where can they be found?**

   Chapter 3 focused on the existing lack of urban water data and builds a new database of drinking water and wastewater utilities. Using open records requests (ORR), treated water volume and its embedded energy were collected from across the United States. Results of the study showcased the challenges of collecting data using open records requests. Additionally, the data collected in this study directly informed and were utilized by each of the next objectives.

2. **What is the current state of U.S. urban energy-water nexus?**

   Energy-for-water is an important component of the energy-water nexus and the urban water cycle. Chapter 4 analyzed urban water and embedded energy data on an annual and monthly scale to identify geographical and temporal trends of the urban energy-water nexus. Continuing the data collection efforts from Chapter 3 and utilizing an urban metabolism perspective, average per capita drinking water and wastewater fluxes were defined and revised national numbers of embedded energy in urban water resources were tabulated. Additionally, these data were published online in an open-access database to further energy-water nexus research and promote a new era of data access and sharing.

3. **How do the direct and indirect components of the urban water cycle compare across U.S. cities?**

   Chapter 5 quantified and compared the direct and indirect water footprints of urban environments within the context of the food-energy-water nexus. Combining data from the United States Census Bureau (USCB), United States Department of Agriculture (USDA), United States Department of Transportation (USDoT), and virtual water research, the indirect water footprints of cities (considering food, fuel, and electricity) were calculated. These water footprints were evaluated across the country through network analysis and compared to their local direct water footprint. Additionally, a subsequent layer of analysis was conducted to calculate the embedded energy and emissions within both direct and indirect water resources.

4. **How have the material demands of cities changed over the past 50 years and what data exist to quantify this change?**

   In a seminal study from the 1960s, Abel Wolman estimated the metabolism of a typical U.S. city of one million people. Since that time, the field of urban metabolism has expanded globally and refined its techniques. However, there have been very few studies since then of U.S. cities. Using a macroeconomic database of domestic material flows from the USDoT and data collected in Chapters 3 and 4, Chapter 6 revisited the metabolism of U.S. cities. Additionally, the methods were expanded to include the embedded energy of materials necessary to facilitate the metabolism of urban environments.

5. **How do social, economic, environmental, and infrastructure factors indicate the use of urban water resources?**

   There have been a variety of studies that attempt to identify indicators of water use, particularly at the residential scale. However, these data are often scarce, temporally constrained, and restricted by privacy concerns. Using the data collected in Chapters 3 and 4, these regression models of residential households were scaled to inform variations of urban water flux at the city-level. The results of Chapter 7 informed the viability of top-down versus bottom-up methods for water resources management.

These questions identify central components of urban water, both direct water and indirect water, to support the overarching research goal of *understanding water demands of urban environments across the United States to promote sustainable policy in an effort to transition towards the blue city.* While this work is inherently interdisciplinary and spans across multiple fields of research, these questions were answered from a civil engineering perspective. An engineering perspective was important in merging these fields due to the inherent quantitative nature of the research and a need to understand the fundamentals of both direct water infrastructure (drinking water and wastewater) and indirect water demand (food-energy-water interactions).

These five research questions and corresponding chapters supported the overarching research goal of understanding both direct and indirect water resources. However, to translate these research statements into actions and policy, it was necessary to understand the sociotechnical nature of the urban system and its mechanisms of transitions and governance. Chapter 8 discussed the process of urbanization and its myriad of socioecological transformations [28–31], requiring management or governance structures that align to principles of sustainability and long-term resource security. Beyond the standard direct water requirements, there is water consumed along the supply chain of materials imported into the urban environment, most notably food, fuel, and electricity [32], often called a water footprint [33, 34]. Water management and water conservation have become complex sociotechnical systems [35, 36], requiring sustainable governance systems to incorporate all social actors and integrate across geographical scales [1].

A shift to include indirect resources in the urban water cycle further expands the actors of the sociotechnical system. Urban water governance studies examine change to the broad socioinstitutional and sociotechnical regime of the current status quo of urban water systems. These water systems require a shift to a more flexible and sustainable policy and emphasize interactions across the various structures and processes [37, 38]. The existing literature widely cite transparency and accountability in the form of information and knowledge as essential pillars in sustainable urban water governance [39–42]. To this end, Chapter 8 discussed and further defined the *blue city* as a motivational moniker within urban water governance, utilizing lessons learned from the five research questions.

CHAPTER 2

BACKGROUND

Understanding the full urban water cycle to include both direct and indirect water resources required an understanding and engagement with multiple research fields. This background section summarizes the key principles of the four major research fields that motivate this work. (1) Urban water data were scattered and scarce, creating a large knowledge gap for a major user of water resources across the country. (2) The food-energy-water nexus provided an opportunity to understand the indirect water and energy consumption embedded within the urban environment. (3) Urban metabolism viewed the city as an organism requiring inputs and discharging waste, creating a framework for evaluating material flows. (4) Urban water governance provided the foundation for envisioning the *blue city* and uncovering challenges in transitioning the urban water sector. The interdisciplinary nature of this research required a synthesis of literature across water resources and the urban environment to identify knowledge gaps associated with systems-level water resources in urban environments.

2.1 Urban Water and Data

Water resources have an extensive collection of publicly available data sets at the national level, not so on a utility scale. Public water supply accounted for 12% of total U.S. water withdrawals in 2010 [43] and in 2015 [44]. As water stress continues to increase and climate change affects the global distribution of water resources [45], it is important to account for this portion of water. Accounting for this public water supply is a challenge as water utilities often do not make their usage data public, necessitating the use of time-consuming open records requests for obtaining these water data (Chapters 3 and 4). Gleick [46] cites this lack of data and a resulting benchmarking tool as a critical barrier for informing long-term policy of sustainable water management. While water is generally a local resource, it is increasingly global in scope due to resource sustainability concerns, population growth and shifts, and interbasin water transfers, direct or virtual [e.g., 47]. There is a significant literature and data gap associated with cataloging, analyzing, and benchmarking the flux of water and its associated embedded energy through urban environments in the United States. The United States Geological Survey (USGS) compiles two major data sets: Water Data for the Nation and the National Water Use Information Program as part of the National Water Census. The USGS Water Data for the Nation is an inventory of surface water and streamflow gauges across the country with many properties having near real-time collection [48]. The National Water-Use Information Program is a county level inventory of water resource consumptions occurring every five years [49]. However, county level inventories only account for water resources that are withdrawn from within the given footprint; the study does not account for intercounty water transfers such as those in major metropolitan areas of New York City, NY, or Los Angeles, CA. At a state level, the California State Water Resources Control Board (WRCB) provides some data regarding water utilities; however, the level of data does not include treated wastewater or considerations of the energy-water nexus [50].

Additionally, the American Water Works Association (AWWA) provides an annual benchmarking tool for water utilities. This benchmarking tool ranges from operational to employee data, including energy consumption data at a utility level [51]. However, these data are not freely available nor does the tool provide data at the individual utility level. Data are aggregated based on responses from surveys and provide coarse estimates at high levels of variance. These types of data are a start towards understanding the urban water cycle and benchmarking for sustainability, but they fall short in terms of a national-scale database. Another non-governmental organization is Circle of Blue, which does an annual survey of water utility pricing for major cities across the United States [52]. While this survey is a relatively comprehensive study, it focuses on the price of water rather than the physical volumes of water at the utility-level.

The urban water cycle includes rainfall, drinking water imports, water loss, water abstraction, wastewater discharge, and stormwater runoff. Figure 2.1 shows these generalized fluxes of water through the urban environment and the multiple interactions of energy with the urban water cycle. Figure 2.1 also indicates water loss (non-revenue water) before being utilized in the urban environment—non-revenue water—and the embedded energy lost as a result. The AWWA and International Water Association define two categories of water losses: apparent and real losses [53]. Apparent losses are due to meter inaccuracies, data errors, and unauthorized consumption; this water is not properly measured, accounted, and paid for. Real losses are the physical losses of water due to leakage, high pressure, or storage overflows that never reaches a consumer. Combining these two losses into a singular category defines non-revenue water (NRW) [54]. These definitions replaced the often-used term, 'unaccounted-for water,' which varied in calculation across the world, misleading research and results. Non-revenue water provides an important layer of understanding to the flux of water in the urban water environment and has significant implications in water savings [55]. Thornton et al. [56] estimated that, on average, U.S. water distribution systems lose 16% of their total treated water. Self audits reported to the Texas Water Development Board in 2014 by utilities in Texas reveal a slightly lower percentage at 12.5% loss [54]. Globally, the International Benchmarking Network for Water and Sanitation Utilities estimates that global non-revenue water percentages are closer to 30% [57], with conservative cost estimates of this non-revenue water at $14 billion annually [58].

One example of a national database that provides a comprehensive look at a utility scale is the United States Department of Energy's Energy Information Administration (EIA). The EIA provides data for power generation, energy pricing, and energy consumption [59]. In addition, the EIA crosses the boundary into the energy-water nexus by including water consumption and withdrawal statistics for individual thermoelectric power plants [60]. Compared to what is available for energy, the resolution and scale of water utility data are lacking. The lack of a comprehensive utility data set for water resources similar to energy utilities leaves a significant knowledge gap as national level comparisons and an understanding of the energy for water requirements continue to gain attention in advancing the sustainability and resilience of water supply and treatment. This dearth of data associated with urban water resources provided the major motivation for this dissertation.

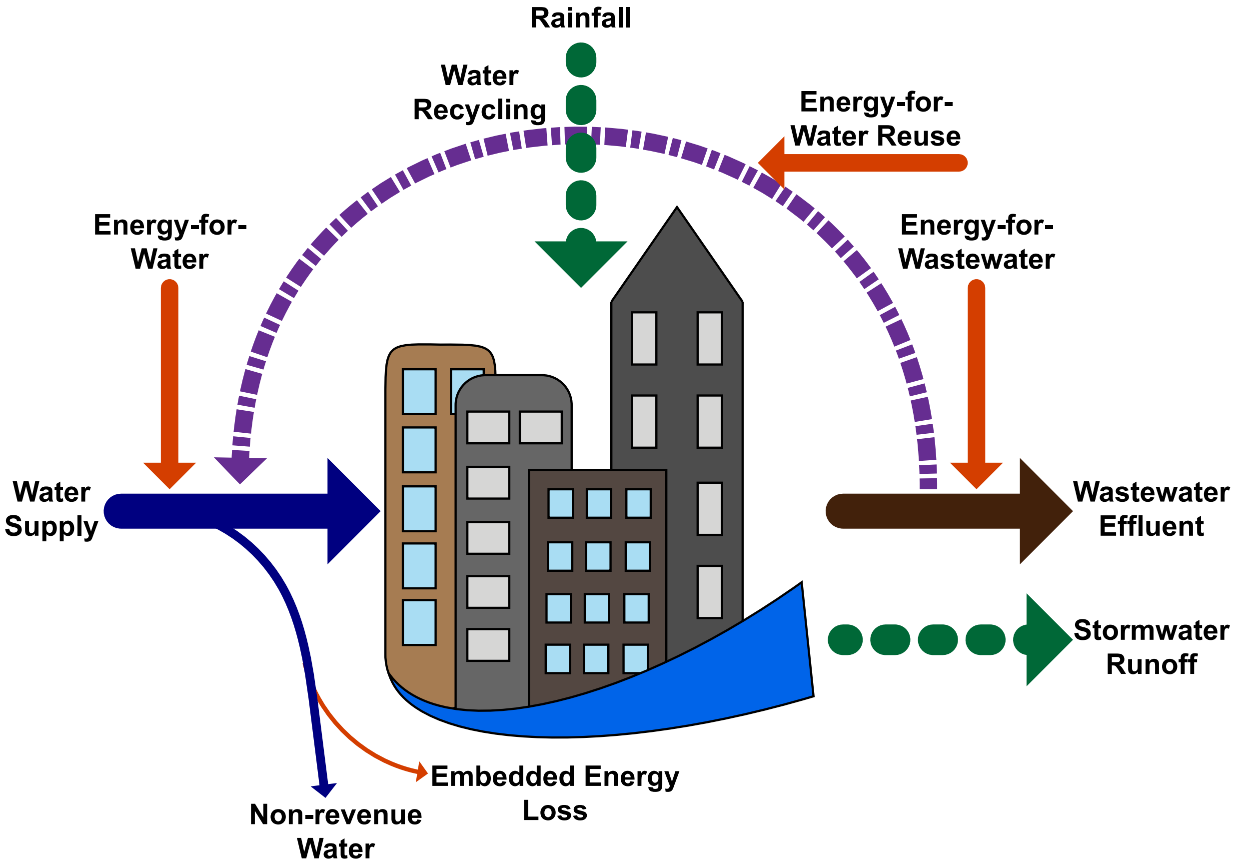

Figure 2.1: The urban environment receives water resources input from both rain and drinking water, discharging both stormwater and wastewater to the environment, with the potential for water reuse to push the linear system towards a cycle (shown with a dash-dot line). Orange arrows entering water flows indicate embedded energies within the system. Dashed lines for stormwater runoff and rainfall indicate flows that are generally excluded from the anthropogenic focus of the study.

2.2 Food-Energy-Water Nexus

The food-energy-water nexus highlights the growing global competition for water resources for direct consumption, food production, and energy production [61]. Within this nexus, there are tradeoffs between these resources that impact the environment, economy, and society. Figure 2.2 shows a few of the interdependent relationships between the competing resources. Within this nexus framework, the concept of embedded resources (or indirect resource consumption) provides a means of understanding the interactions of the resources. Indirect water and indirect energy consumption are two of the major themes throughout this dissertation. Indirect water in food resources carries important implications for food and water security [33, 62]. These embedded water resources in food sources provide a means to assess the water exports and imports of water stressed locations, creating opportunities for globalized redistribution of water resources. Embedded water resources are often referred to as virtual water. Virtual water was a concept developed in the 1990s to assess water consumption of a product through its life cycle [63]. Research on virtual water has since expanded this understanding to further define blue, green, and grey water footprints of a product [33, 34]. Blue water footprints refer to consumed surface or groundwater resources, while green water footprints refer to direct consumption of rain water. Grey water footprints are the amount of water required to assimilate pollutants back into the environments.

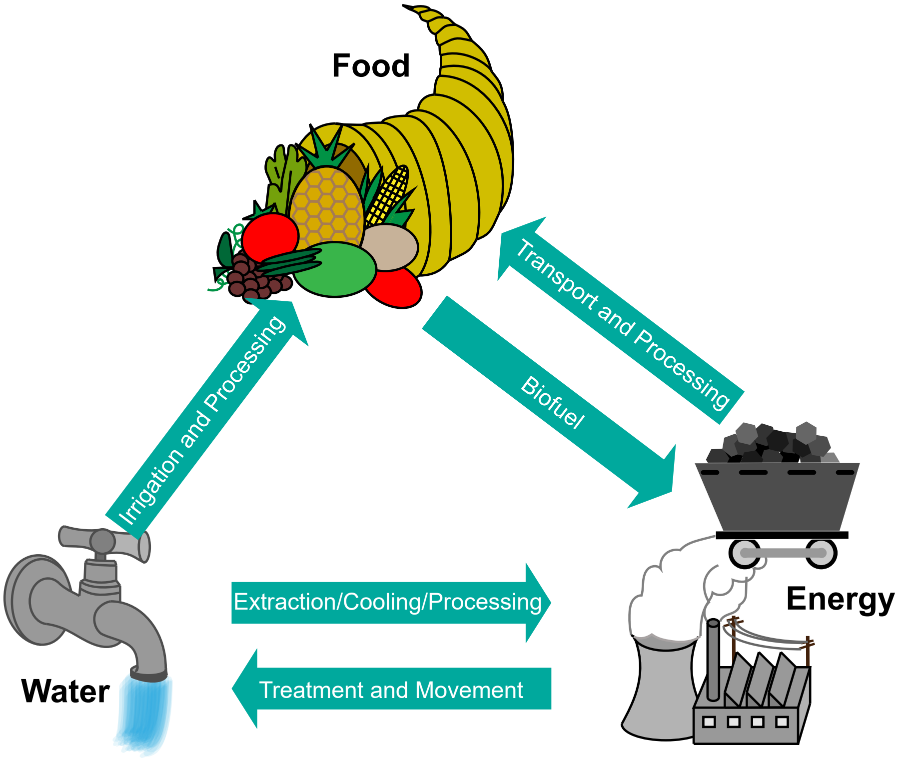

Figure 2.2: The food-energy-water nexus highlights the interdependencies of essential resources and their competition for freshwater resources.

Using these definitions, virtual water research have paired water footprints with product movement to assess virtual water of global food trade [47, 64], country-level food transfers [65], and regional water footprints associated with aquifers [66]. Additionally, previous studies have compared countries or regions within global virtual water trade [67, 68], analyzed the network properties of virtual water trade [47, 64, 69], and quantified urban reliance on aquifers [66]; however, only a few studies have discussed water footprints at an urban scale [70–73]. These studies of indirect water consumption have also included water footprints of other goods such as construction materials, lumber, and rubber [e.g. 74–76].

A subset of the broader food-energy-water nexus, the energy-water nexus, particularly focuses on the bidirectional relationship of energy and water resources. The energy-water nexus generally consists of two components and their associated impacts: energy-for-water and water-for-energy. Water-for-energy evaluates the indirect water consumption associated with both primary (fuel) and secondary (electricity) sources. Indirect water resources are associated with electric power generation and fuel extraction [77–86]. Recent work has expanded the water-for-energy literature to quantify water consumed for hydroelectric power generation [87–91]. The water required for electricity has been extensively studied at a basin, regional, or national scale [79, 91–99] and across the globe [e.g. 87, 100–103]. Also, multiple studies have evaluated the water demands for electricity production based on cooling type, fuel, etc. [77–81, 85, 96, 104, 105].

The other component of the energy-water nexus describes the dependence of water resources on energy (primary and secondary) consumption. Drinking water and wastewater utilities rely upon energy resources to collect, treat, and distribute/discharge water resources. The embedded energy in water resources is a critical component of water resource resilience [106]. Recent studies of the energy-water nexus in the urban environment focus primarily on drinking water resources [27, 107, 108]. However, to provide a full picture of water and energy in the urban environment, it is necessary to also understand corresponding wastewater discharges. Several studies have previously categorized the total energy for water use in the United States with estimates ranging from from 4-16% of total U.S. energy demand [109–111]. The variations in these estimates are based on the inclusion of residential water heating and the inclusion of energy for direct steam usage. James et al. [112] estimate that pumping and treatment of water requires 2-3% of the world's energy. A more recent study estimates energy consumption for water at closer to 1-2% of total energy consumption in the United States and 1.7-2.7% globally [113]. However, these studies utilized limited primary data to extrapolate energy usage and do not concurrently assess water fluxes. With a greater number of cities implementing water reuse strategies and turning to energy-intensive treatment practices, such as desalination, the energy-for-water requirement is projected to increase [113].

2.3 Urban Metabolism/ Material Flow Analysis

Material flow analysis (MFA) is a resource accounting method [114] that focuses on three goals: (i) the analysis of material flows and stocks, (ii) evaluating the importance of these stocks and flows, and (iii) the analysis of flows with respect to goals [115]. MFA is a complementary tool and should be interpreted as part of sustainability goals, environmental standards, or environmental impact assessments [115]. Additionally, MFA is an important tool for the assessment of material consumption trends of a system due to its ability to disaggregate data and quickly identify economic characteristics [116, 117]. MFA is utilized to evaluate the water metabolism of cities across the United States due to its position as an important educational and communication tool [115] and its relevance to describing socio-environmental interactions in support of informing policy and action [118–120]. The study of urban metabolism is principally an application of material flow analysis (MFA) to the city-scale [121].

The concept of urban metabolism originated in the 1960s by Abel Wolman [122] to quantify water and energy flows through a hypothetical city of one million people, focusing on three metabolic problems: (i) water supply, (ii) sewage disposal, and (iii) air pollution. These problems have since expanded in scope with a focus on material balances [123] and embodied energy or ecological footprint concepts [124]. Kennedy et al. [125] define urban metabolism more broadly as the total volume of social, economic, and technical processes within an urban environment, contributing to imports, growth, and waste production. Therefore, an urban metabolism framework considers both upstream and downstream influences [114]. Additionally, the parameters of urban metabolism generally satisfy the selection criteria for sustainability indicators [126, 127]. Urban metabolism is a useful approach for recognition of problems, setting priorities, and policy- and decision-making [115]. Measures of urban metabolism are necessary to address resource concerns within the context of global resource flow [128]. Studies of urban metabolism have shifted from a methodological development in its initial form to an analysis of trends and classification of urban areas [129–131].

Using MFA, studies have tabulated the urban metabolism of cities across the globe from Hong Kong to Sydney to Toronto [132–134]. Many of these studies determine that water resource fluxes through the urban environment dominate and comprise approximately 90%, by mass, of all flows [10, 122, 135]. Rainfall, Figure 2.1, is rarely included in urban metabolism studies; however, its inclusion would lead to water dominating urban metabolism to an even greater extent [136]. These studies often evaluate one city at a specific point in time. Some studies do evaluate cities temporally, across multiple years, [e.g., 125], showing an increasing urban metabolism; however, there are no temporal studies of metabolism within a given year (seasonal analyses). Additionally, according to Kennedy et al. [127] and Zhang [137], there are relatively few studies that have characterized the urban metabolism of cities in the U.S. [135, 138–144].

What constitutes a sustainable urban metabolism? Current speculations on sustainable urban metabolism require a decrease in energy, water, and material flux over time regardless of population increases [9]. However, this definition focuses specifically on the physical and direct material flows of an urban environment. An understanding of indirect or embedded resources in a city's metabolism is suggested as a necessity in advancing to the second generation of urban metabolism studies [124]. A few studies combine the energy-water nexus and urban metabolism [10, 145, 146], but they do not extend the embedded energy to other materials. Finally, a lack of available data leads to urban metabolism studies that focus on a singular aspect or material flow in an urban metabolism study [116]. An approach to urban metabolism that only considers a single sector of materials is inadequate [5]. To understand the role of water in the city, it is necessary to study the flows of energy and all materials entering the city [147].

2.4 Urban Water Governance

Water institutions are embedded within the existing social, cultural, and political structures of the environment [148], creating socioecological and sociotechnical systems, and necessitating a holistic governance perspective. Urban water governance has a rich literature base that started as a largely theoretical perspective [149], but has an increasing number of practical studies [e.g. 150–153]. Water governance is a collective set of policies or actions with diverse actors all towards a common goal [154]. Governance of water resources is distinguished from government or management as they indicate a specific actor [155]; however, they are not separate matters as good governance enables the proper application of management tools within a government structure [39, 156]. Research and implementation of urban water governance must steer away from a sole focus on technological advancements and include a focus on the socio-institutional perspective [157]. As part of this social focus, water resource governance requires:

- a system perspective,
- inclusion of social actors,
- transparent, accessible, efficient, and responsive discourse and analysis, and
- a comprehensive understanding of water sustainability [155, 158–161].

This analytical approach to water resources governance does not constrain management and policies to a limited focus of water infrastructure or the ecosystem and, instead, broadens the discussion of water resources management to include societal stakeholders [155]. Transitions research is a governance approach to evaluate a portfolio of tools and their impact on change towards sustainable development [162]. Under this transitions framework, there are multiple studies on urban experimentation that evaluate the introduction of various policies and programs towards sustainability objectives [163–170]. Often transition literature references the goal of achieving a more sustainable sociotechnical regime that emphasize an adaptive framework with flexible and inclusive practices that empahasize experimentation [171–173]. The ultimate goal of transitions management and urban water governance is to create better water regimes that optimize social and economic welfare in an equitable fashion, support long-term resource sustainability, and promote ecosystem integrity [155].

The approach allows for and encourages a broader definition of urban water to provide a comprehensive understanding of water sustainability including direct and indirect resources.

Traditional urban infrastructure is a large-scale and centralized process operating in a rigid regulatory framework with efficiency and expansion as the key management techniques [38]. Transitioning these existing urban water systems to a more sustainable urban governance regime is often hindered by multiple barriers or institutional inertia [174, 175]. External barriers include economic, environmental, institutional, political, regulatory, and infrastructural challenges that slow change towards sustainability [176]. These barriers are characteristics of the current sociotechnical regime of urban water management and often fall into the institutional, policy, and legal areas [177–180]. There have been many attempts in the literature to identify barriers of transition [e.g., 181, 182]; however, relatively few studies have focused on enabling factors for overcoming this institutional inertia [152].

The institutions that govern material flows and indirect water resources are decentralized and driven largely by economic forces. However, that does not preclude them from similar barriers to sociotechnical transitions. For instance, there are political and institutional barriers associated with implementing food waste or green energy programs, which both lower indirect water demands. Social learning/experimentation and adaptive management are two major fields in the literature that are often suggested to overcome institutional inertia. Social learning utilizes experimentation, as discussed in the previous section, to overcome system barriers and further the restructuring of sociotechnical regimes towards regimes of sustainability [150, 152, 172, 183, 184]. In any setting predicated upon learning, there is a need for robust data to inform and dictate the system. Data provide a necessary means of benchmarking system performance in overcoming institutional inertia as continuing the current status quo persists inefficient and unsustainable resource management practices [185].

CHAPTER 3

THE CURRENT STATE OF WATER DATA

> The greatest challenge in constructing an urban metabolism model is to set up a system that can be efficiently analyzed with available data.
> –Halla Sahely, et al. (2003) [134]

The study of the urban water cycle through the lens of urban metabolism and the energy-water nexus provides opportunities for comparing and analyzing urban environments and their water systems to promote best management practices and sustainability. Data collection is a critical step in the process of material flow accounting, the essential method in urban metabolism [115]. Unlike energy data sets through the United States Department of Energy's Energy Information Administration (EIA) [59], no comprehensive nationwide data set of water utilities exists. This deficit presents a barrier to nationwide water research and limits studies to individual local utilities or regional studies. This study provides insights into the challenges associated with creating a national water and wastewater utility data set for studying the urban water cycle.

The urban water cycle is not limited to the consumption of water resources. The energy-water nexus describes the interaction of water and energy resources with water needed for thermoelectric power generation or fuel refinement and energy needed for water treatment and distribution. Water treatment for both potable water and wastewater requires a significant portion of the United States' generated electricity, around 4-16% of total electric consumption depending on the inclusion of heating water [109–111]. Additionally, Wisniewski [106] names energy reliability as a critical component of resilience in the urban water supply. Therefore, the understanding of energy for water is a critical component of the urban water

---

This chapter was based on work published as: Chini, CM and Stillwell, AS (2017) Where are all the data? The Case for a Comprehensive Water and Wastewater Utility Database. *Journal of Water Resources Planning and Management*, (143)3.

cycle, with no water utility data set complete without the cataloging of energy consumption or, in the case of wastewater, energy generation.

3.1 Collecting New Data

Open records requests (ORR) through each state's version of the federal Freedom of Information Act (FOIA) provide an essential means of data collection for researchers and the public. The ability to request public records is an essential part of the democratic governmental process [186]. This study utilized the open records request process for water utilities across the country to evaluate data collection and availability. Through a standard form, a consistent set of data were collected, with clarification provided upon request. The text included in each utility's records request is included in Appendix A. A single year's worth of data from both water and wastewater utilities in major urban areas across the country were requested. The open records request process was necessary due to the large number of utilities involved in the study. It was not feasible and often not possible to find a personal contact via utility websites to obtain records without this formal channel. However, the formal process was dropped upon request of the utility if it was easier and/or beneficial to provide data in the informal setting.

Urban environments and their respective utilities were selected based on several requirements. The goal of the data requests was to obtain a spatially diverse and comprehensive data set from urban environments; therefore, major cities in each of the fifty states within the United States were the focus. The selected cities had populations greater than 100,000, unless the state does not have a city reaching that threshold; in which case, the largest city(ies) in the state for inclusion in the study were investigated. Not every city with a population of over 100,000 people was included within the study. The goal was to assemble a robust database and sample size for the urban environment in each state, and the requests tended to align with metropolitan statistical areas in each state, as defined by the U.S. Census Bureau.. The approach resulted in 112 targeted cities across the country, see Figure 3.1. However, due to varying operational jurisdictions some cities required two different open records request: one to the operator of the water treatment facilities and the other to the wastewater utility district.

The goal to utilize urban metabolism and energy-water nexus principles in studying the urban water cycle shaped the open records requests to focus on overall water volumes and energy for water data, excluding, to an extent, water quality considerations. Data requests from each utility asked for daily operational data for treated flow in gallons, electricity consumption in kWh, and natural gas consumption in therms. Additionally, for wastewater utilities, the request included biogas generation in therms. Each data request included the caveat that weekly or monthly data were sufficient if daily data were not available. Finally, each utility provided a service population for the reported data.

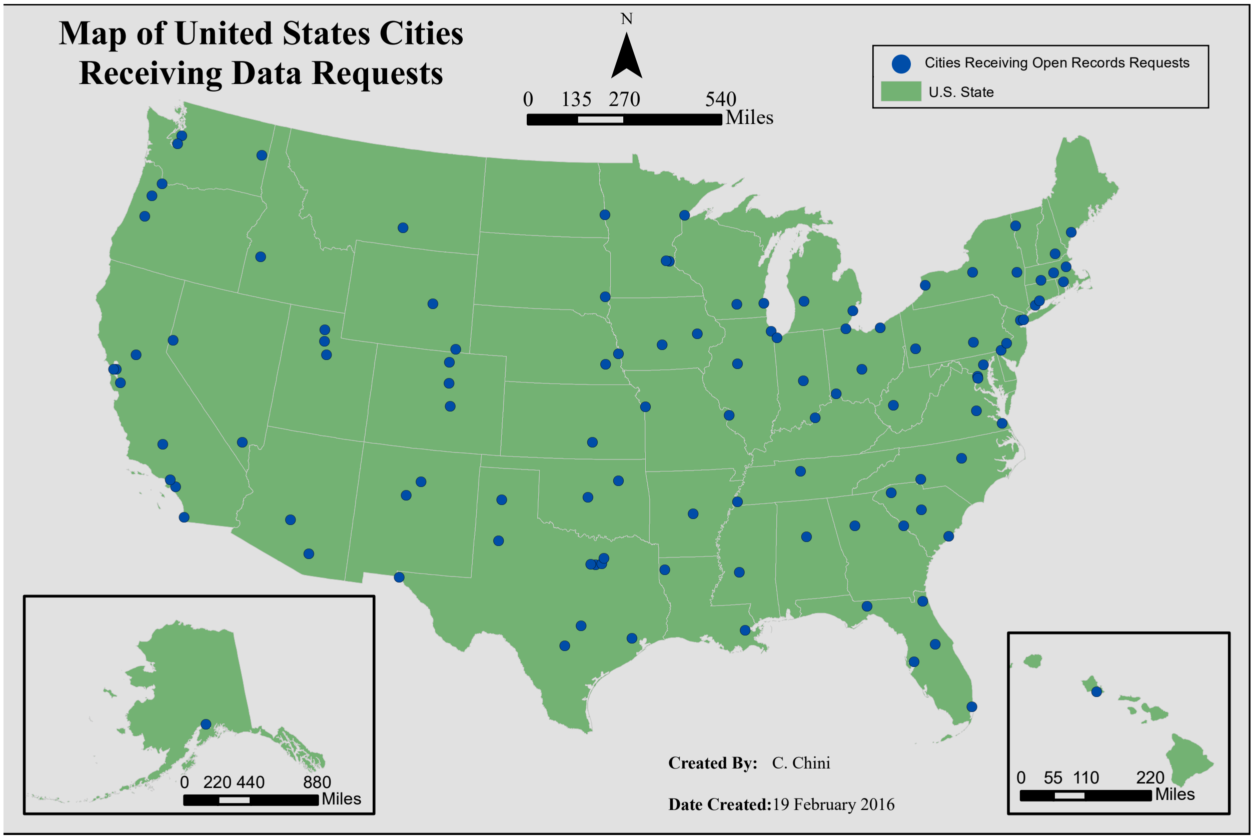

Figure 3.1: Data were requested for cities in each of the 50 states and tended to align with metropolitan statistical areas (MSAs) as defined by the U.S. Census Bureau

3.2 Challenges of Creating a Database

Evaluation of the data collection process began with initial communication attempts through final receipt of the data. There were three categories of evaluation for the study: (i) communication, (ii) data availability, and (iii) data accessibility. The communication section discusses the process of initiating contact and requesting data and its associated challenges. In data availability, a discussion of the issues that arise from a non-standard data collection process is provided. Data accessibility references the ability of researchers and other entities to obtain the data. Through the writing of this chapter, data from 130 drinking water or wastewater utilities were received. The responses represented a population of over 50 million people from both drinking water and wastewater utilities. Chapter 4 further summarized the findings of the database, which was expanded in both responses and requests between the writing of the two chapters.

3.2.1 Communication

To contact utilities, requests were sent through various modes of communication, including standard mail, social media, phone calls, email, and online request forms. Mailing addresses, phone numbers, or social media accounts are often available through utility company's websites, though are sometimes difficult to find. A total of 223 water or wastewater utilities in 112 different cities were contacted multiple times through various forms of media over 8 months. Of these utilities, 97% of the utilities were public with the remaining 3% as private utilities (all drinking water). At the time of writing, 130 utilities responded to the data requests. These responses represented a response rate of 58%, with no responses from private utilities. Figure 3.2 shows the response rate of the drinking water and wastewater utilities. Wastewater utilities had a much higher response rate than that of drinking water utilities.

Communication was a significant barrier to the data collection process. The time-consuming process of scouring utility websites, finding the appropriate contact person, and receiving the data often took multiple back-and-forth conversations via email, telephone, or both. Usually, the FOIA officer was a city-level position who then had to relay requests to the appropriate department and served as the middleman for the data acquisition process, further complicating the communication barrier. This communication barrier is an opportunity to standardize the data collection and distribution process at the utility level, through publication or other means.

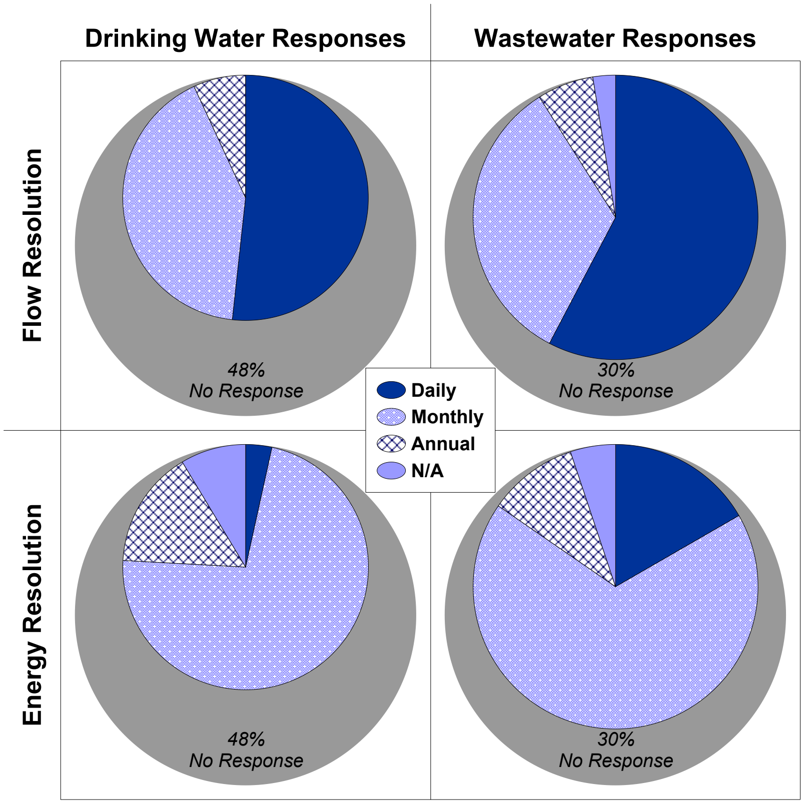

Figure 3.2: Wastewater utility responses compared with drinking water utility responses; wastewater utilities provided more responses at a daily temporal resolution for both water and energy; water flow reporting was more likely than energy reporting to occur at the daily time scale; N/A = utility responses omitting either flow or energy data.

3.2.2 Data Availability

Data availability references the issues that arise from a non-standard data collection process. During personal communications with several facilities, the discussion arose about the ready availability of data from remote sites such as pump stations. Many utilities cited the large number of pump stations in remote locations, all operated on separate accounts and not in direct purview of the contact, as a challenge in procuring the requested data, specifically energy data. Additionally, some utilities responded to data requests with non-digitized energy data documentation (e.g., copies of monthly energy bills), suggesting that energy bills are paid by the water utility's accounting department without any further analysis by the water utility itself. A certain utility, responded to the energy for water request with a 2-inch stack of double-sided, copied energy-bills. The large volume of data in a non-accessible or easily functional format provides challenges for both utilities and researchers to gain understanding of the relationship of water and energy.

Finally, the temporal resolution of energy and water data varies widely between utilities. Figure 3.2 displays the varying temporal resolutions between water and energy data for both drinking water and wastewater services. The figure shows the reporting time step for both water and energy in a format promoting comparisons between the utility and data types. The discrepancy in time scales between water and energy data impedes decision-making opportunities in the energy-water nexus space. Data requests asked for information at the daily scale, preferably, but suggested monthly as an alternative. Wastewater utilities had a higher response rate and were more likely to submit daily flow data than for drinking water utilities. However, there was a large percentage of utilities reporting flow data at a temporal resolution of monthly or greater. In comparison to the energy temporal resolution data, it was more likely that flow data were reported on a daily scale. This temporal resolution for energy is generally in conjunction with monthly utility bills for electricity and natural gas. Discrepancies in time scales between water and energy data impede decision-making opportunities in the energy-water nexus space. Ideally, water and energy would be reported on a daily time scale to facilitate decision-making opportunities using principles of the energy-water nexus and to increase the resilience and sustainability of the system.

3.2.3 Data Accessibility

In addition to data availability, the accessibility of data by research, such as this study, or other entities is a significant barrier to obtaining and creating a national dataset. Within the scope of data accessibility, four barriers to obtaining the data arose during the course of the study: (i) cost, (ii) sensitive data, (iii) state residency, and (iv) private utilities. These four hindrances posed a significant challenge in the study of the utility-level energy-water nexus.

Many utilities charged a fee for the processing and collection of the data request. While this fee is within each state's open records laws, it poses a significant challenge when requesting data from over 200 utilities. From the study, 13 of 223 utilities (6%) required a fee prior to data delivery. This fee ranged from \$0.69 to over \$500.00 for the relatively not-complex data requested. Many additional utilities also requested a fee, which was then waived for the purposes of academic research. The significant cost of these utility's requests did not facilitate accessibility of data for research of the urban water cycle.

The second boundary to accessing data was the sensitive nature of the data. Some utilities refused delivery of data without a non-disclosure agreement or simply declined the request due to sensitivity and potential security risks of the data. Non-disclosure agreements limited the use of these data in subsequent publication. This stance impinges upon the very nature of research; therefore, it was a major hindrance to the collection and publication of a national data set. Additionally, some utilities cited sensitive information requested and, thus, refused the request. A certain utility cited "the potential for revealing potential vulnerabilities to our water system" in the declining of the request (personal communication). In total, 5 public utilities declined to release their data. While the declining of the request due to sensitivity was not an overly large percent of the total requests, it still posed a challenge to the development of a national database.

The final two boundaries to obtaining the data were state residency and private utilities. One public utility would only release data to in-state residents. Fortunately, this singular utility was atypical of the overall study. However, this is a dangerous precedent to set and could impose significant restrictions on national studies. Finally, of the 3% of water utilities contacted that are private, not a single utility responded with data. While public records acts do not apply to private utilities, increased economic stress for maintenance and operation cause more urban environments to turn to private water utilities [187, 188]. This void in data creates opportunities for targeted policies to require the release of such data to the public in private water utilities, similar to the existing structure of (often) private energy utilities [59].

3.3 Discussion

After months of working with utilities for data requests, three significant patterns and trends developed in the responses leading to a single conclusion: the need for a uniform, national database. The trends evident from the study range from first contact to final data receipt including (i) challenges arising from a nonstandard process, (ii) inaccessibility of data, and (iii) temporal resolution for decision-making purposes. Based on the aforementioned challenges, similar strategies need to be developed for water resources data at the utility level as is currently done for utilities in the energy sector.

The non-standard processes of data collection across multiple utilities did not enable efficient collection of data for the purposes of comparison. Non-standard data collection and often less-than-efficient organization of the data hinders decision-making at the systems level considering water resources and energy for water. This widespread problem opens opportunities for targeted policy and the use of data management software. Policy at the state or national level could create standards for utility data collection and encourage utilities to invest in software to manage the data collection process. The efficient collection and organization of data enables a greater understanding of the system and allows for targeted improvements to upgrade the sustainability and resilience of the system.

In addition to the non-standard process of data collection, the collected data are often difficult to obtain through either communication, economic, or security barriers. There are opportunities to fix these concerns by encouraging policy to create requirements for water utility reporting and measurement. Already mandated annual water quality reports or wastewater discharge reports could add more robust data reporting including water volumes and energy consumption. The energy-water nexus has great potential to improve efficiencies and sustainability of water utilities; therefore, open communication and a greater inclusion of water and energy consumption statistics in currently published documentation are recommended.

Finally, the data availability ranged in temporal resolution, often only at the monthly scale for electricity and energy consumption, inhibiting decision-making based on the intersection of water and energy. The intersection of water and energy in water utilities is an important component of sustainability. Energy consumption has implications in detecting system deficiencies, including contributing to climate studies. A more energy efficient system increases resilience to electricity blackouts or energy shortages [106]. Decision-making at the intersection of energy and water requires high temporal resolution data, preferably at the daily level. The use of remote sensing and data collection software to enable daily collection of water volume and energy consumption data is recommended.

Because of these challenges and the time consuming nature of data collection, even at the scale requested, it is necessary for future research to create and maintain a comprehensive water resources database at the utility scale. A national database of utility-level water volumes and consumed energy is essential for the advancement of water utilities and their sustainable operations. Policy is necessary to direct utilities to publish their data similar to energy data within the EIA. This policy could originate from either non-governmental organizations such as the AWWA, national governmental agencies such as the USGS or USEPA, or research groups/universities. The creation of this database has significant potential to enable a wealth of future research, beneficial to water utilities across the country and around the world.

CHAPTER 4

THE STATE OF URBAN WATER IN THE UNITED STATES

> In terms of policy, MFA can be used for early recognition, priority setting, to analyze and improve the effectiveness of measures, and to design efficient material management strategies in view of sustainability.
>
> –Carolyn Hendriks, et al. (2000) [115]

Increasing water stress and climate change affects the global distribution of water resources [45]. As a result, cities face increasing challenges to water management constraints [189]. Over half of the population of the United States is vulnerable to water resources risks [190], and these water resources are integral to the life, economy, and social structure of urban environments [191]. Therefore, understanding the anthropogenic fluxes of water through the built environment (defined in this context as movement into and out of a city) is an important consideration of urban sustainability [1]. Despite this importance of water resources fluxes, there are relatively few sources of publicly available urban water data. The data that are available are scattered and require a significant amount of synthesis [192].

Recent studies of the energy-water nexus in the urban environment focus primarily on drinking water resources [27, 107, 108]. However, to provide a full picture of water and energy in the urban environment, it is necessary to also understand corresponding wastewater discharges. Several studies have previously categorized the total energy for water use in the United States with estimates ranging from from 4-16% of total U.S. energy demand [109–111]. The variations in these estimates are based on the inclusion of residential water heating and the inclusion of energy for direct steam usage. Additionally, James et al. [112] estimate that pumping and treatment of water requires 2-3% of the world's energy. A more recent study

---

This chapter was based on work published as: Chini, CM and Stillwell, AS (2018) The State of U.S. Urban Water: Data and the Energy-Water Nexus. *Water Resources Research*: 54(3), 1796-1811.

estimates energy consumption for water at closer to 1-2% of total energy consumption in the United States and 1.7-2.7% globally [113]. However, these studies utilized limited primary data to extrapolate energy usage and do not concurrently assess water fluxes. Using a more robust set of primary data from drinking water and wastewater utilities and principles of material flow analysis (MFA), this chapter answers the following motivational research question: *What is the current state of the U.S. urban energy-water nexus?*

Anthropogenic fluxes of water including water supply, water loss, and wastewater effluent and their embedded energy were the focus of the study. The urban water cycle includes rainfall, drinking water imports, water loss, water abstraction, wastewater discharge, and stormwater runoff. Figure 2.1 shows these generalized fluxes of water through the urban environment and the multiple interactions of energy with the urban water cycle. Water is lost in the system due to non-revenue water, which provides an important layer of understanding to the flux of water in the urban water environment and has important implications in water savings [55]. The underlying principles of MFA, a resource accounting method that tracks material flows into and out of geographic regions [114], provided the basis for the study. MFA is an important educational and communication tool [115] and is highly relevant to describing socio-environmental interactions in support of informing policy and action [118–120].

While water is generally managed as a local resource, it is increasingly broader in scope due to resource sustainability concerns, population growth and shifts, and interbasin water transfers, real or virtual [e.g., 47]. There is a significant gap associated with cataloging and analyzing water flux and its embedded energy through urban environments in the United States. This chapter filled this knowledge gap by (i) using a unique database of primary data from drinking water and wastewater utilities across the country, (ii) assessing the overall state of the U.S. urban energy-water nexus, with respect to energy-for-water demands, and (iii) promoting data sharing between utilities and researchers through publication of the data in an open-access database. This study provided nationwide statistics of annual and intra-annual water fluxes in cities, non-revenue water, and spatial grouping. Each of these statistics were presented for the first time using a robust database of primary data to provide important insights for understanding urban energy and water sustainability.

4.1 Materials and Methods

This section describes the approach to data collection, synthesis, and analysis of drinking water and wastewater utilities and their water use and embedded energy. Before discussing methodology specifics, it is important to identify geographical and accounting system boundaries. The study of urban water flux utilizes drinking water and wastewater utility level data; therefore, each utility service area provided the geographical boundary for its respective city. As utility boundaries do not necessarily correspond to the political jurisdictions of cities and their city limits, the utilities might service some communities outside the main urban area. For instance, the Detroit Water and Sewerage Department provides service to nearly 40% of the State of Michigan's population, far more than the population of the City of Detroit. Additionally, drinking water and wastewater utilities in the same city do not necessarily have the same service areas. Therefore, in order to define the flux of urban water, the utility water flows were normalized by service population. For the accounting boundary, the study focused on drinking water imports and wastewater exports, excluding the energy-for-water needs of water within buildings (e.g., domestic water heating). Only utility-level energy consumption was considered for the production, treatment, and pumping of water resources into and out of the city. These were necessary boundary adjustments due to the scope of the study and the goal of evaluating the state of the U.S. urban energy-water nexus from a water utility perspective.

4.1.1 Data Collection

This objective's data collection efforts were in conjunction with and a continuation of those from Chapter 3. Open records requests were sent to utilities in 127 cities across the United States over the course of two years, representing 253 distinct water and sewer utilities ($127 \times 2 - 1$; Minneapolis and St. Paul, MN share a wastewater district, and therefore the data are combined). These cities represent major urban environments in each of the 50 states. Each city has a population of greater than 100,000, except in states where there were no cities meeting that criterion. In those states, the largest cities were selected to be a part of the database. Additionally, not every city with a population greater than 100,000 people was included in the data search. The goal was to assemble a robust database and sample size for the urban environment in each state, and the requests tended to align with metropolitan statistical areas in each state, as defined by the U.S. Census Bureau. The data from these open records requests were published at a monthly timescale, when available, in an online and open database through the Consortium of Universities for the Advancement of Hydrologic Science, Inc. (CUAHSI) [193].

Data request methods included standard mail, email, telephone conversations, online forms, and social media. These data requests focused on water flow, energy consumption, service population, and, in the case of wastewater, energy recovery. Data were requested for the year 2012, based on its relative recentness and its correlation to other national datasets, such as the Commodity Flow Survey [194]. The formal request process to receive data from water utilities had substantial communication, data availability, and data accessibility challenges, as described in Chapter 3. Additionally, data were assembled outside of the formal records request process for non-revenue water. These data were available through multiple sources including news reports, end-of-the-year financial reports, state databases, and through first-hand communication via email and social media exchanges (i.e., Twitter).

4.1.2 Data Synthesis

Data received from open records requests came in a variety of forms and temporal scales. Often these data came in multiple formats such as scanned copies of utility bills, hard copies of reports, or in PDF files of tables. Water volume and energy data were aggregated to monthly and annual scales for each utility, as necessary. Energy resources consumed by utilities included electricity, natural gas, fuel oil, and biogas. In previous studies on drinking water supply, only electricity was considered as a component of energy-for-water [108], neglecting other energy sources. Natural gas, fuel oil, and biogas are considered primary energy sources, while electricity is a secondary energy source (i.e., generated from a primary energy source). Distinguishing between these sources and normalizing to a common form was necessary to accurately assess and compare energy-for-water at different utilities.

These energy portfolios were converted to secondary energy (electricity in equivalent kWh) for comparison, Equation 4.1, consistent with Chapter 5.

$$E[kWh_{eq}] = e + 0.45 \times (29.3\left[\frac{kWh}{therm}\right] \times n + 43.9\left[\frac{kWh}{gal}\right] \times f) \tag{4.1}$$

where, e is the electricity consumption in kWh, n is the natural gas consumption in therms, and f is the consumption of fuel oil in gallons. The conversion factors were equivalent to the normalization factors used in the American Water Works Association's *Benchmarking Performance Indicators for Water and Wastewater: 2013 Survey* [51]. A factor of 0.45 was used in the conversion to account for efficiency in using natural gas, biogas, and fuel oil to produce electricity [195]. For biogas, values were often given in standard cubic feet (scf), a conversion factor of 0.61 therm per 100 scf was used to account for the different thermal intensities of biogas and natural gas [196].

4.1.3 Data Analysis

Data analysis consisted of both statistical and spatial components. To calculate a national average of urban environments, a population-weighted average was utilized. In addition, to provide corresponding weighted standard deviations for water volume, it was necessary to artificially create data points based on population and treated volume. For example, a city with a population of 800,000 that uses 400 liters per capita per day would be represented by 800,000 data points with the value of 400 in the sample to capture the weighted standard deviation of an urban citizen in the United States. Therefore, a city with a low population but high water flux will not overly skew the average and standard deviation of the urban water statistics.

A spatial analysis of the database was used to both visualize the data and evaluate the effectiveness of regional benchmarking initiatives. A k-means clustering algorithm with a varied number of groups or clusters determined appropriate geographic regions. The k-means clustering algorithm partitions data into k clusters where each observation belongs to a cluster with the nearest mean. The algorithm seeks to minimize an objective function based on Euclidean distance and variance of a mean. In other words, the algorithm spatially correlates utilities based on proximity to each other and water flux and embedded energy characteristics. A geographic information system provided the basis for this analysis that included drinking water and wastewater flows and corresponding embedded energy values as well as service population. The results of this analysis sought to justify regional or national scale benchmarking for utility level comparison.

4.2 Results and Discussion

Primary flow and energy data were collected, organized, and analyzed from water utilities, representing a drinking water service population of 81.4 million and a wastewater service population of 86.2 million people. Of the 253 utilities for which data were requested, 76% responded with some form of data; Table 4.1 indicates the number of utilities that responded with data in each requested category. Monthly values for both water volume and embedded energy were available for 56 drinking water utilities and 70 wastewater utilities. The following sections discuss four key areas of results from the analysis of the database. The first section shows overall variations in the data across the country from an annual timescale, including national averages, non-revenue water, and a cumulative distribution function of both water fluxes and embedded energy. The second section evaluates and visualizes the data on a spatial scale. Next, intra-annual variations of the data across the country are analyzed. Finally, the impacts of the urban energy-water nexus and its implications on a national scale are discussed.

4.2.1 Annual Water Fluxes and Non-revenue Water

Many urban metabolism studies previously categorized the flux of water through an urban environment. In his seminal study, Wolman [122] described the water metabolism of a theoretical U.S. city of one million residents, estimating a drinking water consumption of 570 liters per capita per day (lpcd) and a wastewater discharge of 450 lpcd. A 1990 assessment of Sydney estimated 490 lpcd of drinking water consumption and 430 lpcd of wastewater discharge [133]. In comparison, the water and wastewater flux of U.S. cities, based on these primary data, were estimated to be 560 lpcd and 500 lpcd, respectively; see Table 4.1. However, these water fluxes are highly variable between cities. To compare the variances of the two fluxes, the relative standard deviation, the weighted standard deviation as a fraction of the mean (σ/μ), were computed. The relative standard deviation for wastewater volume (0.46) was greater than that of drinking water volume (0.37), illustrating a larger relative variation in wastewater versus drinking water fluxes. This difference was most likely due to large volumes of stormwater associated with combined sewers (carrying stormwater and wastewater) in large cities and varying amounts of inflow and infiltration. Of the 104 cities responding with wastewater flow data, 38 cities had combined sewer overflow permits from the U.S. Environmental Protection Agency [197]. Figure 4.1 shows a lower per capita flux of separated sewer discharge when compared to combined sewer systems. A non-parametric statistical test confirms a significantly larger per capita discharge for combined sewer systems over separated sewer systems ($p < 0.01$). However, there was no statistically significant difference in the embedded energies of the two systems. Chapter 7 further investigated the underlying drivers and indicators of variations in the urban energy-water nexus.

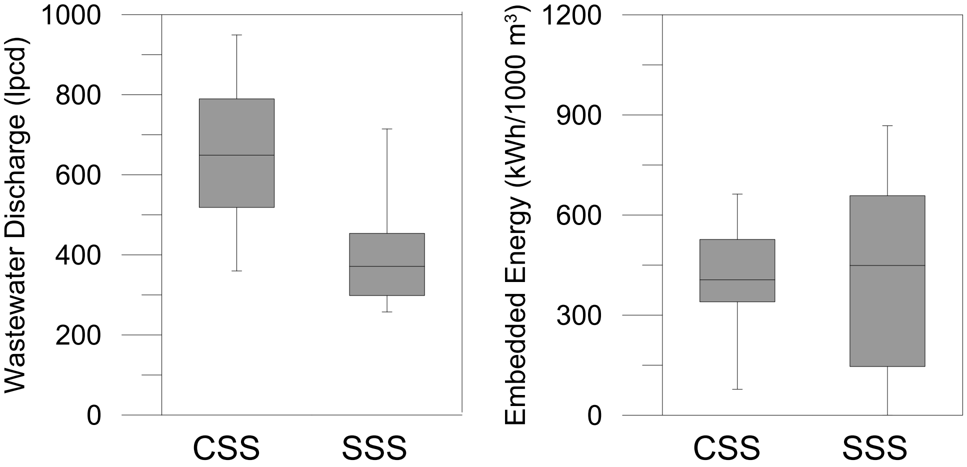

Figure 4.1: Cities with combined sewer systems have significantly more volume, per capita, entering their wastewater treatment plants than separate sewer systems.

Non-revenue water also provided an important layer of understanding to the flux of water in the urban water environment. Of the 16.7 billion m^3 per year of treated drinking water tabulated in this study, 15.7% of this volume was attributed to non-revenue water through primary data. The estimation of non-revenue water closely correlated to previous nationwide estimates by Thornton et al. [56]. Recalculating the mean and standard deviation of drinking water eliminating non-revenue water, resulted in μ =420 lpcd and σ =170 lpcd (n = 70). Non-revenue water accounted for a large portion, nationally, of treated drinking water and, therefore, has important implications in water security and efficiency efforts.

Table 4.1: On average, U.S. cities discharged 88% of their drinking water intake through the wastewater system, not accounting for combined sewers or infiltration and inflow. More energy was required per volume to treat wastewater than drinking water. Wastewater utilities, on average, recover about 14% of their required energy.

	Drinking Water		**Wastewater**		
	Flow (lpcd)	***Energy (kWh/1000 m^3)***	***Flow (lpcd)***	***Energy (kWh/1000 m^3)***	***Recovery (kWh/1000 m^3)***
Sample Size, n	89	73	104	90	45
Mean, μ	556	342	498	432	63
Std. Dev., σ	210	268	227	292	182
25th Percentile	449	148	326	363	48
50th Percentile	519	346	408	463	205
75th Percentile	645	499	641	648	303

No two cities were identical in their water fluxes or embedded energy. To better illustrate the variation of water flux and its embedded energy across the United States, cumulative distribution functions (CDFs) for drinking water, wastewater, and their embedded energy were created. Figure 4.2 shows CDFs for drinking water and wastewater volumes per capita, weighted by service population. The CDFs show a steeper slope when per capita water fluxes are clustered together, with a flatter slope indicating minimal clustering and possible outliers. For drinking water, Figure 4.2 (dark blue line), a majority of urban residents consumed between 400 and 750 lpcd, which centers around the calculated mean; see Table 4.1. The high standard deviation of the sample could be driven by per capita drinking water consumption above 1000 lpcd. Similarly, Figure 4.2 shows a steep slope between 300 and 600 lcpd of wastewater flux (brown line), which encompasses the calculated average; see Table 4.1. These CDFs showed the variability of water flux on a per capita basis. Pairing the individual city values for non-revenue water with their declared water fluxes, the CDF of drinking water excluding non-revenue water was recalculated. Figure 4.2 (light blue line) shows a shifted CDF with lower overall flows. However, there were minimal changes to the shape of the curve, indicating the minimal impact non-revenue water had on the variability of drinking water demand across the country.

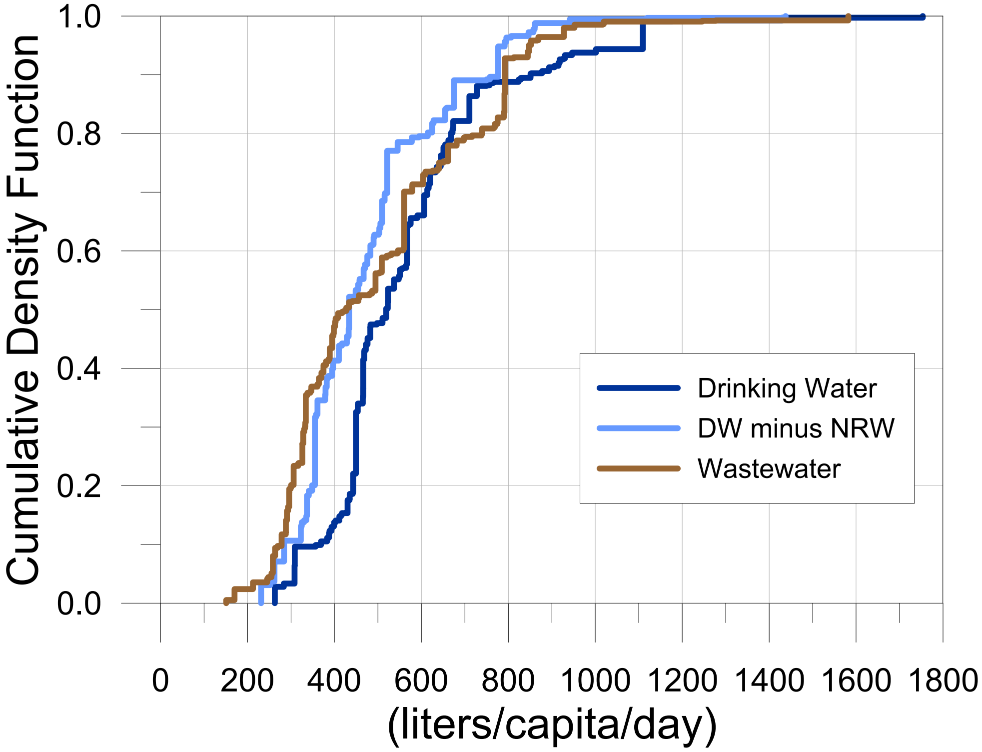

Figure 4.2: The dark blue and brown lines show the distribution of weighted, per capita drinking water and wastewater averages, respectively. The cumulative distribution function was based on the total service populations of utility data received. The light blue line shows the CDF for drinking water flux after removing non-revenue water (NRW).

Similar to water volume, Figure 4.3 shows a CDF of the embedded energy of drinking water and wastewater, weighted by total volume treated. The embedded energy in drinking water had a large slope between 100 and 550 kWh/1000 m^3 before jumping, suddenly, to a value of almost 1000 kWh/1000 m^3, indicating two clusters of energy-for-water consumption per volume among utilities. Similarly for wastewater, there was a steep slope centered around the mean from 300 to 600 kWh/1000 m^3 before a skewed right tail for the remaining 15% of wastewater treated. Finally, plotting the embedded energy of both drinking water and wastewater against their annual volume treated, Figure 4.4, showed minimal economies of scale associated with energy for treatment. In other words, larger drinking water and wastewater utilities did not necessarily have lower embedded energy for treatment and distribution/collection of their resources. This finding reinforced the need for a larger, national database to facilitate utility comparisons as utility size is not an adequate indicator of similar embedded energy.

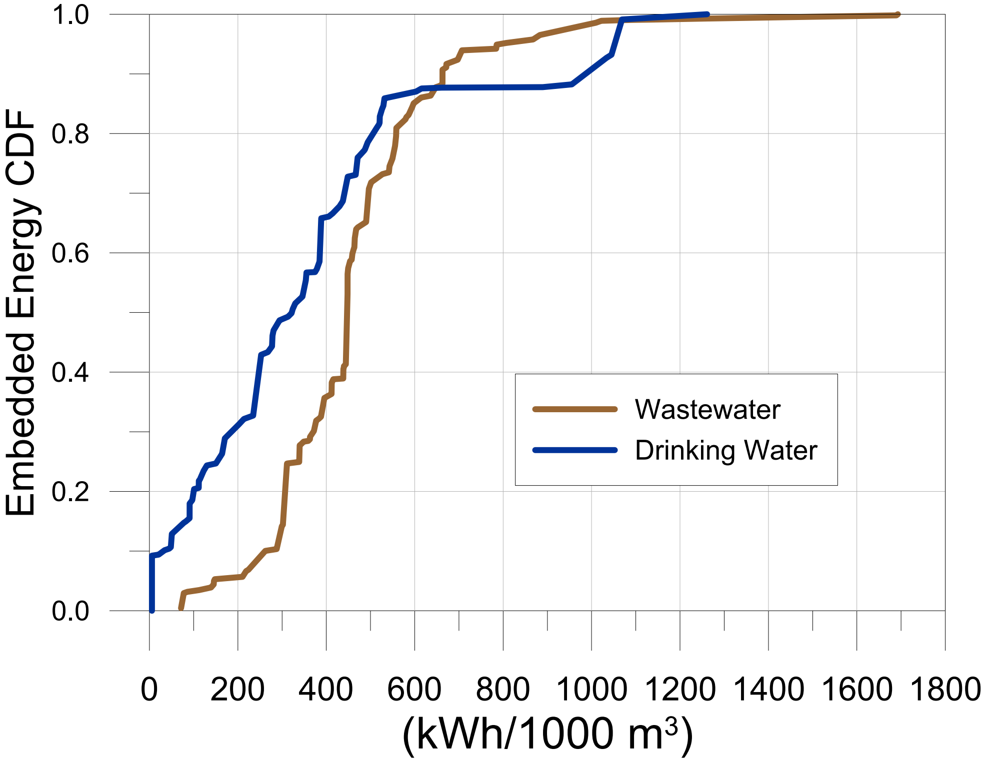

Figure 4.3: Cumulative distribution functions for the embedded energy of drinking water and wastewater resources showed a skewed right tail of the distribution where larger embedded energy is required.

4.2.2 Spatial Analysis of the Urban Energy-Water Nexus

While the CDFs in Figures 4.2 and 4.3 showed the variability of water flux on a per capita basis and embedded energy on a per volume basis, it was also necessary to visualize this variability on a spatial scale. Figures 4.5 and 4.6 display the spatial variability of drinking water and wastewater flux, respectively, across the U.S. cities in the database. The figures show both the volume of water and the embedded energy within the water resources of each city. Visually, there were minimal regional correlations associated with drinking water and wastewater fluxes and their embedded energy. Wastewater fluxes, Figure 4.6, in the northeastern portion of the United States were generally greater than those of the rest of the country, indicating a prevalence of combined sewer systems in a generally older portion of the country.

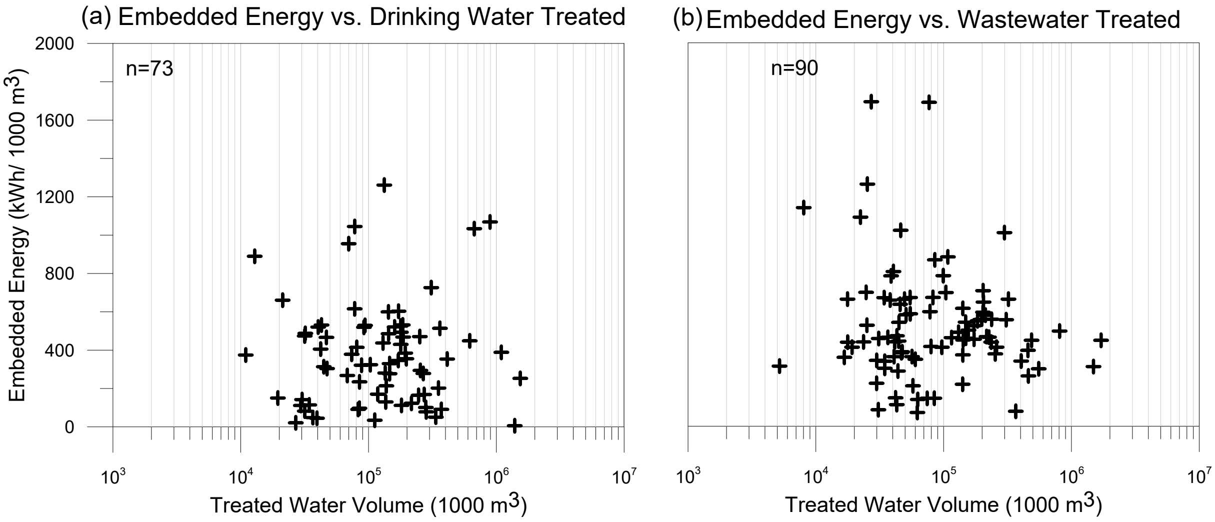

Figure 4.4: The embedded energy demand associated with total treated volume for both drinking water and wastewater did not suggest any economies of scale with regards to energy per treated volume.

The AWWA benchmarks water utilities based on four geographical regions [51], congruent with United States Census Bureau definitions. To determine the appropriateness of these geographical regions, a k-means clustering analysis was conducted on 4 different sets of data: (i) drinking water and embedded energy, (ii) wastewater and embedded energy, (iii) non-revenue water, and (iv) summer peaking factor. The summer peaking factor was the percent increase of the average summer month (June through August) over the average winter month (December through February). For drinking water, there is minimal regional clustering (optimal group number is 12), aside from the Northeastern United States. Wastewater analysis generally grouped utilities into the eastern and western half of the country, aligning with typical wetter/dryer climate patterns. Grouping based on non-revenue water percentage had an optimum of two groups: the Northeastern United States and rest of the country. There was generally lower non-revenue water in the western United States, Figure 4.7.

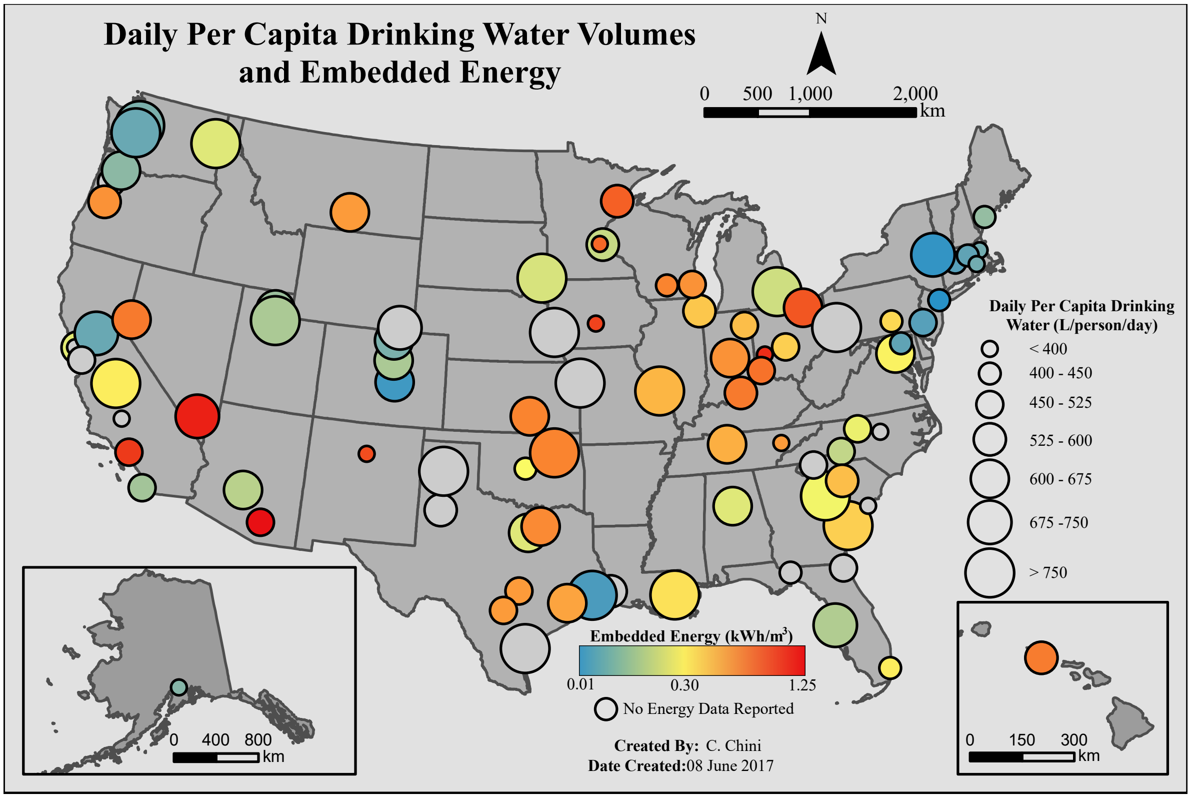

Figure 4.5: The volume of per capita drinking water varied across the country, with minimal correlation between geographic regions. The size of the circle on the map indicates the per capita water flux and the color of the circle represents its energy intensity. The color scheme ranges from blue to red, with blue indicating a lower embedded energy and red representing a higher embedded energy. A grey circle indicates a utility that responded with water volume but not required energy.

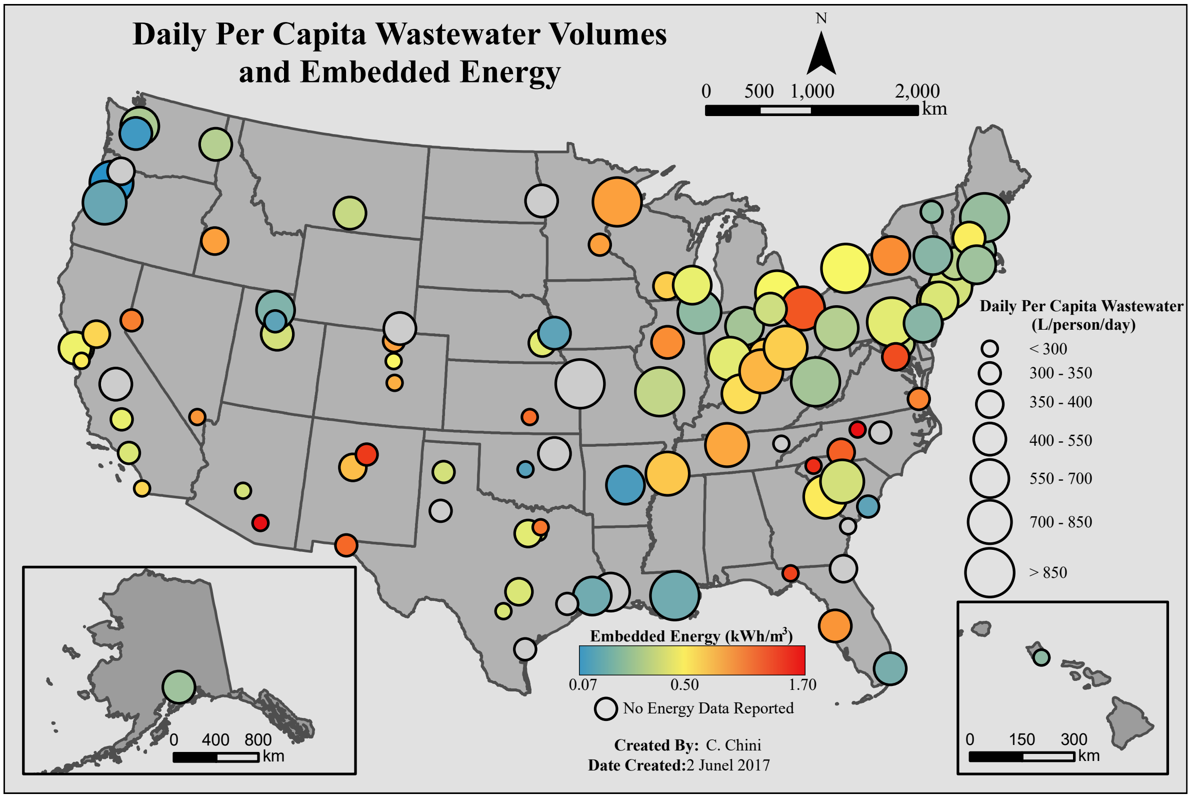

Figure 4.6: The volume of wastewater discharged varied across the country, but there are some instances of correlation with climate (such as in the southwest). Additionally, the embedded energy in wastewater was lower in the Northeast and Northwest. The size of the circle on the map indicates the per capita water flux and the color of the circle represents its energy intensity. The color scheme ranges from blue to red, with blue indicating a lower embedded energy and red representing a higher embedded energy. A grey circle indicates a utility that responded with water volume but not required energy.

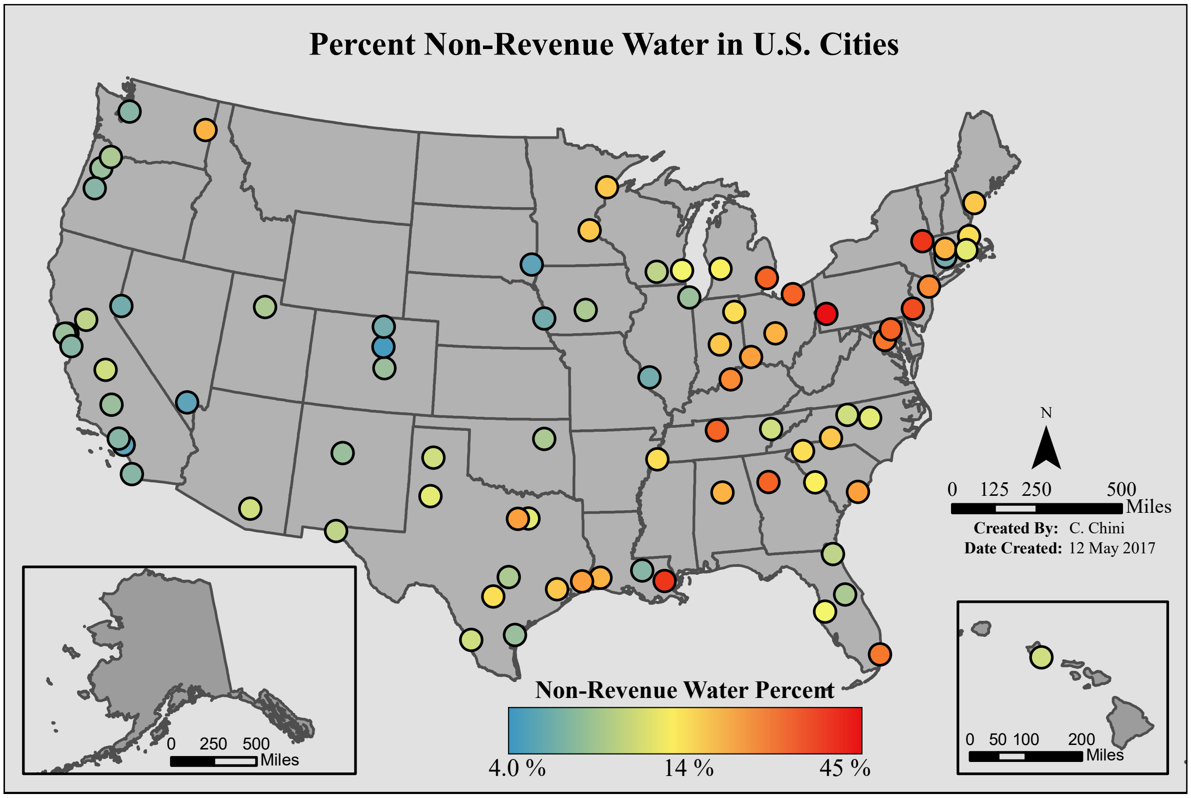

Figure 4.7: Non-revenue water tended to be lower in cities in the western half of the United States than in cities on the eastern half.

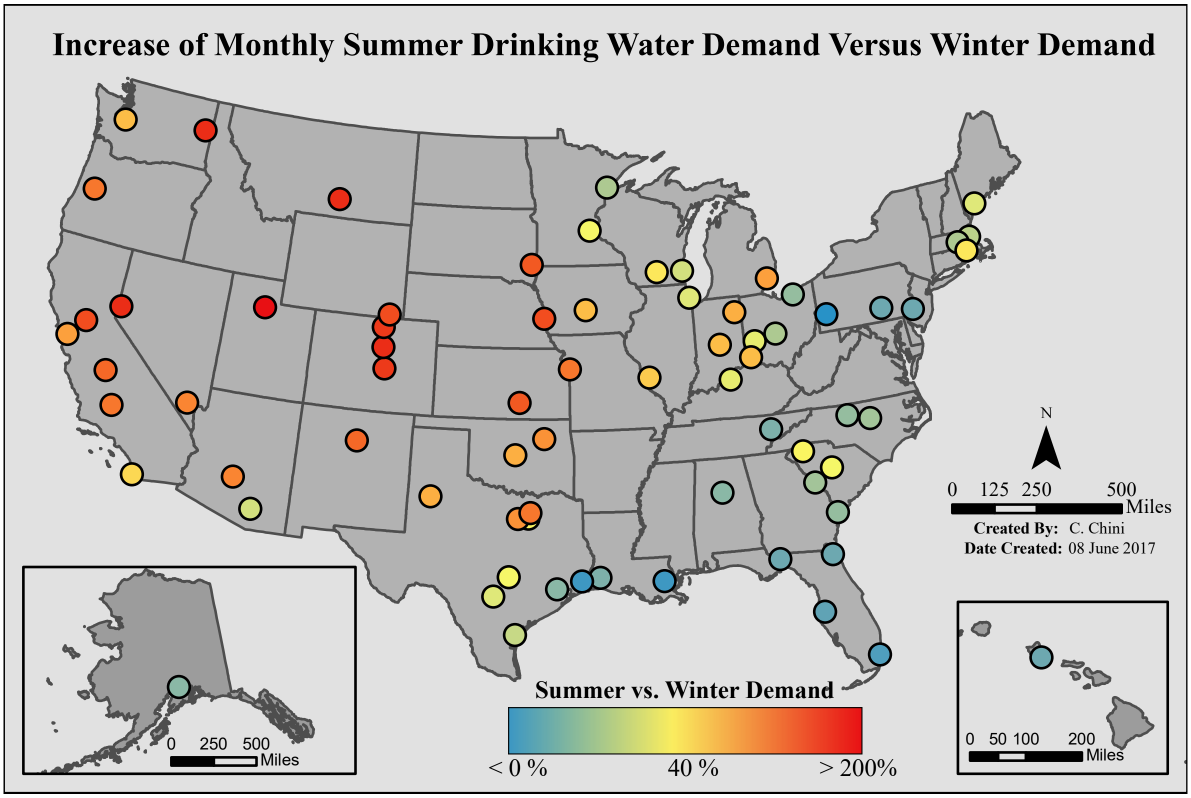

Figure 4.8: There were larger summer peaking factor in western cities than in eastern cities.

Clustering based on summer peaking factor grouped cities into two groups: (i) the eastern states, Texas, and Oklahoma and (ii) the western states. This grouping was visually apparent based on Figure 4.8. It is interesting, however, that cities in Texas and Oklahoma, with similarly arid climates to neighboring southwestern states, exhibited lower increases in summer water demand. Overall, multiple applications of the k-means squared spatial statistics test based on different values of the energy-water nexus revealed limited regional correlation. Outside spatial correlation, there were no visible trends of cities when comparing water flux and embedded energy to service population. Therefore, there was minimal statistical evidence to justify grouping of utilities on a regional geography basis or by utility size with respect to energy and water resource use.

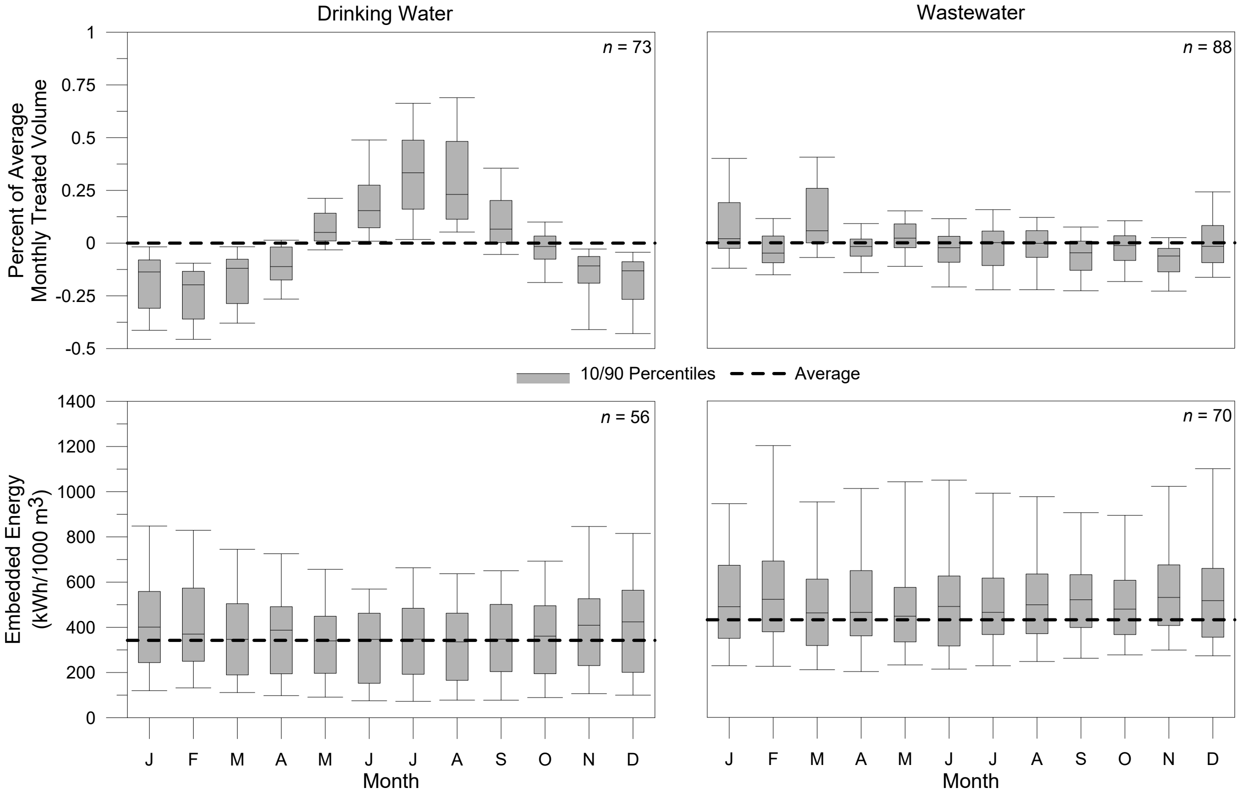

Figure 4.9: Drinking water volume increased across the country during the summer months, with minimal changes in wastewater volume and drinking water and wastewater embedded energy. Monthly treated water volumes were normalized for each city based on their average monthly flow and plotted based on their percent difference from the mean. Embedded energy, due to its inherent normalization based on total volume, is plotted as a strict average across all cities. The box-and-whisker plots show the monthly mean and the 10^{th} and 90^{th} percentiles.

4.2.3 Intra-Annual/Temporal Statistics of the Urban Energy-Water Nexus

Monthly volume and energy data were obtained from over 50 cities for drinking water and 70 cities for wastewater. This sample size provided opportunities to evaluate monthly changes in treated water volume and embedded energy for both drinking water and wastewater utilities. Figure 4.9 shows the monthly variations aggregated across the United States for drinking water and wastewater utilities. As one might expect, drinking water demand in the summer months was greater than the monthly average, with the winter months being lower. This heightened demand in the summer was most likely due to outdoor water demand [198]. The visibility of this difference is interesting at this geographical scale, Figure

4.8, and to this extent, considering the data did not exclude non-residential consumers and were aggregates of cities across the multiple climates in the United States. The visible demand differences in the data supported MFA capabilities to analyze urban water metabolism without large data sets of individual meter readings. Additionally, further supporting the outdoor watering hypothesis, monthly treated wastewater varied minimally across the year, Figure 4.9. Focusing on a few representative cities, Figure 4.10 shows four cities with a relatively constant wastewater discharge and large spikes in drinking water demand during the summer months. These four urban environments—Cheyenne, WY, Denver, CO, Salt Lake City, UT, and cities in North Texas—have generally dry climates.

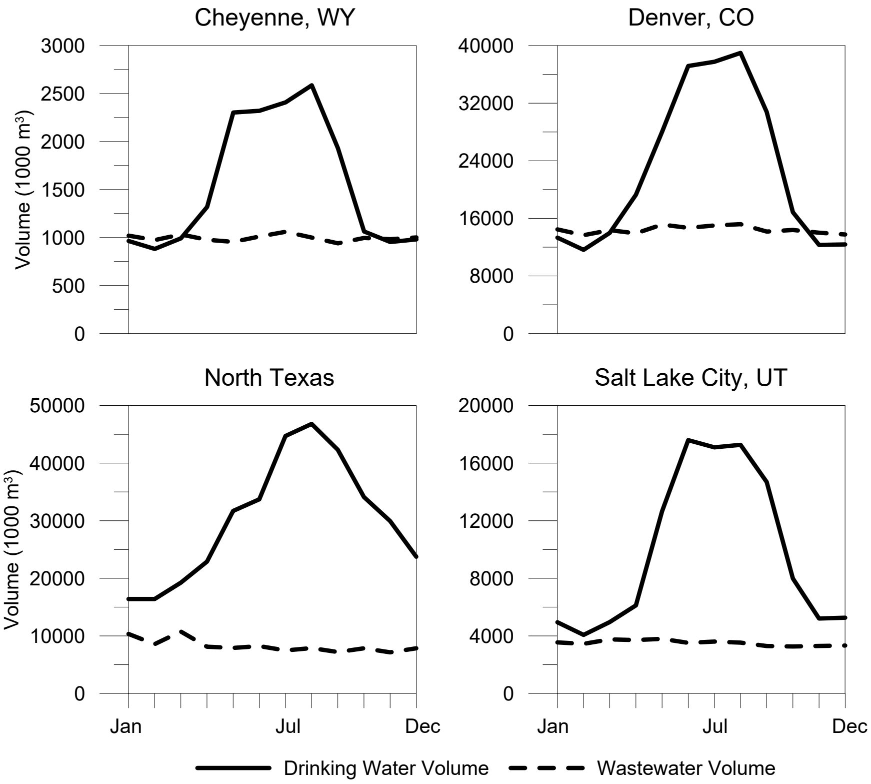

Figure 4.10: Cities in relatively dry climates exhibited large differences in summer drinking water demand and wastewater discharge. North Texas refers to cities served by North Texas Municipal Water District which services 24 communities that are north and east of Dallas including: Plano, Allen, Rockwall, and Frisco.

Figure 4.9 indicated relatively minor variation at a national level for average embedded energy of wastewater and drinking water resources. However, there was a very slight difference in drinking water energy as it lowers minimally during the summer months. When focusing on a few individual cities, this difference in embedded energy for drinking water became more pronounced, Figure 4.11. For cities in colder climates such as Anchorage, AK, Boston, MA, Colorado Springs, CO, and Salt Lake City, UT, there was a pronounced change of embedded energy in drinking water, lowering during the summer months, despite increases in demand. This difference was due to generally lower natural gas demands during the summer months, as natural gas was predominantly used for heating drinking water and treatment facilities. For cities in warmer climates, such as Dallas, TX and Oklahoma City, OK, embedded energy increased during the summer months coinciding with an increase in drinking water demand. Therefore, it was important to consider not just secondary energy (electricity) in the treatment of water resources, but to also consider primary energy (natural gas and fuel oil) when developing a database of energy and water for cities.

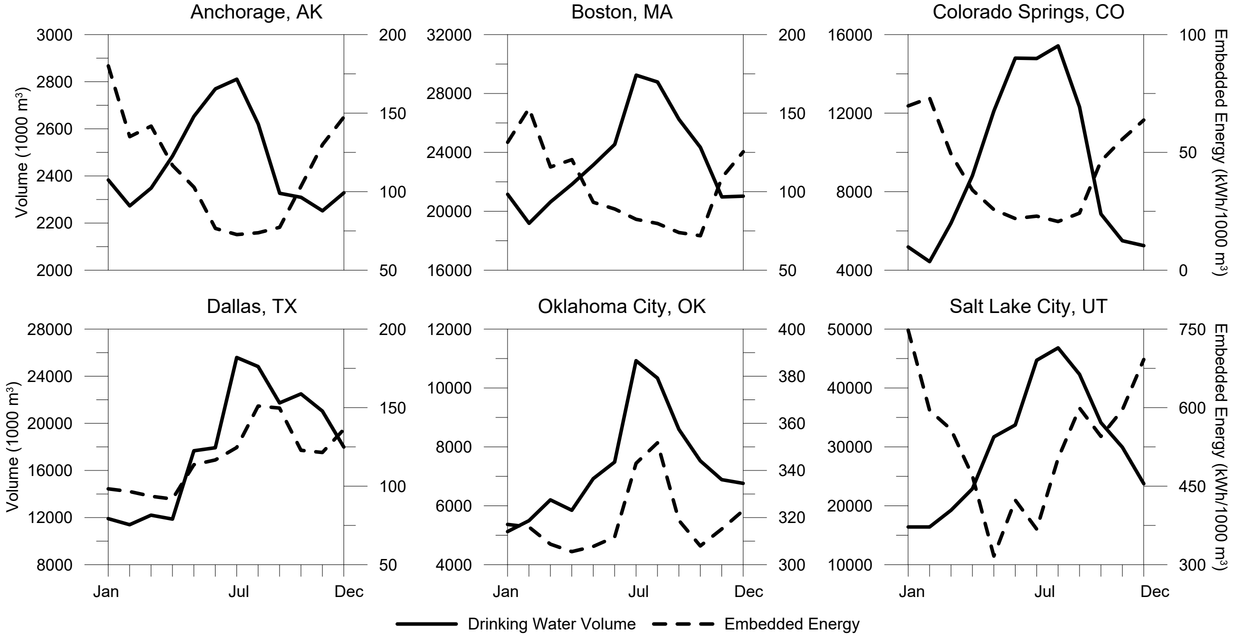

Figure 4.11: Cities in colder climates that use natural gas as part of their treatment and distribution processes for drinking water had much higher embedded energy in the winter than in the summer. The shapes of these curves show the importance of natural gas in the embedded energy of some cities' drinking water.

4.2.4 Impacts of the Urban Energy-Water Nexus

The latest water use survey by the U.S. Geological Survey estimated that 86% of the population was served by centralized drinking water systems [43]. Additionally, about 74% of the U.S. was served by centralized wastewater systems [199]. Acknowledging that there was wide variation in embedded energy within drinking water and wastewater resources, embedded energy were extrapolated to the estimated U.S. population served by public utilities. The primary data for drinking water resources represented 81.4 million people (30.6% of the population served by centralized drinking water systems) and accounted for 16.7 billion m^3 of water (28.9% of total public supply, 2010 estimate) [43, 200]. Extrapolating based on both population and volume yielded a total electricity consumption of 18,800 GWh and 19,900 GWh per year, respectively, for urban drinking water supply, treatment, and distribution. Additionally, wastewater data accounted for 86.2 million people, 37.0% of the population served by centralized wastewater utilities [200]. Extrapolating based on population yielded an estimate of 18,200 GWh per year of electricity consumption for urban wastewater collection and treatment. Combined (37,000–38,100 GWh/yr), the extrapolated energy consumption for U.S. drinking water and wastewater utilities accounted for approximately 1.0% of the total 2012 electricity consumption of the United States [201]. While extrapolating nationally from this sample size of approximately 30% was not ideal, the results were comparable to previous estimates by Liu et al. [113] and James et al. [112], which utilized much smaller sample sizes for their extrapolations. These results increased the robustness of national estimates with the large sample of primary data.

Using similar assumptions of extrapolation, there was a significant impact of non-revenue water. Extrapolating to all public water supply using the calculated 15.7% average, there was an estimated 9.1 billion m^3 of water lost as non-revenue water in the United States, annually. This water loss was equivalent to the water demands of 44.5 million average U.S. urban residents for one year. Additionally, using the calculated average embedded energy and non-revenue water, the United States wasted approximately 3,100 GWh per year through water loss, equivalent to a 360-MW power plant running at full capacity for a year or the annual electricity consumption of nearly 300,000 average U.S. households [202].

Reducing non-revenue water will have substantial benefits towards increasing sustainability of the urban energy-water nexus.

4.3 Broader Impacts

With increasing stress on resources, there was a need for improved resource accounting through data to ensure equitable and safe access of necessary resources for both humans and the environment. Previous studies determined that water resource fluxes through the urban environment dominate and comprise approximately 90%, by mass, of all flows [10, 122, 135]. However, these studies often evaluate one city at a specific point in time. Some studies do evaluate cities temporally, across multiple years, [e.g., 125], showing an increasing urban metabolism; there were no temporal studies of metabolism within a given year (seasonal analyses). Additionally, there are few studies that combine the energy-water nexus and material flow analysis [10, 145, 146]. A monosectoral approach to urban metabolism is insufficient for material and sustainability analyses [5]. This necessary inclusion of principles from the energy-water nexus provided an additional level of understanding and decision making. This objective demonstrated the relevance of input/output frameworks such as MFA to evaluate characteristics of cities with respect to their water flux. The study analyzed the urban energy-water nexus on annual, intra-annual, and spatial scales, providing an important first step in cataloging the trends of U.S. urban water resources and evaluating the effectiveness of urban water conservation and sustainability policies across the country.

Anthropogenic water consumption occupies a central component of the global water system [203, 204], especially considering the large human impact in urban areas. Determining values of anthropogenic urban water flux fills an important knowledge gap associated with the global water system. The study particularly emphasized the relationship of urban water flux and its embedded energy, both primary and secondary energy sources. Studies of the energy-water nexus continue to grow in the literature [81, 205]. It is necessary to promulgate this trend in data collection efforts at a utility level with open access data. These important metrics provide opportunities for academia, utilities, and government to develop and improve the understanding of the urban water cycle. Future studies should include all sources of energy to fully quantify the urban water flux and evaluate the urban energy-water nexus.

Moving forward, there are significant future opportunities for sustainability and resilience studies and initiatives associated with holistic, national analyses of urban water and embedded energy. Expansion of collected data could include on-site electricity generation through biogas turbines, solar panels, or alternative energy sources. Additionally, water quality and water source data would make large contributions to the analysis of urban water and its embedded energy. Water source information could also include water reuse, which is important in creating a circular and sustainable urban economy as opposed to the predominantly linear inputs and waste discharges [206, 207]. Water reuse and recycling have many implementation challenges, including energy intensity. However, recycled water use for non-potable applications remains relatively low in the United States, even for water stressed cities such as Tuscon, AZ (10% of total water supplied) [193] or San Diego, CA (3% of total water supplied) [208].

This objective provided first assessment of the state of the U.S. urban energy-water nexus to study changing demands throughout a year. It is necessary to continue collecting these data either as part of academic studies or as a funded, central database (Chapter 3). The difficulty of acquiring these data necessitate open access data efforts with utilities and either academic, professional, or governmental organizations. To help advance open access efforts in the energy-water nexus, all of the data were published in an open forum through HydroShare by CUAHSI [193]. In the future, periodic updates of the nation's water through an expanded database would allow for continual urban water studies to assess its sustainability. Increases in the collection of these urban water data has significant potential benefits for management of infrastructure and sustainability goals [209]. Gleick [210] highlights data collection as an important strategy in advancing water policy at the national level. Ideally, trends would develop that show decreasing urban water use over time across the entire country, consistent with urban metabolism definitions of sustainability [9].

CHAPTER 5

THE COMPARATIVE WATER FOOTPRINT OF U.S. CITIES

> Comparative urban water research is important for drawing generalizations about sustainability and for adapting lessons from one set of cities for consideration in others.
> –Karen Noiva, et al. (2016) [27]

The water footprint of the urban environment is not limited to direct water consumption (i.e., municipal or local water supplies); embedded water in imported resources, or virtual water transfers, provides an additional component of the urban water footprint. Water resources are integral to the sustainability of an urban system [9, 10], with understanding human modifications to the water cycle important for determining water scarcity [11]. Water footprints measure humans' appropriation of fresh water resources to support a population [33, 34]. The embedded water in these resources carries important implications for food and water security [33, 62]. Using empirical data, this work extended traditional urban water footprinting analysis to compare direct and indirect urban resources for the United States. The indirect component of the urban water footprint includes water indirectly consumed through energy and food, relating to the food-energy-water nexus. This analysis comprehensively quantified the indirect water footprint for 74 metropolitan statistical areas through the combination of various databases, including the Commodity Flow Survey (CFS) of the U.S. Census Bureau (UCSB), the U.S. Department of Agriculture (USDA), the Water Footprint Network, and the Energy Information Administration (EIA).

The inclusion of multiple sectors of the urban water cycle and their underlying processes provided important insights to the overall urban environment, the interdependencies of the

---

This chapter was based on work published as: Chini, CM, Konar, M, and Stillwell, AS (2017) Direct and Indirect Water Footprints of the United States. *Water Resources Research*: 53(1), 316–327.

food-energy-water nexus, and water resource sustainability. Previous studies have not examined the embedded energy or emissions as part of the urban water footprint, nor have they comprehensively analyzed all of the cities within a country. Both direct and indirect water require some amount of energy to either directly treat water and wastewater or transport goods from their origin to the urban environment. The energy required for water footprints is an important component of the energy-water nexus and the urban water cycle. Previous research quantified energy consumption in the water sector at regional and national scales [104, 111, 211, 212]. Both the consumed water and its embedded energy provided important topics of discussion when comparing water footprints of urban environments. Taking the analysis one step further, the carbon emissions associated with embedded energy provided an extra layer of information about the energy-water nexus and the urban water cycle. Spatial heterogeneity in both direct and indirect water footprints was also analyzed, determining the average urban water footprint in the United States to be 1.64 million gallons of water per person per year (6,200 m^3/person/year or 17,000 L/person/day), dominated by indirect water.

5.1 Methods

Prior to discussing methodologies associated with computation of water footprints, it is important to declare geographical system boundaries and the methodological scope. For the purposes of this study, the consumptive direct water boundary was the water that left a drinking water treatment plant and entered the public water supply. This definition of consumptive water was determined as wastewater discharges often occur downstream of the drinking water intake for cities. The water was, therefore, removed from its original source without full replacement at the same location from the individual city perspective. The term 'consumption' was utilized to retain language consistency between direct and indirect water footprints, especially regarding water consumption for electricity. Furthermore, this definition provided an upper bound estimate of the direct water footprint of cities. Direct water system boundaries were determined as the service boundaries of the primary water utility provider of the municipality. Indirect water system boundaries, however, corresponded to the metropolitan statistical areas (MSAs) of the U.S. Census Bureau. These discrepancies in system boundaries were unavoidable and, therefore, required mediation to make appropriate comparisons between the direct and indirect water footprints; see Figure 5.1. Both direct and indirect water footprints were normalized by their respective service populations. For MSAs that extend across multiple states, only counties in which the main city's state was located were considered to remain closer to the boundary of utility districts. The CFS [194] was the limiting factor in the study due to its pentannual publication, narrowing the study to the year 2012 as a representation of recent conditions. For embedded resources accounting, energy for water and emissions associated with energy are included. The study focused on a holistic framework for understanding system interdependencies, reserving detailed life cycle assessment for smaller scale studies. Figure 5.2 provides an overall methodological depiction of the scope of the urban water footprint.

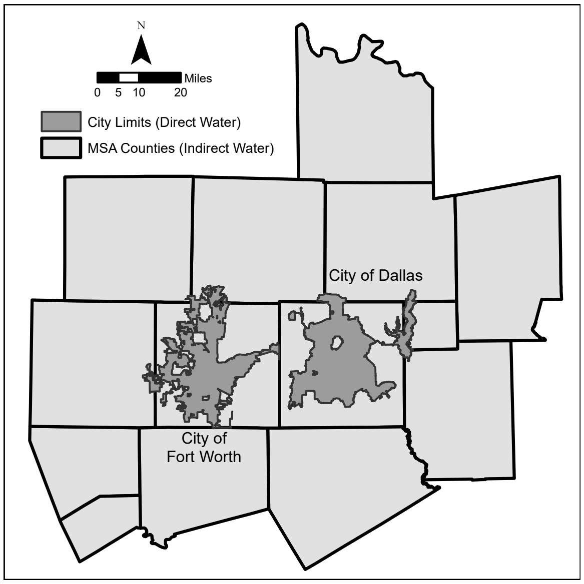

Figure 5.1: Example of differences in utility boundaries and metropolitan statistical areas (MSAs) for the City of Fort Worth and the City of Dallas (Texas). Note that city limits were used to obtain direct water footprints and the MSAs were used to estimate indirect water footprints. Due to discrepancies in these spatial domains, each footprint was normalized by the population of each geographical unit to facilitate comparison.

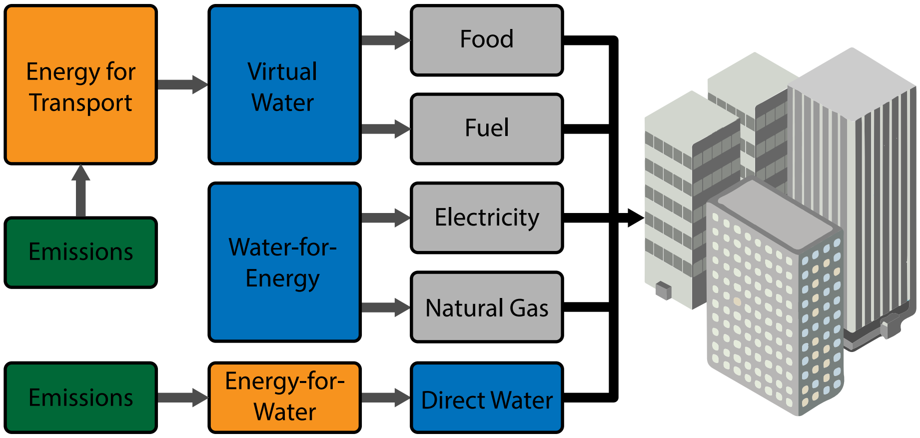

Figure 5.2: Schematic of water and energy resources used to quantify the total water footprint of urban areas in this study. Black arrows represent direct inputs to the city, and grey arrows represent embedded resources. Importation of food and fuel resources leads to indirect water consumption through virtual water, while electricity and natural gas consumption require water-for-energy.

5.1.1 Direct Water Footprints

Direct water in urban environments was assumed to originate from water utilities with minimal contributions from other potential sources of water consumption such as rainwater capture or private well water. Annual pumping and treatment data from open records requests for water utilities provided the necessary accounting for direct water, see Chapters 3 and 4. These records requests represented MSAs as defined by the U.S. Census Bureau and asked for both water volume and energy data for the treatment and distribution of potable water. MSAs such as San Jose-San Francisco-Oakland or Dallas-Fort Worth received multiple requests for each of the municipalities' major drinking water distributors. For the purposes of this study only annual data were necessary. Therefore, daily or monthly data were aggregated into annual values. The aggregated annual water consumption were then normalized by provided service population.

Energy data from the open records requests were similarly aggregated to annual totals, electricity and natural gas consumption were converted to secondary energy in terms of equivalent killowatt-hours (kWh) of consumption based on Equation 5.1, and the energy were then normalized based on 10^6 U.S. gallons (Mgal) of water treated. Imperial units were utilized to correspond to the predominant methods of accounting water resources in the U.S.

utilities. Energy data correspond to the energy required for water extraction, purification, and pumping, where available.

$$E[kWh_{eq}] = e + 0.45 \times (29.3 \times n) \tag{5.1}$$

where, e refers to electricity in kWh and n represents the amount of natural gas in therms. A factor of 0.45 was used to account for efficiency losses between primary and secondary energy, similar to Chini et al. [55].

For the purposes of this study, emissions calculations for drinking water treatment were limited to the embedded energy within drinking water. The purpose of this study was not to perform a full life-cycle assessment on drinking water production and distribution, but to provide a framework for future high-level evaluations of the urban water cycle. Literature quantifying emissions from drinking water production and distribution varies widely depending on the type of treatment system and the quality of raw water [213], with one paper estimating that energy demands account for only 33% of total emissions from treatment processes with the remainder from chemicals in the treatment process [214]. Energy emissions were estimated based on state level emissions data for electricity and natural gas from the EIA [215]. Therefore, the estimation of emissions for drinking water production and distribution was expected to be highly conservative. This assumption, however, corresponds with the boundary scope of the indirect virtual water emissions that only considered emissions from transporting goods (discussed in Section 5.1.2).

5.1.2 Indirect Water Footprints

The components of indirect water were distinguished by data source and labeled: virtual water and water-for-energy. The calculation of these indirect water footprints of the urban environment relied upon the combination of empirical data sets. Virtual water required the CFS and other data sets, while the water-for-energy calculation relies upon the EIA's database. The CFS is a collaboration between the Bureau of Transportation Statistics of the U.S. Department of Transportation (USDoT) and the Census Bureau (USCB) to provide information about the movement of commodities within the United States [194]. The survey tabulates commodity transfers by origin, destination, value, weight, and mode of transportation. This pentannual survey groups transfers based on the Standard Classification of Transported Goods (SCTG) for food, fuel, manufacturing, electronics, and other goods. Table 5.1 shows the groups in the CFS for food and fuel. These data, in conjunction with the methodology employed in Dang et al. [65], provided the foundation for estimating indirect water footprints of cities. However, Dang et al. [65] only included 5 of the 7 food commodity groups within the commodity flow survey, excluding "agricultural products" and "other prepared foodstuffs" (SCTG commodity groups 3 and 7). The study expanded upon this methodology by accounting for the remaining two food commodity groups and fuel commodity groups, (SCTG commodity groups 15-18, refer to Table 5.1). SCTG commodity group 19 ("other coal and petroleum products") was not included due to ambiguity in assigning a virtual water content (VWC) to the products. VWC equals the crop evapotranspiration per crop yield [216], equivalent to the water footprint of each food commodity [33]. Generally, VWC for food commodity groups varies by state of origin, but VWC was on the national or regional scale for fuel products.

For these eleven commodity groups, the VWC of each commodity group was determined based on state of origin using the methodology in Dang et al. [65]. This methodology assumed that the VWC of each commodity group was equal to a weighted average of individual quantities produced by the respective state. For SCTG commodity groups 1-5 and 7, the VWC was calculated based on food production amounts in each state from the United States Department of Agriculture's (USDA) Census of Agriculture [2012]. These agricultural census data are then coupled with the individual food items' VWC described in various databases [67, 219, 220]. When possible, state specific values for virtual water were utilized and national averages were substituted when not available. Equation 5.2 describes the calculation of VWC of each commodity group, adapted from Dang et al. [65].

$$VWC_{c,s} = \frac{\sum_{i\epsilon c}^{I}[(\text{Green}VWC_{i,s} + \text{Blue}VWC_{i,s}) \times \text{Production}_s]}{\sum_{i\epsilon c}^{I} \text{Production}_s} \tag{5.2}$$

where c indicates commodity group, i indicates item, I indicates number of items within c,

$i \epsilon c$ indicates items contained in commodity group c, and s indicates state of production. For the virtual water footprint of cities, both the green and blue virtual water content values were included. Table 5.2 shows the various individual agricultural items included in each commodity group. The VWC for SCTG commodity group 6 ("milled grain products") is directly from Dang et al. [65] without update.

Table 5.1: The Commodity Flow Survey provided information about transfers of goods using the Standard Classification of Transported Goods (SCTG) including food and fuel commodities [217].

SCTG	Full Commodity Group Name
1	Animals and fish[a]
2	Cereal grains (including seed)[a]
3	Agricultural products[b]
4	Animal feed, eggs, honey, and other products of animal origin[a]
5	Meat, poultry, fish seafood, and their preparations[a]
6	Milled grain products and preparations, and bakery products[a]
7	Other prepared foodstuffs, fats, and oils[b]
15	Coal[c]
16	Crude petroleum[c]
17	Gasoline, aviation turbine fuel, and ethanol[c]
18	Fuel oils[c]
19	Other coal and petroleum products not elsewhere classified[d]

[a] "Staple" food commodity groups as defined in Dang et al. [65]
[b] Additional food commodity groups not included in Dang et al. [65]
[c] Fuel commodity groups included
[d] Fuel commodity group not included

The second component of indirect water considered was water-for-energy. In addition to the fuel sources identified in the Commodity Flow Survey, Table 5.1, electricity and natural gas were also included. Various literature sources provide the embedded water resources required for extraction, processing and refining of fuel commodity groups. Mielke et al. [221] estimated the water consumption of coal to be 6 gallons per MMBtu. Generalizing for 19.5 MMBtu per short ton of coal in the year 2012 [222], this conversion equated to a value of 117 gallons per ton of coal. A study by Argonne National Laboratory [223] estimated water intensities of crude oil and gasoline by region, with a United States average of 4.8 gal/gal of crude and 4.6 gal/gal of gasoline, equivalent to 1,100 gal/ton and 7,200 gal/ton, respectively. Fuel oils were assumed to have similar water intensities as crude petroleum.

Table 5.2: The virtual water contents of several different types of crops and livestock are combined to generate aggregate virtual water contents of the commodity groups.

01 Animals	**02 Cereal Grains**	**03 Agricultural Products**		
Bovine	Wheat	Potatoes	Lettuce	Peanuts
Swine	Corn	Sweet Potatoes	Oranges	Rapeseed
Sheep	Rye	Dried Beans	Grapefruit	Sunflower Oil
Goats	Barley	Cabbage	Bananas	Sunflower Seeds
Broiler Chicken	Oats	Cantaloupes	Peaches	Cottonseed
Turkey	Grain Sorghum	Sweet Corn	Pear	Canola
Chicken	Rice	Tomatoes	Apples	Guava
Fish	Millet	Onions	Soy Beans	Papaya

04 Animal Feed, Eggs	**07 Other Foodstuffs**
Eggs	Milk
Hay	Cheese
Alfalfa	Butter
	Cane Sugar
	Sugar Beets
	Cottage Cheese
	Dry Whey

The electricity and natural gas consumption of an urban population is difficult to estimate as electricity grids and gas distributors do not align with political jurisdictions, although there have been recent efforts to determine more spatially refined water footprints [224]. The smallest unit that the Energy Information Administration (EIA) estimates is at the state level. Therefore, the study assumed that per person averages for electricity consumption and the water consumed for electric power generation at a state level adequately represent the embedded water in electricity consumed at the urban level. EIA Form 923 provides electricity generation and water consumption on a production plant level [225]. Since water consumption for hydroelectric power is reported as zero in the EIA database, values for gross water consumption through evaporation for hydropower generation were utilized from Grubert [91]. Similarly, natural gas consumption was normalized on a per person level based on state level estimates from the EIA [226], with embedded water estimated at 3 gallons per MMBtu [85].

The embedded energy of indirect water provided a metric to determine the geographic extent of the water footprint of a city. There were two components of embedded energy: (i) energy for virtual water, and (ii) energy for water-for-energy. The embedded energy in the water-for-energy component comes from pumping cooling water at thermoelectric plants or flowback from natural gas extraction, with these values negligible and highly variable, respectively. Therefore, only the embedded energy of the virtual water component (refer to Figure 5.2) was considered. For this study, the control volume was drawn around the transportation of finished goods for the embedded energy calculation. The Commodity Flow Survey provided commodity information based on transportation type and ton-miles [194]. The Center for Transportation Analysis [227] provided annual estimates of energy (Btu) per ton-mile equivalents for the various modes of travel for the year 2012 (Table 5.3).

Table 5.3: Truck freight traffic requires the greatest energy intensity per ton-mile [227], and the combined truck and air traffic has the greatest global warming potential.

Mode of Travel	**Energy Intensity Btu/ton-mile**	**Global Warming Potential kg CO_{2e}/ton-mile**
Truck Freight[a]	3711	0.1983
Train Freight	294	0.0783
Barge Freight	210	0.0760
Combined Air and Truck[b]	2003	0.9224

[a] Assumes a standard truck weight of 5.8 tons

[b] Averages air and truck energy and emissions intensities per ton-mile

The modes of travel could be further used to determine the emissions of the commodities flowing into the urban system. The EcoInvent database (v3.1) implemented in SimaPro (v8.0.4; PRé Consultants; The Netherlands) provided climate change characterization factors for each mode of travel in equivalent kilograms of CO_2 per ton-mile (kg CO_{2e}). Characterization factors for climate change were available through the U.S. Environmental Protection Agency's (EPA) Tool for Reduction and Assessment of Chemicals and Other Environmental Impacts (TRACI) [228]. Table 5.3 presents values of Btu per ton-mile and climate change characterization factors for each mode of travel.

5.2 Results and Discussion

At the time of writing, data for direct water footprints were available for a population of approximately 47.2 million people, while the estimates of indirect water footprints accounted for 197.2 million people, or 15.0 and 62.8 percent of the 2012 population of the United States, respectively [229]. The large difference in service population was due to the inclusion of suburbs within the metropolitan statistical area for indirect water that often have separate water systems from the main city (refer to Figure 5.1). Additionally, there was a disparity in the number of cities represented by direct water data versus indirect water data (33 versus 74), at the time of writing the chapter. Table 5.4 provides the mean, standard deviation, minimum, and maximum values for direct, indirect, and total urban water footprints.

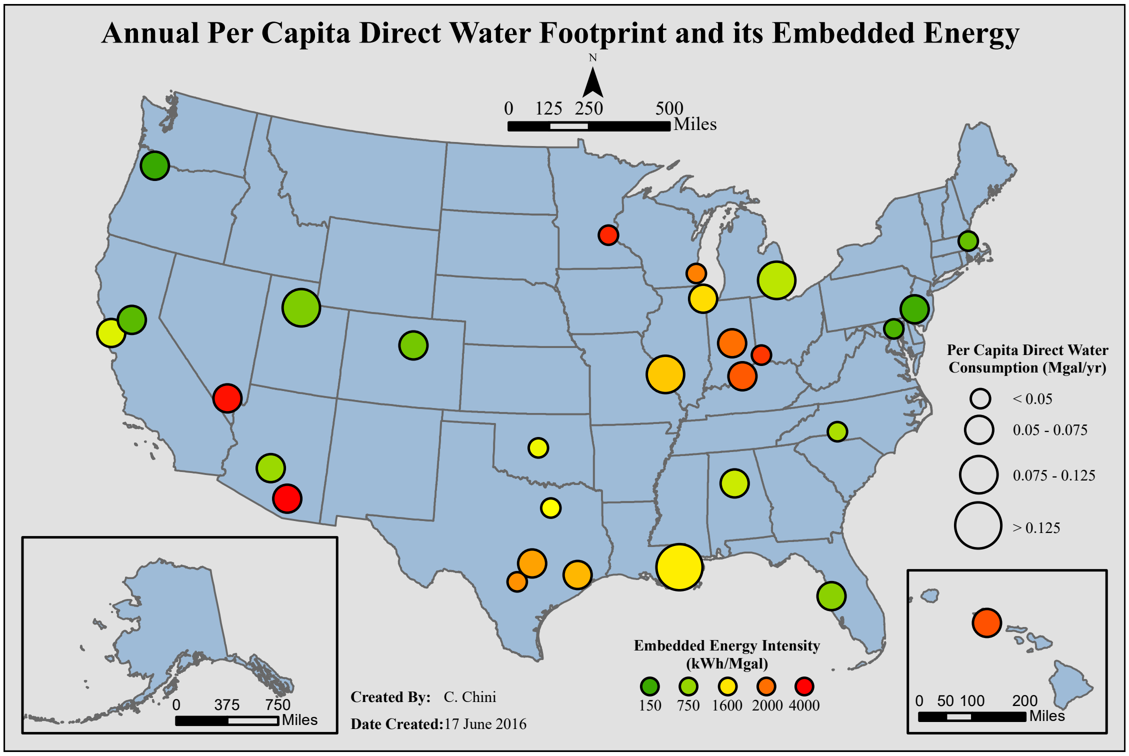

Figure 5.3: Map of per capita direct water footprints for urban areas of the United States. Note that the size of the circle indicates the volume of the annual per capita direct water footprint (ranging from <0.05 to >0.125 Mgal) and the color of the circle indicates the embedded energy intensity (ranging from 150 to 4,000 kWh/Mgal). Direct water footprint information was restricted to 29 urban areas with both direct water footprint volume and energy information available. This figure is a subset of Figure 4.5.

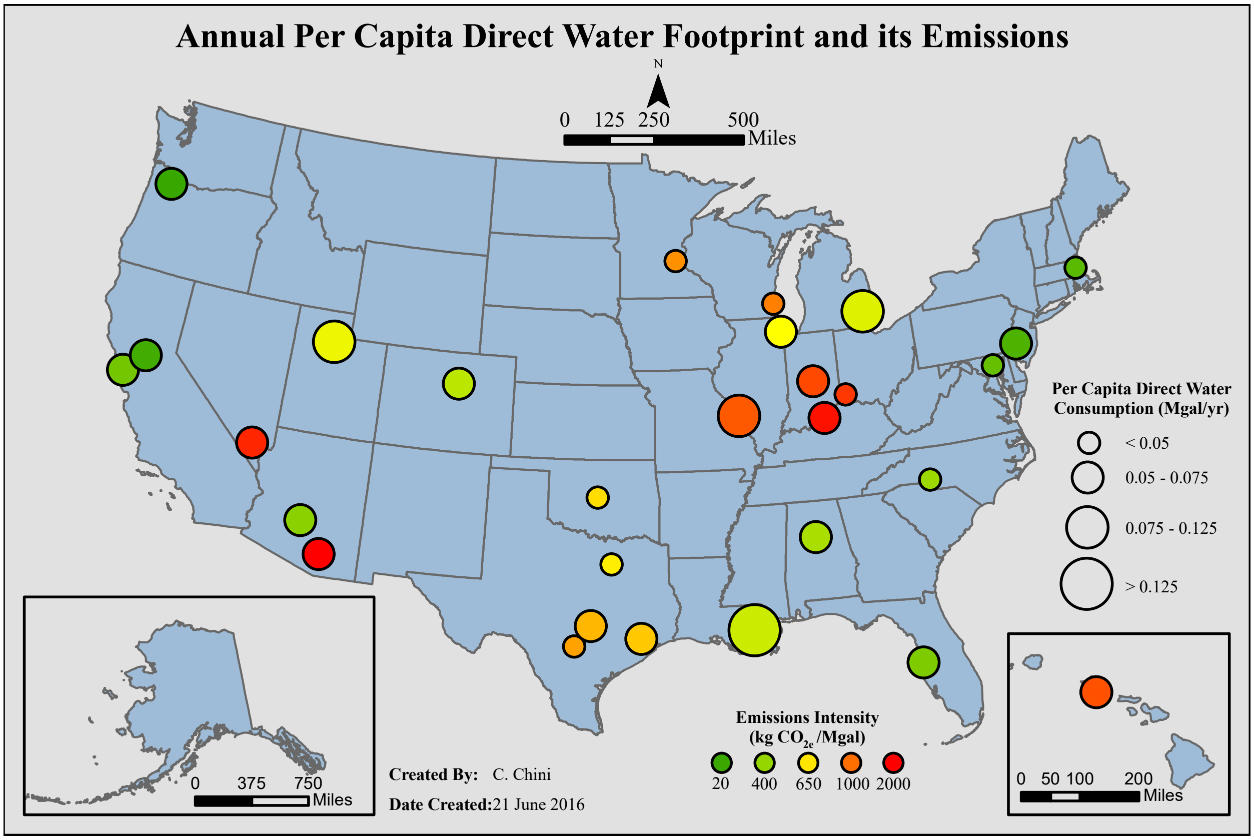

Figure 5.4: Map of emissions as a result of the embedded energy in drinking water resources.

5.2.1 Direct Urban Water Footprints

Direct water footprints and their embedded energy varied widely with respect to geography; see Figure 5.3. Embedded energy for direct water were presented in terms of kWh as opposed to indirect water's embedded energy in MMBtu. At the time of writing, only 29 of the 74 MSAs studied returned drinking water volume and energy through open records requests. Of these 29 cities, the population-weighted average (μ_{pop}) direct water footprint was 58,200 gallons (0.058 Mgal) per person per year, equivalent to 162 gallons per day; see Table 5.4. However, the direct water footprint was highly variable, ranging from 29,700 to 137,880 gallons per year (0.030 to 0.138 Mgal per year), equating to a range of 81 to 378 gallons per day for the cities of Boston and New Orleans, respectively. The center of the country from Texas through Chicago tended towards higher energy intensities with lower embedded energies along the coasts. This trend had two notable exceptions of Las Vegas, NV, and Tucson, AZ, with very large embedded energies (4,000 and 4,700 kWh/Mgal, respectively). In contrast, the average embedded energy for the reporting cities across the country was 1,425 kWh/Mgal, with a standard deviation of 1,091 kWh/Mgal. Additionally, the associated emissions for embedded energy in direct water were 800 kg CO_{2e}/Mgal with a standard deviation of 625 kg CO_{2e}/Mgal. The spatial variability of the emissions are shown in Figure 5.4. The identified trend in the direct water footprint and its embedded energy would benefit from an expanded sample size.

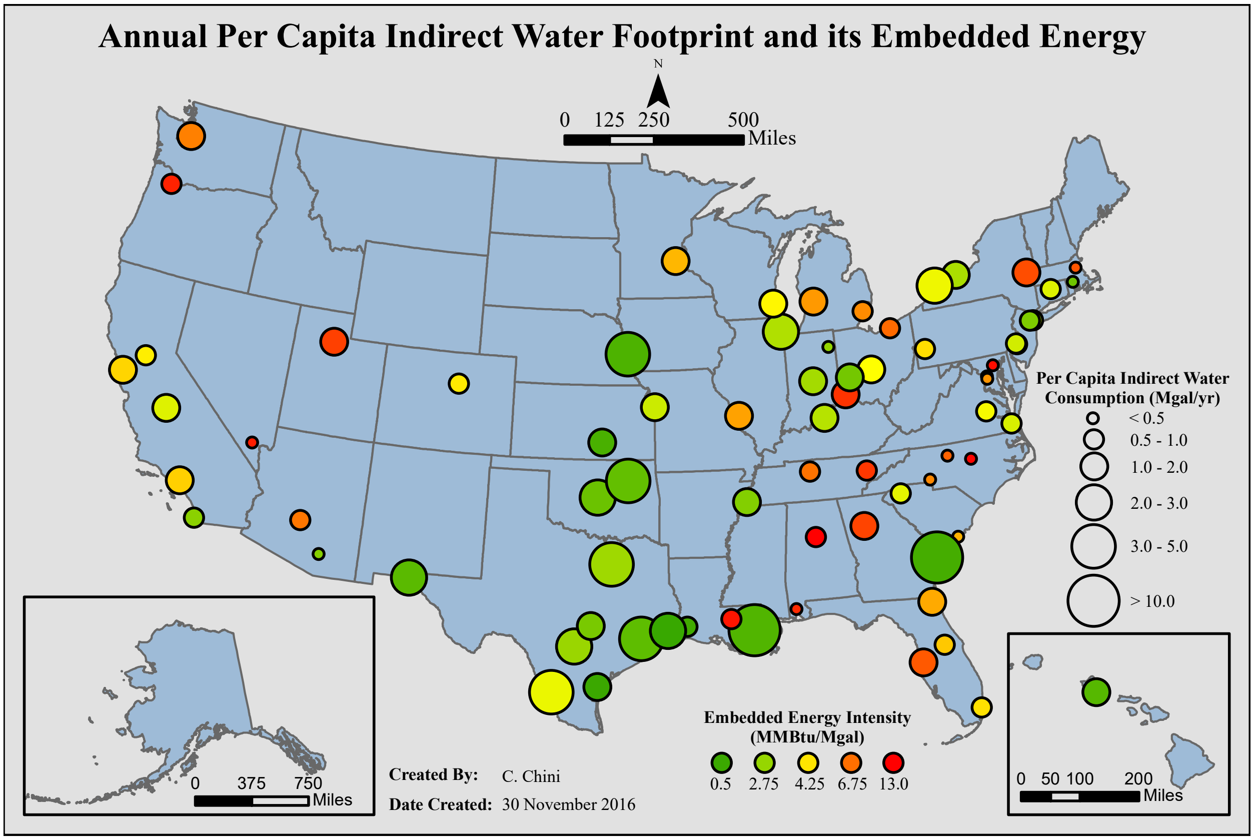

Figure 5.5: Map of per capita indirect water footprints for urban areas of the United States. Note that the size of the circle indicates the volume of the annual per capita indirect water footprint (ranging from <0.5 Mgal to >10 Mgal) and the color of the circle indicates the embedded energy intensity (ranging from 0.5 to 13.0 MMBtu/Mgal). Cites in the center of the United States tended to exhibit a higher indirect water footprint. Indirect water footprints are comprehensively quantified for all 74 metropolitan statistical areas of the United States.

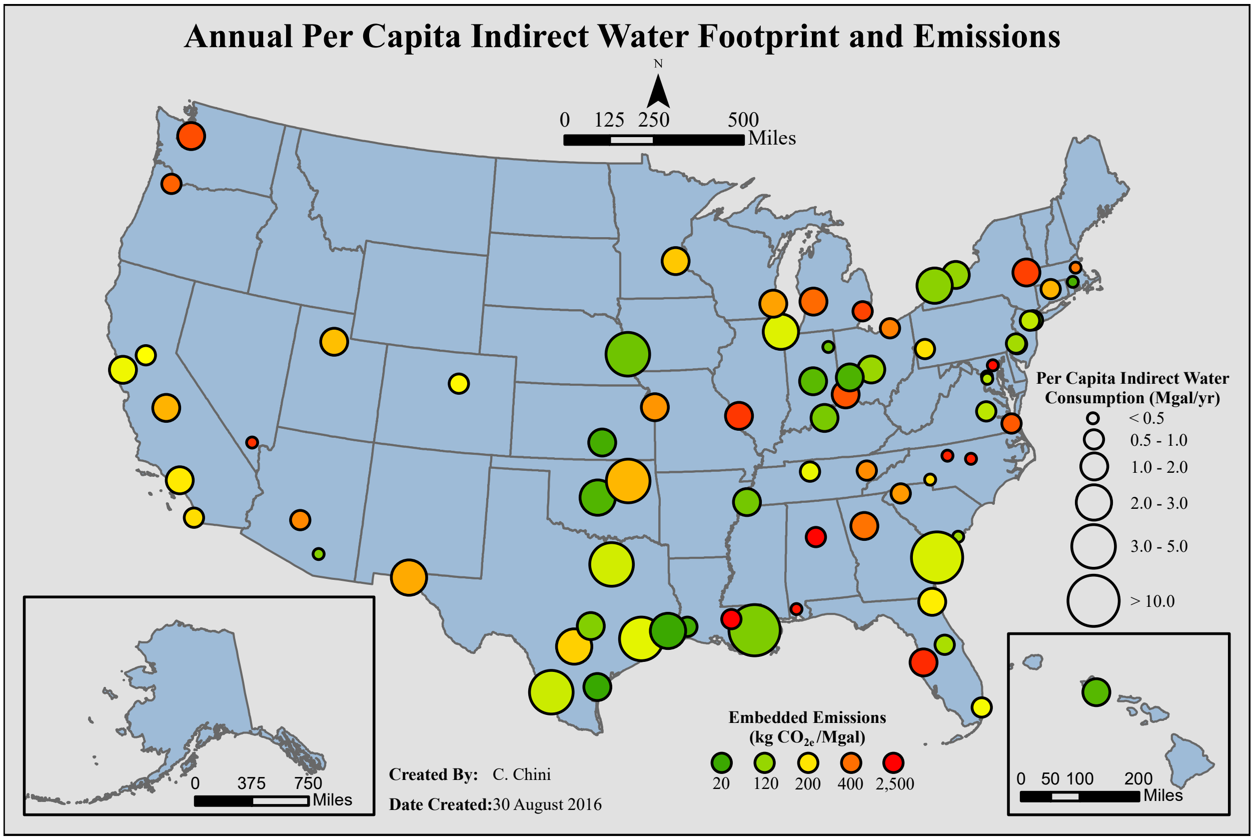

Figure 5.6: Map of emissions as a result of the embedded energy of indirect water footprints of cities.

5.2.2 Indirect Urban Water Footprints

Indirect urban water footprints have water resources originating from outside the area of the city, unlike the predominantly local direct water footprints. Therefore, there were further opportunities for analysis including network and spatial variability. Figure 5.5 shows the per capita indirect water consumption and its embedded energy. New Orleans, LA, and Savannah, GA, both have large per capita indirect water footprints (16.8 and 10.2 Mgal/year, respectively), much greater than the population-weighted average (μ_{pop}) indirect water footprint of 1.33 Mgal/year; see Table 5.4. These high water footprints were both port cities with relatively smaller populations than other port cities, such as Boston, New York, Seattle, or Los Angeles, leading to greater, per capita virtual water inflows. Additionally, relatively large indirect water footprints occurred in the states of Texas and Oklahoma and in Omaha, NE, indicating inflows of goods with higher virtual water contents (i.e., meat products). Interestingly, there was some significant clustering of embedded energy within regions. The southeastern United States tended towards a greater embedded energy within indirect water resources. Additionally, the corridor from Nebraska through Texas exhibited a lower than average embedded energy. This finding was due to a lower travel distance for food and fuel as well as lower intensity modes of travel through the corridor, such as trains. Nationally, embedded energy within indirect water footprints has high variability (μ = 4.6 and σ = 3.0 MMBtu/Mgal). Variability in emissions of the indirect water footprint are similar to the embedded energy; see Figure 5.6. The average embedded emissions from transport of goods for each unit of indirect water is 450 kg CO_{2e}/Mgal with a standard deviation of 480 kg CO_{2e}/Mgal.

As previously stated, the composition of indirect water has four components: i) food and ii) fuel footprints, iii) electricity, and iv) natural gas. Virtual water of food and fuel dominated the composition of the indirect water footprint of cities, which makes sense, as water consumption and not withdrawals are a part of water footprints. The food virtual water footprint constituted 87.6% of the total indirect water footprint and that of fuel constituted an additional 11.9%; see Table 5.4. The water-for-energy required for natural gas and electricity comprised, on average, less than 0.5% of the overall indirect water footprint. The water-for-electricity consumption was heavily dominated by hydroelectricity demands. In states with high evaporation rates and large contributions of hydroelectricity to the energy portfolio, such as those in the Southwest or Southeast United States, the water consumed for hydroelectricity increased. However, the contribution of hydroelectricity to the grid was less than 7% for the United States, yielding high variability of the water-for-electricity footprint. The cities of Mobile, AL, Las Vegas, NV, Tucson, AZ, and Greensboro, NC, had much greater contributions to indirect water footprints from water-for-electricity than the average at 22.2%, 11.5%, 7.8%, and 6.1%, respectively. Many of the other cities in the study have contributions of less than 1%. Using data from the EIA and Grubert [91], the average, by state, embedded water in electricity was 940 gallons per MWh. While this value only considers electricity generation and not heat generation, it is less than 25% of the global water footprint value of 4,100 gal/MWh determined for net electricity and heat generation by Mekonnen et al. [230]. The population-weighted averages of the food, fuel, and overall indirect water footprint were lower than that of the strict average of all cities. This trend indicated that MSAs with larger populations have lower per capita indirect water footprints.

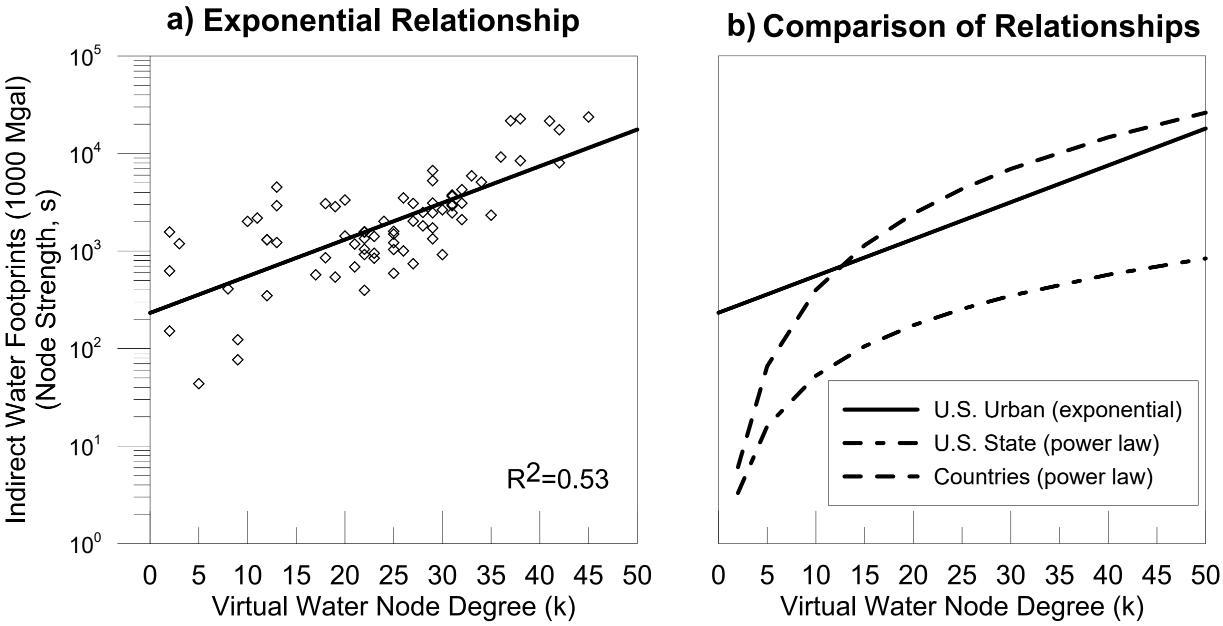

Figure 5.7: The relationship between node strength and node degree for virtual water of cities departed from the power law relationships previously presented in the literature [e.g. 47, 65]. Instead, the relationship between node strength and node degree for urban areas was best represented by an exponential distribution.

Network analysis was employed to compare the node strength and node degree of the indirect water footprint for each of the 74 cities; see Figure 5.7. Previous virtual water studies identified a power law relationship between node strength and node degree for U.S. states [65] and countries [47]. These studies found the exponent for the U.S. states (1.72) to be smaller than the global network exponent (2.6). The presented study was at a finer spatial resolution (urban scale), and the exponent for the best-fit power law is further reduced (1.05). However, as this power law fit is nearly linear, an exponential distribution was determined to be the best approximation of the relationship between urban node degree and strength (see relationship on semi-log plot in Figure 5.7a). Therefore, the trend of a decreasing exponent with geographically smaller nodes continued, leading the power law relationship to break down and yield an exponential fit, Equation 5.3.

$$s = 231.97 \times e^{0.087k} \tag{5.3}$$

Figure 5.7b compares the functional relationships between node degree and node strength for U.S. cities, U.S. states, and countries. The exponential relationship for cities indicated that they were more efficient at obtaining virtual water resources with fewer commodity exchange partnerships (note that the exponential function produces higher node strength, s, values for values of node degree, k, less than 10 in Figure 5.7b). However, fewer exchange partnerships might leave urban areas more vulnerable to disruptions to their supply chains. This analysis highlighted the need for further research to evaluate trade-offs between network efficiency and vulnerability, as well as the scaling properties of commodity and virtual water exchanges.

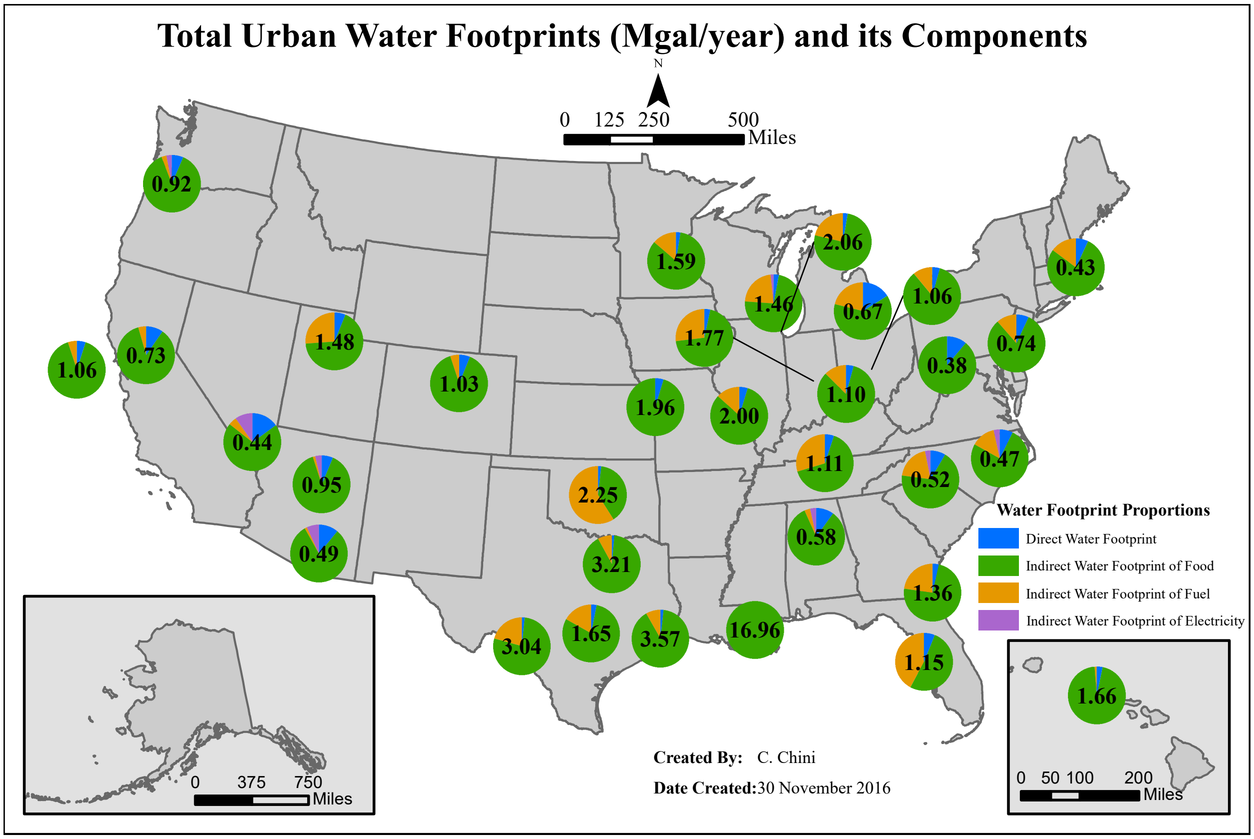

Figure 5.8: Map of total urban water footprint and the contributions of direct and indirect water. Indirect water dominated the water footprint of a city, with food contributing to the bulk of the water footprint. The map was restricted to 33 cities for which direct water footprint volume data are available.

5.2.3 Total Urban Water Footprints

The total urban water footprint was computed as the sum of the direct and indirect water footprints. Figure 5.8 shows the contributions of the indirect and direct water footprints to a city's total footprint and the values, in Mgal/year, of the total urban water footprint. On average, the indirect water was twenty times that of direct water; see Table 5.4. The per capita contributions for both direct and indirect water did not sum to the total water footprint due to the differences in sample sizes. Utilizing the water footprints and associated embedded energies, urban residents within the United States, on average, consumed 1.64 million gallons of water per year, nearly 4,500 gallons per person per day. The total urban water footprint provided a benchmark for urban or regional authorities to create sustainability goals and lower their total water footprint. For example, cities in water-rich environments might focus on lowering indirect water footprints and water-strained cities might focus on direct water. Table 5.5 provides a ranking of the top 5 urban areas with largest and smallest water footprint.

Table 5.4: Urban water footprints were dominated by indirect water; the population weighted average μ_{pop} was lower than the strict average (μ), indicating a lower per capita consumption in cities with a larger population.

		Water Footprint (Mgal/person/yr)				
Urban Water Footprint	**n**	μ_{pop}	μ	σ	V_{min}	V_{max}
Indirect Water Footprint	74	1.34	1.58	2.27	0.09 (Mobile, AL)	16.83 (New Orleans, LA)
Food Virtual Water	74	1.17	1.36	2.25	0.05 (Charleston, SC)	16.80 (New Orleans, LA)
Fuel Virtual Water	74	0.16	0.21	0.34	<0.01 (Newark, NJ)	2.46 (Tulsa, OK)
Direct Water Footprint	33	0.058	0.059	0.022	0.030 (Boston, MA)	0.138 (New Orleans, LA)
Total Water Footprint	33	1.64	1.82	2.84	0.38 (Baltimore, MD)	16.97 (New Orleans, LA)

To consume this volume of water, the required annual energy consumption was approximately 6 MMBtu per person of primary energy for transportation and 80 kWh per person of electrical energy for water treatment and distribution. While this individual energy consumption was relatively small, these values when scaled to a national scale, representing a

U.S. urban population of over 250 million, constituted significant energy investment. The results of the total urban water footprints illustrated the significant water and energy requirements to support urban environments within the United States.

Table 5.5: The analysis uncovered a wide range of urban water footprints across the United States.

	Largest Urban Water Footprints)		**Smallest Urban Water Footprints)**	
Rank	City	(Mgal/person)	City	(Mgal/person)
1	New Orleans, LA	16.97	Baltimore, MD	0.38
2	Houston, TX	3.57	Las Vegas, NV	0.43
3	Dallas, TX	3.21	Boston, MA	0.46
4	San Antonio, TX	3.04	Raleigh, NC	0.48
5	Oklahoma City, OK	2.25	Tucson, AZ	0.50

5.3 Summary

When studying the urban water footprint, quantification of both the direct and indirect virtual water components was important when considering the food-energy-water nexus. Additionally, embedded resources and emissions provided an essential layer of evaluation and understanding for the overall urban water footprint. The study was the first to comprehensively characterize the water footprint of all cities within the United States, leading to three major conclusions: (i) indirect water dominated the total flow of water into an urban environment, (ii) reductions in energy consumption can be realized in both the direct and indirect water footprints, and (iii) benchmarking of total water footprint might inform policy and management of the urban water cycle. These conclusions provided further direction for study of the urban water cycle and its embedded energy.

Indirect water, comprised of virtual water and water-for-energy, dominated the urban water cycle on a per capita basis. On average, indirect water was an order of magnitude greater than direct water consumption. Additionally, virtual water imports associated with food and fuel dominated the indirect water footprint over the water-for-energy component of indirect water. Understanding indirect water and its sourcing is essential for detecting vulnerabilities in the urban water supply [71, 231]. The results of the comprehensive study further supported the need for indirect water calculations to be included in urban water accounting and policy considerations. See Chapter 8 for further discussion on the role of indirect water in policy.

The analysis provided important insights into the food-energy-water nexus at the urban scale, creating opportunities for understanding water and energy savings and efficiency. To promote policy and management of the urban water cycle and its embedded energy and emissions, an overall understanding of the intra-national variations in the urban environment was necessary. The large-scale geographical comparisons of the urban water cycle presented in this analysis provided unique insights that evaluating a single city does not afford. By comprehensively analyzing many urban systems, a foundation for future research to address questions of urban water resources sustainability was built. Evaluating multiple cities in a singular effort, with a unified methodology, enabled future benchmarking and other policy objectives to evaluate the urban water cycle and the urban food-energy-water-nexus.

CHAPTER 6

EMBEDDED ENERGY OF URBAN METABOLISM

> . . . [S]ustainable urban futures will require a fundamental transformation of existing production and consumption patterns in cities...
>
> –Vanessa Castan Broto, et al. (2012) [232]

In the 50 years since Abel Wolman first published an estimate of U.S. urban metabolism, the field of urban metabolism has begun to thrive, with cities outside the United States being much of the focus. The current federal perspective of climate change in the United States has led to a rise in cities and their mayors stepping to the forefront of U.S. climate and sustainability policy. As cities attempt to meet local and international sustainability goals, it is time to revisit the metabolism of cities within the United States. Using existing empirical databases for material flows (the Freight Analysis Framework (FAF)) and a published database on urban water flux, this objective provided a revised estimate of urban metabolism for the typical U.S. city. Median values of metabolism were estimated for a city of one million people for water resources, food, fuel, and construction materials. To facilitate a second generation of urban metabolism, the study extended traditional analyses to include the embedded energy required to facilitate consumption of commodities. This extra layer of urban metabolism has important implications in determining sustainable urban metabolism.

What constitutes a sustainable urban metabolism? Current speculations on sustainable urban metabolism require a decrease in energy, water, and material flux over time regardless of population increases [9]. However, this definition focuses specifically on the physical and direct material flows of an urban environment, requiring an additional layer to urban metabolism: the embedded energy required to facilitate these resource fluxes. This under-

---

This chapter was based on work submitted as: Chini, CM and Stillwell, AS (2018) The Metabolism of U.S. Cities 2.0. *Journal of Industrial Ecology*: submitted standing of indirect or embedded resources in a city's metabolism is suggested as a necessity in advancing to the second generation of urban metabolism studies [124]. A few studies combine the energy-water nexus and urban metabolism [10, 145, 146] but do not extend the embedded energy to other materials. This analysis focused on the embedded energy of urban metabolism utilizing a similar approach to embedded energy, as in Chapter 5, through transportation of goods and treatment of water resources as another indicator of sustainable urban metabolism.

Finally, a lack of available data leads to urban metabolism studies that focus on a singular aspect or material flow in an urban metabolism study [116]. An approach to urban metabolism that only considers a single sector of materials is inadequate [5]. For instance, by mass, water flux accounts for approximately 90% of flow through an urban environment [122, 135, 136]. However, to understand the role of water in the city, it is necessary to study the flows of energy and all materials entering the city [147]. The study relied upon two open-access U.S. databases: (i) the published inventory of urban water fluxes from Chapters 3 and 4, and (ii) the Freight Analysis Framework (FAF) for food, fuel, and other materials.

6.1 Methodology

6.1.1 Material Flows

Two openly available databases of varying geospatial scale, one for water and another for other materials, provided the basis for evaluating urban metabolism at a singular year, 2012. The urban water database developed in Chapters 3 and 4 was utilized for urban water fluxes and their embedded energy. Other material flows were obtained using the FAF, a pentannual macroeconomic analysis of freight travel between states and metropolitan statistical areas (MSAs) across the United States [233]. These data aggregated material flows into and out of an MSA by the Standard Classification of Transported Goods (SCTG), which accounts for 42 categories of materials [194, 217]. These 42 categories range from food to building materials to electronics. Using the MSA as the urban boundary, see Section 6.1.3, the material consumption was calculated as the difference between flows into and out of the

MSA, assuming negligible storage within the MSA for the time period of interest. These categories were combined into four groups (food, fuel, construction materials, and waste) to evaluate major material flows of the urban environment; see Table 6.1. There were 71 MSAs from the FAF database that also had water data from Chapters 3 and 4. Consumption of goods was estimated from the difference between aggregated exports and imports within each category. Median values for material consumption were reported to provide the best estimate of urban metabolism amid data uncertainty.

Table 6.1: The Commodity Flow Survey was divided into four basic categories: food, construction materials, primary fuels, and waste.

Material Category	SCTG Code	Standard Classification of Transported Goods
Food	01	Animals and Fish
	02	Cereal Grains
	03	Agricultural Products
	04	Animal Feed, Eggs, Honey and Other products of animal origin
	05	Meat, Poultry, Fish, Seafood, and their Preparations
	06	Milled Grain Products and Preparations, and Bakery Products
	07	Other Prepared Foodstuffs, Fats and Oils
Construction Materials	10	Monumental or Building Stone
	11	Natural Sands
	12	Gravel and Crushed Stone
	13	Other Non-Metallic Minerals not-elsewhere identified
	14	Metallic Ores and Concentrates
	25	Logs and Other Wood in the Rough
	26	Wood Products
	32	Base Metal in Primary or Semi-Finished Forms and in Basic Shapes
	33	Articles of Base Metal
Primary Fuels	15	Coal
	16	Crude Petroleum
	17	Gasoline, Aviation Turbine Fuel, and Ethanol
	18	Fuel Oils
	19	Other Coal and Petroleum Products
Waste	41	Waste and Scrap

6.1.2 Embedded Energy

Embedded energy determinations required distinct methods between water and other materials. Electricity and natural gas are the predominant embedded energy sources for drinking water and wastewater (Chapter 4). Electricity as a secondary energy source is incomparable with the predominantly primary energy source of embedded energy for other materials. To normalize the embedded energy (in kilowatt-hours of electricity, therms of natural gas, gallons of fuel oil), state-level efficiency factors determined the equivalent energy in gigajoules (GJ) per kWh, Equation 6.1:

$$\text{Energy}[GJ] = c_{state}\left[\frac{GJ}{kWh}\right] \times e + 0.105\left[\frac{GJ}{therm}\right] \times n + 0.158\left[\frac{GJ}{gal}\right] \times f \tag{6.1}$$

where c_{state} is the weighted average of gigajoules to kWh based on fuel composition for electricity for each state, e is the consumed electricity, n is the consumed natural gas in therms, and f is the consumed fuel oil in gallons. Determining state-level values is consistent with previous studies on embedded resources [e.g., 55] and existing policies on low-carbon on renewable fuels standards.

Methods for determining embedded energy resources in materials other than water mirrored those in Chapter 5. The FAF provided transportation type and ton-miles for each import into the city [194]. Combining these data with annual estimates of energy per ton-mile equivalents for each mode of travel from the Center for Transportation Analysis [227] revealed estimated total energy expended for each import into the urban environment in GJ; see Table 6.2. Only the energy to import food, fuel, and construction materials and the energy to export waste was considered. This total energy was then normalized with respect to the mass of consumed resources within the MSA to avoid double counting on city-to-city transport of goods.

Table 6.2: Energy intensity factors were utilized in determining the embedded energy of goods and are adapted from Davis et al. [227].

Mode of Travel	**Energy Intensity GJ/ton-mile**
Truck Freight[a]	$3.92{\times}10^{-3}$
Train Freight	$3.10{\times}10^{-4}$
Barge Freight	$2.22{\times}10^{-4}$
Combined Air and Truck[b]	$2.11{\times}10^{-3}$

[a] Assumes a standard truck weight of 5.8 tons

[b] Averages air and truck energy intensities per ton-mile

6.1.3 Defining Urban Boundaries

The different datasets referenced two different geographical boundaries at the city and the urban scales. Data for water resources defined the urban boundary as the extent of the service area for drinking water and wastewater utilities, which are often different. For material flow data, the FAF provided data based on MSAs. These MSAs included all neighboring suburbs and counties that contribute to the economic workforce of the city. Through this definition, material flows comprised a larger geographical area and population than that of water resources. These disparate definitions of the urban environment and their accompanying boundaries required refinement to achieve some common ground. A service population for water resource and material flow data normalized total material fluxes of the urban environment. Normalization of the data through population was necessary for these large scale comparisons.

6.2 Results and Discussion

6.2.1 Revisiting the Typical U.S. City

For an average city of one million inhabitants, Wolman [122] estimated the relative metabolism of water, food, and fuel inputs and the resultant wastewater (including solids), refuse, and emissions outputs. Figure 6.1 shows a scale representation of the metabolism of cities as suggested and depicted by Wolman [122]. The drinking water inputs equated to an estimated demand of 570 liters per capita per day (lpcd) and a wastewater discharge of 450 lpcd. The representation shows the large influence of water on the urban environment in terms of material flow. This first assessment of metabolism neglected a few of the now-commonplace evaluations including changes in urban stock (physical growth of the city) and other raw materials [e.g., 116, 120]. The values of U.S. urban metabolism from Wolman [122] are comparable to estimates of the average European city by Finco and Nijkamp [234]. The European city of one million people consumes slightly more fossil fuels at 11,500 tonnes per day and signficantly less water at 320,000 tonnes per day [234].

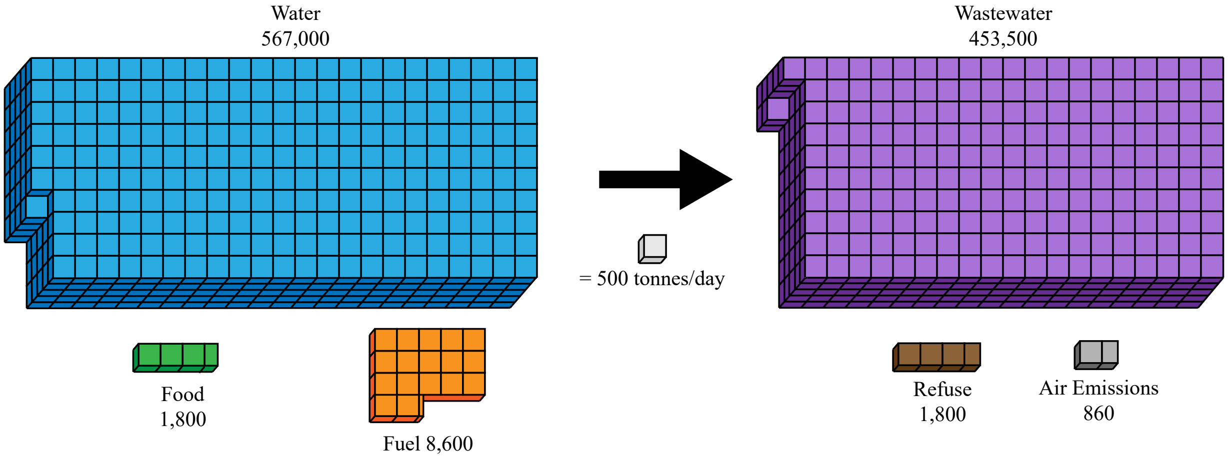

Figure 6.1: The 1965 estimated metabolism of a U.S. city with one million inhabitants in units of metric tonnes (recreated from [122]).

Using the existing visualization approach as a template, the values for urban metabolism were updated and expanded to include the additional category of construction materials. The study focused on physical material fluxes and reserved emissions estimations for further research due to the uncertainty of fuel type and lack of data to disaggregate the SCTG grouping system information with reasonable accuracy. Figure 6.2 shows the updated urban metabolism values for a typical U.S. city of one million residents.

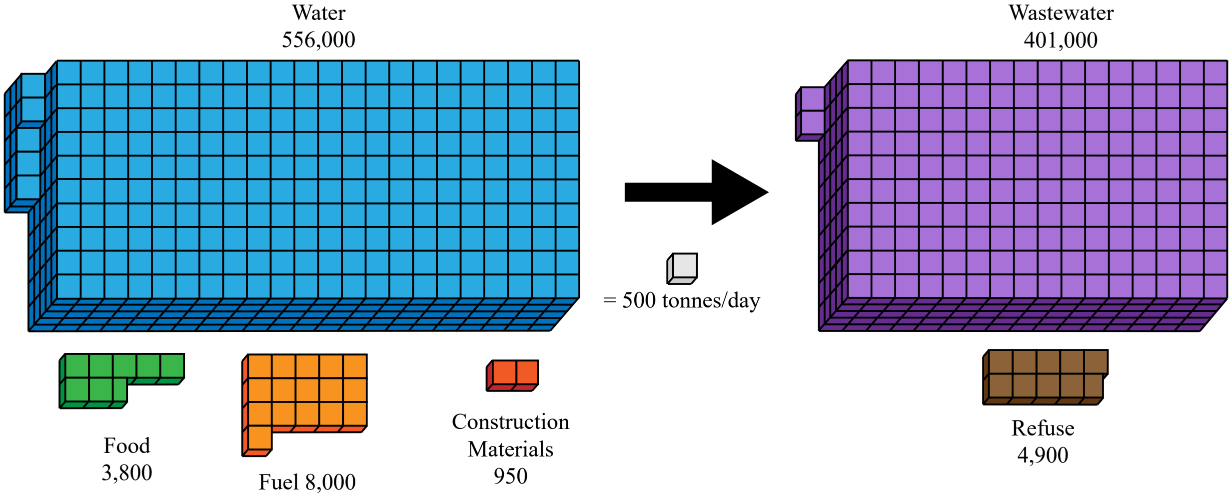

Figure 6.2: The revised estimate of urban metabolism shows increases of food and waste with relatively static fuel and water consumption when compared to Wolman's 1965 estimate.

Comparing Figures 6.1 and 6.2 there was a similar quantity of fuel consumption and drinking water demand, but much higher demands of food and subsequent waste production. There was a slight increase in wastewater flux between the two estimates. The increase in waste production and food consumption in U.S. cities' metabolism was the most striking distinction between the two estimates. This pattern showed an increase in the wastefulness of cities, illustrating an unsustainable trend in material consumption. The accounting of construction material flows illustrated the growth of the urban stock from a metabolism perspective. Overall, water dominated the flux of materials within the urban environment (accounting for 98% and 99% of mass imports and exports, respectively), consistent with previous studies [135, 136].

6.2.2 Regional Patterns of Urban Metabolism

The United States Census Bureau divides the United States into four regions: Northeast, South, Midwest, and West. These four regions are defined by state boundaries and provide another lens in which to view urban metabolism across the United States. Figure 6.3 shows the variations in material consumption from the national median across the United States. Using a similar approach to the national level values, median consumption values for each region were compared to the national average. The Midwestern region showed much higher median urban consumption for non-water resources. The arid West showed higher median values for drinking water demand, while the wetter and more dense Northeast showed lower median water demand. Median urban waste varies minimally across the four regions, while median fuel consumption values varied the most. While there are some variations across the country with respect to region, the variability of the data casts some doubt on the significance of these trends. Due to the geographic centrality of the Midwest within the United States, the data, based on macroeconomic analyses, might overestimate the consumption of cities as a result of material throughput and/or repackaging.

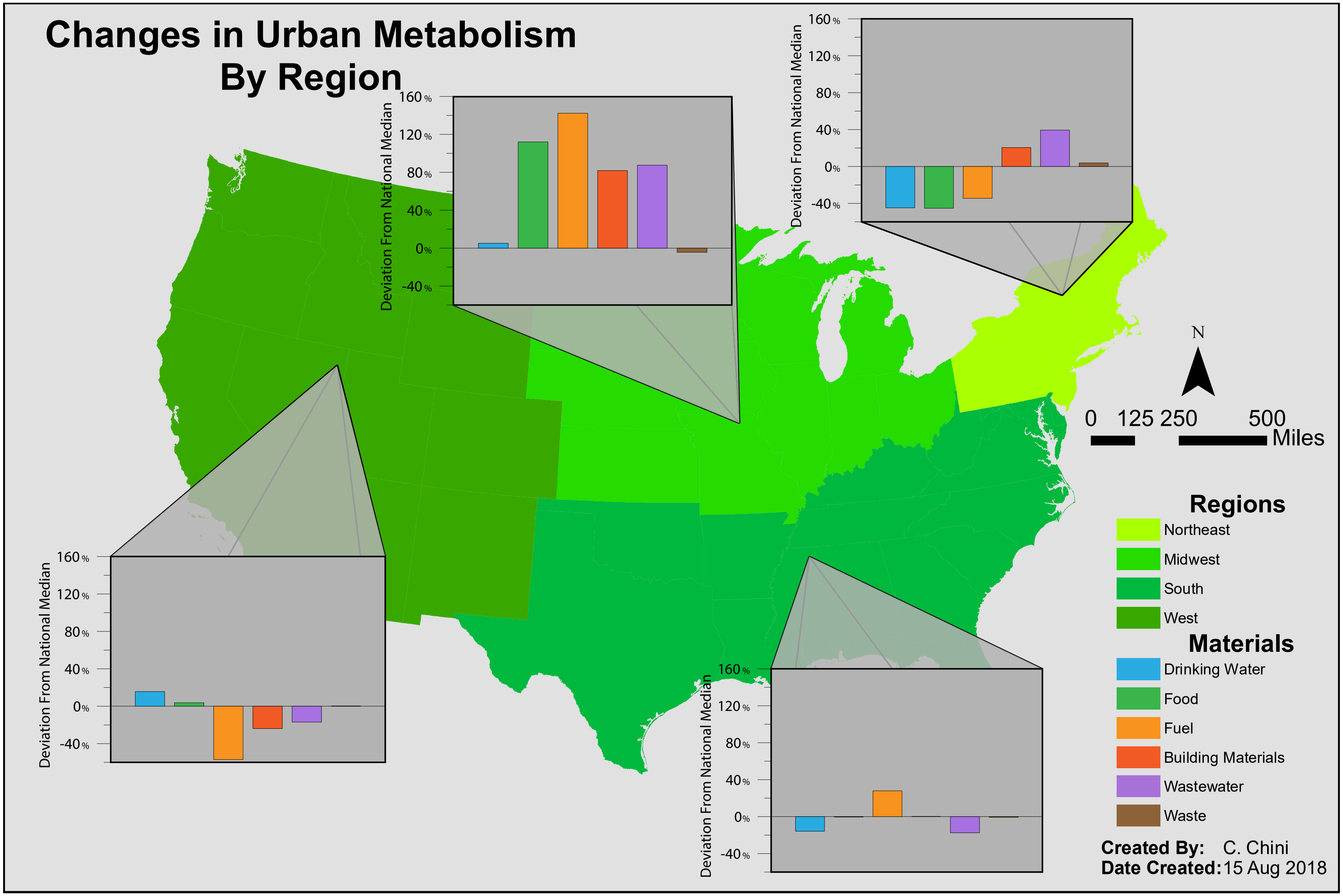

Figure 6.3: There are distinct variations in urban metabolism across the four regions of the United States. Bar charts show the percent variation from the national median material consumption.

6.2.3 Facilitating Urban Metabolism

The embedded energy of urban metabolism is the energy required to enable a city's metabolism to occur. While this energy could be traced back to the production level of goods, the study was limited to the energy required to transport goods from an origin to the urban environment as the destination for direct consumption or transformation. In the case of water resources, the embedded energy was the energy required to collect, treat, and distribute drinking water and wastewater resources. Using the revised estimates of U.S. urban metabolism, the daily energy demand to facilitate urban metabolism was approximately 3,860 GJ/day, represented in Figure 6.4. Of that total required energy, wastewater demanded the largest embedded energy per unit of mass, followed closely by drinking water. The energy required for urban inflows represented 49.6% of total daily energy demand. For comparison, the thermal energy required to generate all electricity in the United States for the year 2012 was 360,000 GJ per million people [235].

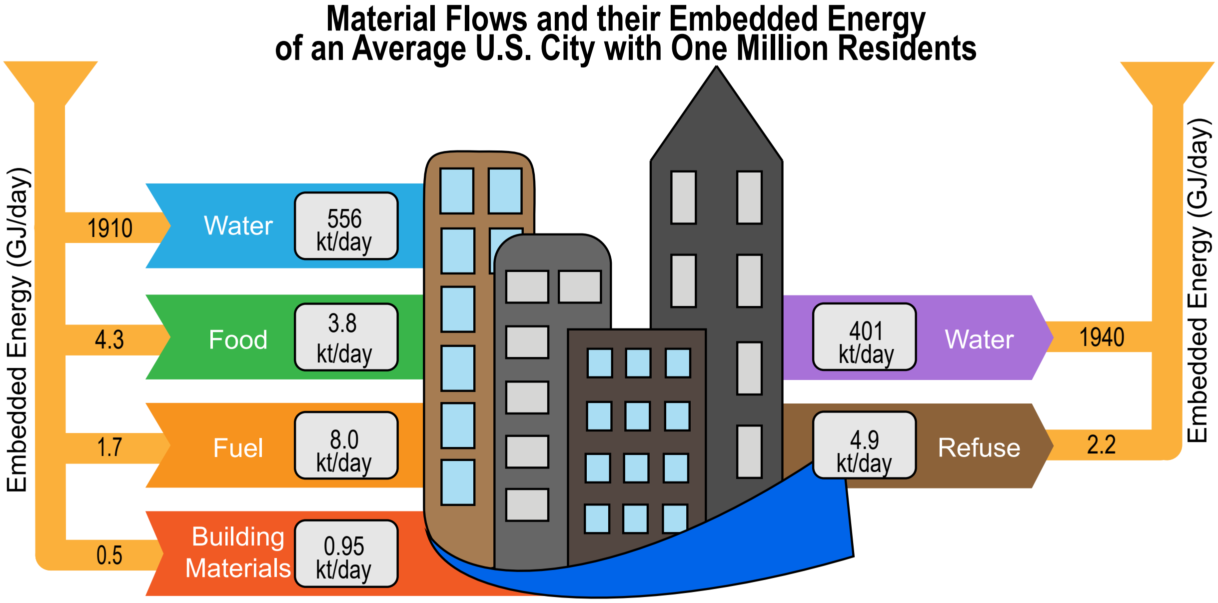

Figure 6.4: The energy required to facilitate urban metabolism is shown for each of the material categories. The energy required for daily water metabolism far surpasses the energy required for the flux of the other analyzed materials.

Water resource flux demanded the largest share of energy, and also had the largest energy intensity of any of the categories (measured as energy per unit mass). Food had the next largest required energy and energy intensity indicative of the highly interconnected network of food transfers throughout the United States [65]. Conversely, fuel and waste had much smaller energy intensities. Fuel is generally transported through train, pipeline, or barge, which are relatively less energy intensive than truck traffic. Refuse, on the other hand, has a lower energy intensity due to the relatively short distances that the material is transported. However, as landfills across the country reach their capacity, especially those in regions with denser populations [236], this energy intensity could change in the future. Additionally, waste-to-energy processes (especially for food waste [237]) could affect both the amount of waste leaving the city and its required energy.

6.2.4 Data Limitations and Uncertainty

MSAs include multiple counties around the central urban environment. Therefore, the boundary of the material flow study could include counties adjacent to the urban environment that are net producers of food and other materials. The estimation of consumption and stock growth, therefore, does not exactly reflect the boundary of the central urban environment. However, to perform an urban metabolism study of the selected magnitude, use of this MSA-based dataset is the most feasible option. There was a high variability in the consumption of cities across the United States. The 25th, median, and 75th percentile values of resource consumption are shown in Table 6.3. The interquartile range for all categories, except for waste, indicated a high heterogeneity in material metabolism across U.S. cities.

The variance associated with these results provides some hesitation when approaching urban metabolism studies. Cities varied substantially in food, fuel, and construction material consumption with some metropolitan areas having a negative consumption, meaning a production of goods within the boundary outweighing the imports. Additionally, there was a large range associated with the embedded energy of urban metabolism, with some cities requiring much higher and lower energy to import commodities. Data collected in Chapters 3 and 4 provides the ranges for water material flux and its embedded energy. For urban metabolism studies to become a useful tool in planning and sustainability studies within the

United States, a better, robust, and comprehensive database than the FAF dataset is necessary. This need not only applies to material flows, but also to flows of water. To create and maintain this database, cities or regions would need to work with retailers and industry to better estimate sold and repackaged resources within their city boundaries. At a minimum, food, building materials, and fuel should be cataloged with resolution on type and origin for the commodity. Data at a monthly, seasonal, or annual scale could better inform quantification of metabolism trends within a city. A database of this scale would enable cities to make decisions on their sustainability through benchmarked consumption and waste habits as well as to determine their local, regional, and global environmental impacts.

Table 6.3: All urban material flows were normalized to daily averages for a city of one million people (e.g., (Chicago Material Flux/Chicago Population)$\times 10^6$). The interquartile range of these flows, aside from waste indicated a highly variable consumption of materials. Additionally, the range of energy intensity for facilitating urban metabolism was provided in gigajoules per 1000 tonnes (GJ/kt).

	Material Flows (tonnes/day)			**Energy Intensity (GJ/kt)**	
Material Category	**Median**	**25^{th} %**	**75^{th} %**	**Mean**	**Std. Dev.**
Drinking Water	556,000	465,000	726,000	3.44	2.52
Food	3,770	1,140	7,610	1.14	0.41
Fuel	8,010	1,250	19,400	0.21	0.15
Construction Materials	950	-1,720	3,060	0.51	0.39
Wastewater	401,000	316,000	659,000	4.83	2.95
Waste	4,930	3,960	5,630	0.44	0.51

6.3 Discussion

The understanding of energy and material flows is essential in the development of sustainable cities [128]. Disorders, such as growth that outpaces the ability of environmental or societal systems to cope, in a city's metabolism indicate potential barriers to sustainable development [114, 137, 238]. Material flow analysis and its application to cities is an important educational and communication tool [115] for describing socio-environmental interactions in support of informing policy and action in urban environments [118–120]. This study revised the estimations of urban metabolism in the United States by Wolman over 50 years ago

[122]. Using the FAF dataset and a database of water fluxes, an updated urban metabolism including water, food, fuel, and construction material consumption was provided. Upon comparison between the two estimates, there were substantial increases to food and refuse streams at a median national scale. Additionally, the study added a subsequent layer of analysis—embedded energy for urban metabolism—that yielded important insights into the material demands of cities and urban sustainability.

In what has been widely cited as the seminal study in urban metabolism, Wolman [122] defines the material consumption of a typical U.S. city of one million people. However, since that time, urban metabolism studies have rarely returned their focus back to the United States. With a need to assess resource sustainability in the face of climate change and localized sustainability initiatives, there are opportunities to provide quantitative assessments of cities and their impacts on the surrounding environment through urban metabolism. Urban metabolism, by quantifying material dependencies, contributes to the discerning of political, social, and economic drivers of consumption [143]. The potential for urban metabolism lies within its diversity and adaptable approach to understanding a city [232]. Urban metabolism is a useful approach for recognition of problems, setting priorities, enacting policy, supporting decision making, and utilizing empirical knowledge to promote sustainability [114, 115]. Urban metabolism studies at a national or regional scale have an important role moving forward in the United States, if data can be made available [144].

Water resources dominated not only material flows, but also the daily energy demand for facilitating urban consumption. Future analysis could benefit from more refined data to rectify geographic discrepancies and to create greater spatially and temporally refined assessments of urban metabolism in U.S. cities. The current level of data available for urban metabolism was inadequate to fully assess changing stocks and consumption of cities. There is a distinct need for policy to create opportunities for data collection of material flows across the United States. As U.S. cities strive to meet their goals within the Paris Climate Accords [239], despite federal divestiture, data-driven urban metabolism studies are an essential tool in a city's sustainability portfolio. Accounting for material and water flows and their embedded energy has direct implications on climate impacts and environmental footprints.

CHAPTER 7

INDICATORS OF THE URBAN ENERGY-WATER NEXUS

> . . . [O]ne goal of establishing a city or region's metabolism is to quantify the material substrate on which a city depends and to unravel the policy and behavioral drivers as well as the differences that might exist among socioeconomic groups...
> –Stephanie Pincetl, et al. (2014) [143]

Urban water systems (drinking water, wastewater, and stormwater) provide integral services to sustaining urban life. The quantity, quality, infrastructure, and management of water resources, as well as perceptions of water, provide an entangled net of policy, science, and engineering that impact the sustainability and resilience of urban water resources. This complex interaction of the life, economy, and social structure of urban environments is predicated upon managing and manipulating water resources [191]. Sustainability in the urban environment is the localization of global sustainability principles pertaining to resource issues and socioeconomic challenges [240]. Cities or urban environments sit at a confluence of technology, resource consumption, population, culture, and economics. With a rescaling of sustainability policy from the regional scale to the urban scale, there is a need to refocus sustainability goals of water resources to the urban environment. This rescaling of policy in sustainability aligns with the predominantly local management of water resources through municipal water utilities. Therefore, urban environments provide the ideal laboratory for study of water resources. In this analysis, the dependence of urban water metabolism on social, economic, environmental, and infrastructure characteristics across over 100 cities in the United States was analyzed. Residential demand studies reflect a bottom-up means of identifying indicators and projecting water demand. However, *does a top-down, city-level analysis identify similar indicators and behaviors to predict urban water metabolism?*

At the residential scale, there have been many studies that quantify drivers of drinking water demand. Income [241–253], age [247, 248, 254], household size [243, 248, 250, 252], urban density/type of development [244, 255–257], and household composition [258, 259] are a few of the identified indicators for predicting water demand. House-Peters and Chang [260] provided a comprehensive review of residential demand studies and their relevant indicators. Household consumption is generally inelastic with respect to price changes. However, high-income households are more price inelastic [261], while seasonal water demand is the most price elastic [262]. Additionally, smaller households are the most sensitive to price changes [263]. Mini et al. [198] found that water restrictions also significantly affect water use. From an environmental perspective, changes in temperature affect residential water demand [246], but they are location dependent [264]. In summary, there are a variety of socioeconomic and environmental factors that affect residential water demand. These characteristics are often evaluated for a singular urban environment and multiple city evaluations are rare [264].

While end-use water meter data are popular sources of data, they are often limited in sample size and duration and have associated privacy and security concerns. Supply-side water meter data, while still somewhat difficult to obtain (Chapter 3), provide an alternative to meter-level data to understand utility-scale water demands. In this study, a top-down approach was utilized with supply-side data to identify indicators of urban water flux and its associated embedded energy. Regression models were scaled from residential studies to the urban environment for identification of similar trends, asking the question: *Do cities exhibit patterns of water use associated with their socioeconomic and environmental conditions?*

7.1 A Socioeconomic Approach

In the early 1900s, Patrick Geddes envisioned the urban environment as a living organism that requires nutrients, creates wastes, and evolves [265]. Odum [266] was the first to apply a metabolic metaphor to a non-biologic organism, which has since been applied to other systems [267]. Metabolism applied to urban environments is a subset of the societal metabolism metaphor [240]. Treating the urban environment as an organism that draws resources from its surrounding ecosystem provides opportunities for city-level resource consumption comparisons, sustainability and resilience indexing, policy opportunities, and benchmarking for the urban water cycle. Building on this metaphor, the field of urban metabolism addresses the material and resource demands and subsequent wastes of cities. Kennedy et al. [125] defines urban metabolism as “the sum total of the technical and socioeconomic processes that occur in cities, resulting in growth, production of energy, and elimination of waste.” Additionally, Bai [4] suggests three dimensions of urban metabolism: (i) inputs, (ii) outputs, and (iii) policies and mechanisms that govern the flows. Based on these definitions, socioeconomic and technical processes are unique to each city, are emblematic of their urban system, and provide insights into the cities’ metabolism. In the case of water resources, attempting to decipher these anthropogenic fluxes through cities is important with respect to urban metabolism and sustainability [1].

The movement of water resources through the urban environment requires extensive infrastructure and is necessary for human consumption and waste disposal. Brian Fagan, in his book *Elixir* [268], describes the evolution of water’s role in civilization morphing from a sacred item in ancient times, to a commodity from the medieval times to the industrial revolution, and now to a finite resource requiring protection. Water plays an essential role and position in the evolution of civilizations and the growth of urban environments [268]. Similarly, Brown et al. [182] discuss the transition of water in cities from a city that simply supplies water to a ‘water sensitive city’ that contains adaptive and multi-functional infrastructure combined with equitable access and resilience to climate change. Urban metabolism provides a means to identify sustainable practices outside the conversation of the role of cities in sustainability [269]. To understand these city characteristics and their impact on urban water metabolism, this study adopted and scaled strategies and models for predicting residential water demand at the household level, treating the city as a distinct and unique system.

Previous analyses from around the globe have attempted to quantify the fluxes of water through cities and, therefore, quantify the physical aspects of urban water metabolism. Understanding and predicting the social and economic changes of urbanization and its affect on nature and society is a global challenge [270]. There are commonalities of the social structure of urbanization across urban systems, with demographic and socioeconomic indicators acting as scaling functions [271]. Urban water systems are large sociotechnical systems and complex adaptive systems [272], requiring an understanding of drivers that explain water use for effective water management strategies [273]. One of the opportunities associated with industrial ecology and urban metabolism is the ability to unravel the social, behavioral, and policy drivers behind a city's material substrate [143].

7.2 Modeling Socioeconomic Indicators

7.2.1 Selecting the Indicators

There are a variety of socioeconomic, environmental, or demographic indicators that pertain to cities. Maclaren [126] describes 12 necessary characteristics of an indicator for the purposes of sustainability, which urban metabolism satisfies. In a recent study, Tanguay et al. [274] identifies 29 important sustainability indicators. Using this list as a starting point, these indicators were adapted for urban water flux and supplemented to define indicators across four groups: (i) economic, (ii) social, (iii) environmental, and (iv) infrastructure.

1. **Economic** indicators included both the city as a whole in the form of per capita gross domestic product (GDP), marginal cost of drinking water and wastewater resources, and sectors of the city's economy (i.e., industry, food, and lodging), as well as median household income.

2. **Social** indicators were selected that contributed to an understanding of both household water use (size and age) and city-wide indicators, including political leaning and median age. Political leaning is linked to energy conservation at both individual [275] and state levels [276].

3. **Environmental** indicators provided an opportunity to place each city within its corresponding environment through an assessment of temperature, rainfall, and number of rainy days.

4. **Infrastructure** networks are essential to understanding metabolic circulation and constitute an important interface between nature and society [277]. Infrastructure net-

works are the 'functional lattice' through which materials flow and the means through which urban metabolism takes place [191, 278]. The size of the distribution/collection system, non-revenue water, and urban density were three potential infrastructure indicators of urban water flux.

Table 7.1 lists the indicators used in the study and their corresponding indicators for household-scale water use. The juxtaposition of these two lists of indicators shows the scaling intent of the study.

Table 7.1: Four categories of indicators for urban water demand were analyzed. The equivalent indicators for household-scale water demand analysis were compared. A value of '-' suggests a non-obvious comparison between the two scales.

Variable Type	Urban Indicators	Equivalent Household Indicators
Economic	Median Household Income	Household Income
	Per Capita GDP	-
	Marginal Pricing	Marginal Pricing
	Make Up of Economy	-
Social	Median Household Size	Size of Household
	Political Leaning	Political Leaning
	Unemployment Percentage	-
	Median Age	Average Age
Environmental	Rainfall	Rainfall
	Average Temperature	Average Temperature
	Number of Rainy Days	Number of Rainy Days
Infrastructure	Miles of Pipe	Size of Home
	Non-Revenue Water	Age of Home
	Population Density	Outdoor Area

7.2.2 Data Collection

Data for the regression model were collected from a variety of sources. Urban water flux and its embedded energy are from the data collected in Chapters 3 and 4. These data are predominantly for the year 2012, and the annual volumes of water and average annual energy intensities were utilized. These data were obtained through open records requests of utilities across the country. For the indicators, a majority of the social and economic data collected were from the United States Census Bureau either for the city proper or the metropolitan statistical area as a whole [279]. Political leaning (either liberal or conservative) is from a 2005 study [280]. While this survey was published seven years before the water data, it was assumed that political leaning remained relatively static over this short time period. For cities not included within the 2005 study, the political affiliation of the citiy's mayor for 2012 was assumed in its place. For the environmental factors of each city, the year 2012 was utilized (a drought year for the Midwestern United States) to correspond to water flux values. These environmental data were collected through historical weather records [281]. Water price and length of pipe networks (drinking water and wastewater networks) were obtained through individual utility websites. Pipe lengths were often found in comprehensive annual financial reports (CAFR) or asset lists of cities or utilities.

7.2.3 Challenges of Data Collection

The process of data collection encountered several unique challenges within the context of the broader goals of this dissertation. Throughout Chapters 3 through 6, a recurring theme is the lack of data. Along a similar vein, data for indicators that accurately represent the urban environment were difficult to obtain. For social and economic data, these challenges were a result of obtaining city-level data as opposed to data from the broader metropolitan statistical area (MSA); see Figure 5.1. Additionally, beyond accessibility, availability, and communication challenges for water volume and energy data discussed in Chapter 3, obtaining information related to a utility's infrastructure was challenging. Specifically, non-revenue water, while a common metric for understanding water loss, was not referenced on multiple utilities' websites, despite many searches. In assessing the length of pipe maintained by either a drinking water or wastewater utility, there were varying degrees of availability. Some utilities clearly stated their assets in terms of length of pipe on their websites, suggesting a sense of pride in the amount of assets managed by the utility. Other utilies, however, had these values within CAFRs or not available at all. In some cases, where the utility is managed as part of the city, these values were a part of the larger city CAFR creating further challenges. An additional problem of obtaining data from utilities was often outdated and confusing interfaces presented on their respective websites. These websites had minimal information or information that was obscured through old interfaces with links that either did not function or were unapparent.

7.3 Results

A linear regression model was chosen to characterize urban water flux and its embedded energy. Five dependent variables were modeled in the study: (i) per capita drinking water volume, (ii) per capita wastewater volume, (iii) drinking water energy intensity, (iv) wastewater energy intensity, and (v) ratio of per capita drinking wastewater volume to per capita drinking water volume. For each of the five dependent variables, subsets of the overall indicators were selected. For instance, to model drinking water volumes, the length of pipes in the wastewater system was not considered. Regression models were completed using the statistical computing software R.

The results of the regression analyses provided some interesting correlations and insight into the urban energy-water nexus. Table 7.2 details the magnitude and significance of each variable in the multi-variate linear regression model. Overall, the regression models had limited explanatory power of the variability in the urban energy-water nexus across the United States, based on the multiple R-squared for each model. However, despite the lack of explanatory power, there were some interesting correlations. The following sections break down results by social, economic, environmental, and infrastructure indicators. Appendix B includes relevant figures for testing regression assumption of multiple linear regression, including normality, heteroskedasticity, and independence, as well as plots comparing predicted and observed values. A two-sided Durbin-Watson test confirmed zero autocorrelation for all models. Variance inflation factors were computed to determine multicollinearity of the model. The model that analyzed the ratio of wastewater effluent to drinking water demand showed some multicollinearity between service population variables.

Table 7.2: The linear regression model for both water flux and its embedded energy explained minimal amounts of the overall variance between cities and their water resource patterns.

		Explanatory Variables				
Regressors	Units	Per Capita DW Demand	DW Energy Intensity	Per Capita WW Effluent	WW Energy Intensity	WW Effluent/ DW Demand
DW Service Population	people	0.00	0.00			0.00
WW Service Population	people			0.00	0.00	0.00
Drinking Water Demand	lpcd			0.20		
Median Household Income	$	0.00		0.00		0.00
Gross Domestic Product	$/person	0.00		0.00		0.00
Marginal Water Price	$/m^3	-48.7	47.1			0.00
Marginal Wastewater Price	$/m^3			-7.06	8.89	0.01
Economy: Manufacturing	%	-90.7		307	103	-0.06
Economy: Food and Lodging	%	270		2070	-1710	0.25
Median Household Size	people/residence	1.56		-104	-280	0.47
Political Leaning	L or C	-123				0.25
Unemployment Percentage	%	1830				0.11
Median Age		5.45				-0.02
Population Density	people/km^2	0.00	-0.01	0.01	-0.03	0.00
Annual Rainfall	cm	-3.66 .	-4.66 *	2.99	-4.73 .	0.01 .
Annual Average Temperature	°C	12.5	16.7	-9.22	183	-0.05 *
Days of Rain (>0.25 cm)		-5.58	4.37	-2.99	3.77	-0.01
Length of DW Pipe	km	-0.04 .	0.03			
Non-Revenue Water	%	1070 *	-43.6			0.52
Length of WW Pipe	km			0.00	0.02	
DW Pipe Density	km/people	51300 *	18600			
WW Pipe Density	km/people			-1660	-19200	
Combined Sewer	Yes or No			254 **	-75.7	0.21
Multiple R-squared		0.34	0.32	0.43	0.20	0.47

Note: DW= drinking water; WW=wastewater

*** : $p<0.001$; ** : $p<0.01$; * : $p<0.05$; . : $p<0.10$

7.3.1 Social and Economic Indicators

In general, the selected social and economic indicators at the urban scale did not significantly explain the variability associated with urban water flux across the United States. Surprisingly, despite many studies at the urban level, median household income was not found to be a statistically relevant indicator. This finding suggests that the heterogeneity within a city with respect to household income was more significant than status of wealth between cities. Additionally, there was no statistical correlation associated with price or economic diversity, indicating the complicated relationship between water resources and the economic system. Interestingly, when per capita drinking water was used as an indicator for per capita wastewater effluent, there was no statistical correlation. The lack of significance could have a two-fold reason: (i) variations in outdoor water use, or (ii) the presence of combined sewer systems. Finally, comparing embedded energy to service population for both drinking water and wastewater utilities illustrates the minimal economies of scale associated with water utilities. The lack of economies of scale was consistent with findings from Chapter 4.

7.3.2 Environmental Indicators

Annual rainfall totals were the most significant indicators for both drinking water and wastewater fluxes. As expected, there was a lower drinking water demand associated with higher rainfall totals, most likely due to decreases in outdoor water use. Interestingly, however, the number of days of rain (>0.25 cm) was not a significant indicator. This trend continued to embedded energy in drinking water, where wetter climates tended towards lower energy intensities. This inclination might be due to the prevalence of lower energy intensity surface water resources, as opposed to groundwater resources, in wetter climates. While annual rainfall was not a significant indicator of per capita wastewater flux, the prevalence of combined sewer systems was a major indicator; see Figure 4.1.

The ratio of per capita wastewater effluent to drinking water demand represents the potential for creating a circular use of water resources, based on available water volume. For instance, a value of 0.90 suggests that 90% of a city's drinking water demand could be supplied by wastewater effluent in the case of direct potable reuse (assuming no system losses in the treatment process). Annual average temperature was a significant indicator of this value across the United States. An increase in the average annual temperature by 1°C was associated with a loss of available drinking water supply through wastewater by five percentage points. This association implies that warmer cities (or, potentially, warming cities [246, 264]) have reduced potential for water reuse as a means of supplementing water supply with current habits. Additionally, an increase in a city's annual rainfall of 1 cm increases the available drinking water supply through wastewater by one percentage point.

7.3.3 Infrastructure Indicators

This analysis focused on several infrastructure variables to explain the variability of the urban-energy water nexus. Previous work has found that type of water source (surface or groundwater) indicates the energy intensity of drinking water resources [282]. For drinking water demand, non-revenue water was a significant indicator, where a one percentage point increase in non-revenue water was associated with a nearly 11 liter per day increase in per capita water consumption. Additionally, a dense distribution network (length of pipe per person) tends towards a higher per capita water demand. The only statistically significant indicator for wastewater volume was the presence of a combined sewer system. Interestingly, length of pipe for both drinking water and wastewater systems was not an indicator of energy intensity or water volume. As wastewater systems are largely gravity fed, a larger system would logically not incur increased energy intensity. However, drinking water systems are pressurized and often driven by a series of pumps. The pumping of water through a pipe system or from a groundwater resource accounts for a majority of the electricity demand of the water utility [109]. Additionally, urban form, considering the network structure of the water distribution system, as well as leakage of pipes play important roles in determining the energy intensity of moving drinking water to the consumer [283, 284]. The lack of correlation of drinking water infrastructure size and energy intensity suggests an inability to disaggregate nuances about the system from a top-down perspective.

7.4 Discussion

Residential water use studies often focus on only one component of water use: drinking water demand. Scaling these studies to the urban scale provided an opportunity to evaluate indicators of the direct urban water cycle from an energy-water nexus perspective. This study evaluated the ability to scale water demand models from a bottom-up method to a top-down method. Using data from Chapters 3 and 4 on the urban energy-water nexus, a selection of indicators from social, economic, environmental, and infrastructure sectors were collected to explain some of the variability in water flux across the United States. In general, the indicators selected within this study did not have large explanatory power of the variability associated with the urban energy-water nexus. These results showcased the importance of both bottom-up and top-down methodologies and their different strengths in evaluating urban water resources. Additionally, minimal correlation by itself is an important conclusion, showing that goals of conservation of urban water resources are not limited by factors such as average age or income. Therefore, socioeconomic indicators are not necessarily a barrier to improving water resources sustainability.

The results of this study are comparable to those found in Mann et al. [285], which evaluated local government's propensity to have a climate change action plan based on various factors, including low income, proximity to risk, crime, and other factors. Through their study of various local governments, they received similar non-correlations between their suite of socioeconomic indicators and action on climate change. In their study, Mann et al. [285] determined only one main factor being a cause for action, proximity to risk, leading to a discussion of barriers to and opportunities for implementing climate action plans. Inversely, however, a study in the United States indicated a clear spatial and statistical profile based on certain socioeconomic factors, emissions, college educated, carbon-intensive industry, etc. for municipalities that joined the Cities for Climate Protection (CCP) program [19]. Brody et al. [19] cite high statistical corollaries between these factors and those that joined the CCP program. However, it is worth noting that neither of these studies consider water resource consumption within their factors. Therefore, those cities that take climate action or participate in global programs are different than the cities included within in this study as no amount of existing policy was evaluated when selecting these cities.

The results of the model suggested that methods from residential-scale analysis do not scale comparably to the urban level. Therefore, there are limits to the use of top-down urban metabolism models in explaining some of the underlying drivers of material consumption. Additionally, due to privacy and availability concerns for household-scale data, bottom-up models are often not a widespread option for conducting research. In suggesting the limits of urban metabolism models, it raises further questions on the balance of data and model scale in urban environments. *What scales (neighborhood, census tract, etc.) balance both data availability, privacy, and model effectiveness to generate sufficient understanding of socioeconomic drivers of the urban water cycle?*

The understanding of the urban water cycle from a socioeconomic perspective and the difficulties in obtaining data suggest a need of policy refocusing within goals of sustainability. Much of the literature on green or sustainable cities focus on energy efficiency, carbon emissions, the literal greening of cities, or social justice issues. However, relatively few sustainability studies highlight the consumption and conservation of water resources as a major focus of policy and innovation. Much work has been published on the competition for a green city or ecocity from a sustainability governance framework [286–290]. This work highlights city-level efforts from a policy standpoint to advertise themselves as a ‘green’ or ‘eco-friendly’ city to attract businesses, people, and a strong tax-payer foundation. However, these efforts can lead to intra-city conflicts that arise from issues such as gentrification or marginalization. These cities often focus on sustainable features that promote a positive outside image while creating sustainability injustices within the city.

Policy directed towards water efficiency and conservation has the ability to ‘trickle-down’ or ‘permeate’ throughout all the citizens of the city due to its materiality. The physicality or materiality of a water resource, even in its indirect form, creates the ability to participate in water conservation efforts. The numbers associated with urban metabolism are relatively meaningless without an understanding of the people behind the demand. These results, or lack thereof, demonstrated the need for incorporating social sciences into an understanding and quantification of the urban water cycle. Anthropogenic uses of water cannot solely be quantified through engineering metrics and necessitate interdisciplinary work in understanding public perceptions [291] and psychological drivers [292] of water use. Specifically, the social sciences are necessary in assessing benchmarks of water use and sustainability indicators, beyond engineering infrastructure assessments of the urban water cycle.

CHAPTER 8

ENVISIONING THE BLUE CITY THROUGH URBAN WATER GOVERNANCE AND OPEN DATA

> . . . [D]ata sharing is not an end in itself, but a means to an end.
>
> –R. Alan Cheville (2016) [293]

Urban water governance is a framework for addressing critical issues in cities' water resources by creating a more inclusive policy for sustainable transitions. The framework provides opportunities to advance and address problems or inequities within urban water. However, there are certain barriers or institutional inertia inherent in traditional urban water systems with an infrastructure or technological approach to management. The traditional urban water cycle considers drinking water supply, treatment, and distribution; wastewater collection, treatment, and discharge; and stormwater management; see Figure 2.1. Each of these components, with the exception of stormwater in many cases, requires some form of embedded energy to treat and/or distribute. The required energy of drinking water is an important component in the direct water cycle of urban environments (Chapters 3 and 4). Beyond these direct water requirements there is water consumed along the supply chain of materials imported into the urban environment, most notably food, fuel, and electricity [32], often called a water footprint [33]. These embedded water resources often greatly exceed direct water demands (Chapter 5), and, while not consumed locally, do provide an indication on the impact that cities have on regional or global water conservation efforts. Virtual water flows are rarely included in decision making as an opportunity to expand a city's water supply, with engineered potable water supplies often being the focus [32].

To promote governance of the full urban water system, the following definition of the *blue city* was proposed as a motivational moniker to consider the full urban water cycle:

This chapter was based on work submitted as: Chini, CM and Stillwell, AS (2018) Envisioning Blue Cities: Urban Water Governance and Water Footprinting. *Journal of Water Resources Planning and Management*: submitted

> A *blue city* promotes policy, public awareness, and economic efficiencies of water resources to provide equitable, secure, and sustainable water systems. These water systems consist of both direct water resources in the form of drinking water, wastewater, and stormwater (and their embedded material requirements, such as energy), while also considering water as an embedded resource in other material fluxes (food, energy, and other materials). The *blue city* considers the greater impacts of its water resource demands and how its citizens are provided these resources, by way of energy and emissions, through an openness of data and sharing of information.

Expanding the framework of urban water governance to promote both sustainable direct and indirect water consumption is necessary for advancing urban environments in the face of climate change and resource constraints. The term *blue city* is a motivational moniker to address this need for urban environments to quantify, benchmark, and manage both direct and indirect water resource consumption. Synthesis of results from sociotechnical transitions, sustainable urban water governance, and water footprints provided opportunities to advance the understanding of urban water consumption and define the *blue city*. This chapter presents the concept of the *blue city*, defines its terminology, and describes one of its main challenges: the need for open-access data to transition towards a sustainable water governance system, considering both direct and indirect resources. Direct and indirect water resources of the Boston Metropolitan Area (Massachusetts) were investigated to highlight data and governance opportunities. The *blue city* moniker illuminates the urban water cycle and provides opportunities for urban environments to make decisions to conserve water resources more broadly. Figure 8.1 illustrates the various components of urban water as part of the *blue city*. Direct and indirect water are imported into the urban environment with wastewater and virtual water discharges leaving the city. Both direct and indirect water have opportunities for internal reuse. Energy is directly required within the urban environment to facilitate direct water fluxes.

As part of sustainable water management, there is an increasing body of literature suggesting a necessary transition in urban water governance to a new sociotechnical regime, while questioning whether the current sociotechnical regime of water management is able to respond to new challenges [38, 150, 182, 294–296]. The term governance is broadly defined as a set of collective actions towards a common goal [154], which includes all social, political, and economic actors related to the system [39]. In regards to urban water governance, this common goal is often the transition to a 'water sensitive' [182] or sustainable city, as sustainability often plays a large role in governance issues [297]. However, these governance issues and discussions are generally focused on direct or physical water resources and their systems, while not considering indirect water demands or the resource demands necessary for urban water. Therefore, there is a need to include the water footprint concept in urban water governance transitions. Water footprints illustrate the global dimensions of consumption patterns in water management and governance [68, 298, 299].

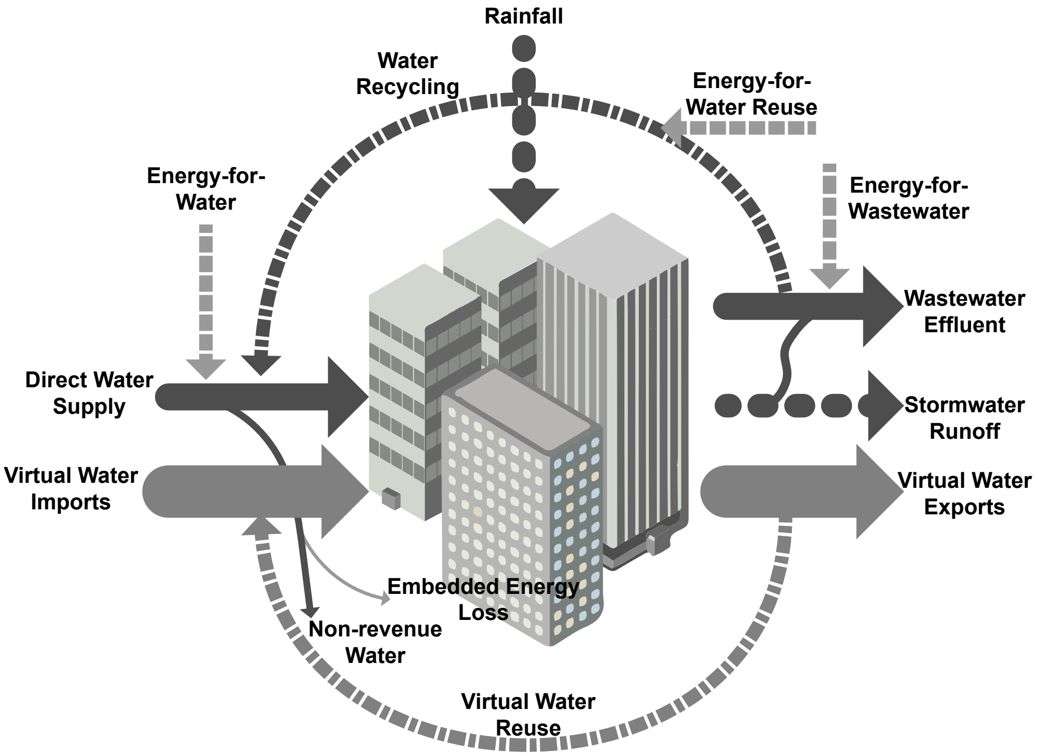

Figure 8.1: The full urban water system includes direct and indirect fluxes of water resources. Energy is required to facilitate flows of direct water resources.

"Business-as-usual" is no longer a sufficient means of urban water governance [300]. Water management and water conservation have become complex sociotechnical systems [35, 36], requiring sustainable governance systems to incorporate all social actors and integrate across geographical scales [1]. A shift beyond direct water resources to indirect resources further expands the actors of the sociotechnical system. Urban water governance studies examine change to the broad socioinstitutional and sociotechnical regime of the current status quo of urban water systems. These water systems require a shift to a more flexible and sustainability policy and emphasize interactions across the various structures and processes [37, 38]. Similar strategies for sociotechnical regime shifts offer opportunities to broaden the spectrum of urban water governance. The existing literature widely cites transparency and accountability in the form of information and knowledge as essential pillars in sustainable urban water governance [39–42]. While data sharing is not an end in itself [293] and data, by themselves, are not necessarily informative [42, 301, 302], data collection, reporting, and analysis can facilitate or enable sustainability transitions that aid multi-level participation, while capturing the current physical or technical state of the system. Open data provide a key translation between environmental science/engineering and policy, linking quantitative and qualitative evaluation of systems. The necessary transition to sustainable urban water governance of direct and indirect water resources requires contribution of data from multiple actors across the private, public, and academic sectors. Collaboration and data sharing are important initiatives to better understand and manage urban water resources to facilitate a transition to the *blue city*.

8.1 Direct Water Footprints of Cities

The Massachusetts Water Resources Authority (MWRA) is the regional water authority for the Boston Metropolitan Area. Through data collection efforts in Chapters 3 and 4, the availability and accessibility of data for MWRA was identified as some of the best in the nation. Data for MWRA came from a combination of online published resources and open records requests. Monthly drinking water demand data were available from from 1984–2016, monthly embedded energy data for drinking water resources were available from 1998–2015, and annual climate emissions estimates were available from 2006–2015. The level of energy-water nexus data provided enabled observation of trends over time, coinciding with policy implementation and governance changes.

MWRA is a public water utility that services 2.5 million people, including Boston, in eastern and central Massachusetts [303]. MWRA offers water services to 51 drinking water and 43 sewerage communities with an average drinking water demand of 215 million gallons (814,000 m^3) per day and wastewater treatment of 350 million gallons (1.3 million m^3) per day (Figure 8.2). The MWRA system includes a series of aqueducts and two protected reservoirs. The current primary drinking water treatment plant that serves the majority of the MWRA communities uses ozone disinfection processes. A second drinking water treatment plant serves a few outlying communities using chlorine disinfection processes. The MWRA operates a treatment plant fed by a both combined and separate sewers with mainly gravity flow connecting two million sewer users. The effluent is discharged at depths up to 120 feet into Massachusetts Bay to ensure mixing and dilution. Sludge from the treatment facility is dewatered and dried for use as fertilizer in agriculture and forestry.

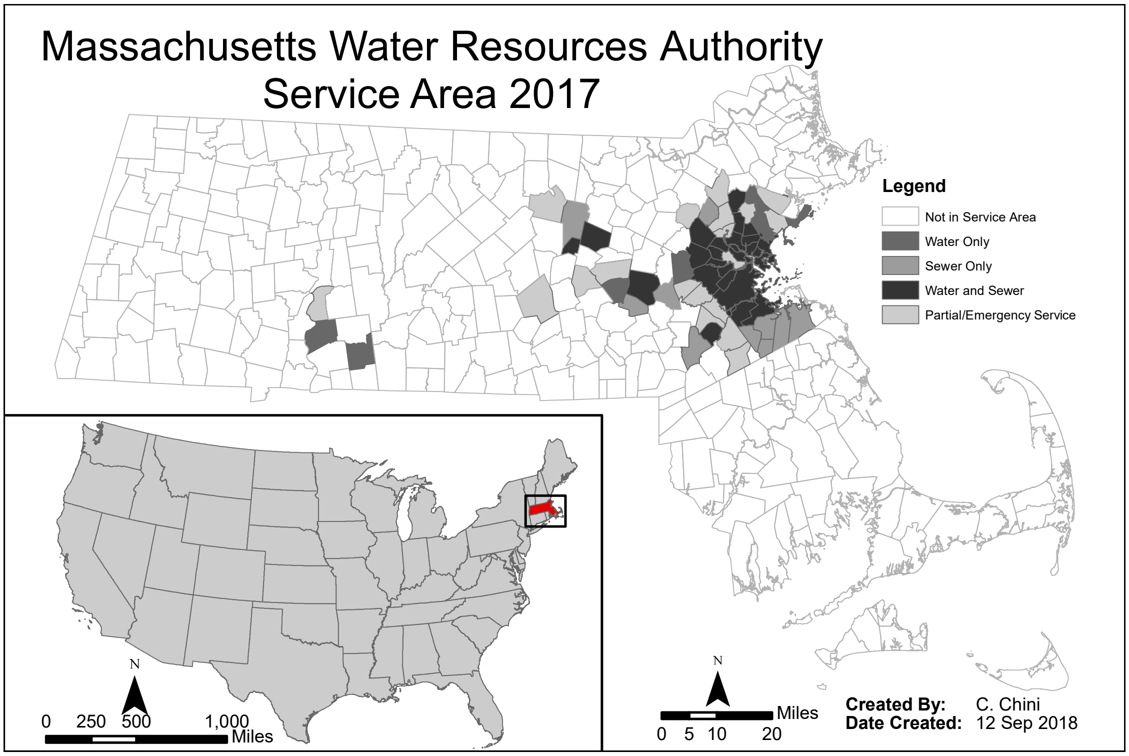

Figure 8.2: MWRA provides drinking water and wastewater portion to the major metropolitan area of Boston, MA (adapted from [303]).

To evaluate the use of data collection and synthesis for sustainable governance strategies for MWRA, interview questions were presented to a representative of the organization. These questions targeted the motivation of data collection, impact of the data on policy and operations, and education of the public through data collection. The interview process and questions were approved by the University of Illinois' Institutional Review Board.

8.1.1 Data Collection and Motivation

"You cannot manage what you do not measure." This statement succinctly summarizes the viewpoint of data within MWRA. Data are collected and organized for four main purposes within the organization: (1) evaluation, (2) efficiency, (3) transparency, and (4) communication.

1. Data collection and organization are necessary for internal ***evaluation*** of various efficiency and green energy production efforts, which contribute to sustainable urban water management. MWRA catalogued data for evaluation of various sustainability efforts that are tested.

2. MWRA organized data to look for trends and possible future initiatives for ***efficiency*** and sustainability.

3. Both community-level and state-level organizations have requested greenhouse gas numbers, computed (in part) through accounting of water and energy flows. To provide a level of ***transparency*** to the operations, these tallies were computed and made available to the public, when requested.

4. MWRA stressed the importance of ***communication*** with the public as a significant consideration for consolidating data. The tabulated information allowed for success metrics to be discussed as part of MWRA's broader environmental mission, beyond safe drinking water and responsible wastewater treatment.

Data available for MWRA included both water volume, energy demand, and greenhouse gas emissions data. These time series data were plotted for drinking water as an example to show distinct trends and a coupling of resource trends, Figure 8.3. Overall, a decreasing trend of drinking water production was observable, with some intra-annual seasonality. Additionally, there was a visible increase of electricity for drinking water resources in the middle of 2005. This spike was a result of bringing online a new drinking water treatment plant that utilizes ozone treatment. Ozone treatment has a greater energy demand than the previous MWRA treatment system. The organization of the energy data was an important component in the calculation of the greenhouse gas (GHG) emissions. The reported annual emissions were relatively constant with a slightly negative trend.

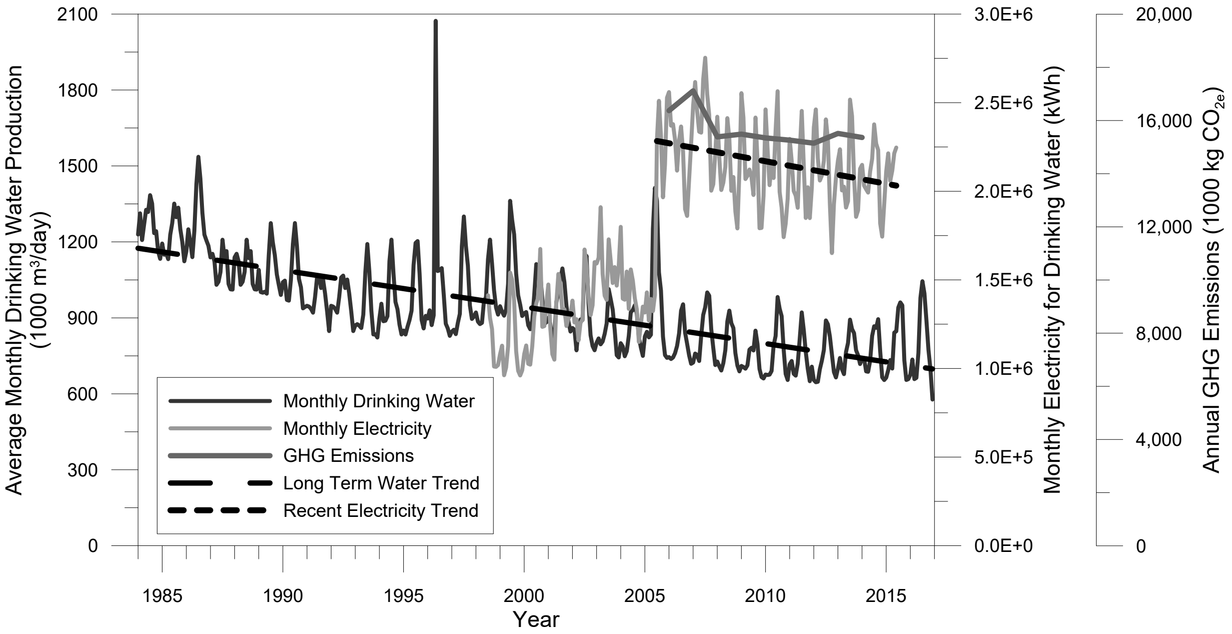

Figure 8.3: Accounting for season variability, there was a long-term trend of decreasing drinking water production and a recent trend of decreasing electricity-for-water.

8.1.2 Learning from the Data

There are many groups within MWRA that use these data to analyze individual programs, look for new opportunities, and consistently and efficiently respond to public data requests. These data are also shared externally to the MWRA. Data are shared with the Massachusetts Department of Environmental Protection, Massachusetts Executive Office of Energy and Environment, and the City of Boston Environment Department on both a regulatory and less formal basis. Additionally, data are provided to smaller communities within the service area by request. Importantly, data for wastewater are exchanged with the U.S. Environmental Protection Agency in relation to the Boston Harbor National Pollutant Discharge Elimination System (NPDES) permit. This vertical transfer of data including federal, state, regional, and local authorities of water volume and embedded energy promotes a culture of resource awareness that is essential in transitioning to sustainable urban water governance.

As part of this overall data collection process, MWRA has noticed several anomalies in the data over time that are often the result of recording errors. Sometimes, however, these anomalies are a signal of change that raises important questions about energy usage and departmental programs. One such departmental program is the advancement of renewable energy integration within the MWRA energy demand. A ten-year program from 2002-2011 resulted in efficiency increases and cost savings from implementing renewable energy. These renewable energy initiatives include constructing photovoltaic solar panels and wind turbines on site, energy recovery from wastewater treatment, and importing external electricity from renewable sources. These programs are a direct result of tracking energy and water resources through data collection and discovering long-term trends. The prerogative of MWRA to understand both water volume and required energy is an important facet of their decision-making and is facilitated through data collection. Collection of and learning from urban water data by MWRA provided opportunities for transitions in urban water governance towards a *blue city*.

8.2 Indirect Water

Water footprints, or indirect water, emphasize a city's dependence on distant water resources [304]. A majority of the U.S. population lives in cities, where most water-related decisions are also made [11, 36, 305, 306]. Conversely to direct water resources, indirect water does not have a regional or local management authority. Instead, water footprints are indirectly managed through resource conservation efforts by various social groups, municipal governments, and the manufacturing and retail sectors. Governance of indirect water resources, therefore, necessitates an understanding of the sociotechnical system as a whole with its variety of actors. These various social actors that would be part of a governance system require a robust and understandable level of data and information to inform actions with respect to indirect water. The water footprint concept was originally intended as a means of identifying global water consumption trends [307]. Applying water footprints to cities incorporates the global dimension of water into governance regimes. Cities have the capacity and autonomy to organize, take initiative, and take action, which makes the urban water footprint concept particularly intriguing [32]. To demonstrate the possibilities of data in enabling a transition to the *blue city*, the study focused on the Boston Metropolitan Area. The data motivations identified for direct water provide the basis for analysis and discussion of the data necessary for implementing urban water governance systems for indirect water.

8.2.1 Data Collection and Motivation

Applying water footprints to the city level is a relatively new assessment approach, with much of the research occurring in the past decade [32, 71, 308–314]. It is not sufficient to simply downscale water footprints from a national or regional scale based solely on population to determine urban water footprints [310, 315]. However, data necessary for computing subnational water footprints are often scarce. Within the United States, the predominant method of assessing urban water footprints is through the Commodity Flow Survey or Freight Analysis Framework [194, 233]. These similar databases estimate flows of goods through macroeconomic analysis at the metropolitan statistical area (a grouping of counties or suburbs around a major city). The material flow data are then paired with estimates of virtual water content [67, 219, 220]. However, these economic data are not generally at a geographic scale congruent with political or utility boundaries. Returning to the four purposes of data collection and analysis from direct water management provides a good starting point to motivate urban data collection for indirect water analysis.

1. **Evaluation**: Developing a data system to evaluate water footprints has implications beyond indirect water and can aid in urban metabolism studies [e.g. 125] or evalu-

ating carbon footprints. An important contribution of data evaluation would be the identification of changes over time.

2. **Efficiency**: Translating water efficiency into urban water footprints involves first creating an understanding of material origin. The water required for the production of commodities varies across the globe and the impact of the consumed water varies based on water scarcity [316, 317]. A city can help increase efficiency and reduce water stress through importing commodities from water efficient locations with sustainable management techniques.

3. **Transparency**: As there is no single entity responsible for managing indirect water resources and the private sector would be a major component of the data collection process. Transparency and privacy could be major concerns in creating a better database for urban water footprinting.

4. **Communication**: An important component of sustainable urban water governance data for water footprints can help in communication efforts to change diets [e.g. 309, 318] and influence patterns of consumption.

The current state of data collection used for urban water footprinting is at the federal level using macroeconomic methods. While these data provide an estimate, data are prepared every five years and exist at geographical boundaries that are often greater than municipal or utility jurisdictions. These data, however, are important in advancing sustainable policy and governance regimes that recognize a city's role in the larger global water cycle. An improved understanding of water footprinting requires open data at the city level to facilitate evaluation, efficiency, transparency, and communication for sustainable urban water management.

8.2.2 Learning from the Data

The Boston Metropolitan Area has been included in two recent water footprint studies [Chapter 5, 313]. Chapter 5 estimated the per capita water footprint of Boston area residents to be approximately 1500 m^3/year. This water footprint assessment included food, fuel, and electricity imports. Over 80% of this water footprint was attributed to food, 15% to fuel, and the remainder to electricity imports. Ahams et al. [313] found a similar water footprint of approximately 1,470 m^3/person/year when including industrial demand and excluding electricity imports. For comparison the cities of Milan, Italy and New York City, NY, USA are estimated to have water footprints of 2,200 m^3/person/year and 2,600 m^3/person/year, both less than the U.S. average of 5,000 m^3/person/year [309, Chapter 5] . These independent assessments of the Boston Metropolitan Area show a comparably low water footprint. However, these estimates are static, both for the year 2012, and, therefore, do not provide insight into trends over time.

There are currently no directives from the City of Boston that directly reference water footprints. However, the City of Boston has implemented a zero-waste policy that emphasizes the circular economy, while engaging community, industry, businesses, and advocacy groups [319]. This directive has implications in reducing food and other material waste that will affect the urban water footprint. Therefore, there are opportunities to approach urban water governance of indirect water resources through other policies. Beyond zero-waste approaches, cities could identify origins of materials, which informs the embedded water within the resource and the relative scarcity and impact the commodity's production on the local environment. Additionally, Boyer and Ramaswami [320] further the role of the city in indirect water resource management, by modeling city-level actions and initiatives on food systems, with the city as part of a larger transboundary system. The analysis revealed significant savings to water resources, among other embedded resources, comparable to savings on production. Collecting data required for water footprint analysis can both aid a zero-waste policy and provide opportunities for further governance transitions with respect to indirect water resources at the urban level.

8.3 Achieving Blue Cities

Brown and Farrelly [185] discuss several distinct barrier types to achieving sustainable urban water governance, including lack of information, poor communication, no long-term strategy, and little to no monitoring and evaluation. Through open data within the energy-water nexus, MWRA attempts to overcome each of these institutional barriers. Information is obtained through organization of open data with long-term goals of evaluating efficiency. Additionally, MWRA utilizes the data to maintain transparency and facilitate communication with various stakeholders. The open information collected by MWRA facilitates adaptive management and comprehensive understanding [150] and sustainable water governance by considering the continual changes of the system [39]. Additionally, open data techniques, such as those employed by MWRA, create opportunities for continuous learning and innovative water management to promote governance transitions in urban areas around the globe [42, 321]. Extending these open data techniques to include indirect water resources for the computation of urban water footprints will enable communication and improve monitoring and evaluation of water resources beyond the boundary of the city.

Ensuring that the full urban water cycle is considered in urban water governance transitions is an important step in promoting sustainable urban environments across city sectors and achieving *blue cities*. The collection, organization, and openness of data facilitate future studies and assessments. For instance, cataloging energy data is an essential component of delivering accurate assessments of greenhouse gas emissions, while also being integral to promoting resilient direct water systems [106]. Electricity demand at MWRA corresponds to about 37% of total tabulated emissions. Therefore, understanding the dynamics of the energy-water nexus at a utility level provides an important component of climate accountability. Recently, the State of Massachusetts Executive Office of Energy and Environmental Affairs [322] completed a pilot study designed to reduce emissions of greenhouse gases and energy for drinking water and wastewater utilities by 20%. These integrative data collection efforts promote long-term strategies and efficiency consistent with goals of sustainable urban water governance. Beyond energy consumption, accounting for other materials consumed within the urban environment has implications not just for water footprinting, but also for economic growth and social equity.

Additionally, there was evidence of public participation through data collection within MWRA, with one of the goals of data organization being communication with stakeholders about successes and advances of the institution. Public participation may take several forms from information supply to stakeholder-governmental collaboration [323]. A program within MWRA to facilitate stakeholder communication and engagement is the Water Supply Citizens Advisory Committee, which provides community input to the operations of MWRA. The management and data sharing structure of MWRA facilitates both horizontal and vertical integration of stakeholders, which is essential for sustainable water governance. For indirect water resources, public and private participation are essential as there is no centralized governing body. Instead, open access data could enable communication across sectors and with the public to facilitate best practices and advance an understanding of water footprinting and impacts on source water resources.

A lack of information and knowledge is one of twelve barriers to urban water governance transitions mentioned by Brown and Farrelly [185]. Water governance, therefore, requires a more comprehensive data set to facilitate information and knowledge growth. Urban data do not simply consist of water flow rates associated with drinking water and wastewater treatment. Both embedded energy and indirect water resources are included as critical data needs in facilitating discussions of a more comprehensive urban water governance system. New conceptual frameworks that are informed by total urban water and energy balances are important for analyzing and monitoring cities and their sustainability [324]. Cities must not only continue to operate within their local water, energy, and carbon budgets [325], but have a responsibility to consider their regional and global impacts on water resources as urban growth continues around the world.

8.4 Summary

Data for drinking water and wastewater treatment plants are scattered and have accessibility and availability constraints (Chapter 3). Open data provide an important basis for assessing the urban energy-water nexus (Chapter 4) and advancing sustainable governance practices. Through a study of the MWRA, this chapter discussed the internal prerogatives of a major drinking water and wastewater utility as they pertain to data. The MWRA's attitude of data organization for the purposes of efficiency, transparency, and communication facilitates a wide range of social actors to become involved within the governance of water resources. Through their data goals, MWRA was able to achieve greater transparency and accountability, which are important foundational elements in transitioning to sustainable urban water governance.

While data for direct water are scattered, they are at least generally tabulated in some form for internal accountability purposes. Data for water footprints are even more scarce, requiring a governance shift in the understanding of the urban water system. To begin tracking urban water footprints and develop governance strategies, there is a need for recognition that the full urban water system consists not only of direct water resources, but also regional and global indirect water resources through the consumption of food, fuel, electricity, and other products. This transition requires collaboration across academic, municipal, and private sectors to facilitate a knowledge exchange and advance policy efforts. From an academic perspective, most water footprint analyses do not consider the broader institutional or social processes that influence and interact with the flows of water [326]. City governments and the private sector can work together to develop consumption estimates that facilitate deeper understanding of urban water demand. These efforts in water footprinting, open data, and direct water accounting enable transitions in governance and policy generation to develop sustainable urban water systems for both direct and indirect water resources, advancing towards the *blue city*.

Sustainable transitions in the water sector are not inevitable and must utilize interdisciplinary perspectives [153]. Sustainable urban water management and governance requires a feedback cycle that is predicated upon data and analysis [170, 182]. To advance urban water transitions, data collection from utilities, the private sector, and public entities are necessary in overcoming institutional inertia, expanding the understanding of the urban water system, and advancing the governance of water resources towards a *blue city*. Finally, this level of open data sharing has potential to facilitate further scientific research and learning, which play a central role in sustainable development goals at local and global scales [327, 328].

CHAPTER 9

CONCLUSION

Blue cities rely upon an enhanced understanding and view of the urban water cycle. There is a need for increased data collection and publicly available data to support this enhanced understanding of the urban water cycle. This dissertation utilized concepts of the food-energy-water nexus, urban metabolism, and urban water governance to provide a deeper level of knowledge related to the urban water cycle, including both its direct and indirect water components. As part of this dussertation, five research questions were proposed and evaluated. Answering these questions provided a clear path forward for advancing *blue cities*: increased open data are needed to facilitate benchmarking and urban water governance. From the five research questions in Chapter 1, the following conclusions were drawn:

1. **What data are required to describe the urban water cycle and where can they be found?**

   The non-standard processes of data collection across multiple utilities did not enable efficient collection of data for the purposes of comparison. Non-standard data collection and often less-than-efficient organization of the data hindered decision-making at the systems level considering water resources and energy for water. This widespread problem opens opportunities for targeted policy and the use of data management software. Policy at the state or national level could create standards for utility data collection and encourage utilities to better manage the data collection process and report data publicly. The efficient collection and organization of data enable a greater understanding of the system and allow for targeted improvements to upgrade the sustainability and resilience of the system.

2. **What is the current state of U.S. urban energy-water nexus?**

Anthropogenic water consumption occupies a central component of the global water system [203, 204], especially considering the large human impact in urban areas. Chapter 4 particularly emphasized the relationship of urban water flux and its embedded energy, both primary and secondary energy sources, finding that energy use for wastewater and drinking water resources equated to 1% of the total generated electricity in the United States. Additionally, non-revenue water loss equaled 15.7% of total treated drinking water across the United States, with enough embedded energy to power 300,000 average U.S. homes, annually. It is necessary to continue data collection efforts at a utility level with open access data. There are significant future opportunities for sustainability and resilience studies and initiatives associated with holistic, national analyses of urban water and embedded energy. Expansion of collected data could include on-site electricity generation through biogas turbines, solar panels, or alternative energy sources.

3. **How do the direct and indirect components of the urban water cycle compare across U.S. cities?**

   Indirect water, comprised of virtual water and water-for-energy, dominated the urban water cycle on a per capita basis. On average, indirect water was an order of magnitude greater than direct water consumption. Additionally, indirect water imports associated with food and fuel were the major components of overall indirect water. Understanding indirect water and its sourcing is essential for detecting vulnerabilities in the urban water supply [71, 231]. The results of Chapter 5 further supported the need for indirect water calculations to be included in urban water accounting and policy considerations.

4. **How has the material demand of cities changed over the past 50 years and what data exist to quantify this change?**

   In what has been widely cited as the seminal study in urban metabolism, Wolman [122] defines the material consumption of a typical U.S. city of one million people. However, since that time, urban metabolism studies have rarely returned their focus back to the United States. With a need to assess resource sustainability in the face of climate change and localized sustainability initiatives, there are opportunities to provide quantitative assessments of cities and their impacts on the surrounding environment through urban metabolism. The potential for urban metabolism lies within its diversity and adaptable approach to understanding a city [232]. Chapter 6 further showed the dominance and subsequent importance of water resources in the urban environment. Additionally, there is a distinct need to promote data collection efforts that refine the understanding of urban material demands to promote sustainable policy and consumption habits.

5. **How do social, economic, environmental, and infrastructure factors indicate the use of urban water resources?**

   Throughout Chapters 4 through 6, the identification of material demands from a top-down perspective were promoted. However, it is necessary to consider these material demands within the context of the urban sociotechnical and socioecological systems. The numbers are not the whole story; the story includes the human component and the demand, behaviors, and waste that people generate. Chapter 7 demonstrated the limits of utilizing a top-down urban metabolism in defining the city and its drivers of growth and material demands. As a bottom-up approach that collects data at the household scale has both privacy and availability concerns, it is necessary to identify appropriate scales for understanding the social dimensions of material demands. What scales (neighborhood, census tract, etc.) balance both data availability, privacy, and model effectiveness to generate sufficient understanding of socioeconomic drivers of the urban water cycle? Importantly, the limits of this research promoted the necessary role of social sciences in engineering and the urban water cycle.

Sustainable transitions in the water sector are not inevitable and must utilize interdisciplinary perspectives [153]. To that end, this dissertation incorporated a wide spectrum of fields to identify and define key components of the urban water cycle. Sustainable urban water management and governance requires a feedback cycle that is predicated upon data and analysis [170, 182]. To advance urban water transitions, data collection from utilities, the private sector, and public entities can help overcome institutional inertia, expand the understanding of the urban water system, and promote sustainable governance of water resources towards a *blue city.* The energy crisis of the 1970s was the impetus behind creating the EIA. What water crisis beyond those in Flint, MI [329] or in Toledo, OH [330] are necessary to spur national response to water data availability [331]? Increased levels of open data have the potential to facilitate further scientific research and learning, central components of sustainable development goals at local and global scales [327, 328].

REFERENCES

[1] Erik Swyngedouw. *Social power and the urbanization of water: Flows of power.* Oxford University Press Oxford, 2004.

[2] World urbanization prospects, the 2014 revision, 2014. URL `http://esa.un.org/unpd/wup/CD-ROM/`.

[3] Peter M Vitousek, Harold A Mooney, Jane Lubchenco, and Jerry M Melillo. Human domination of Earth's ecosystems. *Science*, 277(5325):494–499, 1997.

[4] Xuemei Bai. Industrial ecology and the global impacts of cities. *Journal of Industrial Ecology*, 11(2):1–6, 2007.

[5] M Bruce Beck, Rodrigo Villarroel Walker, and Michael Thompson. Smarter urban metabolism: Earth systems re-engineering. In *Proceedings of the Institution of Civil Engineers-Engineering Sustainability*, volume 166, pages 229–241. Thomas Telford Ltd, 2013.

[6] Konrad Otto-Zimmermann. From Rio to Rio+ 20: The changing role of local governments in the context of current global governance. *Local Environment*, 17(5):511–516, 2012.

[7] Arnulf Grubler, Xuemei Bai, Thomas Buettner, Shobhakar Dhakal, David Fisk, Toshiaki Ichinose, James Keirstead, Gerd Sammer, David Satterthwaite, and Niels Schulz. Urban energy systems. *Global energy assessment: Toward a sustainable future*, pages 1307–1400, 2012.

[8] Richard Dobbs, Sven Smit, Jaana Remes, James Manyika, Charles Roxburgh, and Alejandra Restrepo. *Urban world: Mapping the economic power of cities.* McKinsey Global Institute, 2011.

[9] Francis Pamminger and Steven Kenway. Urban metabolism-improving the sustainability of urban water systems. *Journal of the Australian Water Association*, 35(1):28–29, 2008.

[10] S. Kenway, P. Lant, A. Priestley, and P. Daniels. The connection between water and energy in cities: A review. *Water Science & Technology*, 63(9):1965–1973, 2011.

[11] Robert I. McDonald, Katherine Weber, Julie Padowski, Martina Flörke, Christof Schneider, Pamela A. Green, Thomas Gleeson, Stephanie Eckman, Bernhard Lehner, Deborah Balk, Timothy Boucher, Günther Grill, and Mark Montgomery. Water on an urban planet: Urbanization and the reach of urban water infrastructure. *Global Environmental Change*, 27(7):96–105, 2014.

[12] British Broadcasting Corporation. Paris climate deal: Trump pulls US out of 2015 accord, 2017. URL `http://www.bbc.com/news/world-us-canada-40127326`.

[13] Garry W McDonald and Murray G Patterson. Bridging the divide in urban sustainability: from human exemptionalism to the new ecological paradigm. *Urban Ecosystems*, 10(2):169–192, 2007.

[14] Allan Schnaiberg. Social syntheses of the societal-environmental dialectic: The role of distributional impacts. *Social Science Quarterly*, pages 5–20, 1975.

[15] Charles H Anderson. *The sociology of survival: Social problems of growth*. Dorsey Press, 1976.

[16] William R Catton Jr. Toward prevention of obsolescence in sociology. *Sociological Focus*, 9(1):89–98, 1976.

[17] William R Catton. Why the future isn't what it used to be (and how it could be made worse than it has to be). *Social Science Quarterly*, 57(2):276–291, 1976.

[18] Liliana B Andonova and Ronald B Mitchell. The rescaling of global environmental politics. *Annual Review of Environment and Resources*, 35(1):255, 2010.

[19] Samuel D Brody, Sammy Zahran, Himanshu Grover, and Arnold Vedlitz. A spatial analysis of local climate change policy in the United States: Risk, stress, and opportunity. *Landscape and Urban Planning*, 87(1):33–41, 2008.

[20] Zhenghong Tang, Samuel D Brody, Courtney Quinn, Liang Chang, and Ting Wei. Moving from agenda to action: evaluating local climate change action plans. *Journal of Environmental Planning and Management*, 53(1):41–62, 2010.

[21] Matthew E Kahn. *Green cities: Urban growth and the environment*. Brookings Institution Press, 2007.

[22] Warren Karlenzig. What makes todays green city? In Eugenie Birch and Susan Wachter, editors, *Growing Greener Cities: Urban Sustainability in the Twenty-First Century*, chapter 19, pages 346–363. University of Pennsylvania Press, 2008.

[23] Eugenie Birch and Susan Wachter. *Growing greener cities: Urban sustainability in the twenty-first century*. University of Pennsylvania Press, 2008.

[24] Anders Blok. Greening cosmopolitan urbanism? On the transnational mobility of low-carbon formats in Northern European and East Asian cities. *Environment and Planning A*, 44(10):2327–2343, 2012.

[25] Eva Gustavsson and Ingemar Elander. Cocky and climate smart? Climate change mitigation and place-branding in three Swedish towns. *Local Environment*, 17(8): 769–782, 2012.

[26] Jacob Lederman. Urban fads and consensual fictions: Creative, sustainable, and competitive city policies in Buenos Aires. *City & Community*, 14(1):47–67, 2015.

[27] Karen Noiva, John E Fernández, and James L Wescoat. Cluster analysis of urban water supply and demand: Toward large-scale comparative sustainability planning. *Sustainable Cities and Society*, 2016.

[28] William Cronon. *Nature's Metropolis: Chicago and the Great West.* WW Norton, 1991.

[29] Norris Hundley. *The great thirst: Californians and water, 1770s-1990s.* University of California Press, 1992.

[30] Robert Gottlieb and Margaret FitzSimmons. *Thirst for growth: Water agencies as hidden government in California.* University of Arizona Press, 1991.

[31] Erik Swyngedouw, Maria Kaika, and Esteban Castro. Urban water: a political-ecology perspective. *Built Environment (1978-)*, pages 124–137, 2002.

[32] Willa Paterson, Richard Rushforth, Benjamin L Ruddell, Megan Konar, Ikechukwu C Ahams, Jorge Gironás, Ana Mijic, and Alfonso Mejia. Water footprint of cities: A review and suggestions for future research. *Sustainability*, 7(7):8461–8490, 2015.

[33] Arjen Y Hoekstra and Ashok K Chapagain. *Globalization of water: Sharing the planet's freshwater resources.* John Wiley & Sons, 2008.

[34] Arjen Y Hoekstra, Ashok K Chapagain, Maite M Aldaya, and Mesfin M Mekonnen. The water footprint assessment manual: Setting the global standard, 2011.

[35] Christian Knieper, Georg Holtz, Britta Kastens, and Claudia Pahl-Wostl. Analysing water governance in heterogeneous case studiesexperiences with a database approach. *Environmental Science & Policy*, 13(7):592–603, 2010.

[36] George M Hornberger, David J Hess, and Jonathan Gilligan. Water conservation and hydrological transitions in cities in the united states. *Water Resources Research*, 51 (6):4635–4649, 2015.

[37] Anne Mette Kjær. Governance: Key concepts. *Cambridge, UK*, 2004.

[38] Magan Farrelly and Rebekah Brown. Rethinking urban water management: Experimentation as a way forward? *Global Environmental Change*, 21(2):721–732, 2011.

[39] Cecilia Tortajada. Water governance: Some critical issues. *International Journal of Water Resources Development*, 26(2):297–307, 2010.

[40] Maria Carmen Lemos, Christine J Kirchhoff, and Vijay Ramprasad. Narrowing the climate information usability gap. *Nature Climate Change*, 2(11):789, 2012.

[41] H-M Füssel. Adaptation planning for climate change: concepts, assessment approaches, and key lessons. *Sustainability Science*, 2(2):265–275, 2007.

[42] SHA Koop, L Koetsier, A Doornhof, O Reinstra, CJ Van Leeuwen, S Brouwer, C Dieperink, and PPJ Driessen. Assessing the governance capacity of cities to address challenges of water, waste, and climate change. *Water Resources Management*, pages 1–17, 2017.

[43] Molly A Maupin, Joan F Kenny, Susan S Hutson, John K Lovelace, Nancy L Barber, and Kristin S Linsey. Estimated use of water in the United States in 2010, 2014.

[44] C.A. Dieter, M.A. Maupin, R.R. Caldwell, T.I. Harris, M.A.and Ivahnenko, J.K. Lovelace, N.L. Barber, and K.S. Linsey. Estimated use of water in the United States in 2015: U.S. Geological Survey Circular 1441, 2018.

[45] Taikan Oki and Shinjiro Kanae. Global hydrological cycles and world water resources. *Science*, 313(5790):1068–1072, 2006.

[46] Peter H Gleick. Water use. *Annual Review of Environment and Resources*, 28(1): 275–314, 2003.

[47] M Konar, C Dalin, S Suweis, N Hanasaki, A Rinaldo, and I Rodriguez-Iturbe. Water for food: The global virtual water trade network. *Water Resources Research*, 47 (5):W05520, 2011.

[48] United States Geological Survey. Current water data for the nation, 2016. URL `http://waterdata.usgs.gov/nwis/rt`.

[49] United States Geological Survey. Water use in the United States, 2010. URL `http://water.usgs.gov/watuse/`.

[50] California State Water Resources Control Board, 2016. URL `http://www.waterboards.ca.gov/resources/data\_databases/`.

[51] American Water Works Association. Benchmarking performance indicators for water and wastewater utilities: 2013 survey data and analyses report. Technical report, 2015.

[52] Circle of Blue. Price of water 2015: Up 6 percent in 30 major US cities; 41 percent rise since 2010, 2015. URL `http://www.circleofblue.org`.

[53] American Water Works Association. Apparent and real losses, 2012. URL `http://www.awwa.org/portals/0/files/resources/water%20knowledge/water%20loss%20control/apparent-real-losses-awwa-updated.pdf`.

[54] Texas Water Development Board. Water loss audit, 2016. URL `http://www.twdb.texas.gov/conservation/municipal/waterloss/`.

[55] Christopher M. Chini, Kelsey L. Schreiber, Zachary A. Barker, and Ashlynn S. Stillwell. Quantifying energy and water savings in the U.S. residential sector. *Environmental Science & Technology*, 50(17):9003–9012, 2016.

[56] Julian Thornton, Reinhard Sturm, and George Kunkel. *Water loss control.* McGraw Hill Professional, 2008.

[57] Alexander Danilenko, Caroline Van den Berg, Berta Macheve, and L Joe Moffitt. *The IBNET water supply and sanitation blue book 2014: The international benchmarking network for water and sanitation utilities databook.* World Bank Publications, 2014.

[58] B Kingdom, R Liemberger, and P Marin. *The challenge of reducing non-revenue water (NRW) in developing countries.* The World Bank Washington DC, USA, 2006.

[59] United States Energy Information Administration, 2018. URL `www.eia.gov`.

[60] United States Energy Information Administration. Annual electric utility data, 2015. URL `https://www.eia.gov/electricity/data/eia923/`.

[61] Paolo D'Odorico, Kyle Frankel Davis, Lorenzo Rosa, Joel A Carr, Davide Chiarelli, Jampel Dell'Angelo, Jessica Gephart, Graham K MacDonald, David A Seekell, and Samir Suweis. The global food-energy-water nexus. *Reviews of Geophysics*, 2018.

[62] Miina Porkka, Matti Kummu, Stefan Siebert, and Olli Varis. From food insufficiency towards trade dependency: A historical analysis of global food availability. *PloS one*, 8(12):e82714, 2013.

[63] John A Allan. Virtual water: A strategic resource global solutions to regional deficits. *Groundwater*, 36(4):545–546, 1998.

[64] Carole Dalin, Megan Konar, Naota Hanasaki, Andrea Rinaldo, and Ignacio Rodriguez-Iturbe. Evolution of the global virtual water trade network. *Proceedings of the National Academy of Sciences*, 109(16):5989–5994, 2012.

[65] Qian Dang, Xiaowen Lin, and Megan Konar. Agricultural virtual water flows within the United States. *Water Resources Research*, 51(2):973–986, 2015.

[66] Landon Marston, Megan Konar, Ximing Cai, and Tara J. Troy. Virtual groundwater transfers from overexploited aquifers in the United States. *Proceedings of the National Academy of Sciences*, 112(28):8561–8566, 2015.

[67] Mesfin M Mekonnen and Arjen Y Hoekstra. The green, blue and grey water footprint of crops and derived crop products. *Hydrology and Earth System Sciences*, 15(5): 1577–1600, 2011.

[68] Arjen Y Hoekstra and Mesfin M Mekonnen. The water footprint of humanity. *Proceedings of the National Academy of Sciences*, 109(9):3232–3237, 2012.

[69] Joel A Carr, Paolo D'Odorico, Francesco Laio, and Luca Ridolfi. Recent history and geography of virtual water trade. *PloS one*, 8(2):e55825, 2013.

[70] D. Vanham and G. Bidoglio. The water footprint of Milan. *Water Science and Technology*, 69(4):789–795, 2014. ISSN 0273-1223. doi: 10.2166/wst.2013.759. URL `http://wst.iwaponline.com/content/69/4/789`.

[71] Richard R. Rushforth and Benjamin L. Ruddell. The vulnerability and resilience of a city's water footprint: The case of Flagstaff, Arizona, USA. *Water Resources Research*, 52(4):2698–2714, 2016.

[72] D. Vanham, T.N. Mak, and B.M. Gawlik. Urban food consumption and associated water resources: The example of Dutch cities. *Science of The Total Environment*, 565: 232 – 239, 2016.

[73] D Vanham, S Del Pozo, AG Pekcan, L Keinan-Boker, A Trichopoulou, and BM Gawlik. Water consumption related to different diets in Mediterranean cities. *Science of The Total Environment*, 573:96–105, 2016.

[74] PW Gerbens-Leenes, AY Hoekstra, and R Bosman. The blue and grey water footprint of construction materials: Steel, cement and glass. *Water Resources and Industry*, 19: 1–12, 2018.

[75] Joep F Schyns, Martijn J Booij, and Arjen Y Hoekstra. The water footprint of wood for lumber, pulp, paper, fuel and firewood. *Advances in Water Resources*, 107:490–501, 2017.

[76] Davide Danilo Chiarelli, Lorenzo Rosa, Maria Cristina Rulli, and Paolo D'Odorico. The water-land-food nexus of natural rubber production. *Journal of Cleaner Production*, 172(1):1739–1747, 2018.

[77] Wayne B Solley, Robert R Pierce, and Howard A Perlman. *Estimated use of water in the United States in 1995.* US Geological Survey, 1998.

[78] Ashlynn S Stillwell, Mary E Clayton, and Michael E Webber. Technical analysis of a river basin-based model of advanced power plant cooling technologies for mitigating water management challenges. *Environmental Research Letters*, 6(3):034015, 2011.

[79] J Macknick, R Newmark, G Heath, and KC Hallett. Operational water consumption and withdrawal factors for electricity generating technologies: a review of existing literature. *Environmental Research Letters*, 7(4):045802, 2012.

[80] Molly A Maupin, Joan F Kenny, Susan S Hutson, John K Lovelace, Nancy L Barber, and Kristin S Linsey. *Estimated use of water in the United States in 2010.* US Geological Survey, 2014.

[81] Kelly T Sanders. Critical review: Uncharted waters? The future of the electricity-water nexus. *Environmental Science & Technology*, 49(1):51–66, 2014.

[82] T. A. DeNooyer, J. M. Peschel, Z. Zhang, and A. S. Stillwell. Integrating water resources and power generation: the energy-water nexus in Illinois. *Applied Energy*, 162(1):363–371, 2016.

[83] Carey W King and Michael E Webber. Water intensity of transportation. *Environmental Science & Technology*, 42(21):7866–7872, 2008.

[84] Kelly M Twomey, Ashlynn S Stillwell, and Michael E Webber. The unintended energy impacts of increased nitrate contamination from biofuels production. *Journal of Environmental Monitoring*, 12(1):218–224, 2010.

[85] Emily A Grubert, Fred C Beach, and Michael E Webber. Can switching fuels save water? A life cycle quantification of freshwater consumption for Texas coal-and natural gas-fired electricity. *Environmental Research Letters*, 7(4):045801, 2012.

[86] Meagan S Mauter, Pedro JJ Alvarez, Allen Burton, Diego C Cafaro, Wei Chen, Kelvin B Gregory, Guibin Jiang, Qilin Li, Jamie Pittock, and Danny Reible. Regional variation in water-related impacts of shale gas development and implications for emerging international plays. *Environmental Science & Technology*, 48(15):8298–8306, 2014.

[87] MM Mekonnen and AY Hoekstra. The blue water footprint of electricity from hydropower. *Hydrology and Earth System Sciences*, 16:179–187, 2012.

[88] Georgia Destouni, Fernando Jaramillo, and Carmen Prieto. Hydroclimatic shifts driven by human water use for food and energy production. *Nature Climate Change*, 3(3): 213–217, 2013.

[89] Fernando Jaramillo and Georgia Destouni. Local flow regulation and irrigation raise global human water consumption and footprint. *Science*, 350(6265):1248–1251, 2015.

[90] Fernando Jaramillo and Georgia Destouni. Comment on "Planetary boundaries: Guiding human development on a changing planet". *Science*, 348(6240):1217–1217, 2015.

[91] Emily A Grubert. Water consumption from hydroelectricity in the United States. *Advances in Water Resources*, 96(10):88–94, 2016.

[92] Carey W King, Ashlynn S Holman, and Michael E Webber. Thirst for energy. *Nature Geoscience*, 1(5):283, 2008.

[93] J Macknick, S Sattler, K Averyt, S Clemmer, and J Rogers. The water implications of generating electricity: water use across the United States based on different electricity pathways through 2050. *Environmental Research Letters*, 7(4):045803, 2012.

[94] K Averyt, J Macknick, J Rogers, N Madden, J Fisher, J Meldrum, and R Newmark. Water use for electricity in the United States: An analysis of reported and calculated water use information for 2008. *Environmental Research Letters*, 8(1):015001, 2013.

[95] Robert Alan Holland, Kate A Scott, Martina Flörke, Gareth Brown, Robert M Ewers, Elizabeth Farmer, Valerie Kapos, Ann Muggeridge, Jörn PW Scharlemann, Gail Taylor, et al. Global impacts of energy demand on the freshwater resources of nations. *Proceedings of the National Academy of Sciences*, 112(48):6707–6716, 2015.

[96] Rebecca AM Peer and Kelly T Sanders. Characterizing cooling water source and usage patterns across us thermoelectric power plants: a comprehensive assessment of self-reported cooling water data. *Environmental Research Letters*, 11(12):124030, 2016.

[97] Vincent C Tidwell, Michael Bailey, Katie M Zemlick, and Barbara D Moreland. Water supply as a constraint on transmission expansion planning in the western interconnection. *Environmental Research Letters*, 11(12):124001, 2016.

[98] Uisung Lee, Jeongwoo Han, Amgad Elgowainy, and Michael Wang. Regional water consumption for hydro and thermal electricity generation in the United States. *Applied Energy*, 210(1):661–672, 2017.

[99] Ranran Wang, Julie B Zimmerman, Chunyan Wang, David Font Vivanco, and Edgar G Hertwich. Freshwater vulnerability beyond local water stress: Heterogeneous effects of water-electricity nexus across the continental united states. *Environmental Science & Technology*, 51(17):9899–9910, 2017.

[100] Vasilis Fthenakis and Hyung Chul Kim. Life-cycle uses of water in US electricity generation. *Renewable and Sustainable Energy Reviews*, 14(7):2039–2048, 2010.

[101] Indika Herath, Markus Deurer, David Horne, Ranvir Singh, and Brent Clothier. The water footprint of hydroelectricity: A methodological comparison from a case study in New Zealand. *Journal of Cleaner Production*, 19(14):1582–1589, 2011.

[102] Mesfin M Mekonnen, PW Gerbens-Leenes, and Arjen Y Hoekstra. The consumptive water footprint of electricity and heat: A global assessment. *Environmental Science: Water Research & Technology*, 1(3):285–297, 2015.

[103] Xinxin Zhang, Junguo Liu, Yu Tang, Xu Zhao, Hong Yang, PW Gerbens-Leenes, Michelle TH van Vliet, and Jinyue Yan. China's coal-fired power plants impose pressure on water resources. *Journal of Cleaner Production*, 161: 1171–1179, 2017.

[104] Ashlynn S Stillwell, Carey W King, Michael E Webber, Ian J Duncan, and Amy Hardberger. The energy-water nexus in Texas. *Ecology and Society*, 16(1), 2011.

[105] Rebecca AM Peer and Kelly T Sanders. The water consequences of a transitioning US power sector. *Applied Energy*, 210:613–622, 2017.

[106] Lauren Wisniewski. Take action on all fronts to ensure power resilience. *Opflow*, 41 (12):12–15, 2015.

[107] Ka Leung Lam, Steven J Kenway, and Paul A Lant. Energy use for water provision in cities. *Journal of Cleaner Production*, 143:699–709, 2017.

[108] Robert B. Sowby and Steven J. Burian. Survey of energy requirements for public water supply in the United States. *Journal of the American Water Works Association*, 7 (109), 2017.

[109] Robert Goldstein and W Smith. *Water & sustainability (volume 4): US electricity consumption for water supply & treatment-the next half century.* Electric Power Research Institute, 2002.

[110] Kelly M Twomey and Michael E Webber. Evaluating the energy intensity of the US public water system. *ASME 2011 5th International Conference on Energy Sustainability*, pages 1735–1748, 2011.

[111] Kelly T Sanders and Michael E Webber. Evaluating the energy consumed for water use in the United States. *Environmental Research Letters*, 7(3):034034, 2012.

[112] Kevin James, Stephanie L Campbell, and Christopher E Godlobe. In *Watergy: Taking advantage of untapped energy and water efficiency opportunities in municipal water systems.* Alliance to Save Energy, 2002.

[113] Yaling Liu, Mohamad Hejazi, Page Kyle, Son H Kim, Evan Davies, Diego G Miralles, Adriaan J Teuling, Yujie He, and Dev Niyogi. Global and regional evaluation of energy for water. *Environmental Science & Technology*, 50(17):9736–9745, 2016.

[114] Sabine Barles. Society, energy and materials: The contribution of urban metabolism studies to sustainable urban development issues. *Journal of Environmental Planning and Management*, 53(4):439–455, 2010.

[115] Carolyn Hendriks, Richard Obernosterer, Daniel Müller, Susanne Kytzia, Peter Baccini, and Paul H Brunner. Material flow analysis: A tool to support environmental policy decision making. Case-studies on the city of Vienna and the Swiss lowlands. *Local Environment*, 5(3):311–328, 2000.

[116] Samuel Niza, Leonardo Rosado, and Paulo Ferrão. Urban metabolism: Methodological advances in urban material flow accounting based on the Lisbon case study. *Journal of Industrial Ecology*, 13(3):384–405, 2009.

[117] Samuel Niza and P Ferrão. Material flow accounting tools and its contribution for policy making. In *International Conference of the European Society for Ecological Economics*, pages 14–17, 2005.

[118] Hans Daxbeck, Christoph Lampert, Leo Morf, Richard Obernosterer, Helmut Rechberger, Iris Reiner, and Paul H Brunner. The anthropogenic metabolism of the city of Vienna. *From Paradigm to Practice of Sustainability*, 21:250, 1997.

[119] F Burstrom, Nils Brandt, Björn Frostell, and Ulf Mohlander. Material flow accounting and information for environmental policies in the city of Stockholm. In *Analysis for Action: Support for Policy towards Sustainability by Material Flow Accounting, Proceedings from the Conaccount Conference, Wuppertal, Germany*, pages 153–164, 1998.

[120] Sabine Barles. Urban metabolism of Paris and its region. *Journal of Industrial Ecology*, 13(6):898–913, 2009.

[121] Christopher Kennedy. A mathematical description of urban metabolism. In *Sustainability Science*, pages 275–291. Springer, 2012.

[122] A. Wolman. The metabolism of cities. *Scientific American*, 213(3):179–190, 1965.

[123] Ian Douglas. The urban environment. *Edward Arnold*, 1983.

[124] Benjamin Goldstein, Morten Birkved, Maj-Britt Quitzau, and Michael Hauschild. Quantification of urban metabolism through coupling with the life cycle assessment framework: Concept development and case study. *Environmental Research Letters*, 8 (3):035024, 2013.

[125] C. Kennedy, J. Cuddihy, and J. Engel-Yan. The changing metabolism of cities. *Journal of Industrial Ecology*, 11(2):43–59, 2007.

[126] Virginia White Maclaren. *Developing indicators of urban sustainability: A focus on the Canadian Experience*. ICURR Press, Toronto, ON, 1996.

[127] C. Kennedy, S. Pincetl, and P. Bunje. The study of urban metabolism and its applications to urban planning and design. *Environmental Pollution*, 159(8):1965–1973, 2011.

[128] Christopher Kennedy and Daniel Hoornweg. Mainstreaming urban metabolism. *Journal of Industrial Ecology*, 16(6):780–782, 2012.

[129] Yuliya Kalmykova, Leonardo Rosado, and João Patrício. Urban economies resource productivity and decoupling: Metabolism trends of 1996–2011 in Sweden, Stockholm, and Gothenburg. *Environmental Science & Technology*, 49(14):8815–8823, 2015.

[130] Xuemei Bai, Anna J Wieczorek, Shinji Kaneko, Shaun Lisson, and Antonio Contreras. Enabling sustainability transitions in Asia: the importance of vertical and horizontal linkages. *Technological Forecasting and Social Change*, 76(2):255–266, 2009.

[131] Leonardo Rosado, Yuliya Kalmykova, and João Patrício. Urban metabolism profiles. An empirical analysis of the material flow characteristics of three metropolitan areas in Sweden. *Journal of Cleaner Production*, 126:206–217, 2016.

[132] *The Ecology of a City and Its People: The Case of Hong Kong*. Australian National University, 1981.

[133] Peter Newman and Jeffrey Kenworthy. *Sustainability and cities: Overcoming automobile dependence*. Island Press, 1999.

[134] Halla R Sahely, Shauna Dudding, and Christopher A Kennedy. Estimating the urban metabolism of Canadian cities: Greater Toronto area case study. *Canadian Journal of Civil Engineering*, 30(2):468–483, 2003.

[135] E. H. Decker, S. Elliott, F. A. Smith, D. R. Blake, and F. S. Rowland. Energy and material flow through the urban ecosystem. *Annual Review of Energy and the Environment*, 25:685–740, 2000.

[136] Steven Kenway, Alan Gregory, and Joseph McMahon. Urban water mass balance analysis. *Journal of Industrial Ecology*, 15(5):693–706, 2011.

[137] Yan Zhang. Urban metabolism: A review of research methodologies. *Environmental Pollution*, 178:463–473, 2013.

[138] James Zucchetto. Energy-economic theory and mathematical models for combining the systems of man and nature, case study: The urban region of Miami, Florida. *Ecological Modelling*, 1(4):241–268, 1975.

[139] Lawrence A Baker, Diane Hope, Ying Xu, Jennifer Edmonds, and Lisa Lauver. Nitrogen balance for the Central Arizona–Phoenix ecosystem. *Ecosystems*, 4(6):582–602, 2001.

[140] Marc Melaina and Gregory Keoleian. A framework for urban energy metabolism studies: An Ann Arbor, Michigan case study. In *The Science & Culture of Industrial Ecology Conference of the International Society for Industrial Ecology*, pages 12–14, 2001.

[141] Joel A Tarr. The metabolism of the industrial city: The case of Pittsburgh. *Journal of Urban History*, 28(5):511–545, 2002.

[142] NS Ngo and Diane E Pataki. The energy and mass balance of Los Angeles County. *Urban Ecosystems*, 11(2):121–139, 2008.

[143] Stephanie Pincetl, Mikhail Chester, Giovanni Circella, Andrew Fraser, Caroline Mini, Sinnott Murphy, Janet Reyna, and Deepak Sivaraman. Enabling future sustainability transitions. *Journal of Industrial Ecology*, 18(6):871–882, 2014.

[144] Stephanie Pincetl and Joshua P. Newell. Why data for a political-industrial ecology of cities? *Geoforum*, 85:381–391, 2017.

[145] SJ Kenway, A Binks, J Lane, PA Lant, KL Lam, and A Simms. A systemic framework and analysis of urban water energy. *Environmental Modelling & Software*, 73:272–285, 2015.

[146] Christopher A Scott, Arica Crootof, and Sarah Kelly-Richards. The urban water–energy nexus: Building resilience for global change in the urban century. In *Environmental Resource Management and the Nexus Approach*, pages 113–140. Springer, 2016.

[147] R Villarroel Walker, MB Beck, JW Hall, RJ Dawson, and O Heidrich. The energy-water-food nexus: Strategic analysis of technologies for transforming the urban metabolism. *Journal of environmental management*, 141:104–115, 2014.

[148] R Maria Saleth and Ariel Dinar. Water institutional reforms: Theory and practice. *Water Policy*, 7(1):1–19, 2005.

[149] Kees van Kersbergen and Frans van Waarden. 'Governance' as a bridge between disciplines: Cross-disciplinary inspiration regarding shifts in governance and problems of governability, accountability and legitimacy. *European Journal of Political Research*, 43(2):143–171, 2004.

[150] Claudia Pahl-Wostl. Transitions towards adaptive management of water facing climate and global change. *Water Resources Management*, 21(1):49–62, 2007.

[151] Jeroen Rijke, Megan Farrelly, Rebekah Brown, and Chris Zevenbergen. Configuring transformative governance to enhance resilient urban water systems. *Environmental Science & Policy*, 25:62–72, 2013.

[152] Joannette J Bos and Rebekah R Brown. Governance experimentation and factors of success in socio-technical transitions in the urban water sector. *Technological Forecasting and Social Change*, 79(7):1340–1353, 2012.

[153] Galen Treuer, Elizabeth Koebele, Aaron Deslatte, Kathleen Ernst, Margaret Garcia, and Kim Manago. A narrative method for analyzing transitions in urban water management: The case of the Miami-Dade Water and Sewer Department. *Water Resources Research*, 53(1):891–908, 2017.

[154] Mark Lubell, William D Leach, and Paul A Sabatier. Collaborative watershed partnerships in the epoch of sustainability. *Toward sustainable communities: Transitions and transformations in environmental policy*, pages 255–88, 2009.

[155] Arnim Wiek and Kelli L Larson. Water, people, and sustainability–a systems framework for analyzing and assessing water governance regimes. *Water Resources Management*, 26(11):3153–3171, 2012.

[156] Susana Neto. Water governance in an urban age. *Utilities Policy*, 43:32–41, 2016.

[157] Tony HF Wong. Water sensitive urban design-the journey thus far. *Australian Journal of Water Resources*, 10(3):213–222, 2006.

[158] Beatrice Hedelin. Criteria for the assessment of sustainable water management. *Environmental Management*, 39(2):151–163, 2007.

[159] Elinor Ostrom. A general framework for analyzing sustainability of social-ecological systems. *Science*, 325(5939):419–422, 2009.

[160] Claudia Pahl-Wostl, Georg Holtz, Britta Kastens, and Christian Knieper. Analyzing complex water governance regimes: the management and transition framework. *Environmental Science & Policy*, 13(7):571–581, 2010.

[161] Anik Bhaduri, Janos Bogardi, Afreen Siddiqi, Holm Voigt, Charles Vörösmarty, Claudia Pahl-Wostl, Stuart E Bunn, Paul Shrivastava, Richard Lawford, Stephen Foster, et al. Achieving sustainable development goals from a water perspective. *Frontiers in Environmental Science*, 4:64, 2016.

[162] Niki Frantzeskaki, Derk Loorbach, and James Meadowcroft. Governing societal transitions to sustainability. *International Journal of Sustainable Development*, 15(1-2): 19–36, 2012.

[163] Harriet Bulkeley. Reconfiguring environmental governance: Towards a politics of scales and networks. *Political geography*, 24(8):875–902, 2005.

[164] M Hodson and S Marvin. Making low-carbon England and Wales. *Town and Country Planning*, 79(9):388–392, 2011.

[165] Adam Greenfield. *Against the Smart City: A Pamphlet.* Do projects, 2013.

[166] Emily Boyd and Aditya Ghosh. Innovations for enabling urban climate governance: Evidence from Mumbai. *Environment and Planning C: Government and Policy*, 31 (5):926–945, 2013.

[167] James Evans and Andrew Karvonen. 'Give me a laboratory and I will lower your carbon footprint!'–Urban laboratories and the governance of low-carbon futures. *International Journal of Urban and Regional Research*, 38(2):413–430, 2014.

[168] Philipp Dorstewitz. Planning and experimental knowledge production: Zeche Zollverein as an urban laboratory. *International Journal of Urban and Regional Research*, 38(2):431–449, 2014.

[169] Andrés Luque-Ayala and Simon Marvin. Developing a critical understanding of smart urbanism? *Urban Studies*, 52(12):2105–2116, 2015.

[170] Christopher M Chini, James F Canning, Kelsey L Schreiber, Joshua M Peschel, and Ashlynn S Stillwell. The green experiment: Cities, green stormwater infrastructure, and sustainability. *Sustainability*, 9(1):105, 2017.

[171] Cedo Maksimovic and José Alberto Tejada-Guibert. *Frontiers in urban water management: Deadlock or Hope.* IWA Publishing, 2001.

[172] Carl Folke, Thomas Hahn, Per Olsson, and Jon Norberg. Adaptive governance of social-ecological systems. *Annu. Rev. Environ. Resour.*, 30:441–473, 2005.

[173] THF Wong and RR Brown. The water sensitive city: principles for practice. *Water Science and Technology*, 60(3):673–682, 2009.

[174] Mark T Imperial. Analyzing institutional arrangements for ecosystem-based management: Lessons from the Rhode Island salt ponds SAM plan. *Coastal Management*, 27 (1):31–56, 1999.

[175] RR Brown, Liz Sharp, and RM Ashley. Implementation impediments to institutionalising the practice of sustainable urban water management. *Water Science and Technology*, 54(6-7):415–422, 2006.

[176] Robert C. Brears. *Transitions towards managing natural resources and water*, pages 105–135. John Wiley & Sons, Ltd, 2017.

[177] James Keeley and Ian Scoones. *Understanding Environmental Policy Processes: Cases from Africa*. Earthscan, 2003.

[178] Joyeeta Gupta. "Glocal" water governance: Controversies and choices. In *Water for a Changing World Developing Local Knowledge and Capacity*, page 101. CRC Press, 2008.

[179] Thomas Uhlendahl, Pritam Salian, Claudia Casarotto, and Jakob Doetsch. Good water governance and IWRM in Zambia: Challenges and chances. *Water Policy*, 13 (6):845–862, 2011.

[180] Ganesh B Keremane, Zhifang Wu, and Jennifer Margaret McKay. Is organisational culture a barrier to implementing integrated urban water management in Adelaide? *Journal of the Australian Water Association*, 41(8):27–29, 2014.

[181] Ellen Van Bueren and Jitske De Jong. Establishing sustainability: policy successes and failures. *Building Research & Information*, 35(5):543–556, 2007.

[182] RR Brown, N Keath, and THF Wong. Urban water management in cities: historical, current and future regimes. *Water Science and Technology*, 59(5):847–855, 2009.

[183] Per Olsson, Carl Folke, and Fikret Berkes. Adaptive comanagement for building resilience in social–ecological systems. *Environmental Management*, 34(1):75–90, 2004.

[184] Rutger Van der Brugge and Jan Rotmans. Towards transition management of European water resources. *Water Resources Management*, 21(1):249–267, 2007.

[185] Rebekah R Brown and Megan A Farrelly. Delivering sustainable urban water management: A review of the hurdles we face. *Water Science and Technology*, 59(5):839–846, 2009.

[186] California Legislative Code. Chapter 3.5 inspection of public records [6250-6276.48], 1970. URL `http://leginfo.legislature.ca.gov/`.

[187] Karen J Bakker. A political ecology of water privatization. *Studies in Political Economy*, 70, 2003.

[188] Karen Bakker. *Privatizing water: Governance failure and the world's urban water crisis*. Cornell University Press, 2010.

[189] William J. Cosgrove and Daniel P. Loucks. Water management: Current and future challenges and research directions. *Water Resources Research*, 51(6):4823–4839, 2015.

[190] Julie C. Padowski and James W. Jawitz. Water availability and vulnerability of 225 large cities in the United States. *Water Resources Research*, 48(12):W12529, 2012.

[191] Matthew Gandy. Rethinking urban metabolism: Water, space and the modern city. *City*, 8(3):363–379, 2004.

[192] Scott Elliott, Ethan Decker, Felisa A Smith, Donald R Blake, Isobel J Simpson, and F Sherwood Rowland. Cities in the earth system. *Environmental Science & Policy*, 3 (4):157–160, 2000.

[193] Christopher M. Chini and Ashlynn S. Stillwell. Urban energy-water nexus: Utility-level water flows and embedded energy, Consortium of Universities for the Advancement of Hydrologic Sciences, Inc. (CUAHSI), 2018.

[194] United States Census Bureau. Commodity flow survey. URL `http://www.census.gov/econ/cfs/`, 2012a.

[195] Raphael Semiat. Energy issues in desalination processes. *Environmental Science & Technology*, 42(22):8193–8201, 2008.

[196] U.S. Environmental Protection Agency. Opportunities for combined heat and power at wastewater treatment facilities: Market analysis and lessons from the field. 2011. URL `https://www.epa.gov/sites/production/files/2015-07/documents/opportunities_for_combined_heat_and_power_at_wastewater_treatment_facilities_market_analysis_and_lessons_from_the_field.pdf`.

[197] U.S. Environmental Protection Agency. National pollutant discharge elimination system (NPDES), 2017. URL `https://www.epa.gov/npdes/combined-sewer-overflows-csos`.

[198] C Mini, TS Hogue, and S Pincetl. Estimation of residential outdoor water use in Los Angeles, California. *Landscape and Urban Planning*, 127:124–135, 2014.

[199] American Society of Civil Engineers. Failure to act: The economic impact of current investment trends in water and wastewater treatment infrastructure, 2011. URL `http://www.asce.org/uploadedFiles/Issues_and_Advocacy/Our_Initiatives/Infrastructure/Content_Pieces/failure-to-act-water-wastewater-report.pdf`.

[200] United States Census Bureau. U.S. and world population clock, 2012. URL `http://www.census.gov/popclock/`.

[201] Energy Information Administration. Electric power annual: Table 3.1.a, 2016. URL `http://www.eia.gov/electricity/annual/`.

[202] Energy Information Administration. Electric sales, revenue, and average price: Table 5.a, 2016. URL `http://www.eia.gov/electricity/sales_revenue_price/`.

[203] Charles J Vörösmarty and Dork Sahagian. Anthropogenic disturbance of the terrestrial water cycle. *BioScience*, 50(9):753–765, 2000.

[204] Charles Vörösmarty, Dennis Lettenmaier, C Leveque, M Meybeck, Claudia Pahl-Wostl, John Alcamo, William Cosgrove, Hubert Grassl, Holger Hoff, P Kabat, et al. Humans transforming the global water system. *Eos, Transactions American Geophysical Union*, 85(48):509–514, 2004.

[205] Karen Hussey and Jamie Pittock. The energy–water nexus: Managing the links between energy and water for a sustainable future. *Ecology and Society*, 17(1), 2012.

[206] Tim Cooper. *Beyond recycling: The longer life option.* New Economics Foundation, London, 1994.

[207] Slawomir W Hermanowicz and Takashi Asano. Abel Wolman's the metabolism of cities revisited: a case for water recycling and reuse. *Water Science and Technology*, 40(4-5):29–36, 1999.

[208] Weiwei Mo, Ranran Wang, and Julie B Zimmerman. Energy–water nexus analysis of enhanced water supply scenarios: A regional comparison of Tampa Bay, Florida, and San Diego, California. *Environmental Science & Technology*, 48(10):5883–5891, 2014.

[209] Sven Eggimann, Lena Mutzner, Omar Wani, Mariane Yvonne Schneider, Dorothee Spuhler, Matthew Moy de Vitry, Philipp Beutler, and Max Maurer. The potential of knowing more: A review of data-driven urban water management. *Environmental Science & Technology*, 51(5):2538–2553, 2017.

[210] Peter H. Gleick. Water strategies for the next administration. *Science*, 354(6312): 555–556, 2016.

[211] Afreen Siddiqi and Olivier L de Weck. Quantifying end-use energy intensity of the urban water cycle. *Journal of Infrastructure Systems*, 19(4), 2013.

[212] Matthew D Bartos and Mikhail V Chester. The conservation nexus: Valuing interdependent water and energy savings in arizona. *Environmental science & technology*, 48 (4):2139–2149, 2014.

[213] Philippe Loubet, Philippe Roux, Eleonore Loiseau, and Veronique Bellon-Maurel. Life cycle assessments of urban water systems: A comparative analysis of selected peer-reviewed literature. *Water Research*, 67:187 – 202, 2014.

[214] Alexandre Bonton, Christian Bouchard, Benoit Barbeau, and Stéphane Jedrzejak. Comparative life cycle assessment of water treatment plants. *Desalination*, 284:42 – 54, 2012.

[215] Energy Information Administration. Energy-related carbon dioxide emissions at the state level, 2000-2013. URL `http://www.eia.gov/environment/emissions/state/analysis/`, 2012.

[216] Naota Hanasaki, Toshiyuki Inuzuka, Shinjiro Kanae, and Taikan Oki. An estimation of global virtual water flow and sources of water withdrawal for major crops and livestock products using a global hydrological model. *Journal of Hydrology*, 384(3): 232–244, 2010.

[217] U.S. Department of Transportation. 2012 Commodity Flow Survey: Standard classification of transported goods, 2012. URL `http://www.census.gov/econ/cfs/`.

[218] United States Department of Agriculture. Census of agriculture: National agricultural statistics service, 2012. URL `http://www.agcensus.usda.gov/Publications/2012/`.

[219] Stanley T Mubako and Christopher L Lant. Agricultural virtual water trade and water footprint of US states. *Annals of the Association of American Geographers*, 103(2): 385–396, 2013.

[220] Stanley Tirivanhu Mubako. *Frameworks for estimating virtual water flows among US states*. PhD thesis, Southern Illinois University Carbondale, Carbondale, IL.

[221] Erik Mielke, Laura Diaz Anadon, and Venkatesh Narayanamurti. Water consumption of energy resource extraction, processing, and conversion. *Belfer Center for Science and International Affairs*, 2010.

[222] Energy Information Administration. Approximate heat value of coal and coke, 2015. URL `http://www.eia.gov/totalenergy/data/monthly/pdf/sec13\char‘_5.pdf`.

[223] May Wu, M Mintz, M Wang, S Arora, et al. Consumptive water use in the production of ethanol and petroleum gasoline. U.S. Department of Energy: Office of Scientific and Technical Information, 2009.

[224] Christopher M. Chini, Lucas A. Djehdian, William N. Lubega, and Ashlynn S. Stillwell. Virtual water transfers of the U.S. electric grid. *Nature Energy*, in press, 2018.

[225] Energy Information Administration. Electricity: Form EIA 923, 2012. URL `https://www.eia.gov/electricity/data/eia923/`.

[226] Energy Information Administration. Natural gas annual supply & disposition by state, 2012. URL `http://www.eia.gov/dnav/ng/ng_sum_snd_a_EPG0_FPD_Mmcf_a.htm`.

[227] Stacy C Davis, Susan W Diegel, and Robert G Boundy. Transportation energy data book: Edition 32, Oak Ridge National Laboratory, Center for Transportation Analysis, 2013.

[228] U.S. Environmental Protection Administration. Tool for Reduction and Assessment of Chemicals and Other Environmental Impacts (TRACI), 2016. URL `http://www.epa.gov/`.

[229] United States Census Bureau. Population estimates, state totals: Vintage 2012, 2012. URL `https://www.census.gov/popest/data/state/totals/2012/`.

[230] Mesfin M. Mekonnen, P.W. Gerbens-Leenes, and Arjen Y. Hoekstra. Future electricity: The challenge of reducing both carbon and water footprint. *Science of The Total Environment*, 569570:1282 – 1288, 2016.

[231] Samir Suweis and Paolo D'Odorico. Early warning signs in social-ecological networks. *PLoS ONE*, 9(7):1–6, 2014.

[232] Vanesa Castán Broto, Adriana Allen, and Elizabeth Rapoport. Interdisciplinary perspectives on urban metabolism. *Journal of Industrial Ecology*, 16(6):851–861, 2012.

[233] U.S. Department of Transportation, Federal Highway Administration. Freight analysis framework (faf) version 4, 2018. URL `http://faf.ornl.gov/fafweb/`.

[234] Adele Finco and Peter Nijkamp. Pathways to urban sustainability. *Journal of Environmental Policy and Planning*, 3(4):289–302, 2001.

[235] Energy Information Administration. Energy-related carbon dioxide emissions at the state level, 2000-2013, 2012. URL `http://www.eia.gov/environment/emissions/state/analysis/`.

[236] James Thompson and Rob Watson. Time is running out: The U.S. landfill capacity crisis, Solid Waste Environmental Excellence Protocol, 2018. URL `https://nrra.net/sweep/time-is-running-out-the-u-s-landfill-capacity-crisis/`.

[237] Amanda D Cuéllar and Michael E Webber. Wasted food, wasted energy: The embedded energy in food waste in the United States. *Environmental Science & Technology*, 44(16):6464–6469, 2010.

[238] Peter Baccini. A city's metabolism: Towards the sustainable development of urban systems. *Journal of Urban Technology*, 4(2):27–39, 1997.

[239] Climate Mayors. U.S. mayors demonstrate ambitious, collective climate leadership, 2017. URL `http://climatemayors.org`.

[240] Paulo Ferrão and John E Fernández. *Sustainable urban metabolism.* MIT Press, 2013.

[241] Massimiliano Mazzanti and Anna Montini. The determinants of residential water demand: Empirical evidence for a panel of Italian municipalities. *Applied Economics Letters*, 13(2):107–111, 2006.

[242] Céline Nauges and Alban Thomas. Privately operated water utilities, municipal price negotiation, and estimation of residential water demand: The case of France. *Land Economics*, pages 68–85, 2000.

[243] Fernando Arbues and Inmaculada Villanua. Potential for pricing policies in water resource management: Estimation of urban residential water demand in Zaragoza, Spain. *Urban Studies*, 43(13):2421–2442, 2006.

[244] Elena Domene and David Saurí. Urbanisation and water consumption: Influencing factors in the metropolitan region of Barcelona. *Urban Studies*, 43(9):1605–1623, 2006.

[245] Geoffrey J Syme, Quanxi Shao, Murni Po, and Eddy Campbell. Predicting and understanding home garden water use. *Landscape and Urban Planning*, 68(1):121–128, 2004.

[246] Subhrajit Guhathakurta and Patricia Gober. The impact of the Phoenix urban heat island on residential water use. *Journal of the American Planning Association*, 73(3): 317–329, 2007.

[247] Douglas S Kenney, Christopher Goemans, Roberta Klein, Jessica Lowrey, and Kevin Reidy. Residential water demand management: Lessons from Aurora, Colorado. *Journal of the American Water Resources Association*, 44(1):192–207, 2008.

[248] Joachim Schleich and Thomas Hillenbrand. Determinants of residential water demand in Germany. *Ecological Economics*, 68(6):1756–1769, 2009.

[249] Graeme Dandy, Tin Nguyen, and Carolyn Davies. Estimating residential water demand in the presence of free allowances. *Land Economics*, pages 125–139, 1997.

[250] Fernando Arbúes, Ramón Barberan, and Inmaculada Villanua. Price impact on urban residential water demand: A dynamic panel data approach. *Water Resources Research*, 40(11):W11402, 2004.

[251] Sylvestre Gaudin. Effect of price information on residential water demand. *Applied Economics*, 38(4):383–393, 2006.

[252] Mark Hoffmann, Andrew Worthington, and Helen Higgs. Urban water demand with fixed volumetric charging in a large municipality: The case of Brisbane, Australia. *Australian Journal of Agricultural and Resource Economics*, 50(3):347–359, 2006.

[253] A Sankarasubramanian, JL Sabo, KL Larson, SB Seo, T Sinha, R Bhowmik, A Ruhi Vidal, K Kunkel, G Mahinthakumar, EZ Berglund, et al. Synthesis of public water supply use in the United States: Spatio-temporal patterns and socio-economic controls. *Earth's Future*, 5(7):771–788, 2017.

[254] C Fox, BS McIntosh, and P Jeffrey. Classifying households for water demand forecasting using physical property characteristics. *Land Use Policy*, 26(3):558–568, 2009.

[255] Hug March and David Saurí. The suburbanization of water scarcity in the Barcelona Metropolitan Region: Sociodemographic and urban changes influencing domestic water consumption. *The Professional Geographer*, 62(1):32–45, 2010.

[256] Elizabeth A Wentz and Patricia Gober. Determinants of small-area water consumption for the City of Phoenix, Arizona. *Water Resources Management*, 21(11):1849–1863, 2007.

[257] Heather Cooley and Peter H Gleick. Urban water use efficiencies: Lessons from United States cities. In *The World's Water: 2008–2009. The Biennial Report on Freshwater Resources*. Island Press, 2009.

[258] Fernando Arbués, Marıa Ángeles Garcıa-Valiñas, and Roberto Martınez-Espiñeira. Estimation of residential water demand: A state-of-the-art review. *The Journal of Socio-Economics*, 32(1):81–102, 2003.

[259] Lily House-Peters, Bethany Pratt, and Heejun Chang. Effects of urban spatial structure, sociodemographics, and climate on residential water consumption in Hillsboro, Oregon. *Journal of the American Water Resources Association*, 46(3):461–472, 2010.

[260] Lily A House-Peters and Heejun Chang. Urban water demand modeling: Review of concepts, methods, and organizing principles. *Water Resources Research*, 47 (5):W05401, 2011.

[261] R Quentin Grafton, Michael B Ward, Hang To, and Tom Kompas. Determinants of residential water consumption: Evidence and analysis from a 10-country household survey. *Water Resources Research*, 47(8):W08537, 2011.

[262] Mary E Renwick and Sandra O Archibald. Demand side management policies for residential water use: who bears the conservation burden? *Land Economics*, pages 343–359, 1998.

[263] Fernando Arbues, Inmaculada Villanúa, and Ramón Barberán. Household size and residential water demand: An empirical approach. *Australian Journal of Agricultural and Resource Economics*, 54(1):61–80, 2010.

[264] Betsy Breyer, Heejun Chang, and G Hossein Parandvash. Land-use, temperature, and single-family residential water use patterns in Portland, Oregon and Phoenix, Arizona. *Applied Geography*, 35(1-2):142–151, 2012.

[265] Robert F Young. Interdisciplinary foundations of urban ecology. *Urban Ecosystems*, 12(3):311–331, 2009.

[266] E. P. Odum. *Fundamentals of Ecology.* Brooks Cole, 1953.

[267] Marina Fischer-Kowalski and Walter Hüttler. Society's metabolism. *Journal of Industrial Ecology*, 2(4):107–136, 1998.

[268] Brian Fagan. *Elixir: A history of water and humankind.* Bloomsbury Publishing, 2011.

[269] David Satterthwaite. Sustainable cities or cities that contribute to sustainable development? *Urban Studies*, 34(10):1667–1691, 1997.

[270] National Research Council (US). Committee on Grand Challenges in Environmental Sciences. *Grand challenges in environmental sciences.* National Academy Press, 2001.

[271] Luís MA Bettencourt, José Lobo, Dirk Helbing, Christian Kühnert, and Geoffrey B West. Growth, innovation, scaling, and the pace of life in cities. *Proceedings of the National Academy of Sciences*, 104(17):7301–7306, 2007.

[272] Ifigeneia Koutiva and Christos Makropoulos. Modelling domestic water demand: An agent based approach. *Environmental Modelling & Software*, 79:35–54, 2016.

[273] Bradley Jorgensen, Michelle Graymore, and Kevin O'Toole. Household water use behavior: An integrated model. *Journal of Environmental Management*, 91(1):227–236, 2009.

[274] Georges A Tanguay, Juste Rajaonson, Jean-François Lefebvre, and Paul Lanoie. Measuring the sustainability of cities: An analysis of the use of local indicators. *Ecological Indicators*, 10(2):407–418, 2010.

[275] Thomas Dietz, Christina Leshko, and Aaron M McCright. Politics shapes individual choices about energy efficiency. *Proceedings of the National Academy of Sciences*, 110 (23):9191–9192, 2013.

[276] Jess Chandler. Trendy solutions: Why do states adopt sustainable energy portfolio standards? *Energy Policy*, 37(8):3274–3281, 2009.

[277] Jochen Monstadt. Conceptualizing the political ecology of urban infrastructures: Insights from technology and urban studies. *Environment and Planning A*, 41(8):1924–1942, 2009.

[278] Vanesa Castán Broto and Harriet Bulkeley. Maintaining climate change experiments: Urban political ecology and the everyday reconfiguration of urban infrastructure. *International Journal of Urban and Regional Research*, 37(6):1934–1948, 2013.

[279] United States Census Bureau. Quickfacts, 2018. URL `https://www.census.gov/quickfacts/fact/table/US/PST045217`.

[280] Jason Alderman, Gitanjali G. Borkar, Amanda Garrett, Linday Hogan, Janet Kim, Winston Le, Veronica Louie, Alissa Marque, Phil Reiff, Colin C. Richard, Peter Thai, Tania Wang, and Craig Wickersham. The most conservative and liberal cities in the United States, The Bay Area Center for Voting Research, 2005. URL `https://alt.coxnewsweb.com/statesman/metro/081205libs.pdf`.

[281] Weather Underground. Historical weather, 2018. URL `https://www.wunderground.com/history/`.

[282] Robert B Sowby and Steven J Burian. Statistical model and benchmarking procedure for energy use by US public water systems. *Journal of Sustainable Water in the Built Environment*, 4(4):04018010, 2018.

[283] YR Filion. Impact of urban form on energy use in water distribution systems. *Journal of Infrastructure Systems*, 14(4):337–346, 2008.

[284] Saeed Hashemi, Yves Filion, and Vanessa Speight. Identification of factors that influence energy performance in water distribution system mains. *Water*, 10(4):428, 2018.

[285] Sarah Mann, Rebecca M Briant, and Maurizio Gibin. Spatial determinants of local government action on climate change: An analysis of local authorities in England. *Local Environment*, 19(8):837–867, 2014.

[286] Erin Goodling, Jamaal Green, and Nathan McClintock. Uneven development of the sustainable city: Shifting capital in Portland, Oregon. *Urban Geography*, 36(4):504–527, 2015.

[287] Aidan While, Andrew EG Jonas, and David Gibbs. The environment and the entrepreneurial city: Searching for the urban 'sustainability fix' in Manchester and Leeds. *International Journal of Urban and Regional Research*, 28(3):549–569, 2004.

[288] Samuel Walker. Urban agriculture and the sustainability fix in Vancouver and Detroit. *Urban Geography*, pages 1–20, 2015.

[289] Michele Acuto. Ain't about politics? The wicked power-geometry of Sydney's greening governance. *International Journal of Urban and Regional Research*, 36(2):381–399, 2012.

[290] Anne M Rademacher. Fluid city, solid state: Urban environmental territory in a state of emergency, Kathmandu. *City & Society*, 20(1):105–129, 2008.

[291] Shahzeen Z Attari. Perceptions of water use. *Proceedings of the National Academy of Sciences*, 111(14):5129–5134, 2014.

[292] Bradley S Jorgensen, John F Martin, Meryl W Pearce, and Eileen M Willis. Predicting household water consumption with individual-level variables. *Environment and Behavior*, 46(7):872–897, 2014.

[293] R Alan Cheville. Data sharing from a policy perspective. *Advances in Engineering Education*, 5(2):n2, 2016.

[294] D Butler and C Maksimovic. Urban water management-challenges for the third millennium. *Progress in Environmental Science*, 1:213–236, 1999.

[295] Richard Ashley, D Blackwood, David Butler, John Davies, Paul Jowitt, and Heidi Smith. Sustainable decision making for the UK water industry. In *Proceedings of the Institution of Civil Engineers-Engineering Sustainability*, volume 156, pages 41–49. Thomas Telford Ltd, 2003.

[296] Richard M Ashley, David J Balmforth, Adrian J Saul, and JD Blanskby. Flooding in the future–predicting climate change, risks and responses in urban areas. *Water Science and Technology*, 52(5):265–273, 2005.

[297] René Kemp, Saeed Parto, and Robert B Gibson. Governance for sustainable development: Moving from theory to practice. *International Journal of Sustainable Development*, 8(1-2):12–30, 2005.

[298] Alessandro Galli, Thomas Wiedmann, Ertug Ercin, Doris Knoblauch, Brad Ewing, and Stefan Giljum. Integrating ecological, carbon and water footprint into a “footprint family” of indicators: Definition and role in tracking human pressure on the planet. *Ecological indicators*, 16:100–112, 2012.

[299] Davy Vanham and Giovanni Bidoglio. A review on the indicator water footprint for the EU28. *Ecological Indicators*, 26:61–75, 2013.

[300] Asit K Biswas and Cecilia Tortajada. Future water governance: problems and perspectives. *International Journal of Water Resources Development*, 26(2):129–139, 2010.

[301] Chaim Zins. Conceptual approaches for defining data, information, and knowledge. *Journal of the Association for Information Science and Technology*, 58(4):479–493, 2007.

[302] Jennifer Rowley. The wisdom hierarchy: representations of the DIKW hierarchy. *Journal of Information Science*, 33(2):163–180, 2007.

[303] Massachusetts Water Resources Authority (MWRA), 2017. URL `www.mwra.com`.

[304] Carle Folke, Asa Jansson, Jonas Larsson, and Robert Costanza. Ecosystem appropriation by cities. *Ambio*, 25(3), 1997.

[305] Farhat Chowdhury, Christopher Lant, and Benedykt Dziegielewski. A century of water supply expansion for ten US cities. *Applied Geography*, 45:58–76, 2013.

[306] Janet G Hering, T David Waite, Richard G Luthy, Jorg E Drewes, and David L Sedlak. A changing framework for urban water systems. *Environmental Science & Technology*, (19):10721–10726, 2013.

[307] Arjen Ysbert Hoekstra. Human appropriation of natural capital: A comparison of ecological footprint and water footprint analysis. *Ecological economics*, 68(7):1963–1974, 2009.

[308] H. Hoff, P. Döll, M. Fader, D. Gerten, S. Hauser, and S. Siebert. Water footprints of cities: Indicators for sustainable consumption and production. *Hydrology and Earth System Sciences*, 18(1):213–226, 2014.

[309] Davy Vanham and G Bidoglio. The water footprint of milan. *Water Science and Technology*, 69(4):789–795, 2014.

[310] Alessandro Manzardo, Anna Mazzi, Andrea Loss, Michelle Butler, Anahita Williamson, and Antonio Scipioni. Lessons learned from the application of different water footprint approaches to compare different food packaging alternatives. *Journal of Cleaner Production*, 112:4657–4666, 2016.

[311] D Vanham, S Del Pozo, AG Pekcan, L Keinan-Boker, A Trichopoulou, and BM Gawlik. Water consumption related to different diets in Mediterranean cities. *Science of the Total Environment*, 573:96–105, 2016.

[312] D Vanham, TN Mak, and BM Gawlik. Urban food consumption and associated water resources: The example of Dutch cities. *Science of the Total Environment*, 565:232–239, 2016.

[313] Ikechukwu C Ahams, Willa Paterson, Susana Garcia, Richard Rushforth, Benjamin L Ruddell, and Alfonso Mejia. Water footprint of 65 mid-to large-sized US cities and their metropolitan areas. *JAWRA Journal of the American Water Resources Association*, 53(5):1147–1163, 2017.

[314] D Vanham, BM Gawlik, and G Bidoglio. Food consumption and related water resources in Nordic cities. *Ecological Indicators*, 74:119–129, 2017.

[315] Kuishuang Feng, Ashok Chapagain, Sangwon Suh, Stephan Pfister, and Klaus Hubacek. Comparison of bottom-up and top-down approaches to calculating the water footprints of nations. *Economic Systems Research*, 23(4):371–385, 2011.

[316] Arjen Y Hoekstra, Mesfin M Mekonnen, Ashok K Chapagain, Ruth E Mathews, and Brian D Richter. Global monthly water scarcity: Blue water footprints versus blue water availability. *PLoS One*, 7(2):e32688, 2012.

[317] Mesfin M Mekonnen and Arjen Y Hoekstra. Four billion people facing severe water scarcity. *Science Advances*, 2(2):e1500323, 2016.

[318] Mika Jalava, Matti Kummu, Miina Porkka, Stefan Siebert, and Olli Varis. Diet changea solution to reduce water use? *Environmental Research Letters*, 9(7):074016, 2014.

[319] City of Boston. Zero waste Boston, 2018. URL `https://www.boston.gov/departments/environment/zero-waste-boston`.

[320] Dana Boyer and Anu Ramaswami. What is the contribution of city-scale actions to the overall food system's environmental impacts?: Assessing water, greenhouse gas, and land impacts of future urban food scenarios. *Environmental Science & Technology*, 51 (20):12035–12045, 2017.

[321] M Giuliani, F Pianosi, and A Castelletti. Making the most of data: an information selection and assessment framework to improve water systems operations. *Water Resources Research*, 51(11):9073–9093, 2015.

[322] Massachusetts Department of Energy Resources. Massachusetts drinking water & wastewater facilities energy management pilot, 2007. URL `http://www.mass.gov/eea/agencies/massdep/climate-energy/energy/water-utilities/massachusetts-energy-management-pilot.html`.

[323] Dave Huitema, Erik Mostert, Wouter Egas, Sabine Moellenkamp, Claudia Pahl-Wostl, and Resul Yalcin. Adaptive water governance: Assessing the institutional prescriptions of adaptive (co-) management from a governance perspective and defining a research agenda. *Ecology and Society*, 14(1), 2009.

[324] Steven J Kenway and Paul A Lant. How does energy efficiency affect urban water systems? In *Understanding and managing urban water in transition*, pages 615–631. Springer, 2015.

[325] S Kenway, J McMahon, V Elmer, S Conrad, and J Rosenblum. Managing water-related energy in future cities–a research and policy roadmap. *Journal of Water and Climate Change*, 4(3):161–175, 2013.

[326] Maria J. Beltran. Response–what do virtual water and water footprint conceal? *Water Alternatives*, 9(1):162–164, 2016.

[327] Y Lu, N Nakicenovic, M Visbeck, and AS Stevance. Policy: Five priorities for the un sustainable development goals-comment nature, 520 (7548).

[328] Stuart E Bunn. Grand challenge for the future of freshwater ecosystems. *Frontiers in Environmental Science*, 4:21, 2016.

[329] Associated Press. Water crisis in Flint declared public health emergency, 2015. URL `http://www.cbsnews.com/news/water-crisis-in-flint-michigan-declared-public-health-emergency/`.

[330] University of Toledo. Lake Erie Center, 2014. URL `http://www.utoledo.edu/nsm/lec/`.

[331] Charles Fishman. Water is broken. Data can fix it, New York Times, 2016. URL `http://www.nytimes.com/2016/03/17/opinion/the-water-data-drought.html`.

APPENDIX A

OPEN RECORDS REQUESTS

To request data from cities, the following letter was sent to each utility:

"My name is Christopher Chini and I am a graduate student at the University of Illinois at Urbana-Champaign in the Department of Civil and Environmental Engineering. I am writing to formally request data related to the _____. Specifically, I am looking to obtain data from the drinking water and wastewater processes for the City of _____.

My research involves the study of the urban water cycle. As part of this, I am gathering water and wastewater data from several cities across the United States for comparisons. I hope to include data from the City of _____ as part of the study. I would like to get data at a daily time step for both water treatment and distribution as well as wastewater collection and treatment. This data would ideally include daily operational data for both of these processes, including:

- Treated Flow (gallons)
- Electricity Consumption (kWh)
- Natural Gas Use (therms)
- Bio Gas Generation (if applicable)

This data will be analyzed to determine the relationship between water and energy in the water and wastewater sectors. If daily data is not available, weekly or monthly data would be beneficial in my study as a substitute. I am specifically looking for the year 2012.

Additionally, I need an estimate of the total population that this data serves. Thank you very much for reading my request. I appreciate your time. If you have any questions, please do not hesitate to contact me via my information, below."

APPENDIX B

EVALUATING THE INDICATOR MODELS OF URBAN ENERGY AND WATER

This appendix provides supporting figures for Chapter 7. Figures presented in this appendix include a comparison of predicted versus observed values (Figure B.1), Q-Q plots for normality of residuals (Figure B.2), and a visual display of residuals for testing heteroskedasticity. Figures B.3, B.4, B.5, and B.6 show the residuals for each of the independent variables that were determined to be statistically significant ($p < 0.10$) from the models in Chapter 7. In the case of the wastewater effluent model, no residual plot was included as the only statistically significant variable determined was the presence of a combined sewer system (yes or no).

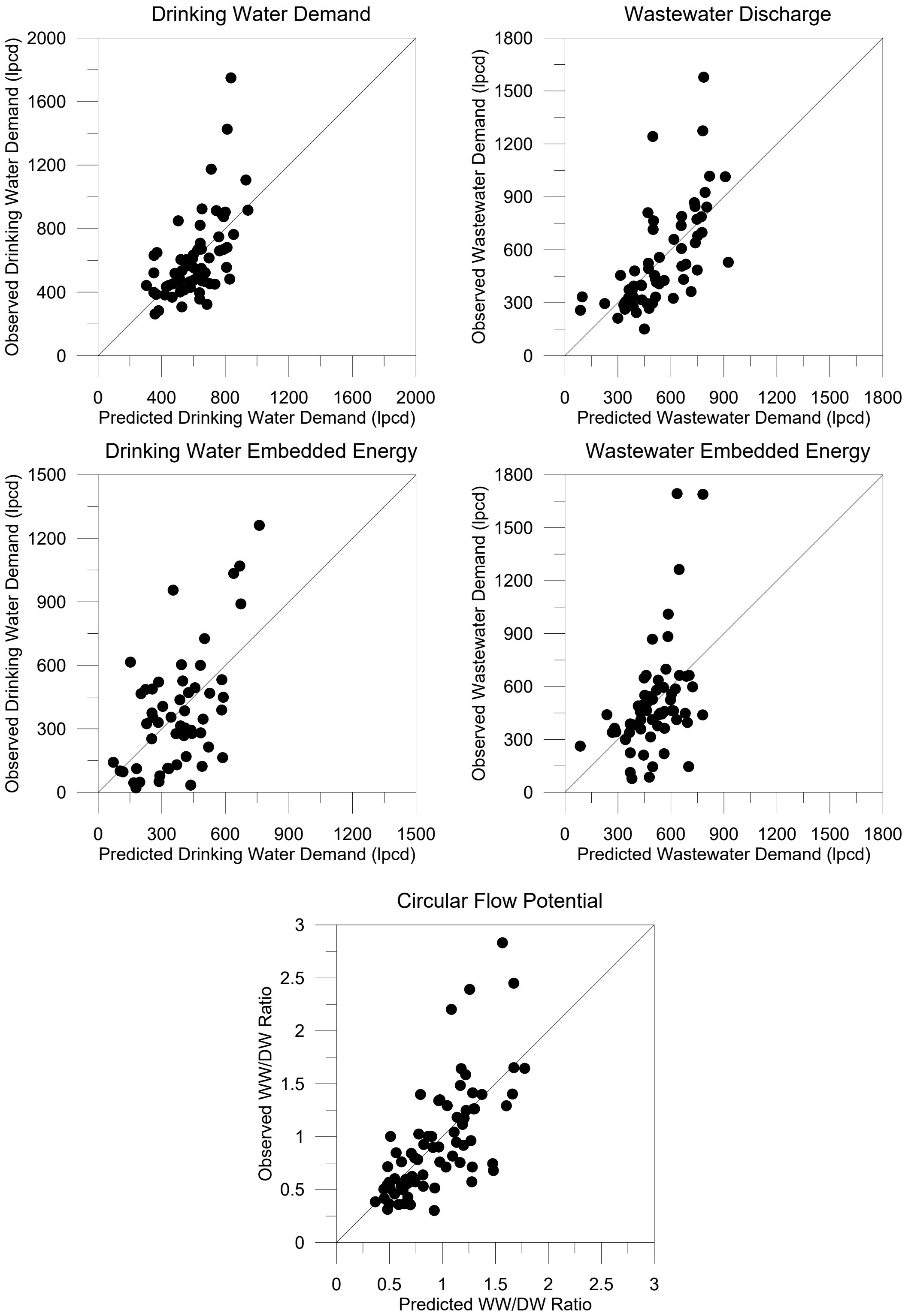

Figure B.1: Plotting the predicted values from the linear models against the observed dependent variables shows the limited explanatory power of the model.

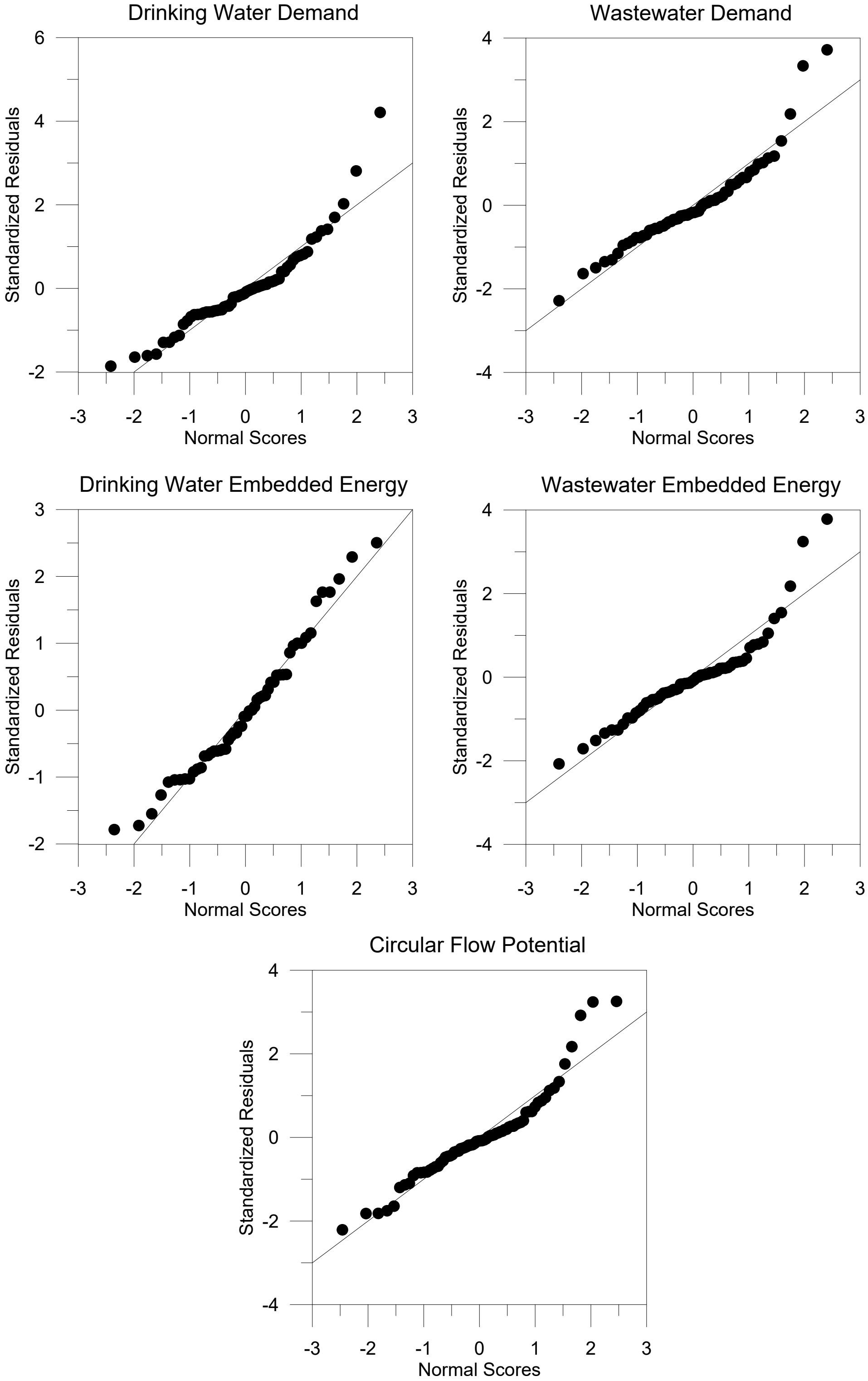

Figure B.2: The Q-Q plots of residuals show skew for the higher and lower extremes.

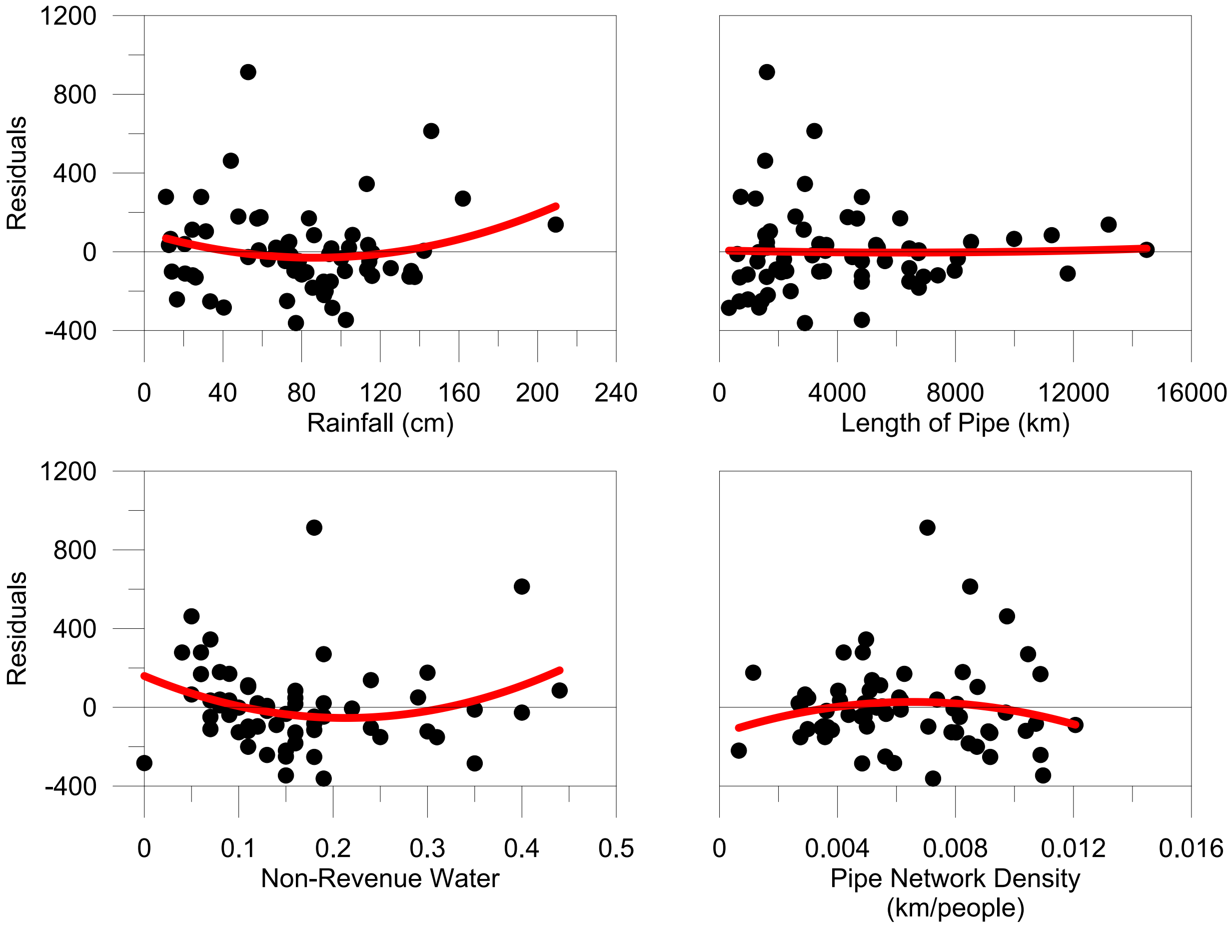

Figure B.3: There were four statistically significant variables associated with determining per capita drinking water demand with varying degrees of heteroskedasticity.

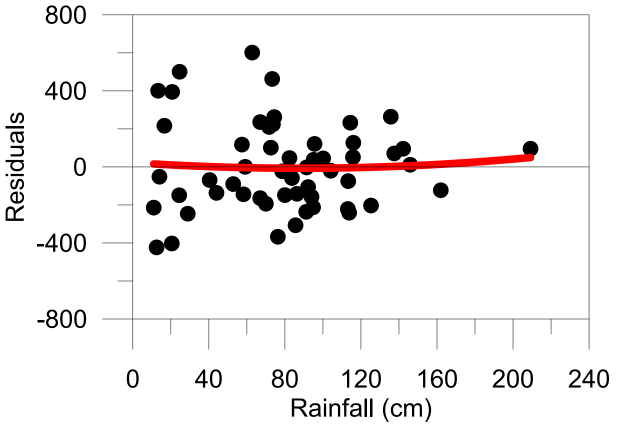

Figure B.4: Rainfall was the only statistically significant variable for drinking water embedded energy and had relatively constant spread of residuals.

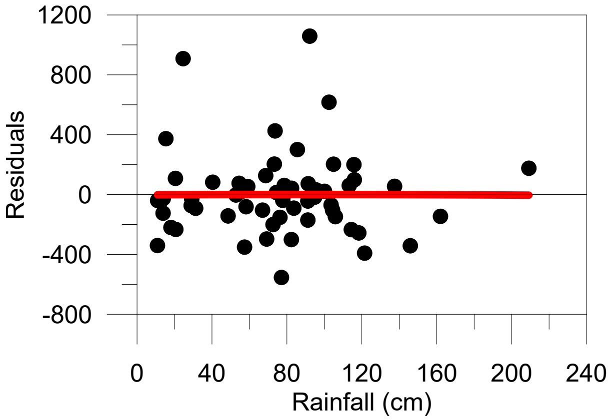

Figure B.5: Similarly to drinking water embedded energy, the embedded energy of wastewater resources was statistically dependent upon annual rainfall totals.

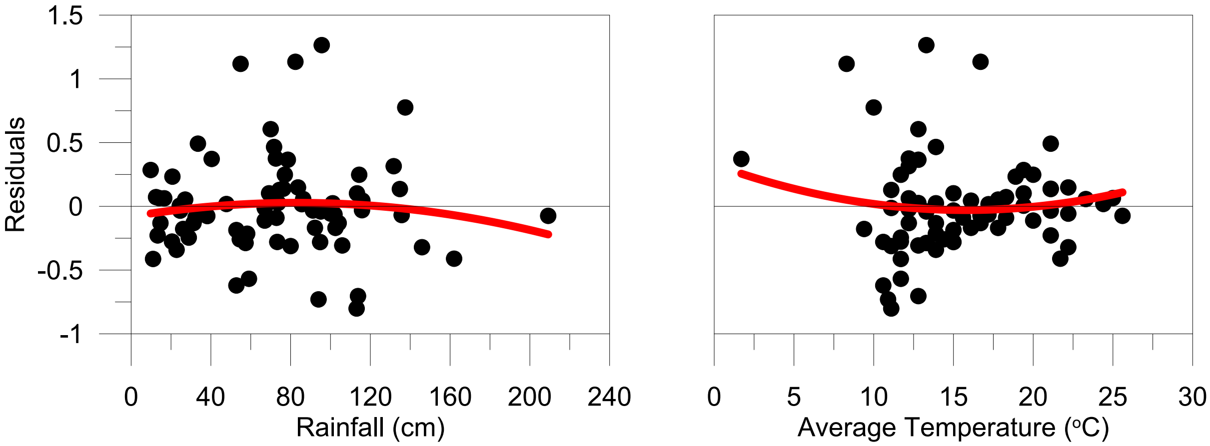

Figure B.6: Average temperature and annual rainfall totals had the most explanatory power in assessing the ratio of wastewater discharge to drinking water demand.