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ABSTRACT

Cities are hotspots of commodity consumption, with implications for non-local water resources. Water flows virtually into cities through this commodity exchange, meaning that local water issues have a global context. This form of water ‘teleconnection’ is being increasingly recognized as an important aspect of water decision making at the national scale. In cities and urban areas, virtual water flows are rarely acknowledged. The emphasis is on the hydrologic and engineered water balances. Through an extensive literature review of water footprint studies the potential and importance of evaluating water footprints and taking virtual flows into account in cities is evaluated. Specifically, the Water Footprint Assessment, life cycle assessment, and environmentally extended input-output methodologies are examined. As water is just one flux entering and leaving the urban boundary, the urban metabolism framework is investigated as a complimentary framework for analysis. Further, impacts of our trade decisions are discussed in terms of water scarcity and degradation and studies are reviewed for how they evaluate these impacts. Key themes and priorities for future research are also identified and discussed for urban water footprint analysis.
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ACKNOWLEDGEMENTS

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INDEX OF TERMS

CF – Carbon Footprint
CSA – Combined Statistical Area
EEIO – Environmentally Extended Input-Output
EF – Ecological Footprint
EFA – Environmental Footprint Assessment
ERA – Embedded Resource Accounting
IRIO – Inter-Regional Input-Output
LCA – Life Cycle Assessment
MFA – Material Flow Analysis
MRIO – Multi-Regional Input-Output
MSA – Metropolitan Statistical Area
POV – Point of View
SRIO – Single Region Input-Output
UM – Urban Metabolism
WF – Water Footprint
WFN – Water Footprint Network
WFA – Water Footprint Assessment
“We must hold fast to the realization that our cities are for people and unless they work well for people they are not working well at all. As the people of the world learn what is possible, they will demand that their cities be geared to the humane and beautiful”

- Jim Rouse, Social Visionary
Chapter 1
Introduction

Cities are the hub of global economic forces (Dobbs et al., 2012; Dobbs et al., 2011; van Vliet, 2002) with their links to distant and proximate locations through extensive exchange networks. Globally, cities are home to more than half of the world’s population and are expected to support nearly two-thirds by 2050 (United Nations, 2014). While this increased urbanization has come to signify greater socioeconomic opportunity and improved social welfare (Léautier, 2006; UNCHS, 2001), it is also creating additional stress on our water resources and the ecosystems they support (Averyt et al., 2013; McDonald et al., 2014; Moore et al., 2013; Savenije et al., 2014). This is further compounded by the environmental degradation that can result from aging and/or inadequate water infrastructure in cities (Charles, 2008; Hijdra et al., 2014; Rahm et al., 2013; Wu and Gao, 2010). Thus, it is becoming increasingly clear that cities hold the key to achieving sustainability targets because of their potential to address, and have an impact on global issues revolving around climate change (Georgescu et al., 2014; Marcotullio et al., 2013; Zhao et al., 2014), biodiversity loss (Aronson et al., 2014; McKinney, 2006; Seto et al., 2012a), and water resources (Grimm et al., 2008; Groffman et al., 2014; Kennedy et al., 2012; Kennedy and Hoornweg, 2012; Satterthwaite, 2011; Shuster and Garmestani, 2014).

To track the trajectory and assess the resilience of pathways toward urban sustainability goals, there is a growing need to define and quantify flexible indicators and common standards for cities, both to account for their
unique conditions and for comparable results (Aronson et al., 2014; Ferrão and Fernández, 2013; Folke et al., 1997; Kennedy and Hoornweg, 2012; Milman and Short, 2008; Satterthwaite, 2011; Tanguay et al., 2010; UN-Habitat, 2009). In addition, the use of city indicators to track and characterize flows and states could contribute to representing the connections (networks) among cities as well as with the global economy (Kennedy and Hoornweg, 2012). A variety of sustainability indicators have been proposed and implemented (see, e.g., (Singh et al., 2012) and references therein). The footprint family is a group of accessible and synthetic indicators that connect our consumer habits and production demands to the Earth’s resources (Ewing et al., 2012; Fang et al., 2014; Galli et al., 2012; Rushforth et al., 2013), and are creating ways to measure these impacts as a means to evaluate beyond normative economic benchmarks such as gross domestic product (GDP) (Ewing et al., 2012; Feng et al., 2011a; Kubiszewski et al., 2013).

The footprint family is comprised of the ecological footprint (EF), carbon footprint (CF), and water footprint (WF), among other environmental footprints. The WF quantifies the total volume of freshwater used to produce the goods and services consumed by an individual, region, or nation (Hoekstra and Chapagain, 2007). It includes both the amount of direct and virtual, also referred to as indirect or embedded (Allan, 1998; Allan, 1993; Hoekstra and Chapagain, 2007; Hoekstra and Hung, 2002), water consumption. The virtual water component quantifies the physical amount of water needed to produce goods in one region that are then exported to the region of consumption. The WF is expressed in volumes of water, typically cubic meters (m³), and is an analogue of the EF (Hoekstra, 2009; Rees, 1992; Wackernagel and Rees,
1998). The EF assesses the amount of biodiverse land needed, in equivalent units of global hectares, to provide for a region’s demands and ability to assimilate waste flows. The CF focuses on evaluating greenhouse gas emissions in terms of carbon equivalents, evaluated in kilograms or tons.

These environmental footprint assessments (EFAs) (Hoekstra and Wiedmann, 2014) have been estimated at the global, national, and regional scales as well as for products and businesses (see e.g. (Fang et al., 2014; Francke and Castro, 2013; Hoekstra and Mekonnen, 2012; Huijbregts et al., 2008; Moore et al., 2013; Sovacool and Brown, 2010; Wackernagel et al., 1999; Wiedmann and Minx, 2008)). The EF is unique in that it accounts for the biosphere’s capacity, whereas the WF and CF do not explicitly have capacity measurements built into their methodology (Ewing et al., 2012). However, the concept of the ‘planetary boundary’ can be thought of as the limit of the global footprint of humanity (Gerten et al., 2013; Hoekstra and Wiedmann, 2014; Rockstrom et al., 2009b), thus serving as an estimate of global capacity. For carbon, the reference to global capacity of 350 ppm has gained popularity as a threshold number to be targeted which has already been surpassed (Hansen et al., 2013; Hansen et al., 2008; Rockstrom et al., 2009b).

The capacity for the CF has been stated as 18-25 giga-tonnes of carbon equivalency which is in a range that would ensure that the global temperature does not rise more than 2 degrees (Hoekstra and Wiedmann, 2014; Pandey et al., 2011). The global capacity for freshwater use, estimated by Gerten et al. (2013), is 2,800 km$^3$/year. The planetary boundary for freshwater may be useful for informing international policy and governance but water scarcity is a very regional and local issue, due to the spatiotemporal variability of the
hydrological drivers and processes (Averyt et al., 2013; Mubako et al., 2013b; Poff et al., 2010; Steffen et al., 2015). Indicators of water sustainability ultimately will need to account for the spatial and temporal dependency of water scarcity as well as cross-scale interactions with the global system.

In urban areas, it is common to identify two different kinds of water balances in terms of direct flows: engineered and hydrologic. Hereafter the term urban area and city are used interchangeably to encompass different urban boundaries and spatial scales such as a central or satellite business district, suburb, or larger metropolitan area. The engineered water balance is controlled by the water demanded by and supplied to the city and the subsequent wastewater that is generated, while the hydrologic balance accounts for all of the natural inflows, outflows, and changes in storage in the urban basin. There are also important interactions between the engineered and hydrologic balances, e.g. combined sewer overflows, leakage from aging infrastructure, among others (Bhaskar and Welty, 2012; Padowski and Jawitz, 2012). Analogously, in the context of the WF of an urban area, a third water balance can be identified in relation to the virtual water flows entering and leaving the urban area through the products consumed and produced within.

The concentration of people and economic activity in urban areas leads to an imbalance between the virtual water inflows and outflows, with a bias towards certain commodities and economic sectors as compared with the average patterns of the complete national economy. For example, there is a concentrated demand for food and agricultural products in urban areas while limited land space is available for food production. Further, when evaluating the urban WF, the focus can be on the flows and networked exchanges coming
across the boundary or it can be on the interflows within the urban boundary. In either case, the characteristics of these flows, including flows associated with energy use (Kenway et al., 2011) and food consumption, and their implications for urban water supply decision making are poorly known.

The goals of this review are to identify and clarify the need for WF analysis at the urban scale, assess the strengths and drawbacks of current WF methodologies in the context of urban WF analysis, and identify and suggest areas for future research in urban WF analysis. It is recognized that the original intent of the WF methodology was to examine global trends of consumption and water use (Hoekstra, 2009). The urban WF seeks to incorporate this global dimension while recognizing that, in cities, water impacts and decision making are highly localized. The paper is structured as follows. Chapter 2 provides background information from previous WF studies done at different spatial scales. National scale studies are included in this chapter because these have been thus far the primary focus of WF analysis and they serve to highlight the need for studies at subnational and urban scales. Chapter 3 reviews the main methodologies used for WF analysis. Chapter 4 discusses urban metabolism and water scarcity in the context of WF analysis. Chapter 5 is a discussion of key themes for the WF and recommendations are made for future research in urban WF analysis. The need for a water footprint is addressed, and discussion of setting appropriate boundaries with constrained data is examined for this analysis. Urban sustainability and resilience are also discussed in terms of WF analysis. Finally, in chapter 6, main findings are summarized.
Chapter 2

Background: Water footprint studies at different spatial scales

The WF can be viewed solely through a lens of water use in production, which includes the physical direct flows within a region as well as indirect flows through the inputs used in production, or it can be viewed from a consumption perspective that incorporates the water being virtually consumed by a region. To account for different water sources and levels of water quality, the WF can be expressed in terms of green, blue, and grey water footprints (Table 1) (Falkenmark and Rockstrom, 2006; Hoekstra and Mekonnen, 2012). Overall, the emphasis of WF analysis has been on the blue and green water components and only a few studies have considered grey. In terms of the spatial scale, the majority of the studies have been performed at the national scale (Chen and Chen, 2013; Hoekstra and Mekonnen, 2012; Konar et al., 2012; Konar et al., 2011; Oki and Kanae, 2004), partly because of the need for national water security, the ability of nations to potentially affect major economic trade patterns, and the availability of bilateral trade data (Chapagain et al., 2006a; Hoekstra, 2011; Steen-Olsen et al., 2012). However, the number of regional and urban scale studies is steadily increasing. Thus, in this chapter WF studies at the national, subnational (includes single regions and interstate transfers within a country), and urban scale are reviewed. Throughout, intra-national exchanges are referred to as "transfers" – those that are explicitly within the national border – and international exchanges are referred to as "trade". "Trade" is also used when the delineation is mixed.
Table 1. Blue, green, grey water footprints.

<table>
<thead>
<tr>
<th>Blue water</th>
<th>Green water</th>
<th>Grey water</th>
</tr>
</thead>
<tbody>
<tr>
<td>Volume of water sourced from surface water or groundwater/ baseflow. Determined by modeling evapotranspiration for irrigated water or a consumptive water coefficient is applied for water withdrawals.</td>
<td>Volume of water that is determined by the moisture in the soil - water evapotranspired through plants and soils. Evaluated for agricultural processes.</td>
<td>Volume of water necessary to assimilate waste flows. Primarily evaluated for nitrogen and phosphorous content in return flows. Determined by dividing pollutant load by the difference in the maximum acceptable concentration and the natural concentration of the receiving water body.</td>
</tr>
</tbody>
</table>
2.1 National scale

Allan (1993) first introduced the concept of embedded water to address food scarcity concerns for arid regions. He asserted that countries with average low incomes face issues of food scarcity whereas wealthy countries have access to international trade markets and can import water rich products. The national scale studies have shown the ability of virtual water flows to transport water indirectly to water scarce regions and also that wealthier countries are typically net virtual water importers even without having issues of scarcity (Hoekstra and Mekonnen, 2012).

The first global virtual water estimates evaluating international crop trade were determined by Hoekstra and Hung (2002), and the Water Footprint Network (WFN) (http://www.waterfootprint.org) publishes independent reports, in addition to the UNESCO-IHE report series, which provide updated national and global WFs. The global WFs are obtained by aggregating the national estimates. Hoekstra and Mekonnen (2012) analyzed the blue, green, and grey WF of humanity and showed the variability of national WF by evaluating trends in both production and consumption footprints. They found that one fifth of the WF of global production was for exported goods, thereby highlighting the global role played by virtual water flows. Konar et al. (2011) used network statistics to evaluate international virtual water trade, highlighting that nations which export large volumes of water are also more likely to trade with other large water exporters. They also identified the major virtual water importers: United States (U.S.), Argentina, Brazil, and exporters: Japan, China, and Netherlands for crop and livestock products in 2000. Dalin et al. (2012) evaluated the temporal evolution of the network statistics of
global virtual water trade, showing that countries not only increased their trade partners but increased, to a greater extent, their virtual water trade and savings. For example, they found that the volume of water tied to trade more than doubled over a 22 year period from 1986 to 2007. Tamea et al. (2014) found that population, gross domestic product (GDP), and distance between trade nations are all correlated with the amount of imported virtual water. In addition, their results indicate that domestic water resources are not a critical concern in international trade, as of yet, and instead economic factors are the main driving force. Using a global multi-region input-output (MRIO) approach, Lenzen et al. (2013) determined the virtual water trade among 187 nations. A more complete description of the MRIO method is provided in the next chapter. To account for water scarcity, Lenzen et al. (2013) first weighted the blue water use according to a national water stress index. This study showed that some water rich countries import from water scarce countries. Many other studies have been conducted at the national scale (see e.g. (Chen and Chen, 2013; D’Odorico et al., 2012; Kumar and Singh, 2005; Roth and Warner, 2008; Wheida and Verhoeven, 2007; Yang and Cui, 2014; Yang and Zehnder, 2007)), only a few have been selected to underscore key findings.

Overall, national scale findings support that the movement of goods is creating a shift in our water resource use that is economically driven rather than resource driven. Cities play a central role in globalization since they are key drivers of economic activity through their production processes and their concentrated use of food, energy, and manufactured goods (Beaverstock et al., 2000; Capello, 2000; Taylor, 2001). To account for this, WF analysis at the urban scale is required. To support water resource management, the city and
intra-urban scales are ideal given the range of direct (e.g., water supply, water quality, water reuse, flood control, etc.) and indirect (e.g., consumption of goods and services) water decisions that are made by cities and their citizens.

2.2 Subnational scale

The studies that have estimated the WF and virtual water flows at the subnational scale are summarized in Table 2 and mapped in Figure 1. Among the subnational studies, Dang et al. (Dang et al., 2015) quantified the WF of food flows within the U.S., building upon earlier work by Lin et al. (Lin et al., 2014). Lin et al. (Lin et al., 2014) employed network analysis, together with freight shipment data for the U.S. (i.e., Commodity Flow Survey (CFS) data), to examine food flows within the US. The study by Dang et al. (Dang et al., 2015) provides insight into how the water footprint of trade varies across scales, by directly comparing the volume of the WF of domestic food transfers with the total volume of water embodied in international food trade. They found that the WF of food transfers within the U.S. is equivalent to 51% of international trade, which is slightly higher than the corresponding food value and mass shares, due to the fact that water-intensive meat commodities comprise a much larger fraction of food transfers within the U.S.

A number of case studies at the subnational scale have identified that water rich regions import from water scarce regions (Guan and Hubacek, 2007; Ma et al., 2006; Verma et al., 2009; Yu et al., 2010). This can be likened to the Leontief paradox, which opposes the notion of trade being determined by comparative advantage, and showed that the U.S., a capital-intensive market, actually imported capital goods and exported labor-intensive goods. For example, Yu et al.’s (2010) study of the United Kingdom (UK) found that
the relatively dry southeast utilizes more water in agriculture than the water rich northeast. Guan and Hubacek’s (2007) and Ma et al.’s (2006) studies showed a greater amount of virtual water transfers from the water scarce north China to the water rich south China. This was also found in Verma et al.’s (2009) study where arid eastern India was exporting goods to the wetter western India. The presence of this paradox can be explained further by considering other elements that can crucially influence interstate transfers and international trade such as climate, land use conditions, and socioeconomic differences among regions. For instance, the climate for year round growing is beneficial for water scarce northern China (Guan and Hubacek, 2007). Also, south China has seen a shift from a mostly agricultural society to a now industrial society which has created land use changes that have diminished the potential for food production there (Ma et al., 2006). While in eastern India farmers are heavily subsidized in comparison to the western region (Verma et al., 2009).

The subnational case studies, when contrasted with the national studies, support the need for performing WF analysis using different boundaries, as the behavior of virtual water flows and the interpretation of the WF changes with the boundary. For example, the transfer of virtual water from water scarce to water rich regions is most evident at the subnational scale. It remains to be clarified how cities contribute to this trend.
Table 2. Summary of studies that have evaluated the water footprint at the subnational scale and urban scale.

<table>
<thead>
<tr>
<th>City/region</th>
<th>Study by</th>
<th>Water footprint</th>
<th>Methodology**</th>
</tr>
</thead>
<tbody>
<tr>
<td>C Berlin, Delhi, and Lagos</td>
<td>(Hoff et al., 2013)</td>
<td>Green and blue†</td>
<td>WFA</td>
</tr>
<tr>
<td>R California</td>
<td>(Fulton et al., 2014)</td>
<td>Green, blue, and grey</td>
<td>WFA</td>
</tr>
<tr>
<td>B Heihe River basin, China</td>
<td>(Zeng et al., 2012)</td>
<td>Green and blue†</td>
<td>WFA</td>
</tr>
<tr>
<td>B Yellow River basin, China</td>
<td>(Zhuo et al., 2014)</td>
<td>Green and blue</td>
<td>WFA</td>
</tr>
<tr>
<td>B European River basins</td>
<td>(Vanham, 2013)</td>
<td>Net virtual water</td>
<td>WFA</td>
</tr>
<tr>
<td>S Interstate transfers in the U.S.</td>
<td>(Mubako and Lant, 2013)</td>
<td>Green and blue</td>
<td>WFA</td>
</tr>
<tr>
<td>S Interstate transfers in the U.S.</td>
<td>(Dang et al., 2015)</td>
<td>Green and blue</td>
<td>WFA</td>
</tr>
<tr>
<td>S Interstate transfers in India</td>
<td>(Verma et al., 2009)</td>
<td>Green and blue</td>
<td>WFA</td>
</tr>
<tr>
<td>S North and South China</td>
<td>(Ma et al., 2006)</td>
<td>Green and blue (surface and groundwater)</td>
<td>WFA</td>
</tr>
<tr>
<td>S Interprovincial transfers in China</td>
<td>(Dalin et al., 2014)</td>
<td>Green and blue</td>
<td>WFA</td>
</tr>
<tr>
<td>S Interprovincial transfers in Indonesia</td>
<td>(Bulsink et al., 2010)</td>
<td>Green and blue, and grey</td>
<td>WFA</td>
</tr>
<tr>
<td>S Western U.S. states</td>
<td>(Ruddell et al., 2014)</td>
<td>Blue</td>
<td>WFA/MRIO/ERA</td>
</tr>
<tr>
<td>C Beijing and China</td>
<td>(Hubacek et al., 2009)</td>
<td>Blue</td>
<td>SRIO</td>
</tr>
<tr>
<td>C Beijing</td>
<td>(Wang and Wang, 2009)</td>
<td>Blue</td>
<td>SRIO</td>
</tr>
<tr>
<td>C Beijing</td>
<td>(Zhang et al., 2011)</td>
<td>Blue</td>
<td>IRIO</td>
</tr>
<tr>
<td>C Beijing</td>
<td>(Wang et al., 2013)</td>
<td>Blue and grey</td>
<td>SRIO</td>
</tr>
<tr>
<td>B Haihe River basin, China</td>
<td>(White et al., 2015)</td>
<td>Blue†</td>
<td>MRIO</td>
</tr>
<tr>
<td>B Haihe River basin, China</td>
<td>(Zhi et al., 2014)</td>
<td>Blue</td>
<td>SRIO</td>
</tr>
<tr>
<td>B Haihe River basin, China</td>
<td>(Zhao et al., 2010)</td>
<td>Blue</td>
<td>SRIO</td>
</tr>
<tr>
<td>B Yellow River basin</td>
<td>(Feng et al., 2012)</td>
<td>Green and blue (rural and urban WF)</td>
<td>MRIO</td>
</tr>
<tr>
<td>S Liaoning Province, China</td>
<td>(Dong et al., 2013)</td>
<td>Blue</td>
<td>SRIO</td>
</tr>
<tr>
<td>S Shandan County China</td>
<td>(Deng et al., 2014)</td>
<td>Blue</td>
<td>SRIO</td>
</tr>
<tr>
<td>S North and south China</td>
<td>(Guan and Hubacek, 2007)</td>
<td>Blue (considers wastewater)†</td>
<td>IRIO</td>
</tr>
<tr>
<td>S Interprovincial trade in China</td>
<td>(Feng et al., 2014)</td>
<td>Blue†</td>
<td>MRIO</td>
</tr>
<tr>
<td>S Interprovincial trade in China</td>
<td>(Jiang et al., 2014)</td>
<td>Blue</td>
<td>MRIO</td>
</tr>
<tr>
<td>S The southeast and northeast UK</td>
<td>(Yu et al., 2010)</td>
<td>Green and blue</td>
<td>MRIO</td>
</tr>
<tr>
<td>S Domestic UK</td>
<td>(Feng et al., 2011b)</td>
<td>Green and blue</td>
<td>MRIO</td>
</tr>
<tr>
<td>C Sydney and Melbourne</td>
<td>(Lenzen and Peters, 2010)</td>
<td>Indirect impacts of blue water use</td>
<td>MRIO</td>
</tr>
<tr>
<td>S Victoria, Australia</td>
<td>(Lenzen, 2009)</td>
<td>Blue</td>
<td>MRIO</td>
</tr>
<tr>
<td>S Andalusia, Spain</td>
<td>(Velázquez, 2006)</td>
<td>Blue</td>
<td>SRIO</td>
</tr>
<tr>
<td>S California and Illinois</td>
<td>(Mubako et al., 2013a)</td>
<td>Green and blue (also saline water)</td>
<td>IRIO</td>
</tr>
<tr>
<td>C Beijing</td>
<td>(Huang et al., 2014)</td>
<td>Blue and grey†</td>
<td>LCA</td>
</tr>
</tbody>
</table>

† Study evaluates scarcity
*C – City, B – Basin, and S – Subnational
**Water Footprint Assessment (WFA), Life Cycle Assessment (LCA), Embedded Resource Accounting (ERA), Input-Output (IO), Single-Region (SRIO), Inter-Regional (IRIO), and Multi-Regional (MRIO)
Figure 1. Map of water footprint studies at the subnational and urban scale.
2.3 Urban scale

The urban WF studies have been performed for Beijing, China (Huang et al., 2014; Ma et al., 2006; Wang and Wang, 2009; Wang et al., 2013; Zhang et al., 2011); London, UK (Feng et al., 2011b); as well as Berlin, Delhi, and Lagos in Germany, India, and Nigeria, respectively (Hoff et al., 2013). Hoff et al. (2013) found that Berlin imported more than 60% of its virtual water from abroad whereas the virtual water for the developing cities of Delhi and Lagos primarily came from domestic sources. There were also high variations in the cities virtual water import per capita with the total virtual water import of Delhi (~434 m³/yr) being lower than that of Berlin (~643 m³/yr). This was attributed to diet choices. For example, the high water-intensive meat (evaluated in the form of livestock feed) and coffee consumption of Berlin results in higher virtual water flows there. Lagos, however, had almost double the virtual water import of Berlin with 1210 m³/yr per capita, which was attributed to their consumption of millet, sorghum, and cassava, as well as their low crop water production. Ultimately, Hoff et al. (2013) conclude that evaluating the WF and virtual water flows at the regional and city scale provides insight into water consumption that is lacking at the national level.

As shown in Table 2, the majority of the urban studies have employed approaches that rely on input-output (IO) tables and most are for a single city, Beijing, which is in a heavily water stressed region (Jiang et al., 2014). For instance, Wang et al. (2013) compared the WF of Beijing for the years 2002 and 2007 using sector level data. They found that over this period there was a decline in industrial and agricultural water use and that the city was a virtual water importer. In addition, using grey water estimates, they indicated that
water shortages are a bigger concern in Beijing than water pollution. Zhang et al. (2011) assessed the blue WF for Beijing for 2002 and found that 51% of Beijing’s WF was from virtual water imports. Overall, results from the Beijing studies in Table 2 indicate that virtual water flows are an important source of water stress alleviation for this city; however, products are often sourced from water strained regions in northern China (Feng et al., 2014). Thus, understanding the dynamics of virtual flows, together with the consideration of different boundaries, has important implications for urban and regional water management in the case of Beijing.

One of the main conclusions from urban scale studies is that there can be large differences between national and urban scale estimates of WF. For example, Feng et al. (2011b) found that the consumption WF per person for London, UK, was 58% higher for direct water use and 79% higher for virtual water use than the national average. Feng et al. (2012) also found this for urban household’s in the Yellow River basin identifying a 50% higher WF than rural households. Ultimately, to account for the heterogeneity and unique characteristics of each city (e.g., size, population, infrastructure, diet, industries, income distribution, quality of life, etc.), WF analysis at this scale needs to incorporate local (city and finer scale) data. This is one of the key challenges for urban WF analysis which is discussed further in chapter 5. The urban studies in Table 2 incorporate local assumptions and data in different ways. For instance, together with assumptions for the local redistribution of production surpluses and deficits, Hoff et al. (2013) downscaled national WF estimates based on gridded population. While Feng et al. (2011b) combined national (e.g., IO tables) and local (e.g., income, demographic, and
consumption estimates) data when determining the WF at the local authority level in the UK. The need to relate and employ datasets from different scales (e.g., national, subnational, and local) in urban WF studies can be an important source of WF variability and uncertainty that needs to be further understood and quantified. In a regional context, this was also stressed by Fulton et al. (2014) as part of their WF analysis for California and by Zhuo et al. (2014) in their case study for the Yellow River basin, China.

Urban WF studies have not clearly distinguished the different urban water flows. They have tended to emphasize direct water uses and virtual flows, leaving out flows such as stormwater. Figure 2 identifies the different direct and indirect urban water flows and further categorizes them as local or external if they originate from within or outside the urban boundary, respectively. In this figure, water is transferred into the urban boundary from external (imported water; e.g., water from a supply reservoir outside of the urban boundary) and internal (pumped surface/ground water; e.g., ground water withdrawn from within the urban boundary) sources. This transferred water either exits the boundary as wastewater, pipe leakage, or infiltration and inflow, or it remains within the boundary as recycled water. Virtual water is imported through the consumption of goods and services and exported by producing goods and services that are consumed outside of the urban boundary. In relation to stormwater, assuming that this is separate from the sewer system, the natural water balance components are all relevant such as precipitation, evapotranspiration, streamflow, and recharge. The way to include stormwater effects in urban WF analysis needs to be further investigated; it likely involves the net impact on internal and external surface
and ground water stocks resulting from urban land development and hydrologic alterations. In the case of imported water and pumped surface water, a consumptive coefficient is applicable but this does not apply to stormwater. Assessing a grey water volume for the amount of water needed to assimilate these flows may be appropriate. In any case, a complete urban WF analysis will need to consider all the flows identified in Figure 2.
Figure 2. Conceptual diagram of the direct and indirect water flows of an urban area. The recycling of direct and indirect local water within the urban boundary is also identified. Flows are defined as local and external if they originate from within or outside the urban boundary, respectively. Flows are also identified as hydrologic, engineered, and economic.
Chapter 3

Methodologies for water footprint analysis

Methodologies employed for WF analysis can generally be separated into bottom-up (product level approaches) and top-down (sector level approaches) (Daniels et al., 2011; Feng et al., 2011a). Among the top-down and bottom-up approaches, the following three main methodologies have been identified which have been used for regional or urban studies (Figure 3): (1) Water Footprint Assessment (WFA) which tends to be employed at the product/commodity level, (2) environmentally extended input-output (EEIO) which uses economic IO tables and thus considers sector level data, and (3) life cycle assessment (LCA) which relies heavily on standardization and databases to estimate the environmental, including water, and health impacts of products along their full life. Hybrid approaches of these methods have also been utilized to integrate scales and available datasets. Table 2 identifies which of the three methodologies was employed by each of the different subnational and urban studies reviewed. A more expanded version of this table can also be seen in Appendix A, Table A1.
Figure 3. Summary diagram of the common methodologies for water footprint analysis and urban metabolism.
3.1 Method 1: Water Footprint Assessment (WFA)

The WFA includes both direct and indirect water flows to determine the WF. This method was established by the Water Footprint Network (WFN), which provides the commonly used WaterStat database. This method can potentially be applied at any scale as well as for any economic sector and product to determine the consumptive use of freshwater resources. A complete description of this method is provided by the WFN’s Water Footprint Assessment Manual (2011). To apply this method to an urban area, one would ideally have data to quantify all the flows identified in Figure 2.

Due to the large share of global freshwater that goes into agricultural production, the interest in considering green water, and the ability to directly incorporate hydrologic modeling outputs, the WFA method has most prevalently been applied to estimate the WF associated with agricultural commodities, including livestock (see full equations in Appendix B). Global models are primarily used to estimate product water footprints. Key models that have been used include CROPWAT (FAO, 2013; Hoekstra and Hung, 2002; Ma et al., 2006; Mubako and Lant, 2013), the Global Crop Water Model (GCWM) (Hoff et al., 2013; Siebert and Döll, 2010), and the H08 (Dalin et al., 2014; Hanasaki et al., 2010), among others (Table 3). Once the virtual water content of agricultural commodities is found, a subsequent step is to determine the destination of this virtual water. Part of this virtual water may be consumed locally while the remainder is transferred to another region. This separation between local and transferred virtual water is strongly dependent on the boundary or scale definition.
Table 3. Commonly used crop water models for WFA.

<table>
<thead>
<tr>
<th>Model</th>
<th>Global Crop Water Model (GCWM)</th>
<th>H08 – Hydrological model</th>
<th>CROPWAT</th>
</tr>
</thead>
<tbody>
<tr>
<td>Resolution</td>
<td>5 arcmin (9 km x 9 km)</td>
<td>0.5 degree (55 km x 55 km)</td>
<td>Not mapped</td>
</tr>
<tr>
<td>Time step</td>
<td>Daily</td>
<td>Daily</td>
<td>Daily</td>
</tr>
<tr>
<td>Scope of the model</td>
<td>Simulates crop water use and</td>
<td>Evaluates land surface</td>
<td>Determines crop water and irrigation</td>
</tr>
<tr>
<td></td>
<td>crop yields. Yield decrease in</td>
<td>hydrology, river routing,</td>
<td>requirements, irrigation schedules, and</td>
</tr>
<tr>
<td></td>
<td>rainfed agriculture due to</td>
<td>crop growth,</td>
<td>calculates scheme water supply for</td>
</tr>
<tr>
<td></td>
<td>water scarcity can also be</td>
<td>reservoir operation,</td>
<td>different crop patterns.</td>
</tr>
<tr>
<td></td>
<td>simulated.</td>
<td>environmental flow</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>requirements,</td>
<td></td>
</tr>
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<td></td>
<td></td>
<td>and anthropogenic water</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>withdrawal. Subannual</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>timescale. Water and</td>
<td></td>
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<tr>
<td></td>
<td></td>
<td>energy balances on the</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>land surface are closed.</td>
<td></td>
</tr>
<tr>
<td>Inputs/Data</td>
<td>MIRCA 2000 dataset on crop</td>
<td>Meteorological forcing</td>
<td>Local climatic datasets (when</td>
</tr>
<tr>
<td></td>
<td>growing area, sowing and</td>
<td>data, runoff parameterization, land</td>
<td></td>
</tr>
<tr>
<td></td>
<td>harvest dates. Rainfed crop</td>
<td>surface type, soil</td>
<td>critical depletion level, yield</td>
</tr>
<tr>
<td></td>
<td>areas based on 2000 data.</td>
<td>moisture. Incorporates</td>
<td>response factor.</td>
</tr>
<tr>
<td></td>
<td>Irrigation statistics from many</td>
<td>SWIM (soil water integrated model) for 50 different</td>
<td></td>
</tr>
<tr>
<td></td>
<td>sources including FAO.</td>
<td>different crop types.</td>
<td></td>
</tr>
<tr>
<td>Reference $\text{ET}_\text{o}$</td>
<td>Priestley-Taylor method</td>
<td>Function of potential ET and soil</td>
<td>FAO Penman-Monteith approach (Allen et al., 1998)</td>
</tr>
<tr>
<td></td>
<td>(Priestley and Taylor, 1972)</td>
<td>moisture- refer to</td>
<td></td>
</tr>
<tr>
<td></td>
<td>or the FAO Penman-Monteith</td>
<td>Hanasaki et al. (2008a, b)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>approach (Allen et al., 1998)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Evaluates blue and green</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>water</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Additional methodology</td>
<td>Siebert and Döll (2010) and</td>
<td>Hanasaki et al. (2008a, b)</td>
<td>Allen et al. (2006) and Hoekstra and</td>
</tr>
<tr>
<td>Origin</td>
<td>Goethe Universität</td>
<td>National Institute for</td>
<td>United Nations Food and Agricultural</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Environmental Concern – Japan</td>
<td>Organization (UN FAO)</td>
</tr>
</tbody>
</table>
To determine where the virtual water is transferred to, two different approaches have been implemented at the subnational scale. The first approach models commodity transfers using the assumption of commodity mass balance. For example, Hoff et al. (2013), using a regional top-down disaggregation, allocates virtual water from cells (~10 x 10 km² or 5 arcmin) with a production surplus to cells with a deficit by iteratively redistributing the virtual water to neighboring cells. The cell values prior to redistribution were obtained by downscaling national estimates of consumption and production to the grid level based on gridded population and land area, respectively. This approach requires minimal local or intra-urban data and thus it is most useful for global studies. Since it does not account for the actual food miles traveled by commodities, it could lead to bias estimates for some regions and cities. A similar commodity balance approach was implemented by Ma et al. (2006) for provinces in China and by Mubako and Lant (2013) for states in the U.S. The second approach uses empirical data on commodity transfers. For example, Dang et al. (2015) use CFS freight shipment data to determine virtual water transfers within the U.S. Due to the anomalous consumption characteristics of cities it would be desirable to use empirical data on commodity transfers to quantify the water footprint of cities. However, this high resolution data is not available for all countries. As cities are extremely heterogeneous and specialize in a highly biased portion of the hydro-economy (i.e. services and manufacturing), top-down disaggregation approaches that begin at a regional or national scale, and include the agricultural and energy economies, are not likely to yield accurate information about the urban water footprint unless they are constrained by detailed urban water use and trade data.
The WFA method has been extended as a basis for Coupled Natural Human System (CNH) network analysis and modeling. This extension has provided opportunities to gain insight into the spatial scaling and process drivers behind transfers, to better understand the dependence among different regions and the importance of different transfer paths (network links), and to identify water savings. Ultimately, network and multi-network approaches provide a mathematical framework for the integrated characterization and modeling of the system (e.g., producers, consumers, transfers, etc.) underlying the WF (Ruddell et al., 2014; Rushforth et al., 2013). Additional information and examples of the application of network analysis to WF studies can be found elsewhere (Dalin et al., 2014; Dang et al., 2014; Konar et al., 2011; Ruddell et al., 2014).

3.2 Method 2: Environmentally Extended Input-Output (EEIO)

EEIO analysis evaluates the interdependencies between sectors by tracking monetary flows along the supply chain that are then connected with environmental consumption coefficients (Wiedmann, 2009; Zhou et al., 2010). In the context of WF analysis, this means that EEIO allows the determination of the amount of virtual water, typically in units of a water volume per dollar of commodity value, that is transferred between two process nodes in the trade network (see full equations in Appendix B). For this, one needs the amount of consumptive water used by each sector in the IO tables. The IO tables quantify the value of the economic transactions between the different sectors. The IO approach to economic data was introduced by Nobel prize winner Wassily Leontief. Since its inception, it has become a standard economic tool and is primarily used in assessing employment impact from investments across
sectors (Blackhurst et al., 2010). EEIO has been implemented at the single-region (SRIO) (Velázquez, 2006), inter-regional (IRIO) (Guan and Hubacek, 2007; Mubako et al., 2013a), and multi-regional (MRIO) (Lenzen, 2009; Wiedmann et al., 2011) level (Table 2). A detailed account of the IO framework and its extensions is given by Miller and Blair (2009).

In comparison with the WFA, EEIO takes advantage of available IO economic data to provide more detail about the interdependence of economic sectors. Although EEIO is mainly applied at the sector level, it can be applied at the product level when data is available (Miller and Blair, 2009). As is the case with WFA, the method does not necessarily imply a given scale of analysis, instead the lack of data has dictated the scope and boundaries for which EEIO is most often employed. A SRIO analysis was employed by Velázquez (2006) by evaluating the direct and indirect water flows for Andalusia, Spain, where the indirect flows are the virtual water components that are being used across the life cycle but the exports to and imports from other regions were not evaluated. IRIO analysis is between two regions, which was demonstrated in Mubako et al.’s (2013a) study of the linkages between Illinois and California and Guan and Hubacek’s (2007) study of north and south China. When accounting for multiple spatially distinct regions, MRIO analysis is used and transfer or trade data between locations is required. Hence, the trend has been to employ MRIO at the national scale where trade data is more readily available (Lenzen, 2009; Wiedmann et al., 2011). EEIO is more general than WFA and is a superset of the WFA technique in the sense that the IO tables allow the direct incorporation of additional information such as multiple environmental impacts (Miller and Blair, 2009).
The use of sector data can be both an advantage and limitation in WF analysis. As indicated by Daniels et al. (2011), the sector data can be useful for detecting sectors and life cycle stages that could be further analyzed and to geo-position environmental stresses. However, a limitation of using the top-down sectoral analysis, relative to product level implementations, is the current need to aggregate products or commodities into sectors, which can result in aggregation errors (Chen and Chen, 2013; Lenzen et al., 2013). It can also be noted that the temporal scale for IO tables does not always overlap with the available water use data (see Appendix A, Table A1). At the city scale, knowing the virtual water transfer between the urban economic sectors could be useful for assessing the water inter-dependency and efficiency of sectors. This in turn could provide valuable information for urban water management, e.g., to assess trade-offs in the economic performance of sectors under different water-related constraints. Such an intra-urban intra-sectoral virtual water trade analysis is especially important in the presence of core industries that dominate a city’s water use profile, economic productivity, and water supply planning. Assessment of the interdependence of such a sector on the residential and other commercial entities within the city yields a picture of water use spillover benefits and of the ‘water multiplier’ indicating the degree of virtual water recycling between sectors within the city.

### 3.3 Method 3: Life Cycle Assessment (LCA)

LCA has been a primary tool in evaluating industrial supply chains in an effort to understand the full life of a product and the environmental impacts which accrue along its path. LCA has only recently been used to evaluate freshwater use across products and sectors (Bayart et al., 2010; Berger and
Finkbeiner, 2010; Hester and Little, 2013; Kounina et al., 2013; Pfister et al., 2009). A large emphasis of LCA is in creating, implementing, and maintaining a standard for comparatively assessing the systemic human and environmental impact associated with the creation of products which are outlined by the International Organization for Standardization (ISO). As part of this standardization, they support the development of inventory databases that streamline the assessment process. Some of the commonly used databases are ecoinvent, GaBi, and Quantis. The latter two are tracking freshwater consumptive use whereas ecoinvent currently tracks withdrawals (Goldstein et al., 2013; Kounina et al., 2013). It should be noted that LCA databases contain information that can be specific to regions or industrial processes that may only be applicable in certain conditions and may have limited use as general water consumption factors. An advantage of LCA is its explicit consideration of human and environmental impacts. In regards to water resources, LCA research is working towards standardizing the quantification of impacts to water resources, such as distinguishing between withdrawals and consumptive use, and the effect of water degradation (Boulay et al., 2011; Pfister et al., 2009).

In general, the emphasis of LCA on implementing ISO standards and employing inventory databases makes it different from WFA and EEIO. Notwithstanding that all three methodologies when applied to water resources will be affected by similar data challenges and limitations. LCA identifies the need to develop indexed impact assessments for each product whereas the WFA takes a macro approach for assessing impacts and making overall sustainability recommendations. Additional discussion on the differences in
LCA and WFA as applied to impact assessments is further addressed in section 4.2.

LCA analysis consists of four separate phases: goals and scope, inventory analysis, impact assessment, and interpretation. The impact assessment can further be broken into midpoint and endpoint phases that evaluate and quantify the impacts of production. Milà i Canals et al. (2009) refers to midpoints as impact pathways which include assessing scarcity and considerations of water deprivation from other systems. Endpoints, referred to as areas of protections, include human health effects and changes in ecosystems (Milà i Canals et al., 2009).

LCA has been used to assess the WF of products and commodities in Beijing. Huang et al. (2014) analyzed the consumptive water for tomatoes, maize, and wheat in Beijing markets and included consumed water for the supply chain inputs of transportation, packing materials, and additional farm inputs. To assess impacts, they applied a water stress index (Pfister et al., 2009) for each crop dependent on the source region and determined water degradation footprints by assessing a grey water footprint and aquatic eutrophication footprint. However, this analysis is limited in scope to single products. To account for the city scale, each product consumed and produced in the city will need to be analyzed, thus requiring a level of detail that may be unfeasible in many cases. Hybrid approaches combining LCA and EEIO have been implemented to address multi-scale issues for carbon footprint assessments at the urban scale (Virtanen et al., 2011; Wiedmann et al., 2011) and this hybridization could be extended to WF assessments. Also, an important step in the more routine implementation of LCA for WF analysis
was the release of the ISO 14046 guidelines. These guidelines define standards for WF analysis. In addition the UNEP/SETAC collaborative is also promoting further work on the LCA WF and seeks to use life cycle analysis to support sustainable societies (Bayart et al., 2010; Ridoutt and Pfister, 2010b).

In the context of urban areas, where the ability to incorporate and maximize the use of local data is crucial, the strong dependence of LCA on standardized inventory databases may result in an undesirably standardized and under contextualized characterization of water sustainability. Specifically, a city may wish, for socio-political reasons, to apply nonstandard and contextual impact characterizations and accounting techniques that violate the assumptions inherent in the ISO implementations of water LCA. This lack of subjective contextualization could be an important limitation for urban water LCA studies that aim to inform a specific municipality’s planning and water supply decision making process.
Chapter 4

Complementary frameworks for water footprint analyses

In this chapter, current research on urban metabolism (UM) and water scarcity are reviewed in terms of their applications with WF analysis. High concentrations of goods pass through the urban boundary and the UM creates a framework to capture this. The WF concept at the urban scale fits within this framework as direct water use is already accounted for, thus an added consideration is to explore the virtual water that is embedded in the goods crossing this boundary. The external impacts of these flows are at the core of the UM framework but are not easily quantified (Zhang, 2013). The main assessments for WF have been to evaluate grey water footprints, water degradation, and scarcity. Understanding the impacts on distant water resources in terms of the social implications and ecological impacts is necessary for the WF to be an indicator for relevant policies. Therefore, scarcity needs to be evaluated at the appropriate temporal and spatial scale and linked to the consumptive nature of the city.

4.1 Urban Metabolism (UM)

The UM concept combines perceptions of the built environment with our social interactions by quantifying material and product fluxes passing through an urban boundary (Broto et al., 2012). UM, by analogy, looks at an urban area as a living organism that requires certain material flows, as energy and inputs, to maintain the structure and functions it performs and to sustain the life it supports. This method further interprets the efficiency of the urban
system in assimilating inputs and generating outputs in the form of waste flows (Pincetl et al., 2012). The use of the word ‘metabolism’ to interpret urban processes has been criticized (Golubiewski, 2012; Newman, 1999), as this implies that cities are single organisms whereas in reality multiple ecosystems are involved. Nonetheless, UM researchers do tend to recognize and consider linkages among different ecosystems when studying and analyzing cities (Kennedy, 2012). As a quantitative approach to cities, UM was brought into popularity by Wolman (1965), an educator and water resources and sanitary engineer. Wolman (1965) described and quantified the inputs and outputs (e.g., supply of water, disposal of sewage, and the amount of air pollution) associated with a hypothetical U.S. city of 1 million inhabitants. Since this initial work, UM has been expanded to account for flows of energy, various materials, and nutrients (Kennedy and Hoornweg, 2012) (See Figure 4 for general UM stocks and flows).

Within UM studies, a variety of complementary modeling approaches have been implemented (Ferrão and Fernández, 2013; Kennedy et al., 2007; Kennedy et al., 2011; Moore et al., 2013; Pincetl et al., 2012). Zhang (2013) completed a comprehensive review of UM studies which illustrates the many different views that have been taken on this. For example, some UM studies emphasize social factors (Gandy, 2004; Newman, 1999) while others quantify the physical inputs and outputs of a city using mass balance, material flow analysis (MFA), LCA, or energy-equivalents (known as emergy (Odum, 1983)) (Pincetl et al., 2012; Zhang, 2013). Many of the methods have been coupled. MFA has been thought of as a toolbox of techniques for UM (Ferrão and Fernández, 2013) and has utilized the ecological footprint to gain a
Figure 4. Inflows and outflows through the urban boundary including virtual water flows.

physical understanding of how much productive land would be needed to provide for a city’s demand (Browne et al., 2012; Dakhia and Berezowska-Azzag, 2010; Moore et al., 2013; Wackernagel et al., 2006; Yang et al., 2012). EEIO is also one input-output technique that MFA incorporates. Figure 3 highlights these main quantitative methodologies. Gonzalez et al. (2013) presented the application of the UM approach in developing a decision-support system for assessing potential impacts of urban development on metabolism components. The “metabolism modeling” term is sometimes used to emphasize the complexity of interactions in urban environments that need to be included in assessing the complete socio-technical system for sustainable urban water management (Makropoulos, 2014). Generally, UM studies highlight the dominance of linear patterns in our means of production, consumption, and disposal, whereas in nature the patterns tend to be circular (Huang et al., 2013). As a unifying goal, UM seeks to track these linear patterns to find a means to create a closed loop, sustainable system.

Limitations of and recommendations to improve the UM approach have been made by several authors. Pincetl et al. (2012) expresses the need for an expanded framework to track the various human-natural interactions that create the urban metabolic relationship. They urge for an interdisciplinary approach to address the most pressing issue of “how to sustain the quality of life for humans without permanently exhausting planetary resources or altering the planetary dynamics that support civilization”. An important and omnipresent hindrance to UM studies, including expanding their scope, is the limitations in data availability (Goldstein et al., 2013; Kennedy and Hoornweg, 2012; Moore et al., 2013; Zhang, 2013). Kennedy et al. (2007)
compared 8 different UM studies to identify trends across time and found limitations due to the different types of data and approaches used. Warren-Rhodes and Koenig (2001) also make a strong argument for having a framework to make UM calculations more routine for cross-comparison purposes. Thus far, the European Union has created the most robust material flow accounts under Eurostat as a means for tracking environmental accounts with economic IO data (Barles, 2009; Kennedy and Hoornweg, 2012).

For urban WF analysis, the data and results from existing and future UM studies could provide support for urban WF studies, as direct water use is commonly tracked in UM. UM in its historical usage is explicitly not a “virtual” or LCA analysis, and deals with physical flows. Further, Kennedy et al. (2007) found a tendency for the metabolism (i.e., the rate of change of an urban flow with population) of cities to increase with population. In the context of urban scaling, discussed in section 5.4, this implies nonlinear behavior and provides a means for inter-relating results from urban scaling and UM. This behavior could be further verified and expanded using the different components of the urban WF.

In terms of sheer mass, water is typically the largest flow through the urban boundary out of the various flows in UM studies (Huang et al., 2013; Kennedy et al., 2007). Water flow in the UM framework has primarily been tracked in terms of direct use and waste flows passing the urban boundary and has not been a primary concern for UM studies where energy analysis has been dominant (Huang et al., 2013; Kennedy et al., 2007). Huang et al. (2013) indicate that an urban water metabolism efficiency indicator system is needed and should be extended to account for both virtual and physical water flows.
They suggest the need to incorporate the so-called ‘social water cycle’ that is relevant within the urban boundary. This notion consistently emerges out of urban sustainability studies, i.e. the need to integrate human and biophysical systems (Liu et al., 2015; Pincetl et al., 2012; Ramaswami et al., 2012; Ruddell et al., 2014). A way of achieving and quantifying this integration is through WF analysis, which includes but distinguishes direct and indirect water footprints, by incorporating both UM and virtual flows. WF analysis provides a synthetic measure of both the water being used, which ultimately depends on its natural cycling, as well as the human activity driving this water use, directly and virtually (see Figure 2).

4.2 Water Scarcity

Urban regions face a variety of water stresses for their direct water use which will be discussed further in chapter 5. Incorporating social and environmental impacts for direct as well as virtual flows is a necessary step for the WF of an urban region and is a focus of WF assessments (Hoekstra et al., 2012; Lenzen et al., 2013; Mubako et al., 2013b). The LCA framework incorporates both water stress (as a midpoint indicator) and damages to human health (as an endpoint) in terms of water degradation. The LCA approach creates an indexed or weighted WF to account for those impacts whereas the WFA framework evaluates a volume based WF and considers the sustainability impacts as a separate assessment at the regional or process level. These differences are clearly defined by Boulay et al. (2013) and are further discussed elsewhere (see e.g., (Berger and Finkbeiner, 2013; Boulay et al., 2013; Chenoweth et al., 2014; Hoekstra, 2015; Hoekstra et al., 2009)). Both
methods evaluate stresses caused by resource consumption, concluding that water quality and quantity are important considerations when evaluating a WF.

For WFA, water quality is assessed in terms of grey water footprints and has been a primary indicator for nitrogen and phosphorous loads by using national averages of acceptable levels for blue water (Liu et al., 2012a). Although, it has been argued that grey water footprints, determined as a volume for assimilating pollutants, does not imply any real impacts to the system as it does not represent a measured volume (Chenoweth et al., 2014). LCA evaluates degraded water and the grey water footprint is just one indicator that can be assessed in addition to eutrophication, acidification, and eco-toxicity (Berger and Finkbeiner, 2013).

Falkenmark (2013) notes that in terms of water quantity, scarcity is typically discussed from two perspectives, that of ‘water crowding’ which is driven by population whereas the more common term ‘water stress’ is measured by evaluating use-to-availability and is dependent on demand related scarcity. For direct water use at the urban level, ‘water crowding’ is an important consideration and has driven many urban water projects to seek imported water from outside of their boundary (McDonald et al., 2014). A commonly used definition and index of water stress at the global scale is to determine the ratio of total annual freshwater withdrawals to hydrologic availability (WTA), where moderate stress is above 20% while severe stress is above 40% (Berger and Finkbeiner, 2010; Falkenmark, 2013; Pfister et al., 2009; Vorosmarty et al., 2000). This is typically given as yearly averages and does not adequately capture the seasonal dynamics of scarcity. The LCA and WFA methodology support the use of analyzing consumption to available
water (CTA) at the watershed scale (Hoekstra, 2015). The water stress index (WSI) proposed by Pfister et al. (2009) has demonstrated (Berger and Finkbeiner, 2013; Ridoutt and Pfister, 2010a) that when impact assessments are accounted for, the quantification and interpretation of the regional or product-specific WF can change quite significantly and discounts the actual volume of water being impacted. Many approaches have been used to assess quantitative impacts, some taking into account qualitative assessments where some account for ecosystem services. The LCA community is determining which indicator models are most appropriate for standard assessments (Boulay et al., 2015).

Although many of the regional studies in Table 2 discuss the need to address scarcity concerns, it has not consistently been evaluated. Hoff et al. (2013) critique the evaluation of virtual flows by observing that current methodologies do not include opportunity costs of food production in the source region or a standard means of interpreting scarcity in that region. For Berlin, Hoff et al. (2013) evaluated total (green plus blue) water stress by determining food water availability to food water requirements (Rockstrom et al., 2009a) for the source country. The WSI has been popularly used for determining water stress in China studies. As was discussed, the WSI was utilized by Huang et al. (2014) to determine water stress for regions supplying products to Beijing markets. It was also utilized for the comprehensive MRIO assessment of provincial transfers in China (Feng et al., 2014) and the Haihe River Basin (White et al., 2015) which identified stress levels for each province. Zeng et al. (2012) evaluated blue water stress, using a monthly watershed based assessment, outlined by Hoekstra et al. (2012), and found that
the production WF for the Heihe River Basin faced blue water stress for at least 8 months out of the year. The overall trend in studies on China is that virtual water does play an important role in offsetting strained water resources at the national scale. Virtual water analysis creates a framework to understand this demand-impact relationship and may play a role in mitigating this but decreasing demand, restructuring production, and improving water productivity are all key factors that need to be considered (Jiang et al., 2014; White et al., 2015). For urban WF analysis progress and refinement can also be made by integrating water resources impacts such as pollution, floods, and flow regime changes (Berger and Finkbeiner, 2013).
Chapter 5
Discussion

This chapter is a discussion of key themes that are helpful for guiding and motivating future research in urban WF analysis.

5.1 The need for a water footprint of cities

It is expected that in the near-future more urban areas are going to experience water stress. In urban areas, water stress may originate and be driven by a variety of factors including climatic variability, water pollution, population growth, increases in water demand due to economic growth, poor planning, inadequate or failing infrastructure, national or regional transboundary issues, and more (Childers et al., 2014; Grimm et al., 2008; Satterthwaite, 2011; Seto et al., 2012b). Contemporary urban areas are at the center of the so-called environmental paradox of economic growth (Raudsepp-Hearne et al., 2010; Reid et al., 2005) where increasing economic expansion, which normally translates to increasing or more intense urbanization, tends to lead to increasing environmental degradation. A fundamental aspect of this paradox is the sense that it can be reversed by a combination of actions including: technological advancements, creating a more natural urban water cycle, creating the right set of policies, offering economic incentives, and informing and engaging the public. Indeed, this is a pre-requisite or central motivating belief behind environmental policy and action at both the national and local level. However, this paradox and its proposed solutions often ignore a fundamental characteristic of urban areas, i.e. their dependence on resources from outside their boundaries. It is necessary to account for this dependence to
fully understand and quantify an urban area’s true influence on the environment.

Satterthwaite (2011) highlights the assumption, many times implicit in this environmental paradox, that cities and high consumption lifestyles need to go hand-in-hand. He suggests that this does not necessarily have to be the case. He further indicates that many of the contributions of urban areas such as culture (e.g., artistic activities) and diversity, particularly in large cities, provide a higher quality of life but do not imply a high consumption lifestyle. Nonetheless, it is noted that understanding the transition between various consumptive patterns requires tracking how such patterns interact with the environment. Virtual flows are one way of quantifying this interaction. Other quantitative approaches are being proposed and implemented in the context of urban and landscape ecology (Grimm et al., 2008; Wu, 2010). It is further recognized that even though the general tendency is for demand-driven supply chains (O'Rourke, 2014), producers (e.g., companies) can also have a big influence on the WF of an urban area. In some urban areas, it may actually be the producer that has the most control over the WF and virtual flows. This will need to be considered when performing WF analysis in an urban area.

5.2 A general approach for urban water footprint analysis

In chapter 3, advantages and drawbacks were addressed for the three main methods that have been implemented for WF analysis at the regional and urban scale. On the basis of that review, it is recognized that any of these methods can be implemented for urban WF analysis. The selection of a given method will depend on the scope of the study, the available datasets, and other preferences. It was discussed that hybrid approaches have been used (see e.g.
(Ewing et al., 2012; Fang et al., 2014; Wiedmann et al., 2011)) and as new datasets are collected and developed for urban areas, the ability to combine these methods will increase. To do this effectively and consistently, it is desirable to have an overarching framework that clearly articulates the connections among the methods.

Recently, a generalization of EEIO and EFA methods (WF, EF, CF), termed Embedded Resource Accounting (ERA), was proposed and developed (Ruddell et al., 2014; Rushforth and Ruddell, 2015; Rushforth et al., 2013). ERA makes explicit the assumptions implied by the different footprint standards and methods, and in doing so it not only adds transparency to footprint analysis but it also facilitates the integration of different datasets (e.g., IO tables, commodity flows, etc.). For example, ERA uses the notion of equivalency to clarify that typically, in WF analysis, the same volume of water in two different countries is treated as being the same or having similar value, whereas in reality the local context will modify the value of water. The rich conceptual scheme of ERA (Ruddell et al., 2014; Rushforth et al., 2013), which borrows ideas, tools, and terms from systems, network, economic and sustainability science, among others, allows the consideration of point of view (POV). The POV may represent the person or business doing the accounting, a given standard, a management priority, etc., each with its own boundary definition. POV captures the fact that different water users will subjectively alter the boundaries of the system, selectively including or excluding some water impacts, for the purpose of their WF analysis and decision making. Ultimately, this affects the accounting of direct and indirect water and has implications for the attribution of water impacts. The attribution of impacts is
strongly dependent on the legal and political context and the social objectives of the decision makers. ERA was recently applied to the case of water embedded in the electrical energy trade in the western U.S. (Ruddell et al., 2014), demonstrating how water use in one location may be discounted compared with water use in another location. Overall findings from this study highlighted that water stressed states in the Colorado River basin are net virtual water exporters. POV is represented by considering the water manager and water consumer. With current WF assessments these parties may discount the externalities associated with their trade. By incorporating POV into the decision framework these effects would be considered.

Accounting for POV may provide useful for urban WF analysis. Urban areas have many different water stakeholders (e.g., urban residents, utilities, businesses, city officials, environmental groups, etc.), each with its own POV. ERA has many of the key characteristics required for a complete method for urban WF analysis.

5.3 Spatial scale or boundary for urban water footprint analysis

Identifying the boundaries of an urban area can be challenging (Oliveira et al., 2014; Rozenfeld et al., 2008). This is mainly because urban areas tend to be strongly connected to their hinterlands and other distant areas for their subsistence and functioning. Also, the spatial extent of urban areas changes in time, cities can grow or shrink, independently of their defined administrative boundaries (Batty, 2014). The changing form of urban areas and their connectivity with other distant areas makes any definition of an urban boundary relative. Hence, the dynamics and fluxes of an urban area will crucially depend on the definition of its boundary. It thus seems reasonable
and perhaps necessary to account for different spatial scales when dealing with fluxes in urban areas for example the potable water supply boundary, the watershed or aquifer boundary, and the state or regional government boundary.

Because water footprints and virtual water flows have the potential to employ a network conceptualization to describe the hydro-economy, a way to consider different spatial scales is through allometric or scaling relationships. These scaling models must not ignore the presence of decision boundaries that determine the form of the network, breaking processes at specific and sometimes politically arbitrary scales. In the context of urban areas, scaling have been used to study and characterize different urban metrics such as innovation, wealth, crime, personal income, road surface (Bettencourt et al., 2007; Bettencourt et al., 2010; Fragkias et al., 2013; Fuller and Gaston, 2009; Oliveira et al., 2014); green areas (Fuller and Gaston, 2009; Oliveira et al., 2014); CO₂ emissions (Fragkias et al., 2013); among others. Hereafter this approach is referred to as urban scaling. Urban scaling could be employed to study the dependence of the urban water footprint and its different components on spatial scale. This could include examining the amount of blue and green water, surface and groundwater withdrawals, or, with reference to Figure 2, the amount of direct/indirect local and external urban WF components as a function of urban scale.

In practice, for research purposes, the definition of the urban scale tends to be arbitrarily set by the dataset employed and the agency in charge of data collection. For instance, in the U.S., the Census Bureau uses several definitions of urban areas: metropolitan statistical area (MSA), microstatistical
area, and combined statistical area (CSA). The MSA has a rural to urban gradient as it consists of one or more counties around an urban core that has a population greater than 50,000 while the microstatistical area is a densely populated region with more than 10,000 residents but less than 50,000 (U.S.Census). The CSA is a grouping of adjacent MSAs. Furthermore, river basins, population, and population density have all been used to define boundaries for urban areas (Marcotullio et al., 2013; Padowski and Jawitz, 2012; Vanham, 2013). There are multiple definitions of an urban area and this should be taken into consideration in urban WF analysis, because per the logic of ERA these different boundaries imply different points of view and different accounting of footprints and values. For example, in the context of urban scaling, Oliveira et al. (2014) found for cities in the U.S. that the scaling of CO₂ emissions with population may suffer from endogeneity bias when using the MSA as the boundary.

Urban scaling is one way of dealing with multiple urban scales. However, since it deals with average characteristics or properties of cities, it does not allow direct exploration of intra-urban heterogeneity and dynamics. For this, detailed (i.e. at the parcel and individual establishment and residential level) urban WF studies are needed. Detailed urban WF studies could help explain the nature of any urban water scaling relationship, account for the spatial variability of WF components within each city and their unique interaction with the different direct and indirect water supplies, and help assess the implications of local urban impacts on the WF. Bringing the urban WF to the parcel and individual establishment and residential level will create the most opportunity for urban citizens, stakeholder groups, and water managers
to operationalize the urban WF metric, and it will provide the most reliable information for informing urban planning and policy decisions. Also, the dependence of cities on their hinterlands can be more precisely captured in a detailed study than by performing urban scaling, as the scaling could end up averaging dissimilar properties. Additionally, by considering the internal dynamics of cities, unique behavior may be revealed that is not possible by the averaging involved in urban scaling, such as the ability to have spatially overlapping WFs in the city due to trade with common suppliers.

Ultimately, economic decisions do not follow boundaries. With the direct and indirect water being moved by economic forces, the definition of an urban boundary for WF analysis will entirely depend on the scope of the study and the datasets used. To guide future research, two approaches have been proposed for urban WF analysis that assume different urban boundaries. Urban scaling considers the whole city scale while the detailed approach needs to be implemented at the local parcel and establishment level. Both of these approaches could be extended with network analysis and modeling tools to consider the links among cities and regions and, in the detailed studies, to consider intra-urban links. This has already been suggested by Ruddell et al. (2014) in the context of ERA.

5.4 Required datasets for urban water footprint analysis

Limitations of accessible data are pervasive in urban studies (Fulton et al., 2014; Kennedy et al., 2007; Padowski and Jawitz, 2012; Pincetl et al., 2014). This section reviews available and required datasets for urban WF analysis from parcel to city scale with a focus on the U.S.; however, many of these constraints are found for all cities. Within an urban area, the primary
focus of WF studies has been on commodity transfers and transfers among economic sectors (see Table 2). Recent work has identified a second distinct WF, associated with labor (commuting flows) (Rushforth and Ruddell, 2015). The WF associated with interstate transfers and international trade emphasizes industrial and commercial linkages and dependencies between geographic areas that may be disrupted by or vulnerable to water shortages and scarcity, while the WF associated with the movement of labor emphasizes the interdependency between economic production, environmental quality, and shared critical infrastructure. The minimal hypothetical data required for the urban transfer WF may be separated into 3 broad groups: direct urban consumptive water use, virtual water coming into the city, and virtual water leaving the city, which closely aligns with the WFA methods and definitions. Commuting flows, may not be the dominant water users at the national scale, but provide insight into the movement of water near to and within the urban boundary.

Water withdrawal data necessary to systematically calculate the WF of multiple urban areas exist, but the data varies by quality and validity (Gleick, 2003). Table 4 summarizes available datasets for urban WF analysis that were selected for their potential applicability for a comparative analysis of U.S. cities. Harmonizing datasets poses the largest challenge to creating a system to calculate the WF of trade for each city in a country. As mentioned, the needs of the agency collecting data dictate the geographic boundaries within the dataset and the frequency at which data is collected (see the spatial and temporal scale columns in Table 4). Therefore, geographic and temporal harmonization is an ongoing challenge for urban WFs of trade.
The USDA Census of Agriculture (Table 4) provides relevant data about agricultural products and land under crop production, but the scales at which data are available pose disaggregation challenges. This farm census data is available at the zip code level and is associated with a city name, but not necessarily located within the city, which creates a positive bias in WFs; associating too many water consuming activities to the city. The USGS MRDS (Table 4) contains comprehensive data of the location of, and production from, all known mining, mineral benefaction, and sand and gravel operations in the U.S. and world. However, verification of these data is necessary to ensure that the locations listed in the MRDS are still operating prior to calculating the water footprint of mineral demand in an urban area.

For many commodity flow and trade databases (e.g., CFS, FAF3, and FAOSTAT in Table 4), the finest-scale geography for trade data is the metropolitan area or county and reliable methods need to be developed to disaggregate trade data to the city-scale (Bujanda et al., 2012; Federal Highway Administration; Opie et al., 2009; Southworth et al., 2010; U.S. Census, 2006; Viswanathan et al., 2008). Employment and establishment counts collected at the city level and located in national census (U.S. Census and CES in Table 4) provide one method of disaggregation, but readily available public data may be unreliable due to data omissions to protect the identity of businesses that may be singled out in census data. Trade databases contain agricultural and mineral commodity flows, but commodity flows may not sync with where the actual production of commodities occur, requiring validation of commodity flow data.
<table>
<thead>
<tr>
<th>Source</th>
<th>Type of data</th>
<th>Spatial scale</th>
<th>Temporal scale (most recent)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Expenditure Survey (CES)</td>
<td>20A collects expense estimates for food and beverages per household.</td>
<td>and MSA</td>
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<tr>
<td>Bureau of Financial Analysis (BEA)</td>
<td>National Input-Output datasets. Also provides regional Input-Output modeling</td>
<td>National</td>
<td>Every 5 years (2007 – years with 2 and 7)</td>
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<td></td>
<td>system (RIMS) multipliers, GDP analysis at state, metropolitan area, industry,</td>
<td>state, MSA, and zip code</td>
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<tr>
<td></td>
<td>etc., and details on imports and exports at national level.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>IMPLAN</td>
<td>Economic databases and methodologies to construct input-output tables. Creates</td>
<td>National, state, county,</td>
<td>Yearly</td>
</tr>
<tr>
<td></td>
<td>IMPLAN multipliers that are a ratio of total impacts divided by direct impacts.</td>
<td>MSA, and town</td>
<td></td>
</tr>
<tr>
<td>US Census</td>
<td>Population and socioeconomic statistics, including income and GDP.</td>
<td>National, state, county, MSA, city, and town</td>
<td>Every 10 years (2010)</td>
</tr>
<tr>
<td>USDA Census of Agriculture</td>
<td>Farm and Ranch Irrigation Survey for blue water calculations. Harvested</td>
<td>State, county, and zip code</td>
<td>Every 5 years (2012)</td>
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<tr>
<td></td>
<td>cropland by size of farm and acres harvested. Inventory and sales of livestock.</td>
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<tr>
<td>Bureau of Transportation Statistics -</td>
<td>Primary source on domestic freight shipments by American establishments in 42</td>
<td>National, state, and MSA</td>
<td>Every 5 years (bilateral only in 2007, 2012)</td>
</tr>
<tr>
<td>Commodity Flow Survey (CFS)</td>
<td>sectors. Provides a modal picture of national freight flows, and represents</td>
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<td></td>
<td>the only publicly available source of commodity flow data for the highway</td>
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<tr>
<td></td>
<td>mode. Used to track commodity to source region. Data are provided on the</td>
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<tr>
<td></td>
<td>types, origins and destinations, values, weights, modes of transport, distance</td>
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<tr>
<td></td>
<td>shipped, and ton-miles of commodities shipped.</td>
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<tr>
<td>Federal Highway Administration and Bureau of</td>
<td>This dataset integrates data from the most recent CFS and a variety of</td>
<td>National, state, and MSA</td>
<td>Every 5 years. Last finalized data were</td>
</tr>
<tr>
<td>Transportation Statistics- FAF3</td>
<td>sources to create a more comprehensive picture of freight movement among</td>
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<td>2007, interim data for 2012 and projected to</td>
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<td></td>
<td>states and major metropolitan areas. Provides estimates for tonnage, value,</td>
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<td>2040</td>
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<td></td>
<td>and domestic ton-miles by region of origin and destination, commodity type,</td>
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<td></td>
<td>and mode for 2007, the most recent year, and forecasts through 2040.</td>
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<tr>
<td>USGS</td>
<td>Water withdrawal data for 8 sectors: public supply, domestic, irrigation,</td>
<td>National, state, and county</td>
<td>Every 5 years (2010)</td>
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<tr>
<td></td>
<td>livestock, aquaculture, industrial, thermo-electric-power generation, and</td>
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<td></td>
<td>mining.</td>
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</tr>
<tr>
<td>USGS MRDS</td>
<td>Mineral resources data for production of all minerals, including sand and</td>
<td>GPS coordinates of each</td>
<td>(2011) Developing a new data product for the</td>
</tr>
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<td></td>
<td>gravel, from the U.S. and locations of mining operations worldwide.</td>
<td>mining site in the world</td>
<td>coterminous U.S.</td>
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<td></td>
<td>Food balance sheets can be created to provide the pattern of a country’s food</td>
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<td>supply during a given reference period. These show for each food item, such</td>
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<td>as primary consumption, which corresponds to the sources of the supply and its</td>
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<td>utilization.</td>
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</table>
The data challenges discussed provide relevant information on industrial and some commercial water uses (e.g., economic activities that create industrial production and water used for power plant cooling) (Gleick, 2003). However, most commercial activities are supplied by water utilities; this requires a data disaggregation step to separate commercial activities from domestic water consumption. Utility-level data of industrial and commercial (IC) consumption and residential (domestic) consumption (Mayer et al., 1999) provide directly observed water consumption data, but the frequency and validity of water consumption reporting by water utility varies by utility (Gleick, 2003). Utility-level water consumption data are relevant to both the WF of trade and labor. Industrial and commercial data from water utilities is often publically-available, but only for broad water use categories and has not been tied to commodity codes or industrial classification systems such as the North American Industry Classification System (NAICS), Standard Classification of Transported Goods (SCTG), Standard Industrial Classification (SIC) System, and Harmonized System (HS).

Harmonizing industrial and commercial water utility account-level data with industrial classification systems is possible, but burdensome for large water utilities. Further, industrial and commercial accounts may have multiple meters that provide water for business activities that produce different commodity classes as well as support service-sector activities, making it difficult to associate trade, value, and consumption to specific commodity types, and vice versa. Recent progress in addressing this issue can be found in (Morales and Heaney, 2014). The major data hurdles facing the systematic calculation of urban WFs of labor are the existing commuting data and
domestic water use data. In the U.S., the USGS water withdrawal data (USGS in Table 4) provides county-level specificity of domestic water uses, but this may not provide the relevant detail for most cities. Commuting data collected by federal-level governments provide an estimation of daytime population changes. Further work is required to create a system that matches account-level attributes (industrial, commercial, and residential) to commuting flows and economic production and the aforementioned industrial classification systems. This process will allow for the attribution of an urban area’s WF down to the parcel level, similar to advances with carbon emissions and carbon footprints (Gurney et al., 2007; Gurney et al., 2009; Gurney et al., 2012).

The systematic calculation of urban WFs of trade and labor are constrained by the extant datasets. Due to this, there will be a lag between current water management needs and the urban WF calculations. Given the timeliness of trade and water data publication, urban WFs can be calculated systematically every ten years and possibly every five years (see e.g. the datasets in Table 4). Limitations in linking water consumption data to industrial classification will limit the detail of WFs to the economic sector (NAICS, SCTG) and economic megasectors (Fisher, 1939; Kenessey, 1987). However, this level of detail will still allow for the identification and management of long-tail sources and exporters of virtual water that are the critical linkages in the virtual water trade network.

A recommendation for data collection could include participatory research with key stakeholders within the city. Challenges have been identified in translating urban sustainability research findings into specific policy and
city actions and Zborel et al. (2012) highlight the need to develop synthetic and more communicable research results. The WF metric could provide a synthetic and interpretable result for decision makers at different city levels. This participatory research model suggested by Zborel et al. (2012), proposes that researchers and city staff collaborate and work together in clarifying key research questions and problems, preparing datasets, and implementing solutions informed by research. This collaborative approach can also better inform researchers about local datasets that may exist that might not be in the public sphere or easily accessible. Participatory research could also provide the means to create new datasets with a focus on environmental indicators as much of the datasets have been tailored to economic perspectives. The ability to develop new datasets linking trade, productivity, and social value with water use at the city level is indispensable for the progress of this type of urban water sustainability planning policy and decision making.

5.5 Applications of urban water footprint analysis: addressing sustainability and resilience

Sustainability can be defined in many ways. A popular definition is that of providing for the needs of today without compromising the needs of future generations (Bruntland, 1987). Typically, the various definitions have to do with the endurance of an interacting human-biophysical system and its processes. The goal with urban sustainability is to actively make urban areas better places to live while addressing multiple values, such as human well-being, ecosystem services, social and environmental justice, etc. Indeed, making cities more sustainable, by reducing or changing material flows and
waste generation, is one of the key challenges of sustainability science (Childers et al., 2014; Pickett et al., 2013; Pincetl, 2010; Seto et al., 2012b).

Several models of the sustainable city have been proposed, including both interdisciplinary as well as transdisciplinary approaches (Bosselmann, 2008; Pickett et al., 2013). Childers et al. (2014) suggest that analyzing the interactions of key resources across cities may help in understanding the constraints and resource consumption/production patterns that could make some urban areas more sustainable than others. They also suggest that the transition of an urban area to more sustainable pathways will involve changes, in particular trade-offs, in the resource consumption/production patterns of the city. This has been explored already in the context of UM but mostly, as it was mentioned earlier, by examining the direct dependence of urban areas on different resources (Kennedy et al., 2012). Alternatively, through virtual water flows, the level of dependence and impact that an urban area’s consumption/production pattern has on distant water sources could be quantified. This in turn has potential for creating a stronger synergy between the city, i.e. its people, and the biophysical environment and natural resources needed by the city. The fostering of such synergy has been suggested to be at the heart of urban sustainability (Childers et al., 2014).

Urban water managers are most concerned with the direct water flows that are under their control. Thus, cities are actively and innovatively implementing green infrastructure, water reuse techniques, and other technologies to ameliorate direct impacts on water resources (Grant et al., 2012; NRC, 2009; Sedlak, 2014). For example, the city of Philadelphia in the U.S., as part of its green city initiative (Philadelphia Water Department, 2011),
has identified as a key target the need to protect and restore the quality of its urban basins and rivers. Other areas, such as Las Vegas or Maricopa County in the U.S. southwest, have implemented extensive water recycling and water conservation measures along with zero-runoff stormwater and recharge programs. Similar initiatives for restoring the quality of local waters and improving urban water services are underway in several other U.S. cities and in cities across the globe (EPA, 2010; Van Leeuwen and Sjerps, 2014).

Targets require metrics, which are water use sustainability and productivity benchmarks. The development and adoption of appropriate metrics and benchmarks will be the key to advancing city and business decision making and planning in the area of water sustainability. For instance, Van Leeuwen and Sjerps (2014) developed and employed an urban water index comprised of measures covering a range of water-related categories such as water security, water quality, drinking water, sanitation, infrastructure, climate robustness, biodiversity and attractiveness, and governance. They applied this index to 30 cities covering 22 different, mainly European, countries. Within the broader context of urban sustainability, the United Nations started the Global Urban Indicators Database (http://ww2.unhabitat.org/programmes/guo/guo_indicators.asp) in 1993, which includes data from over 237 cities, to track the livability of the participating cities using measures of wastewater treatment, air pollution, waste disposal methods, disaster management, and environmental planning. The World Bank developed the Eco² Cities Guide to help city planners build comprehensive sustainability plans within a participatory process (World Bank, 2013). As part of this plan, they propose using different water-related indicators derived from
MFA. Siemens (2011) developed a green city index to measure the environmental performance of 27 cities in the U.S. and Canada. This index is comprised of several water indicators such as the per capita water use, water system leakages, stormwater policy, and water quality.

Key to a city’s direct water management is also acknowledging its resiliency, to absorb shocks and stresses while retaining identity, structure, and key processes (Alliance, 2007; Godschalk, 2003; Gunderson and Hollins, 2002). When viewed as processes, urban sustainability and resilience have shared as well as distinct objectives (Redman, 2014). It is possible for a city to be resilient while being on a long-term pathway that is perceived by the urban citizens as not being very sustainable. Pickett et al. (2014) highlight that resilience can be used to promote and operationalize sustainability. For instance, adaptive capacity and robustness may enable a city to remain, in spite of shocks and stresses, on a pathway deemed as sustainable to meet target goals. In this context, urban WF analysis can contribute to understanding and specifying urban water resilience. Indeed, a city’s link with other distant places may help in identifying vulnerabilities (the potential to experience change or harm due to exposure to a shock or stress), which is missing from all of the initiatives and studies just described above. For example, water-related vulnerabilities could be identified as a city that is preferentially linked to water stressed regions or as a city in a water stressed region that has only few links. To examine urban water resilience in the context of vulnerability, a useful research direction is to employ a comparative analysis framework where the WF of different cities, and its components (e.g., direct versus virtual water, water used for consumption versus water used for
production, etc.), are compared against each other. With this framework, cities could be classified into different groups according to their water footprint and/or other characteristics (e.g., size, population, network-related properties, etc.). These groups could then be used to test different hypothesis regarding the role of geographic location and climate on water-related vulnerabilities, or the role of scale and regional boundaries on the identification of shocks and stresses (Ruddell et al., 2014).

It is also critical, per economic theory, to incorporate the production of value into water benchmarks, because it is equally important to use water beneficially as it is to conserve water. This creation of value as a means to maximize social welfare, may or may not lead to the conservation of water but warrants further investigation. This is an area where WF analysis can lead to tangible and measurable applications as these kind of metrics are often employed to inform high-level investment and management decisions, develop environmentally-sensitive plans, and inform the public (Ruddell, 2012; Rushforth et al., 2013). In particular, WF analysis could be employed to develop different value intensity metrics (Ruddell, 2012; Ruddell et al., 2014). Value intensity examines the dollar spent per volume of water consumed and is not meant to evaluate water pricing but is more of a means to determine the worth placed on our resource consumption. Value intensities are particularly useful for urban areas since they can consider the values held by multiple stakeholders and they can account for the direct and/or indirect components of the urban WF.

Applied research for urban WF analysis could also be focused on designing, implementing, and testing ways in which a city may effectively
incorporate WF-based metrics into its operational activities and decisions. The Urban Water Footprint project in the EU (http://www.urban-wftp.eu/en/) is a recent initiative that will monitor and measure the WF at different spatial scales (e.g., city, neighborhood, and building) in the cities of Vicenza (Italy), Innsbruck (Austria), and Wroclaw (Poland). As part of its goals, the project seeks to implement WF analysis as a tool to improve and assist with urban water management as well as to assess and predict the influence of local policy on urban water. This is an emerging area of application for WF analysis. Beyond any specific research outcomes, this kind of project offers an opportunity to engage city stakeholders and urban citizens, and directly measure the societal value of research outcomes. This can be made possible not only by the research being conducted in cities but also by the ability of the urban WF concept to integrate diverse water-related information.

Other potential applications of urban WF analysis are: increasing recycling of virtual water within city boundaries, which is an indirect effect of recycling more products, i.e. reuse within the city; assessing waste flows generated by the city, specific to virtual water being wasted in agricultural goods; value-based benchmarking to identify the most and least water-productive direct water uses; and identifying the most and least sustainable virtual connections to manage, incentivize, and/or regulate them to improve the resiliency of a city’s water supplies. Additionally, Badruzzaman et al. (2014) outline several WF applications specifically for water utilities. Some of the applications that they highlight are: assessing operational/supply chain water use, assessing water-related regulatory and business risks, selecting between future development options, and identifying unsustainable WFs in the
supply chain. Additional and more specific applications of WF to water utilities can be found in their report (Badruzzaman et al., 2014).

Ultimately, the autonomy of cities, their capacity for self-organization and self-initiative, as well as their ability to take action is what makes WF analysis at the city scale appealing and potentially meaningful to an action-oriented audience.
Chapter 6

Summary

The WF provides a conceptual framework to tie our consumptive use to the water resources that are being directly and indirectly affected. This review identifies main findings from national, subnational, and urban studies to highlight the benefits of this framework. Common methodologies were also identified for the urban and subnational scale and relevant themes were discussed as they apply to WF analysis with recommendations for future research.

Trends at the larger extent identify that water is just one factor of production that does not govern our trade patterns. These flow patterns are not impacted by the effects of our water consumption nor do they consider a region’s comparative advantage in terms of water usage. This is especially prevalent at the subnational scale with findings of water scarce regions exporting to water rich regions. Defining appropriate scales and boundaries to understand these transfer exchanges assists with identifying key trends which may warrant further investigation. This has been identified for regions already such as arid northern China.

The main WF methods identified have overlaps with urban metabolism methodology and could be coupled to fully understand the direct and virtual water flows related to all UM fluxes. This is a crucial area of research as all flows are intertwined and giving attention to each flux assists in identifying overlapping needs. At the city scale, top-down EEIO methods provide for full analysis of blue water flows for the economy but building regional IO tables is
tedious and creates room for error which is also prevalent in the aggregation of sector data. LCA, while being developed as a standardized method for product water usage, can be limited by the databases defining water usage. Also, the evaluation of impacts, which is at the root of this assessment, is still in development in terms of analyzing the source water quality and quantity. WFA is fully developed in terms of analyzing crop water usage and provides overarching goals and recommendations for determining a WF at any scale, but by focusing on the bottom up commodity level key water users may not be identified for an urban region.

The city scale is optimal for WF analysis given cities dominance in economic trade which has global implications. The city is also a hub for cultural exchange and through effective planning and permitting has a unique ability to make impactful localized decisions with broad socio-economic consequences. As our natural resource use is driven by decisions at an economic level, the WF methods discussed provide insight to the value exchange occurring through this movement of water that is not being adequately addressed. By utilizing a consistent approach to identify water fluxes, both real and virtual, through an urban boundary and within, the WF creates opportunities for benchmarks and cross comparisons. This proves difficult with a toolbox of methods to choose from, with each method having data and boundary limitations; however, participatory research to identify key datasets needed at the urban scale may provide improvements.

The following recommendations were discussed as important steps for future research:
• The need to track the transition between various consumptive patterns and how such patterns interact with the environment.

• Utilizing an overarching framework that clearly articulates the connections among WF methods.

• Consistently defining an urban boundary for WF analysis within the constraints of limited datasets.

• Distinguishing direct and indirect water footprints as they relate to engineered, hydrologic, and economic flows.

• Incorporating virtual water flows within a UM framework to evaluate all environmental impacts of a city’s fluxes.

• A focus on design, implementation, and testing ways in which a city may effectively incorporate WF-based metrics into its operational activities and decisions.

• Further refinement to determine how water stress affects a city’s resilience and how sustainability initiatives can be determined to overlap with resiliency efforts.

By utilizing the suggestions above future work should explore opportunities to use the WF concept to assist decision makers as we move into a new era of understanding the role of cities in the context of water scarcity.
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Redman, C. L., 2014, Should sustainability and resilience be combined or remain distinct pursuits?, *Ecology and Society* 19(2).


Rushforth, R. R., Ruddell, B. L., 2015, Virtual water transfers within a large US metropolitan area: a case study of the Phoenix Metro area, *Sustainability* (7).


U.S. Census, United States Census Bureau.

UN-Habitat, 2009, Global Urban Indicators - Selected statistics, UN-HABITAT.


World Bank, 2013, Eco²Cities Guide.


## Table A1. Expanded Table 2 - Summary of studies that have evaluated the water footprint at the subnational scale and urban scale.

<table>
<thead>
<tr>
<th>Article</th>
<th>City/Region</th>
<th>Study by</th>
<th>Methodology</th>
<th>Data</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>WF of cities; Indicators for sustainable consumption and production</strong></td>
<td>Berlin, Delhi, Lagos</td>
<td>(Hoff et al., 2013)</td>
<td>WFA – national scaling – green and blue WF. Evaluated 19 major crop groups that make up 71% of global harvested crop land. Spatial interpretation – mapped crop production and consumption at the grid cell level (5 arcmin) and determined cities virtual flows of water by what was brought in from neighboring cells. Determined key sources of exported goods and evaluated scarcity in those regions.</td>
<td>1998-2002 UN - COMTRADE binational trade data. 2000 - UN FAO crop production statistics. GCWM – Global Crop Water Model (Siebert and Döll, 2010) - to calculate crop production, crop water use, and the virtual water content. LPJmL model - water availability for scarcity assessment.</td>
</tr>
<tr>
<td><strong>WF outcomes and policy relevance and change with scale considered: Evidence from California</strong></td>
<td>California</td>
<td>(Fulton et al., 2014)</td>
<td>WFA – green, blue, and grey WF. Used a variety of datasets to estimate the WF of California (CA) for crop commodities, livestock and industrial goods. Evaluated virtual flows entering and exiting CA from other states and from international trade. Compared CA WF to national US WF.</td>
<td>1998-2005 CA Department of Water Resources (CDWR) - 20 crop categories for green and blue water use. 2007 – USDA Census of Ag – production of animal products. 1995 – CDWR – survey of industrial blue water use. 2007 - Commodity Flow Survey (CFS) – import and export data. 2007 and 2008 - US Census Bureau – international trade data.</td>
</tr>
<tr>
<td><strong>Assessing WF at river basin level: a case study for Heihe River Basin in northwest China</strong></td>
<td>Heihe River basin, China</td>
<td>(Zeng et al., 2012)</td>
<td>WFA – green and blue WF. Focus was on production WF within the river basin. 12 crop groups and 4 animal categories. Evaluated blue water scarcity. Combined city and county data for crop area and production in the basin. Water consumption ratios for industry assumed 36% and 67% for domestic use. Water scarcity evaluated using method by (Hoekstra et al., 2012).</td>
<td>WFA manual for methodology (Hoekstra et al., 2011) CROPWAT model - blue and green virtual water content for each crop. MIRCA2000 database for crop distribution maps at 5 arcmin. 2011 - FAO – Animal Production and Health data for density of livestock. Ministry of Water Resources of China – withdrawal data.</td>
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<tr>
<td><strong>Sensitivity and uncertainty in crop WF accounting: a case study for the Yellow River basin</strong></td>
<td>Yellow River basin, China</td>
<td>(Zhuo et al., 2014)</td>
<td>WFA – green and blue WF. Determined WF for maize, soybean, rice, and wheat production within the basin for the period of 1996-2005. Used a grid-based daily water balance model to assess crop water usage – focus was on uncertainty of the model. Uncertainty was analyzed using Monte Carlo simulation. Seven parameters were modified.</td>
<td>Climate Research Unit Time Series (CRU-TS) Gridded monthly climatic data taken from (Harris et al., 2014) UN FAO - Crop temperature data Crop coefficient values from a number of sources. MIRCA2000 – irrigated and rainfed areas and yields for each crop.</td>
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<td>Topic</td>
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<tr>
<td><strong>Agricultural VW flows within the USA</strong></td>
<td>Interstate transfers in the U.S.</td>
<td>WFA – green and blue WF. Evaluate the virtual water footprint of 5 broad food commodity groups. Food transfer data obtained from the CFS. Evaluates blue and green virtual flows for cereal and milled commodity groups and livestock within the U.S. and compared to international trade.</td>
<td>2007 – Commodity Flow Survey (CFS) – U.S. Department of Transportation 2007 – USDA - Agricultural production data 1996-2005 average - virtual water content for U.S. states (Mekonnen and Hoekstra, 2011; Mubako and Lant, 2013; Mubako, 2011)</td>
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<tr>
<td><strong>Going against the flow: A critical analysis of inter-state VW trade in the context of India’s National River Linking Program</strong></td>
<td>Interstate transfers in India</td>
<td>WFA – green and blue WF. Used Kampman’s (2007) spatial analysis of 16 primary crops, representing 87% of total water use and 86% of total crop land to evaluate trends of virtual water flows with interstate trade across India.</td>
<td>Virtual water estimates from Kampman, 2007 study which used CROPWAT for green and blue WF and 1997-2001 FAOSTAT crop production data</td>
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<td><strong>Virtual versus real water transfers within China</strong></td>
<td>North and South China</td>
<td>WFA – green and blue WF with blue WF separated into surface and groundwater sources. Evaluated six categories (condensed from 25 crops and 6 meat categories): grain, vegetable, fruit, meat and poultry products, eggs, milk and dairy products. Used UNFAO commodity balance sheets to determine production, stock changes, and demand for each region in China.</td>
<td>1999 - Virtual water content of crops within China - used CROPWAT, Climwat, and FAOSTAT data. 1995-1999 - virtual water flows between China and other nations taken from (Hoekstra, 2003)</td>
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<tr>
<td><strong>Water resources transfers through Chinese interprovincial and foreign food trade</strong></td>
<td>Interprovincial transfers in China</td>
<td>WFA – green and blue WF. Used H08 model for VW – modeled interregional trade (31 divisions) and trade with the rest of the world (ROW) Evaluated VWT for 4 main crops – corn, rice, soybean, wheat and 3 livestock products – ruminant, pork, poultry. Used linear optimization model to downscale data from 8 regions to provincial level. Applied network modeling to determine key trends in interprovincial and international transfers and water savings.</td>
<td>2005 - CHINAGRO – Extensive Interregional trade data of agricultural production – tracked for 8 regions 2002-2007 average for crop water use (blue and green) – found by H08 – Global hydrologic model - 0.5° (30 arcmin) spatial resolution MIRCA2000 used for crop area</td>
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<tr>
<td><strong>The WF of Indonesian provinces related to the consumption of crop</strong></td>
<td>Interprovincial transfers in Indonesia</td>
<td>WFA – green, blue, grey WF. Evaluated 10 primary crops that represent 86% of water use and cropland. Virtual flows between 30 provinces determined by commodity mass balance – used average per capita consumption in the nation. Assumed trade happens within close island groups and then excess was distributed to other islands.</td>
<td>2000-2004 Climwat used to determine ET. Crop Parameters taken from a number of sources. 1996-2005 - Virtual water entering Indonesia (Mekonnen and Hoekstra, 2010) FAO data on fertilizer content, leaching factor (Chapagain et al., 2006b), max acceptable level of nitrogen (EPA, 2005) for grey water assessment.</td>
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<td><strong>Embedded resource accounting for coupled natural-human systems: An application to water resource impacts of the western U.S. electrical energy trade</strong></td>
<td>Western U.S. states</td>
<td>WFA/MRIO/ERA – blue WF – applied an ERA analysis to evaluate net direct impacts which is a relationship between direct water use, indirect water use, and equivalency factor determined by the point of view of the accountant. Applied ERA to water usage in the electric energy sector for 11 western states to determine virtual water trade embedded in energy use. Assessed dollar intensity for each states local energy consumption as a $/gallon to compare the equivalency ratios (gallon/gallon) between states.</td>
<td>2009 - U.S. Energy Information Administration (USEIA) – import and export energy data and pricing for each state and water withdrawal and consumption which also relies on USGS data. Consumption to Availability water stress index calculated at county level based on methods from Tidwell (Tidwell et al., 2012)</td>
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<td>Study Title</td>
<td>Location</td>
<td>Methodology</td>
<td>Data Sources</td>
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<tr>
<td>Analysis of WF of Beijing in an interregional IO framework</td>
<td>Beijing</td>
<td>SRIQ – blue WF. Used a 60 sector IO table, aggregated into 33 sectors for comparable water use data. Evaluated the local and external WF for Beijing municipality (Beijing city and 5 counties) and determined the source regions from the surrounding provinces. External WFs were determined by the water savings present for Beijing had they produced the goods in Beijing – not the water use in the source region.</td>
<td>2002 – National Bureau of Statistics of China – Interregional IO Tables 2001 – Beijing Municipal Water Conservation Office – Sectoral Water Requirement Quotas for 33 sectors for freshwater intake</td>
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<tr>
<td>A hydro-economic MRIO analysis of the Haihe River basin’s waterfootprint and water stress</td>
<td>Haihe River basin, China</td>
<td>MRIO – blue WF for the HRB. Used proportional scaling for each province’s IO table to delineate the basin. MRIO analysis linked basin to 25 provinces and 2 city regions. Evaluated direct and indirect WF as well as source of each flow in and region where items were exported to. Scarcity assessed using WSI (Pflster et al., 2009)</td>
<td>2008 – China Economic Census Yearbook – sector water withdrawals China Statistical Yearbook on Environment – household withdrawals Water Resource Bulletins – water consumption ratio for sectors IO tables (Liu et al., 2012b)</td>
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<tr>
<td>Decomposition analysis of water footprint changes in a water-limited river basin: a case study of the Haihe River basin, China</td>
<td>Haihe River basin, China</td>
<td>SRIQ – blue WF assessed for the HRB for 2002 and 2007. Generated regional IO tables (GRIT) from method by (Jensen et al., 1979) to obtain the basin IO tables using administrative boundary IO tables. Economic contributions are assumed to be proportional to the land in the basin. 17 sectors evaluated.</td>
<td>2003 and 2008 – Beijing, Tianjin, and Hebei Statistics Bureau – IO tables 2003 and 2008 – Haihe River Water Conservancy Commission and Ministry of Water Resources – water use data</td>
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<tr>
<td>An Extended IO table compiled for analyzing water demand and consumption at county level in China</td>
<td>Shandan county, China (Deng et al., 2014)</td>
<td>SRI – blue WF. Evaluated primary industries within the county. Used non-survey and partial-survey methods to derive IO tables at county level. 144 sectors at province level aggregated to 12 at county level. Divided cropland into irrigated and non-irrigated to determine the economic value of water demand for irrigated land.</td>
<td>1992-2007 Statistics Yearbook of Gansu Province. Water Department of Shandan County and Chinese Academy of Sciences.</td>
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<tr>
<td>Assessment of regional trade and VW flows in China</td>
<td>North and south China (Guan and Hubacek, 2007)</td>
<td>IRI – blue WF. Evaluated eight hydro-economic regions and established water accounts for each. Merged 7 provincial IO tables into two regional IO tables for North and South China – evaluated trade across 40 sectors and examined water availability, fresh water use and created water consumption coefficient and wastewater coefficients to evaluate total exports in terms of virtual water.</td>
<td>1997 - State Statistical Bureau of China – Input-Output (IO) Tables Climate and Human Activities – sensitive Runoff Model (CHARM) - water availability/supply 1997 – China’s Regional Water Bulletins 1999 - Regional Water Statistics Yearbook, annual hydrology reports 1995 - Third National Industrial Survey – wastewater discharge</td>
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<tr>
<td>Virtual Scarce Water in China</td>
<td>Interprovincial trade in China (Feng et al., 2014)</td>
<td>MRIO – blue WF. Evaluated a consumptive water footprint and a scarce water footprint using average grid cell values of WSI to determine provincial scarce water consumption. Water consumption ratios applied for each river basin. IO table includes 26 provinces and 4 city regions with 30 sectors which were aggregated to 16 water use sectors.</td>
<td>IO tables (Liu et al., 2012b) 2008 China Economic Census Yearbook – sector water withdrawal 2008 – China Water Resources Bulletin-for river basins – determined water consumption ratios.</td>
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<tr>
<td>Assessing regional and global WF for the UK</td>
<td>The southeast and northeast UK (Yu et al., 2010)</td>
<td>MRIO – blue and green WF. Examined domestic WF looking at southeast and northeast UK. Merged regional IO tables to evaluate full MRIO analysis with two UK regions and three world regions. Evaluated supply chains for 28 sector to determine domestic WF. Used forward and backward linkages to determine which sectors have the greatest influence on water consumption.</td>
<td>2001 UK Office for National Statistics (ONS) – national IO tables and household expenditure EUROSTAT – international trade Global Trade Analysis Project (GTAP) – international IO tables 1990 - REWARD - Regional and Welsh Appraisal of Resource Productivity and Development - water consumption industry and service Agricultural water use (Hoekstra and Chapagain, 2007) 2001 - Environment Agency - household water use</td>
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<tr>
<td>Topic</td>
<td>Location/Sectors/Regional/Timeframe</td>
<td>Method/Approach</td>
<td>Water Footprint (WF), Water Footprint Assessment (WFA), Life Cycle Assessment (LCA), Embedded Resource Accounting (ERA), Input-Output (IO), Single-Region (SRIO), Inter-Regional (IRIO), and Multi-Regional (MRIO)</td>
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</table>
| Spatially Explicit Analysis of Water Footprints in the UK            | Domestic UK (Feng et al., 2011b)   | MRIO – blue and green WF. Distinguished consumption and production WF. For consumption - evaluated geo-demographic consumer segmentation data at the local authority level. Computed direct water coefficients and assessed virtual water flows within UK and trade with regions of other countries. 123 sectors from IO tables aggregated to 28 water consumption sectors. | 2004 – UK national IO table  
Global Trade Analysis Project Foreign IO tables  
2006 – UK ONS - water consumption Office for Official Publications of the European Communities – water use data for countries  
MOSAIC database - UK geo-demographic consumption data |
| How City Dwellers Affect Their Resource Hinterland                   | Sydney and Melbourne (Lenzen and Peters, 2010) | MRIO – indirect impacts of blue water use. Evaluated consumption, industrial production, economic turnover, employment, water use, and greenhouse gas emissions to evaluate the cities impacts on source regions. State level water use data was disaggregated using statistical local area based proxy data using a number of factors: land use, employment, etc. | Australian Bureau of Statistics:  
2006 – Water use accounts  
2007 – Household expenditure survey  
2001 – Census data by location |
| Understanding VW flows: A MRIO case study of Victoria               | Victoria, Australia (Lenzen, 2009)   | MRIO – blue WF. Linked water use accounts with MRIO model using 344 sectors across 8 Australian states to evaluate the VW flows for Victoria. Assessed water savings for Victoria – assumed production in each region would use same amount of water had it been produced in Victoria. | Australian Bureau of Statistics:  
2008 – National and state IO tables  
2006 – Electricity, gas, water and sewerage operations – companion data  
2004-05 – Water use accounts |
| An IO model of water consumption: Analysing intersectoral water relationships in Andalusia | Andalusia, Spain (Velázquez, 2006) | SRIIO – blue WF. Evaluated direct and indirect water use/consumption within Andalusia for 25 sectors. | 1996 – Andalusion Environmental Input-Output Tables (TIOMA) by Consjeria de Medio Ambiente – includes quantity of water consumed per sector (cubic meters)  
1995 - Andalusion Input-Output Tables (TIOAN) by Instituto Estadistico de Andalusia – production of each sector (currency) |
| Input-output analysis of VW transfers: Case study of California and Illinois | California and Illinois (Mubako et al., 2013a) | IRIO – blue and green WF. 440 sectors aggregated into 8 sectors to match USGS water consumption categories. Compared virtual water flows between California and Illinois, other U.S. states and internationally. | 2008-IMPLAN IO tables for 440 US sectors  
2008-USGS water consumption data for 8 sectors  
Green WF (Mubako, 2011) |
| Water Footprint of Cereals and Vegetables for the Beijing Market    | Beijing (Huang et al., 2014)        | LCA – blue and grey WF. Determined the blue water footprint for maize, wheat, and tomatoes. Compared Beijing’s WF to other source regions for each crop by applying the WSI to investigate scarcity and water degradation. | 2010-National Development and Reform Commission in China – Farm input expenditures: Chinese Reference Life Cycle Database to convert to indirect water use of farm inputs and provided water use in transportation and packing materials. |
APPENDIX B

Mathematical representation of WF methods

B1. Water Footprint Assessment:

The following equations have been modified from (Mekonnen and Hoekstra, 2011) and (Mubako and Lant, 2013) to express the equations and approaches taken by the Water Footprint Network to express the consumptive water footprints for a region. Production water footprint equations can be found in the respective resources.

The consumptive water footprint for a chosen region can be expressed as the following:

\[ WF_{cons} = WF_{cons, direct} + WF_{cons, indirect} (agricultural) \]  \hspace{1cm} (1)

\( WF_{cons} \) is comprised of the direct water consumption, \( WF_{cons, direct} \), and the indirect water consumption, \( WF_{cons, indirect} \), for the region. \( WF_{cons, direct} \) includes household, and industrial direct water use as well as direct water use for crops minus any exports leaving the region. \( WF_{cons, indirect} \), includes both indirect water use for both crop and livestock commodities that are used in production outside of the chosen regions boundary. It should be noted that a consumptive coefficient is usually applied to withdrawal water for direct water use for industry and domestic blue water footprints when return flow data is unavailable (Hoekstra et al., 2011). Crop water consumption is assumed using crop water models discussed in chapter 3.

The indirect water footprint in terms of a regions virtual water imports and exports is
\[ WF_{\text{cons\_indirect}} = VW_{\text{import}} - VW_{\text{re\_export}} \]  \hspace{1cm} (2)

The indirect water footprint for consumption, \( WF_{\text{cons\_indirect}} \), is equivalent to the virtual water of the imported commodity, \( VW_{\text{import}} \) (m\(^3\)/yr), minus the virtual water that is re-exported, \( VW_{\text{re\_export}} \) (this is included where data is available).

\( VW_{\text{import}} \) (crops) is found by multiplying the crop commodity yield in the source region, \( C_{\text{import}} \) (ton/yr) by the volume of water used for the commodity from the source region, \( WF_{\text{import\_prod}} \) (m\(^3\)/ton) expressed as

\[ VW_{\text{import}} \) (crops) = C_{\text{import}} \times WF_{\text{import\_prod}} \]  \hspace{1cm} (3)

The \( VW_{\text{import}} \) (livestock) is comprised of the virtual water used to produce the feed that is consumed over the life of the animal, \( VW_{\text{feed}} \), plus \( VW_{\text{withdrawal}} \) which represents any other consumptive water needs for the livestock. This is expressed as

\[ VW_{\text{import}} \) (livestock) = VW_{\text{feed}} + VW_{\text{withdrawal}} \]  \hspace{1cm} (4)

**B2. Environmentally Extended Input-Output:**

IO methodology expressed below has been modified from (Miller and Blair, 2009; Mubako et al., 2013a; White et al., 2015; Yu et al., 2010). EEIO studies listed in Table 2 may also be referenced for their representation of these methods. IO methods begin with an \( n \times n \) matrix, \( n \) representing the number of sectors in the economy. For each sector, \( i \), there is total output, \( x_i \), final demand, \( y_i \), and monetary flows across sectors \( i \) and \( j \), \( X_{ij} \). This relationship can be expressed as

\[ x_i = \sum_{j=1}^{n} X_{ij} + y_i \]  \hspace{1cm} (5)

83
\[ a_{ij} = \frac{X_{ij}}{x_j} \]  

expresses the technical coefficients that represent the inter-industry sales, \( X_{ij} \), divided by the total input of a sector \( j, x_j \). A substitution of \( a_{ij} x \) can be made to equation 5. The matrix of technical coefficients is expressed as \( A \) and is represented as

\[ x = Ax + y. \]  

(7)

Solving for \( x \) creates the following relationship,

\[ x = (I - A)^{-1} y. \]  

(8)

The Leontief inverse matrix, \((I-A)^{-1}\), signifies the total production every sector must generate to satisfy one unit of final demand. \( I \) represents the \( n \times n \) unity matrix.

For a multiregional (MRIO) assessment the technical coefficient matrix must be applied across regions represented as

\[ x = (I - A^r)^{-1} y. \]  

(9)

The region of analysis is denoted by \( r \) and \( rr \) implies trade within the same region. This can be extended to e.g. \( rm, rn \) to denote trade with multiple regions.

The final step is to create a net water consumption coefficient for each sector

\[ w_j = \frac{c_j}{x_j}. \]  

(10)

By taking the amount of consumed water for sector \( j, c_j \), and dividing it by the total input to that sector, \( x_j \). The water consumption coefficient, \( w_j \), is expressed as a volume per dollar. This differs from the consumptive water use coefficient mentioned in section B1. The consumptive water use coefficient
does need to be applied to withdrawal data to create the water consumption coefficient, \( w_j \). Then, \( w \), a \( 1 \times n \) matrix representing \( w_j \) across all sectors is multiplied by the inverse Leontief to create the total water consumed throughout the production chain

\[
W = w(I - A^w)^{-1}y. \tag{11}
\]

This method can be extended to any environmental flow.