HUMAN MEDIATED PHYSICAL AND VIRTUAL WATER TRANSFERS OF THE UNITED STATES: WHO USES THE WATER?

BY

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DISSERTATION

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ABSTRACT

Globalization has strengthened and expanded connections between consumers and distant water resources used in production, by enabling consumer demand in one location to be fulfilled with production and resource use in another. Many of the environmental consequences associated with water-intensive production, particularly agricultural production, are not felt by those consuming the products but are left as an artifact for producing communities. Consumers increasing dependency and influence on nonlocal water use decisions can lead to water scarcity, groundwater depletion, or other environmental impacts, but, with better understanding, can provide an opportunity for innovative and sustainable solutions to local water issues.

The primary goal of this dissertation is to better understand the telecouplings between nonlocal consumers, which drive transfers of water and water-intensive goods, and the overexploitation of local water resources. There is an incongruity between the scale at which water is studied and managed and the scale that water dependencies and impacts coalesce; this dissertation begins to resolve this mismatch of scale. This work provides an important first step toward empowering producers, consumers, water planners, and decision makers to manage water resources more holistically and at the appropriate scale by linking understanding of local production water consumption with new knowledge of virtual water transfers — that is, the water embedded in the production of traded commodities.

We draw upon publicly available data on agricultural production, water withdrawals and consumption, water infrastructure, and trade, as well as modeled estimates of agricultural water requirements to quantify virtual water transfers between producers and consumers. A novel dimension of this research is the fine spatial, temporal, commodity, and water source resolution made possible through empirically-based datasets and our unique methodological approach.

In this dissertation, we quantify and track agricultural virtual groundwater transfers from the overexploited Mississippi Embayment, High Plains, and Central Valley aquifer systems in the United States to their final destination. Specifically, we determine which US metropolitan areas, US states, and international export destinations are currently the largest consumers of these critical aquifers. Next, we study drought impacts to food and virtual water transfers from the Central Valley of California and examine the linkage between distant consumption of virtual water resources and local water impacts. More broadly, this study elucidates how local climate shocks reverberate through the global food system, highlighting the importance of complex interactions in the coupled climate-food-water system, and the critical role of local groundwater depletion. A comprehensive, high-resolution database was also created that estimates the water footprint of US production and the virtual water contents of food, energy, services, manufacturing, and mining products produced within the US. This work elucidates how different water sources within the US support the country's economy, explicitly relating these water sources to over 500 different industries and products. Finally, an interdisciplinary framework to mitigate the complex social and natural barriers to physical water transfers is put forth.

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Chapter 1

INTRODUCTION

1.1 Motivation & Overview

Mankind has left an indelible imprint on many of the world's watersheds and aquifers, fundamentally altering their natural hydrologic response. In turn, the evolution of hydrologic systems has altered societal behavior. Together, this interplay between humans and water represents an example of a coupled human-nature system. The distant interactions between people and places are commonly referred to as 'telecoupling' (*Liu et al.*, 2013), which represent a specific case of the complex interactions that arise between coupled human and natural systems. These nonlocal interactions are increasingly widespread and lead to unanticipated outcomes with profound implications for resource consumption and sustainability (*Liu et al.*, 2015). In this light, there is a growing recognition that hydrologic systems should no longer be studied only within a local context but instead examined through the lens of the nonlocal socioeconomic system that shape it (*Hoekstra and Hung*, 2005; *Sivapalan et al.*, 2014; *Liu et al.*, 2015; *Sanderson and Frey*, 2015).

Human activity, such as the global food trade system, scales in unexpected and unpredictable ways, producing water sustainability challenges that must be integrated into novel water science and management approaches (*Dalin et al.*, 2012; *Sivapalan et al.*, 2014; *Dang et al.*, 2015). Socio-hydrology offers a framework to study cross-scale interactions between humans and hydrologic systems (*Sivapalan et al.*, 2012, 2014; *Levy et al.*, 2016). *Konar et al.* (2016) conceptualizes (see Fig 1.1) how the traditional study of physical flows and stocks of water within a watershed can be coupled with human-mediated flows and stocks of freshwater. In the proposed socio-hydrologic framework, humans shape the hydrologic system through internal modifications (*e.g.,* land-use change, infrastructure), physical water transfers, and virtual water transfers. The inclusion of external water transfers beyond the physical boundaries of the watershed (which is where most studies end), allows for the exploration of telecouplings between humans and specific water resources. In addition, this framework enables the identification of local and/or nonlocal causes of unsustainable water use and can guide sustainable water management across scales.

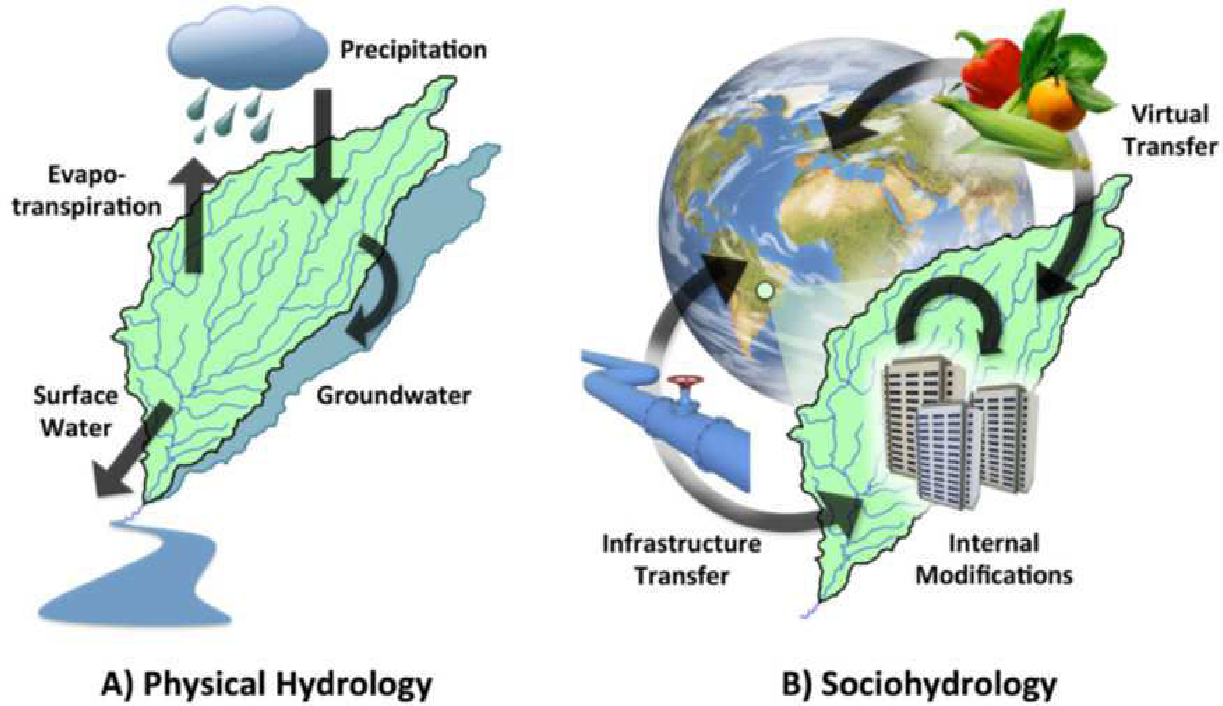

Figure 1.1: Conceptual diagrams of (A) physical hydrology and (B) socio-hydrology. The socio-hydrology framework incorporates human-mediated flows and stocks of freshwater that extend beyond the watershed, the principle unit of analysis in traditional hydrology. From *Konar et al.* (2016) with permission.

Nonlocal interactions between freshwater and humans are commonly made evident by physical and virtual water transfers. Both types of water transfers are shaped by complex exchanges between natural processes and social institutions, policies, norms, and economics, which may potentially lead to unexpected water transfers and associated impacts. Physically transferring water between users or locations (*i.e.,* water reallocation) is an alternative approach to more traditional water supply and demand management strategies. Water reallocation is the transfer of water between users who are committed formally or informally to a certain amount of water when the existing allocation is physically infeasible, economically inefficient, or socially unacceptable. Alternatively, virtual water transfers represent the water embedded within the production of traded goods or services. Virtual water transfers couple local water use to the final, often nonlocal, consumer who is ultimately driving its consumption.

One of the underlying mechanisms relating and driving both virtual and physical transfers of water is ineffective water pricing. Water users often only pay water's delivery cost, and many pay even less than that. Water prices rarely reflect opportunity cost associated with other water uses, whether by humans or the environment. Disregarding the value of some water uses, as well as the full cost of water use, leads to negative externalities, such as groundwater depletion and water degradation. Moreover, water, or the goods — especially agricultural goods — being produced with water, are heavily subsidized. The use of subsidies and pricing water below its marginal cost creates market distortions. These distortions can partially explain the seemingly paradoxical water-intensive production patterns in arid areas and transfers of water from water scarce to water rich regions (*Dalin et al.*, 2014; *Zhao et al.*, 2015).

The goal of this dissertation is to demonstrate how distant demands for water and water-intensive goods shape local water use and sustainability. More specifically, we will evaluate how the global food trade system links consumers around the world to major aquifers in the United States. A particularly novel dimension of this dissertation is the explicit linking of specific water bodies (namely, aquifers of the US) to distant consumers using empirically-based, sub-national commodity transfer data. Previous sub-national studies have used modeled trade estimates (*e.g.*, *Verma et al.* 2009; *Dalin et al.* 2014), which typically require several crude assumptions and introduce additional errors. This work captures the source of water use by specific commodities, evaluates indirect consumption patterns, and assesses impacts of water use at fine spatial resolutions. A comprehensive database was also created that estimates the water footprint of US production and the virtual water contents of over 500 food, energy, services, manufacturing, and mining products produced within the US. Lastly, an interdisciplinary framework to mitigate the complex social and natural barriers to physical water transfers is overviewed and advanced.

In this introductory chapter, I discuss relevant background and concepts, outline the research objectives and questions of this research, and demonstrate the original contributions of this work. Lastly, I layout the organizational structure of the remaining dissertation.

1.2 Background

Mankind's growing production and consumption patterns have moved the planet beyond its boundaries of sustainable resource use (*Rockström et al.*, 2009). Urbanization and globalization have caused a decoupling of human consumption from the local resource base, making it difficult to link and appropriate consumers to the carry capacity of far-off landscapes. This may create dependencies that are not ecologically sustainable or geopolitically stable. The seminal work of *Rees* (1992) first introduced the 'footprint' concept to relate consumers to distant 'elsewhere' and link their resource demand to the negative impacts on the areas that support them. Since then, the footprint concept has evolved from broadly concerning ecology (*Rees and Wackernagel*, 1996; *Wackernagel and Rees*, 1998) to more specific measures of mankind's mark on atmospheric greenhouse gases (carbon footprint; *Wiedmann and Minx* 2008), mineral cycles (nitrogen and phosphorus footprints; *Wang et al.* 2011; *Leach et al.* 2012), and water resources (water footprint; *Hoekstra* 2003), among others.

This dissertation's focus is on the various ways society uses and impacts water both locally and nonlocally. The water footprint (WF), which is a measure of fresh water appropriated to different societal purposes, is the foundational concept upon which we build our work. Water footprints can be used to describe the direct water use of producers (WF of production) or the indirect water use by consumers of water-dependent goods (WF of consumption). Additionally, the water footprint of a product (PWF, also known as virtual water content, VWC) is the amount of water used to produce that commodity, good, or service, typically normalized by the product's weight or price. The Water Footprint Assessment (WFA) framework, along with the accompanying 'virtual water' (VW) concept that relates water used in production to specific distant consumers of those goods, provides a quantitative methodology to assess direct and indirect water use. This framework has been used to trace indirect dependencies on distant water resources (*Dalin et al.*, 2012), connect far-removed consumers to out-of-sight water issues which they play a part (*Marston et al.*, 2015; *Marston and Konar*, 2017), and to assess exposure to nonlocal water related shocks (*Rushforth and Ruddell*, 2016).

The following sections will describe how the field has progressed since its inception and how this dissertation pushes the literature forward. Attention will be given to how the literature has evolved with regards to i) water source partitioning (*e.g.,* blue vs. green; surface water vs. groundwater; scarce vs. non-scarce), ii) spatial resolution, iii) industry and commodity specification, and iv) temporal resolution of analysis. This dissertation makes advances on all four fronts and contributes unique methodological approaches to answer outstanding questions in the field (refer to 1.3 and 1.4 sections).

1.2.1 Methodological advances and overview

Water footprint and virtual water studies have advanced considerably since the concept was first introduced by *Hoekstra* (2003). Initial studies were generally performed at the global level, tracing VW flows between nations. Water footprints were assumed to be uniform over large spatial areas and product types, ignoring local climatic and production difference. Likewise, inter- and intra-annual variation in climate and crop yields were ignored in favor of longer term averages. Local water scarcity, degradation, and overexploitation of different water sources were not considered.

The first methodological framework to quantify VW trade and WF of consumers and producers was a process-based, bottom-up approach called the WFA developed by *Hoekstra et al.* (2009). This has since been followed by a similar bottom-up approach developed within the life cycle assessment (LCA) literature. Top-down approaches have been widely used as well and typically use environmentally extended input-output (EEIO) models to trace VW through the entire supply chain. We employ and build-upon the WFA methodology throughout this dissertation because of its ability to accommodate finer commodity, water source, and spatial resolution than other methodological frameworks.

Water source delineation

Water has different opportunity cost and sustainability implications depending on its source. In recognition of this, the concept of blue and green water (*Ringersma et al.*, 2003; *Falkenmark and Rockström*, 2004; *Falkenmark and Rockström*, 2006) was adopted early in the WF and VW literature. Green water refers to water in the root zone made available by precipitation, while blue water is comprised of surface water and groundwater. Although some studies do not distinguish between green and blue water (*e.g., Islam et al.* 2007; *Zeitoun et al.* 2010; *Vanham* 2013), this is increasingly rare. Several studies, especially those following the LCA or the top-down EEIO approaches, do not consider green water, insisting its use has little environmental or economic implications (*Chenoweth et al.*, 2014). However, a growing number of researchers contend that green water should not be overlooked (*Rockström*, 2001; *Allan*, 2006; *Falkenmark and Rockström*, 2006; *Rost et al.*, 2008; *Aldaya et al.*, 2010).

Empirical measurements of water consumption are sparse. This is particularly true of agricultural water consumption, which is responsible for 92% of global water consumption (*Hoekstra and Mekonnen*, 2012). Therefore, most studies utilize a crop water model to calculate the consumptive water requirements (*i.e.,* evapotranspiration, ET) and crop yields necessary to determine the crop's VWC (ET/crop yield). These models partition crop ET between green water and blue water based on climatic, crop, and management conditions. CROPWAT is the most used model, while models such as the Global Crop Water Model (GCWM) (*Siebert and Döll*, 2010), H08 (*Hanasaki et al.*, 2010; *Dalin et al.*, 2014), and AQUACROP (*Chukalla et al.*, 2015; *Zhuo et al.*, 2016) have been widely utilized as well.

Researchers are gradually distinguishing between blue water sources as well. Recent studies have included environmental flow requirements and monthly variations in water availability to differentiate blue water use based on its scarcity and sustainability impacts (*Hoekstra et al.*, 2012; *Hoekstra and Mekonnen*, 2016; *Zhuo et al.*, 2016; *Yano et al.*, 2016). Additionally, a handful of studies have partitioned blue water into blue surface water and blue groundwater sources, since the implications of using each is different (*Aldaya and Llamas*, 2008; *Aldaya et al.*, 2010; *Dumont et al.*, 2013; *Schyns and Hoekstra*, 2014; *Schyns et al.*, 2015; *Yano et al.*, 2015, 2016; *Marston et al.*, 2015; *Marston and Konar*, 2017). *Mayer et al.* (2016) goes even further, attributing water use of specific sectors to shallow groundwater, deep groundwater, tributary surface water, and Great Lakes water. Recently, *Dalin et al.* (2017) distinguished between renewable and non-renewable groundwater footprints of global crop production. *Marston et al.* (2015), *Marston and Konar* (2017), and now *Dalin et al.* (2017), are the first to connect groundwater extraction to the nonlocal consumers driving their overexploitation. Throughout this dissertation, we distinguish between groundwater, surface water, and green water sources. This represents the state of the science and begins to bring more meaning to VW and WF analysis.

Grey water was introduced several years after blue and green water by *Hoekstra and Chapagain* (2011) as an indicator of freshwater pollution caused by a production process. It represents the volume of water needed to dilute pollutants (namely, nitrogen and phosphorous) to a given water quality standard. The use of grey water footprints has not been as widely adopted as blue and green water footprints because it is a theoretical rather than actual consumptive measure and the assumptions behind it are questionable (see *Chenoweth et al.* 2014). Therefore, grey water is not considered in this dissertation.

Spatial resolution

Water footprint and virtual water trade assessments range from the urban (*Hoff et al.*, 2014; *Rushforth and Ruddell*, 2015, 2016) to global scale (*Hoekstra and Mekonnen*, 2012;

Wang and Zimmerman, 2016). Most studies evaluate WFs and VW trade between nations since data is more abundant (particularly bilateral trade data) at the national level than the subnational level. Sub-national VW studies typically pair VWC with modeled estimates of commodity transfers (*e.g., Verma et al.* 2009; *Dalin et al.* 2014, 2015) or multi-regional input-output (MRIO) models (*e.g., Shi and Zhan* 2015; *Zhi et al.* 2014; *Dong et al.* 2014). This dissertation builds on the work of *Lin et al.* (2014) and *Dang et al.* (2015) who first utilized empirical sub-national commodity transfer data to explore the properties of the US food and VW trade networks. Our use of empirically-based commodity transfer data within this dissertation reduces the uncertainty in VW trade estimates and contributes to the novelty of our work.

Regional (*Vanham*, 2013; *Vanham and Bidoglio*, 2014) and subnational studies can provide more detail and offer greater insight into sustainability challenges faced within a basin or locality than global studies. For example, *Aldaya and Llamas* (2008) and *Aldaya et al.* (2010) determined the green, surface, and groundwater footprint of Spain's Guadiana Basin under different climate conditions (*e.g.,* wet, normal, dry) and made suggestions on the implications of water related policies. Likewise, *Rushforth and Ruddell* (2015) and *Rushforth and Ruddell* (2016) use US county level estimates of water withdrawals per sector to determine the exposure and resilience of two US cities to direct and indirect water risk. In many areas though, further spatial refinement in WF and VW estimates is limited by a lack of data.

Subnational, national, and global studies are gradually moving toward using both hydrologic and geopolitical boundaries in their analysis (*Aldaya and Llamas*, 2008; *Vanham*, 2013; *Vanham and Bidoglio*, 2014; *Schyns and Hoekstra*, 2014; *Pfister and Bayer*, 2014; *Wang and Zimmerman*, 2016; *Zhuo et al.*, 2016). Increasingly, studies are attempting to relate VW trade and WF to local environmental sustainability and hydrologic conditions. The shift toward more impact oriented VW and WF assessments necessitates analysis at a finer spatial resolution in order to reveal sustainability challenges, which most often manifest themselves at local levels. The river basin has traditionally been used in water science and management as the chief spatial unit to assess water scarcity and environmental sustainability associated with water use. Therefore, most WF and VW trade studies considering environmental sustainability and/or water scarcity use the river basin as the fundamental spatial boundary of analysis. This shift in the literature also moves the field closer to offering more policy relevant results (*e.g.,* water footprint caps per river basin; *Hoekstra and Wiedmann* 2014).

Hoekstra et al. (2012) first assessed water scarcity at a global scale using the river basin as the foundational unit of analysis. They found that 201 of the 405 basins evaluated worldwide face severe water scarcity at least one month per year due to human consumption and climate patterns. *Wang and Zimmerman* (2016) quantified the impacts of VW trade on water use and stress at both the national and basin scale (over 12,000 basins were analyzed). Their study showed that some of the world's most water stressed basins would require 10-80% more renewable water resources if not for virtual water trade. Although the general findings and methodological advances of these studies are noteworthy, results for individual basins should be interpreted with great care since several assumptions were needed to disaggregate national level statistics down to the basin level. Moreover, monthly water scarcity was determined without considering infrastructure in place to buffer against intra-annual water variability. For water footprint assessments and studies of virtual water trade to provide meaningful and actionable findings they must be presented at the scale useful for water governance. This dissertation makes progress in this regard by providing results along specific water resource boundaries (*i.e.,* aquifers) and at county and city delineations. In fact, we were the first to analyze the WF and VW trade along aquifer boundaries (*Marston et al.*, 2015). We avoid many of the assumptions (and the corresponding uncertainty) of other studies that downscale national water use statistics to finer resolutions (*e.g., Wang and Zimmerman* 2016; *Hoekstra and Mekonnen* 2012) by leveraging high-resolution, empirically-based data at the point, county, and sub-state scales.

Insufficient data often leads to several assumptions in WF and VW trade studies. These assumptions may result in significant, yet difficult to quantify, uncertainty in WF and VW trade estimates. Moreover, WF and VW studies often employ datasets from different spatial scales, adding further uncertainty in estimates. The work presented in this dissertation is not immune to the issues of data scarcity, scale, and uncertainty prevalent within the field. We partially address the uncertainty within our results by comparing our findings against other studies and performing sensitivity analysis of critical and highly variable parameters.

Industry and commodity specification

Most WF and VW trade studies have only focused on the agricultural sector since it is responsible for over 90% of consumptive water use globally. Top-down approaches quantify one or two WF values for the entire agriculture sector, but the more common bottom-up (or process-based) approaches estimate water use of each crop and livestock animal. In recent years, there has been growing interest in calculating WF and VW trade of other sectors of the economy. Limited data availability and significant variability between sectors and locations has hindered progress in this area, however.

Estimates of water use across the entire economy typically employ a top-down approach, using environmentally extended MRIO tables and other regional data to allocate water use to each economic sector. Estimates using this approach are almost exclusively in China (*e.g., Guan and Hubacek* 2007; *Shi and Zhan* 2015; *Zhi et al.* 2014; *Dong et al.* 2014; *Zhao et al.* 2010; *Feng et al.* 2012; *Zhang and Anadon* 2014) since the country produces MRIO tables needed for this type of analysis. The direct water footprints provided by these studies represent anywhere from 6 to 40 sectors, with agriculture represented by one or two of these sectors.

Mekonnen and Hoekstra (2011a) provide global coverage of industrial water use at 5' x 5' resolution and their work provides the basis for several subsequent papers (*e.g., Hoekstra and Mekonnen* 2012; *Lenzen et al.* 2013; *Schyns et al.* 2015). Yet, *Mekonnen and Hoekstra* (2011a) make several assumptions that raise serious concerns about the accuracy of their estimates. First, all industrial water uses are lumped into one broad category, glossing over the significant variability in water use among sectors. Second, the authors arbitrarily assumed a uniform and global industrial water consumption coefficient of 5%. Lastly, subnational differences in water use due to water prices, scarcity, technology, and conservation are ignored. Instead, national level statistics were disaggregated to 5' x 5' grid cells based on population.

Wang and Zimmerman (2016) introduce a hybrid approach that seems to combine the greater sectoral detail of process-based studies with the top-down I/O-based analysis, which captures intermediate and final consumption of commodities and the water embedded within them throughout the entire supply chain. This work provides greater sectoral specification (45 non-agriculture sectors are assessed) and uses different consumption coefficients for various industries (still, the same values are used for entire continents and/or multiple sectors). Nonetheless, *Wang and Zimmerman* (2016) rely on national estimates of industrial water use, which they then downscale to the basin level based on water use proxy variables, such as population, irrigated area, nighttime lights, and electric power generation.

Blackhurst et al. (2010) offers the greatest sectoral specification of any study to date. The authors use direct requirement coefficients related to I/O tables to estimate water withdrawals and intensities for 428 sectors in the United States. This work provides estimates at the national level, ignoring local differences in water use and assuming a uniform water price across the nation. Furthermore, water use is reported in terms of water withdrawals, not water consumption as standard in the WF literature.

In our research, we match the sectoral and crop level detail of *Blackhurst et al.* (2010) and *Mekonnen and Hoekstra* (2010) (both representing the state of the art, respectively), while overcoming many of their shortcomings. Unlike previous work, this study does not rely on global maps of crop area or irrigation but uses county level census data to estimate agriculture production and irrigated area. By leveraging county and sub-state data on economic activity and water withdrawals, we estimate direct water footprints of production for over 500 goods and economic sectors at the state, sub-state, and/or county scale. Furthermore, water footprints of non-agriculture sectors are bounded by locally reported water withdrawals and transfers and estimated by local economic activity and water use coefficients. In this way, we improve upon previous approaches and reduce some of the uncertainty in WF and VW trade estimates attributable to these studies' critical downscaling assumptions.

Temporal resolution

The current literature has not given considerable attention to the inter- and intra-annual variability of VW transfers and WF. A finer temporal resolution can permit exploration of changes in water consumption within a given year but also under precipitous (*e.g.,* export bans, drought) and gradual (*e.g.,* population growth, technological improvements) changes across several years. Furthermore, inter- and intra-annual variations can reveal sustainability challenges that may not be evident when evaluating average values. Authors have generally removed temporal variability by assuming average climate, production, and trade patterns over a 5- or 10-year period.

Although sparse, some recent studies have begun to examine the temporal dynamics of VW transfers and WF. For instance, several authors investigated the temporal evolution of the global VW trade network (*Carr et al.*, 2012; *Dalin et al.*, 2012; *Clark et al.*, 2015; *Roson and Sartori*, 2015). These inter-annual global studies revealed that VW trade more than doubled between 1986–2007, demonstrating the increasing connection between international trade and local water use (*Dalin et al.*, 2012). *Dalin and Conway* (2016) quantify international VW trade from southern Africa from 1986–2011. Their study reveals how socio-economic change and climatic variability in southern Africa propagated through the global food and VW trade network. Basin level WFA in Spain (*Aldaya and Llamas*, 2008; *Aldaya et al.*, 2010; *Dumont et al.*, 2013), Cyprus (*Zoumides et al.*, 2014) and China (*Zhuo et al.*, 2016) show how climate, agricultural production, and trade shape water use in a basin over time.

The third chapter of this dissertation reveals a severe drought's impact on agriculture water use and consumer's indirect water dependencies over time. A particularly novel aspect of this work is that it demonstrates inter-annual shifts in consumer behavior (represented through changes in trade patterns) and farmer decision making (represented by irrigation and cropping decisions). These important findings would be smoothed over if we represented long-term average values as is typical in the literature.

1.2.2 Physical water transfers and reallocation

Globally, the volume of VW transfers dwarfs those of physical water transfers. However, at a local level physical water transfers, or the reallocation of water between places and/or uses, can play a significant role in meeting local water requirements. The fifth chapter of this dissertation is related to physical water transfers, thereby providing a comprehensive assessment of how society depends on and shapes water resources (refer to Fig 1.2).

Researchers (*Molle and Berkoff*, 2006; *Hadjigeorgalis*, 2009), practitioners (*Johnson et al.*, 1990), and politicians (*Committee on Western Water Management*, 2012) have proposed water reallocation as an adaptive water management strategy that can reduce the economic, social, and environmental harm caused by water scarcity. The reallocation of water has resolved conflict among, and balanced the needs of, multiple water users while improving local and regional economic robustness (*Zhu et al.*, 2015; *Rosegrant and Binswanger*, 1994). It has been used to serve multiple purposes, such as improving water quality and ecosystems (*Debaere et al.*, 2014), directly meeting water demands (*Palomo-Hierro et al.*, 2015), enhancing system flexibility and reliability (*Molle and Berkoff*, 2009), and decreasing water supply cost (*Lund and Israel*, 1995). When current supplies are inadequate and further source development is infeasible, reallocation has been shown to be one of the most cost-effective means of supplying water to the highest priority users (*Bathia et al.*, 1995; *Gomez et al.*, 2004) and, in some cases, can reduce water shortage vulnerabilities by diversifying users' water sources (*Kasprzyk et al.*, 2009). The cost effectiveness of water reallocation was demonstrated by (*Firoozi and Merrifield*, 2003) who used a theoretical model that showed a water portfolio including water reallocation could delay the construction of costly reservoirs.

Despite the cited benefits of water reallocation, it has not been as broadly implemented or effective as expected (*Eden et al.*, 2008; *Giannoccaro et al.*, 2013). Wide-spread and effective application of water reallocation is still fraught with major obstacles due to complex and poorly understood human-water interactions and feedbacks. These barriers are often interrelated and can only be overcome through a systems approach due to the complex interactions between nature and society, which cannot be easily disentangled. Yet, the current literature reduces problems to fit within disciplinary silos and ignores the complex couplings and feedbacks often seen in examples of water reallocation. The current body of literature was found to primarily focus on the institutional, economic, and social obstructions to water reallocation, with little consideration to the natural and physical dimensions of water reallocation. This deficiency in the current literature motivated our investigation into the major barriers to water reallocation and the development of an interdisciplinary research framework to overcome them.

1.3 Research Objectives and Questions

The overarching research question behind this research plan is *how is water use shaped by anthropogenic influences within and beyond the water's hydrologic boundary?* The following four objectives, along with their accompanying research questions, will be the focus of this dissertation. Fig 1.2 provides a schematic overview of how these four objectives relate together.

Objective 1 Comprehensively quantify and trace virtual groundwater and food transfers from overexploited aquifers in the US to their destination of final use.

Research Question(s): What locations are most responsible for — and currently most reliant upon — depletion of critical and overexploited aquifers in the US?

Objective 2 Improve understanding of how drought reverberates through the global food system. More specifically, quantify drought impacts to the water footprint of agricultural production and virtual water transfers from California's Central Valley.

Research Question(s): (i) How do agricultural production water footprints in California evolve with drought? (ii) How does drought impact food and virtual water transfers from California? and (iii) How is global demand for California agriculture contributing to unsustainable local water use?

Objective 3 Quantify the direct water footprint of food, energy, services, manufacturing, and mining products produced within the United States at an unparalleled sectoral detail and spatial resolution.

Research Question(s): (i) How much surface water, groundwater, and green water is used to support production of different industries and products across the United

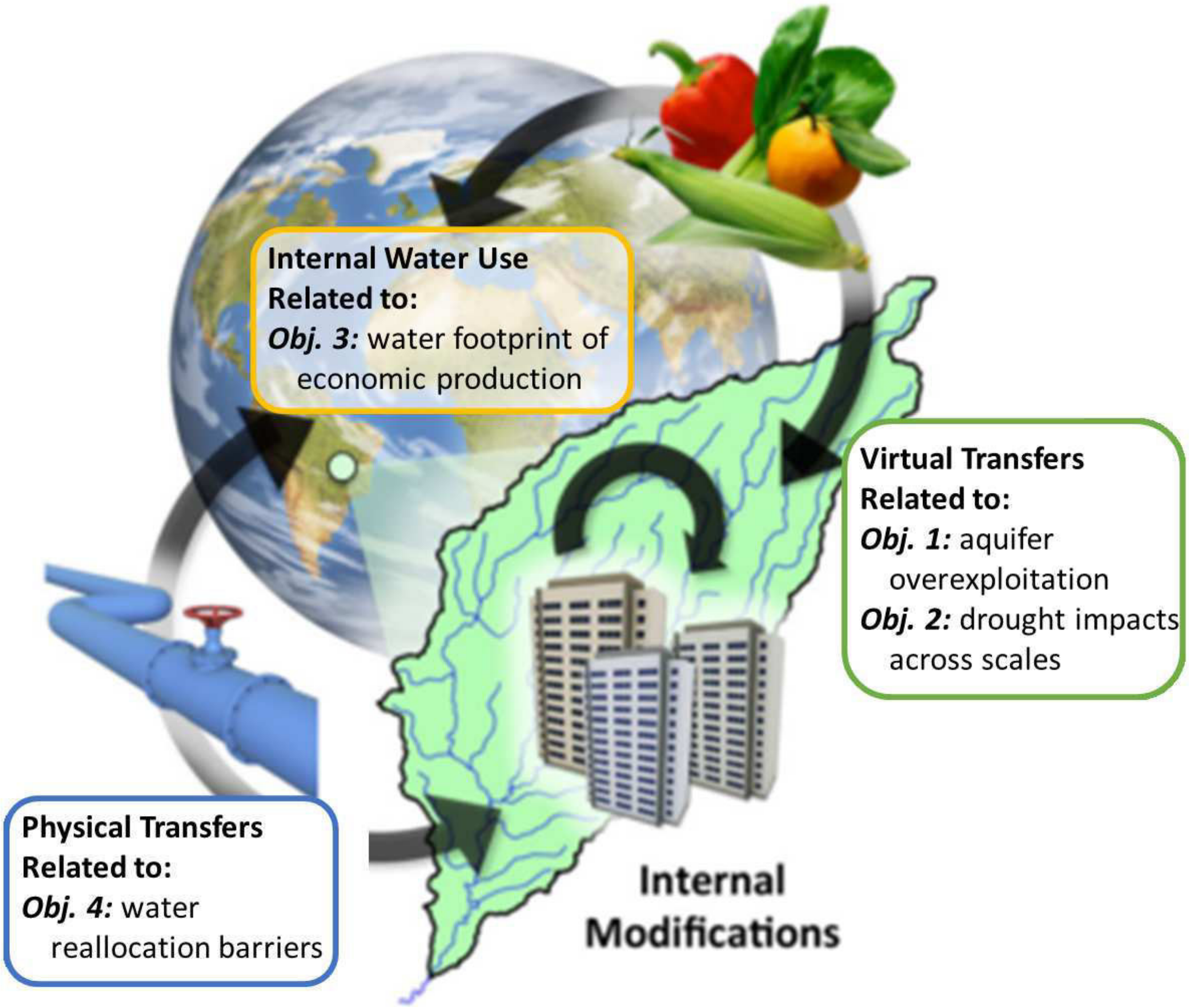

Figure 1.2: Dissertation framework. Collectively, the four objectives of this dissertation highlight how local water use and sustainability are shaped by local and nonlocal anthropogenic forcings via virtual and physical water transfers.

States? (ii) To what degree does spatially explicit and industrial specific PWF improve water use estimates compared to PWF estimates at coarser spatial or industrial resolutions? (iii) How much do PWF vary across different industries and locations?

Objective 4 Derive the major barriers to physical water transfers from an extensive literature review. Propose interdisciplinary framework to overcoming major barriers to physical water transfers.

Research Question(s): (i) What are the major obstacles to physical water transfers? (ii) Why do these barriers still exist and how can they be overcome?

1.4 Research Contributions

The original contributions of this dissertation are as follows:

1. A novel dimension of this research is the fine spatial, temporal, commodity, and water source resolution we provide, which is made possible through empirically-based datasets and our unique methodological approach. Moreover, the incorporation of empirical subnational commodity transfer data differentiates this work from other studies, which typically utilize crude trade models to replicate intra-national trade flows.

2. We are the first to quantitatively demonstrate the role of distant consumer demands on local groundwater sustainability and the fact that aquifer depletion must be considered within its global context.

3. It is shown how local climate shocks reverberate through the global food system, highlighting the importance of complex interactions in the coupled climate-food-water system, and the critical role of local groundwater depletion.

4. We provided the most comprehensive and detailed water footprint assessment of a country to date. Hotspots of United States water consumption, by crop and industry, are identified through improved disaggregation techniques.

5. This work provides a comprehensive and high-resolution dataset of water use requirements of food, energy, services, manufacturing, and mining products produced within the US. This database will be useful for water management and modeling, environmental life cycle assessments, water footprint assessments, benchmarking water use, and demand forecasting and planning.

6. We contribute an interdisciplinary framework to overcome the complex social and natural barriers to water reallocation.

1.5 Dissertation Structure

The chapters of this dissertation are organized as follows:

- In Chapter 2, we comprehensively quantify and track agricultural virtual groundwater transfers from overexploited aquifers of the United States to the major U.S. cities, U.S. states, and countries that are currently most reliant upon them. Much is understood about local food production and groundwater use in these critical aquifer systems. Here, we evaluate the consumption side of the story and determine where these resources are being demanded.

- In Chapter 3, we examine drought impacts to food and virtual water transfers from the Central Valley of California and examine the linkage between distant consumption of virtual water resources and local water impacts. More broadly, this study elucidates how local climate shocks reverberate through the global food system, highlighting the importance of complex interactions in the coupled climate-food-water system, and the critical role of local groundwater depletion.

- In Chapter 4, we estimate the water footprint of production of the United States, the largest producer and consumer of goods and services in the world. This study is the most detailed, comprehensive water footprint analysis of a given country to date. We present direct water footprints of production and product water footprints for around 140 agricultural products and the production of over 375 other goods and services, including energy generation. This study broadly contributes to our understanding of water in the US economy, enables supply chain managers to assess direct and indirect water dependencies, and provides opportunities to reduce water use through benchmarking.

- Chapter 5 explores the potential of water reallocation as an adaptive water management strategy in places of real or institutional water scarcity. By taking a high-level view of the existing body of knowledge, we put forth an interdisciplinary research framework to overcome the primary hindrances to broader implementation of water reallocation. The proposed framework demonstrates how the social sciences, natural sciences, and engineering fields can integrate their unique perspectives so to overcome each of the major barriers impeding wider and more effective water reallocation. We outline some of the specific problems an integrated research approach could be applied to and what the anticipated outcomes would entail.

- Chapter 6 highlights conclusions of this research, notes limitations of our work, and suggest broader implications and paths for future research.

The core chapters of this dissertation either have or will be submitted for publication. Below are the full references:

I. Marston, L., Konar, M., Cai, X., and Troy, T.J. (2015). Virtual groundwater transfers from overexploited aquifers in the United States. *Proc Natl Acad Sci*:201500457. doi:10.1073/pnas.1500457112.

II. Marston, L., and Konar, M. (2017). Drought impacts to water footprints and virtual water transfers of the Central Valley of California. *Water Resources Research.* doi: 10.1002/2016WR020251

III. Marston L., Yufei, A., Konar, M., Mekonnen, M. M., and Hoekstra, A. Y., High resolution production water footprints of the United States. To be submitted.

IV. Marston, L. and Cai, X. (2016) An overview of water reallocation and the barriers to its implementation. *Wiley Interdiscip Rev Water.* doi:10.1002/wat2.1159.

Chapter 2

VIRTUAL GROUNDWATER TRANSFERS FROM OVEREXPLOITED AQUIFERS IN THE UNITED STATES

2.1 Introduction [1]

Globalization has strengthened and expanded connections between socioeconomic systems and distant resources, by enabling consumer demand in one location to be fulfilled with production and resource use in another. The distant interactions between people and places are commonly referred to as 'teleconnections' (*Seto et al.*, 2012), which represent a specific case of the complex interactions that arise between coupled human and natural systems (*Liu et al.*, 2013). These non-local interactions are increasingly widespread and lead to unanticipated outcomes with profound implications for resource consumption and sustainability (*Liu et al.*, 2015). The global food trade system is a clear example of a teleconnected system that connects local resource use with distant consumer demands. Agricultural production is a particularly water-intensive sector of the economy (*Vörösmarty*, 2000; *Hoekstra and Chapagain*, 2011; *Hoff*, 2009), such that trade of agricultural products connects local water use for irrigation to the end consumer of the commodity, in a 'virtual water trade' (*Allan*, 1998; *Hoekstra and Hung*, 2005). In this paper, we seek to understand how distant food demands are linked with non-sustainable local agricultural water use.

Groundwater plays a critical and ubiquitous role in human society (*Konikow and Kendy*, 2005), providing an estimated 36%, 42%, and 27% of global domestic, agricultural, and industrial water uses, respectively (*Döll et al.*, 2012). Population growth, socioeconomic development (*Vörösmarty*, 2000; *Konikow and Kendy*, 2005), and to a lesser extent, climate change (*Vörösmarty*, 2000; *Döll*, 2009), are expected to increase future demand for groundwater resources. Unsustainable groundwater withdrawals will limit future groundwater availability (*Gleick and Palaniappan*, 2010; *Wada et al.*, 2010; *Gleeson et al.*, 2012; *Castle et al.*, 2014), with implications for food security (*Hanjra and Qureshi*, 2010), since

[1] This chapter is published as an article in the Proceedings of the National Academy of Sciences, 2015 (*Marston et al.*, 2015)

approximately 40% of global irrigated agriculture relies upon groundwater. Importantly, approximately 42% of irrigated agriculture in the U.S., one of the largest food producers and the largest exporter globally, depends on groundwater (*Kenny et al.*, 2009). Furthermore, groundwater depletion will affect the ability of urban areas, over half of which are located in water scarce basins (*Richter et al.*, 2013), to meet normal water demands and cope with climate variability, against which groundwater acts as a buffer (*Konikow and Kendy*, 2005).

The Central Valley (CV), High Plains (HP), and Mississippi Embayment (ME) aquifer systems (mapped in Fig. 2.5) enable agricultural production that is critical to local economies and contributes to U.S. and global food security. In 2007, roughly one-fifth of the $300 billion agricultural industry in the United States came from these aquifer regions (*USDA*, 2014; *Scanlon et al.*, 2012). The lands overlying the CV (52,000 km^2), HP (450,000 km^2), and ME (202,000 km^2) make up 8% of U.S. land area, yet comprise 16% of U.S. cropland. More than 17 million people live within the boundaries of these three aquifers. In addition, 25.7% of all U.S. irrigation and livestock withdrawals and 61.1% of all groundwater irrigation and livestock withdrawals come from these three aquifers (*Kenny et al.*, 2009). Despite their importance, these aquifers are being managed unsustainably: 67% of U.S. groundwater depletion from 1900-2008 and 93% of groundwater depletion from 2000-2008 is attributed to these three aquifers (*Konikow*, 2013).

2.2 Methodology

Here, we detail the data used in this study and explain how data sources at different spatial and commodity resolutions were integrated to quantify virtual groundwater transfers from the CV, HP, and ME aquifers.

Water use data

The United States Geological Survey (USGS) publishes a report estimating county-level water use in the United States for years ending in zero and five. Estimates of county-level groundwater and total irrigation were taken from the 2005 report (*Kenny et al.*, 2009). The USGS estimates irrigation withdrawals using information from state and federal crop reporting programs, canal companies, irrigation districts, and incorporated management areas (*Kenny et al.*, 2009).

Agricultural production data

County-level agricultural production data was collected from the United States Department of Agriculture (USDA) 2007 Census of Agriculture (*USDA*, 2014). The USDA sent surveys to nearly all of the recorded 3.2 million farms (defined as producing and selling more than $1,000 of agricultural products annually) across the United States. The response rate for the 2007 Census of Agriculture was 85.2%, with a minimum response rate of 75% for all counties.

Virtual water content data

Virtual water contents (commonly referred to as water footprints) for 126 crops and more than 200 crop derived products were collected from WaterStat (*Mekonnen and Hoekstra*, 2011b). A high resolution, spatially explicit, dynamic water balance model was used to calculate crop water use from 1996-2005. The reported state-level virtual water contents are the average crop water requirement over this time period, taking into account climatic conditions and daily soil water balance within each grid cell. WaterStat reports state-level green, blue, and grey water footprints; we use the blue water footprints for our study.

Domestic transfer data

Data on food transfers [tons] were collected from the Commodity Flow Survey (CFS) for the year 2007 (*US Census Bureau*, 2014a).The CFS data was collected from quarterly surveys and provides data on the origin and final destination of goods, including their weight, value, and mode of transportation. Bilateral food transfer data is provided for 123 CFS areas within the U.S. and for seven agricultural commodity groups (as defined by the Standard Classification of Transported Goods, SCTG). These commodity groups are listed in Table 2.1 and the complete composition of each SCTG category can be found with *US Census Bureau* (2007).

The CFS database provides information on bilateral commodity transfers between CFS Areas. CFS Areas are comprised of Metropolitan Areas and Remainder of States. Metropolitan Areas are delineated by county boundaries based on the size of business activity within the area or the area's importance as a transportation hub. The Remainder of States are the state areas that exclude the Metropolitan Areas. In some cases the Remainder of State is the entire state (*e.g.,* Idaho and Nebraska).

International export data

Port-level export data of agricultural commodities was collected from the foreign trade division of the Census Bureau for the year 2007 (*US Census Bureau*, 2014b). Harbors of the U.S. were spatially linked with CFS areas. Ports within the U.S. that exported more than 1,000 tons of agricultural goods were included within our study. The Census Bureau's port-level records provide data for air and barge transport, which comprises the vast majority of all foreign exports. Exports to Mexico and Canada, however, predominately rely on truck and rail. To capture exports to Mexico and Canada, we matched major overland ports from the Commodity Flow Survey with interstate and railroad maps, in order to trace the path of agricultural goods from the point of production to overland shipments to Mexico and Canada. The quantity of each agricultural commodity group exported to Canada and Mexico by air and barge was subtracted from the total amount sent to each country, as specified by the USDA Global Agricultural Trade System (*USDA*, 2014). The difference was assumed to be the tonnage exported via truck or rail. This export tonnage was distributed amongst the previously identified land ports. This was done proportionally to the incoming agricultural tonnage to the port's corresponding CFS Area (*i.e.,* CFS Areas with greater incoming tonnage are assumed to also export a greater fraction of tonnage sent to either Canada or Mexico). In this way, food transfers were traced from CFS areas overlying the aquifers to U.S. ports, and then to international export destination.

Data integration across spatial resolutions

The data used in this study are provided at multiple spatial scales. USGS water withdrawal data (*Kenny et al.*, 2009) and USDA agricultural production data (*USDA*, 2014) are at the U.S. county scale (*e.g.,* Fig 2.1A). Virtual water content data (*Mekonnen and Hoekstra*, 2011b) is at the U.S. state scale (*e.g.,* Fig 2.1B). International export data of food commodities (*US Census Bureau*, 2014b) is available at the U.S. harbor scale (*e.g.,* Fig 2.1C). Domestic food commodity transfers (*US Census Bureau*, 2014a) are at the CFS Area scale (*e.g.,* Fig 2.1D).

County, state, and port scale data were mapped to the CFS Area scale. We use the CFS Area scale as the primary spatial unit because this is the highest resolution food transfer information available. County-level water use and agricultural production data were scaled to CFS Areas by aggregating the county level data within a CFS Area. State-level virtual water contents were attributed to all CFS Areas within a given state. Each of the 92 ports

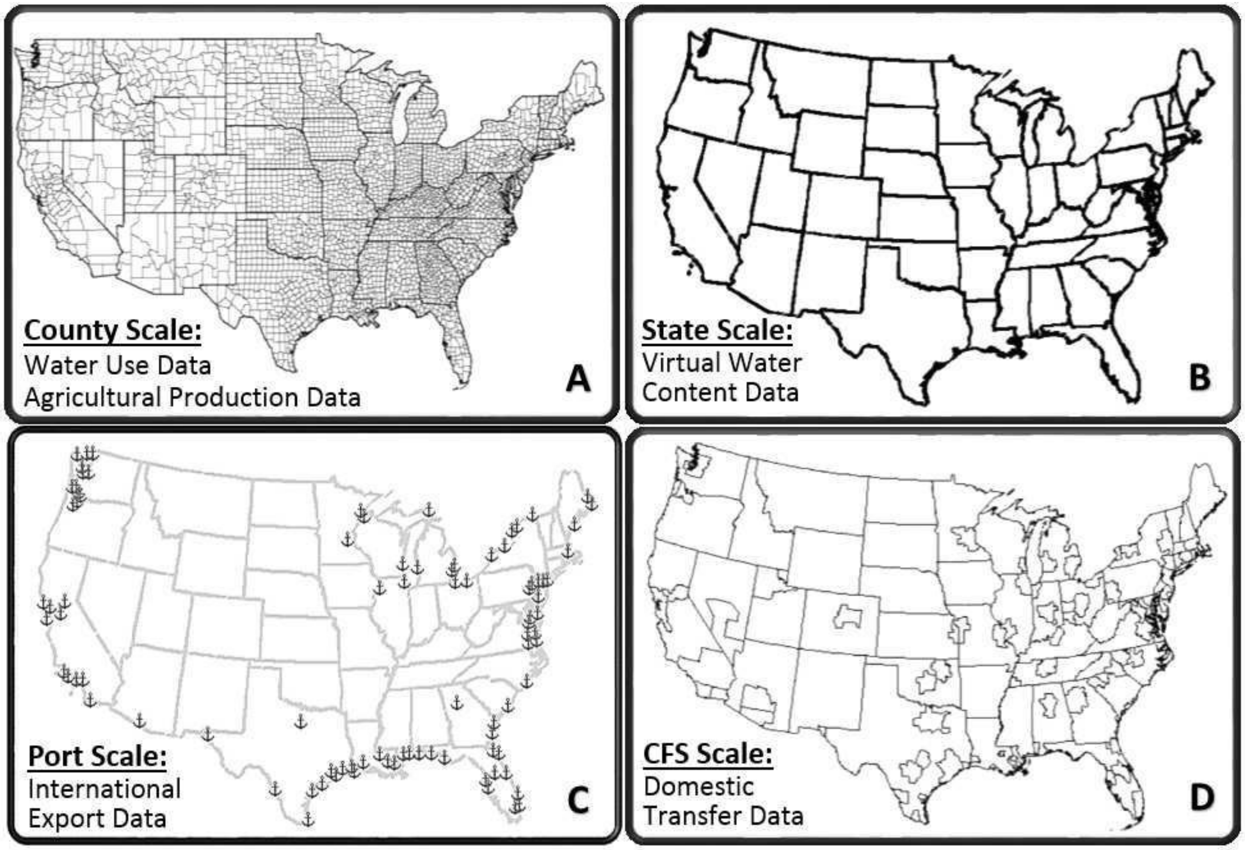

Figure 2.1: Spatial scale of each data source. (A) County: Water use data and agricultural production data; (B) State: Virtual water contents; (C) Port: International export of agricultural commodities; (D) CFS Area: Domestic transfers of food commodities.

of the United States was assigned to the CFS Area within which it is located.

Data integration across commodity resolutions

The data used in this study are provided at multiple commodity resolutions. Agricultural production and virtual water content data are provided for individual agricultural items. However, when tracking the shipments of goods, the U.S. Census Bureau uses the Standard Classification of Transported Goods (SCTG) to group similar products. We use item-specific water use and virtual water content information to weight the transfers of SCTG food commodity groups. Every food commodity that used groundwater was assigned a weight based on the tonnage of the item-specific production data relative to the total production tonnage of the SCTG category within a CFS Area. This assumes that the composition of each SCTG category reflects the agricultural production within a CFS Area. Note that some items were either not produced in the aquifer regions or their production tonnage was negligible.

2.2.1 International virtual groundwater transfers

In this section, we explain how we trace international virtual groundwater transfers. First, agricultural transfers are tracked from all agriculture producing CFS Areas to CFS Areas containing a port. Then, we assume that the export composition matches the composition of transfers coming into the port. For example, if 80% of all cereal coming into the Seattle-Tacoma-Olympia, WA CFS Area is from the HP, we assume that 80% of cereal exports from each of the three ports within the Seattle-Tacoma-Olympia, WA CFS Area originates from the HP. For each commodity group and importing country, the fraction of transfers from each aquifer region is multiplied by the total international export tonnage to get the total exports to each country originating from each aquifer. This tonnage is then multiplied by the corresponding virtual groundwater content (VGC) to arrive at the virtual groundwater transfers to each country for each commodity group.

2.2.2 Determination of cereal supply reliant on overexploited aquifers

Here, we detail how we determine the fraction of each country's cereal supply that is dependent on the CV, HP, and ME aquifers. First, we calculate the fraction of cereal grown using groundwater irrigation, excluding cereal grown without irrigation or grown using strictly surface water irrigation. Next, groundwater-dependent cereal transfers are tracked from the CV, HP, and ME aquifers to CFS Areas containing a port. Cereal transfers dependent on groundwater are assumed to be proportional to the tonnage of cereal production reliant on that aquifer. For example, if 57% of the tonnage of cereals produced in Nebraska used groundwater from the HP, it is assumed that the same percentage of all cereals transferred from Nebraska are reliant on the HP aquifer (*i.e.,* the percentage of transferred cereals that are reliant on groundwater is the same as the percentage of cereals grown that are reliant on groundwater).

For each port, all incoming cereal tonnage dependent on the same aquifer (*i.e.,* either the CV, HP, or ME) was summed. This was then divided by all incoming cereal transfers from across the U.S. to arrive at the fraction of incoming cereal transfers to each port area that are reliant on each aquifer. The aquifer dependent fraction was then multiplied by the cereal tonnage exported from the port to each country. For each country, the sum of all exported cereals reliant on the CV, HP, and ME aquifers that were sent to that country was taken across all U.S. ports, according to the following equation:

$$CT_{A,I} = \sum_{P}(CT_{P,I} * \frac{\sum_{O \in A} CT_{O,P} * GF_O}{\sum_O CT_{O,P}}) \tag{2.1}$$

where CT is cereal transfers [tons] and GF is the fraction of total cereal production dependent on groundwater from a study aquifer. A is a CFS area overlying an aquifer (*i.e.,* either the HP, CV, or ME), I is the international destination of cereal exports, P is a U.S. port, and O is all origin CFS Areas.

2.2.3 Virtual groundwater content estimates

The virtual water content (VWC) refers to the total water required for crop evapotranspiration and incorporation within the product divided by the crop yield (Equation 2.2).

$$VWC = \frac{ET + IW}{CW} \tag{2.2}$$

where ET refers to crop evapotranspiration [m^3], IW refers to water incorporated within the harvested crop [m^3], and CW refers to crop weight [tons]. The VWC is comprised of two components: green and blue VWC, which correspond to rainfall and surface and/or groundwater, respectively. The ET and IW of the green VWC is attributed to rain water, whereas the ET and IW of the blue VWC ($BVWC$) is from surface water and/or groundwater sources. This study focuses on the unsustainable groundwater component of $BVWC$, the virtual groundwater content (VGC).

SCTG 02 and 03: State-level estimates of $BVWC$ for items within SCTG commodity groups 02 and 03 were collected from *Mekonnen and Hoekstra* (2011b). Note that *Mekonnen and Hoekstra* (2011b) presents conservative estimates of $BVWC$ since evapotranspiration only is considered and return flows are excluded. County-level irrigation withdrawals from the USGS in the year 2005 were utilized to calculate the fraction of irrigation supplies from groundwater (GF) for each irrigated crop produced within aquifer boundaries. County-level production data (*USDA*, 2014) was used to determine a production-weighted average VGC across items within an SCTG commodity group:

$$VGC_{SCTG,CFS} = \frac{\sum_c BVWC_{C,CFS} * GF_{CFS} * P_{C,CFS}}{P_{SCTG,CFS}} \tag{2.3}$$

where VGC refers to virtual groundwater content, $BVWC$ refers to blue virtual water content, GF refers to groundwater fraction, and P refers to agricultural production [tons]. Subscripts C, $SCTG$, and CFS refers to commodity item within SCTG commodity group, SCTG commodity group, and CFS area, respectively.

SCTG 06 and 07: All methods follow those of SCTG commodity groups 02 and 03, but now production-based weights are modified. Categories SCTG 06 and 07 are comprised of processed and milled goods, but the production volumes of the individual products are not available. However, the product composition of SCTG 06 and 07 can be estimated based on the production of the primary crops within the CFS area that are used in the production of the processed goods. To avoid over-estimating exports of virtual groundwater embodied in SCTG 06 and 07, the processed goods that require primary crops not produced within the CFS area are not given weight in the SCTG category's overall VGC, while products whose primary inputs are crops widely grown in CFS area are weighted according to production data. This approach discounts the exports of processed commodities whose primary crops are not grown locally.

SCTG 04: The feed VGC was calculated in conjunction with the livestock and meat VGC. Feed requirements per head of the primary livestock raised within the aquifer areas (*i.e.,* cattle, equine, goats, hogs, sheep, chickens (layers and broilers), turkeys, pheasants, and quail) were collected from *Chapagain and Hoekstra* (2003). The number of livestock head produced and sold in 2007 was collected from *USDA* (2014). The feed requirement per head of livestock was multiplied by the number of head sold to arrive at feed requirements. The amount of feed imported into the CFS area was subtracted from the CFS area's feed requirement to get the total feed that needed to be produced within the CFS area. The vast majority of required feed (97%) was produced locally. It was assumed that SCTG 04 consists of the same feed composition as the feed required for livestock inside the CFS area. To determine VGC of feed, the required tonnage of each crop within the feed composition was multiplied by its VGC and then summed to get the total volume of virtual groundwater of feed. The total virtual groundwater volume attributed to feed was divided by the total tonnage of the feed crops to get the feed VGC for each CFS area.

SCTG 01 and 05: The volume of virtual groundwater of the required feed was divided by the total tonnage of livestock to get the feed component of the VGC of animal production within each CFS area. The required water for drinking and for servicing of livestock (from *Chapagain and Hoekstra* 2003) was multiplied by the fraction that was taken from groundwater (*Kenny et al.*, 2009) to get the amount of groundwater used per head of each animal. This was then multiplied by the number of each animal sold in 2007 (*USDA*, 2014) to get the volume of groundwater required for drinking and servicing for each animal type. The required groundwater volume for each animal type was summed and then divided by the total animal tonnage to get the component of the animal production VGC within each CFS area attributed to drinking and servicing. This was added to the corresponding VGC of feed production to arrive at the total VGC for all livestock sold from within the CFS area boundaries. The $VGCs$ differ between SCTG 01 and SCTG 05 because the virtual groundwater volume is divided by the live animal tonnage for SCTG 01, whereas it is divided by the edible fraction (per *Gerbens-Leenes et al.* 2011) for SCTG 05. In this way, the VGC corresponding to SCTG 01 and SCTG 05 are weighted by the tonnage sold or butchered of each animal type within the CFS area.

$$\begin{aligned} VGC_{SCTG,CFS} = & \frac{(FR_{CFS} - FI_{CFS}) * VGC_{SCTG04,CFS}}{P_{SCTG,CFS}} \\ & + \frac{\sum_C WR_{C,CFS} * GF_{CFS} * P_{C,CFS}}{P_{SCTG,CFS}} \\ & + \frac{\sum_C SR_{C,CFS} * GF_{CFS} * P_{C,CFS}}{P_{SCTG,CFS}} \end{aligned} \tag{2.4}$$

where FR refers to feed requirement [tons], FI refers to imported feed [tons], WR refers to livestock water requirement [m^3/ton], and SR refers to livestock servicing requirement [m^3/ton]. All other acronyms and subscripts follow those above.

2.2.4 Virtual groundwater transfers

The food transfer data was multiplied by the virtual groundwater content to arrive at virtual groundwater transfers:

$$VGT_{SCTG,O,D} = VGC_{SCTG,O} * FT_{SCTG,O,D} \tag{2.5}$$

where VGT indicates virtual groundwater transfer [m^3], VGC indicates virtual groundwater content [m^3/ton], FT indicates food transfers [tons]. Subscripts $SCTG$, O, and D indicates food commodity group, origin CFS area, and destination, respectively. In this way, VGT volumes are tracked from aquifer areas to their final destination.

2.3 Results and Discussion

2.3.1 Total virtual groundwater transfers

According to the U.S. Geological Survey (*Kenny et al.*, 2009), irrigation withdrawals from the HP, ME, and CV systems totaled 23.38 km^3, 13.59 km^3, and 9.34 km^3, respectively (refer to the size of the circles in Fig. 2.2). Of these agricultural withdrawals, approximately 27% is lost to irrigation inefficiencies and return flows, while the rest is virtually embodied within crops and livestock (*i.e.,* directly used for crop growth or livestock production). The groundwater footprint of a commodity is the volume of water that is virtually embodied throughout the production process of that commodity, which is also referred to as the virtual groundwater content (*i.e.,* the volume of groundwater per commodity unit, VGC; refer to Methods). Note that VGC varies by commodity and aquifer (see Table 2.1). The total volume of virtual groundwater transfers (VGT; refer to Methods) across all aquifers is comparable to the capacity of Lake Mead (35.7 km^3), the largest surface reservoir in the United States. Between 45% (HP) and 58% (CV) of agricultural groundwater withdrawals that are virtually transferred are not utilized within the States overlying the aquifers, but transferred either elsewhere within the U.S. or exported abroad. The vast majority of VGT remains within the U.S., with 9% exported abroad.

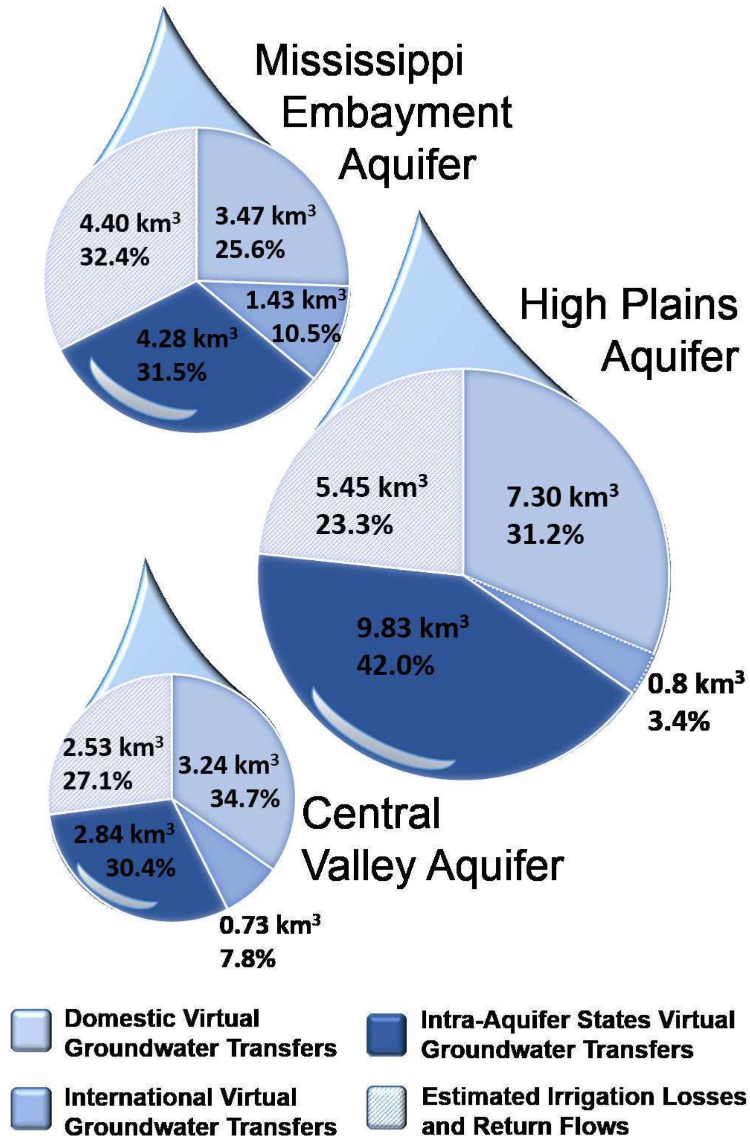

Figure 2.2: Consumption of overexploited aquifers in the United States. The size of each circle indicates the volume of groundwater withdrawals for agriculture from each aquifer as given by the U.S. Geological Survey. In 2005, 23.38 km^3, 13.59 km^3, and 9.34 km^3 of groundwater was withdrawn for irrigation from the High Plains, Mississippi Embayment, and Central Valley aquifer systems, respectively. Each circle shows the proportion of groundwater withdrawals that go to irrigation losses and return flows, intra-aquifer state transfers, domestic transfers, and international exports. The Mississippi Embayment aquifer ships the largest proportion of its virtual groundwater abroad (10.5%), compared with 3.4% in the High Plains and 7.8% in the Central Valley.

Table 2.1: Virtual groundwater content (VGC) estimates for the SCTG food commodity groups in each of the overexploited aquifers of the United States. Units are cubic meters of groundwater consumed per metric ton of production.

Standard Classification of Transported Goods (SCTG)		Central Valley	High Plains	Mississippi Embayment
SCTG Code	SCTG Name	m^3/ton	m^3/ton	m^3/ton
01	Animals & Fish (live)	1,570.7	303.7	9.0
02	Cereal Grains (including seed)	271.5	68.4	129.4
03	Agricultural Products Except for Animal Feed (other)	66.2	63.9	76.5
04	Animal Feed & Products of Animal Origin	134.7	27.9	1.3
05	Meat, Fish, and Seafood & Their Preparations	1,984.7	367.4	9.9
06	Milled Grain Products & Preparations, & Bakery Products	107.7	47.1	369.7
07	Other Prepared Foodstuffs, & Fats & Oils	246.5	164.7	57.2

2.3.2 Domestic virtual groundwater transfers

The annual volume of VGT between States overlying the aquifers is 16.9 km^3, which is comparable to the annual average flow volume of the Colorado River into Lake Mead (approximately 18 km^3/y). There are 7.30 km^3, 3.24 km^3, and 3.47 km^3 transferred out of the HP, CV, and ME aquifer boundaries, respectively, which remains within the United States. This equates to four (CV, ME) to ten (HP) times more groundwater being transferred out of the aquifer regions to other domestic locations than is being withdrawn for local municipal and industrial purposes combined.

Urban areas are key recipients of VGT. Cities in California receive the largest share of domestic VGT: Los Angeles and San Francisco-Oakland receive 12.7% of all VGT. San Francisco-Oakland, Los Angeles, and Sacramento are the recipients of 40.5% of all VGT from the CV. To put the transfer volumes in perspective, around 3.4 km^3 of water was physically transferred in 2007 from the Sierra Nevada Mountains to Los Angeles via the Los Angeles Aqueduct system; in that same year, 1.76 km^3 of groundwater from the CV aquifer system was virtually transferred to Los Angeles solely in agricultural commodities.

The CV and ME aquifer systems both have one or two metropolitan areas that receive relatively large shares of VGT (refer to Fig. 2.3). However, this is not the case for the HP aquifer system where the transfers are more dispersed. This is likely because the HP aquifer system extends across much of the central U.S., where there is more than one or two major cities or ports that would be viable principal consumption or transfer locations. This may also be due to the fact that cereals comprise a large share of agricultural production in the HP, which can be stored and widely distributed, compared with the large quantify of fresh items produced in the CV, such as vegetables and meat (refer to Fig. 2.4).

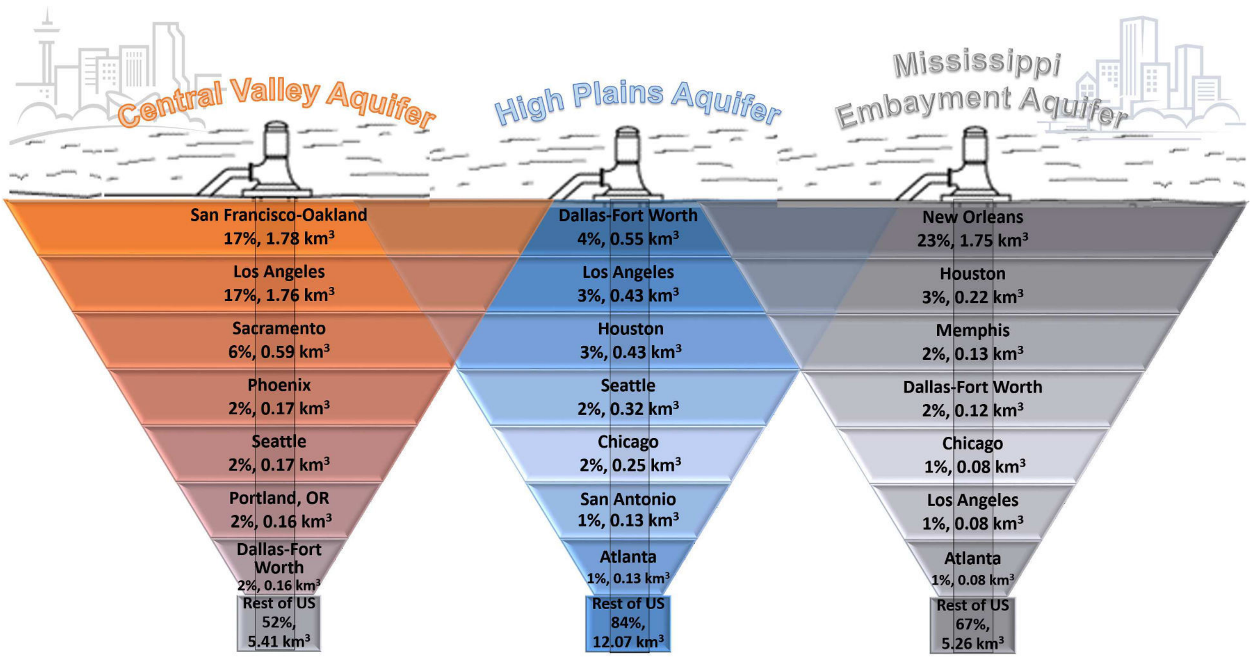

Figure 2.3: Ranking of U.S. Metropolitan areas that currently most rely on virtual groundwater from each of the Central Valley, High Plains, and Mississippi Embayment aquifers. The total volume virtually transferred to each U.S. Metropolitan area is provided, as well as the fraction of groundwater withdrawals from each aquifer system that this represents. Note that the triangles are provided for graphic representation only and are not scaled according to size. Importantly, cities in the Western U.S. are relatively reliant upon virtual groundwater transfers from the Central Valley aquifer, while virtual groundwater transfers from High Plains aquifer tend to be more dispersed across the U.S., as they are with the Mississippi Embayment aquifer, with the exception of the major shipping port of New Orleans.

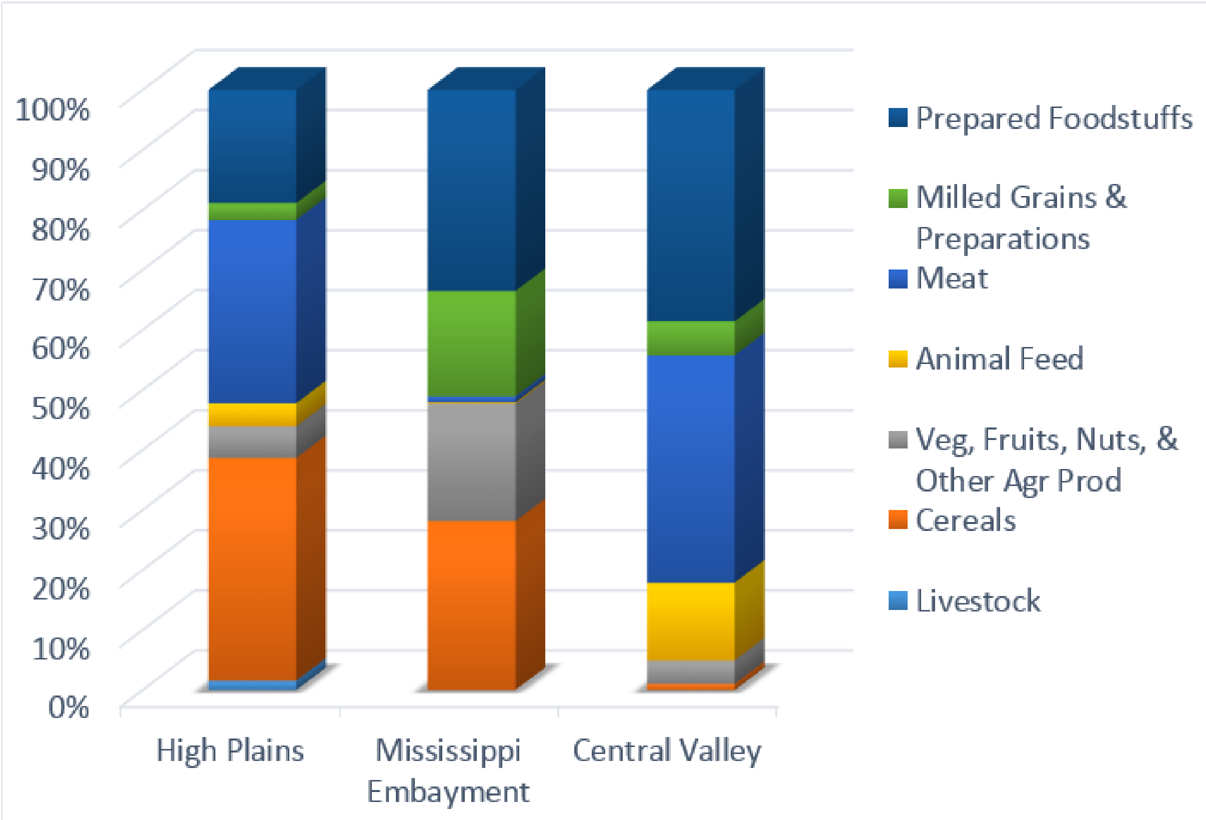

Figure 2.4: Percentage of virtual groundwater transfers attributed to each of the seven commodity groups in the High Plains, Mississippi Embayment, and Central Valley aquifers.

With increased intersectoral demands for water, economic development, and climate change, water is projected to become more scarce in many locations (*Gleick*, 2003). Conflicts have arisen between rural and urban areas, U.S. states, and countries regarding renewable surface water allocations. Reallocation of water from rural agriculture to urban uses is a politically charged issue, but a growing trend nonetheless (*Falkenmark and Molden*, 2008; *Molle and Berkoff*, 2009). These results demonstrate that water use in rural areas already largely serves urban areas by providing food (*i.e.,* virtual water flows to cities through food commodities).

Fig. 2.5 highlights that domestic VGT are predominantly to population centers, wealthy areas, and between areas that are close in distance, as we would expect from the gravity model of trade (*Tinbergen*, 1962). Since large volumes of water are virtually transferred within the U.S., the socioeconomic and environmental challenges in both sending and receiving locations should be considered in future water supply discussions. Going forward, stakeholders may want to evaluate teleconnections that impact their local water balance, considering not only physical water allocations, but how water relates to the broader economy and virtual water transfers between sectors and locations.

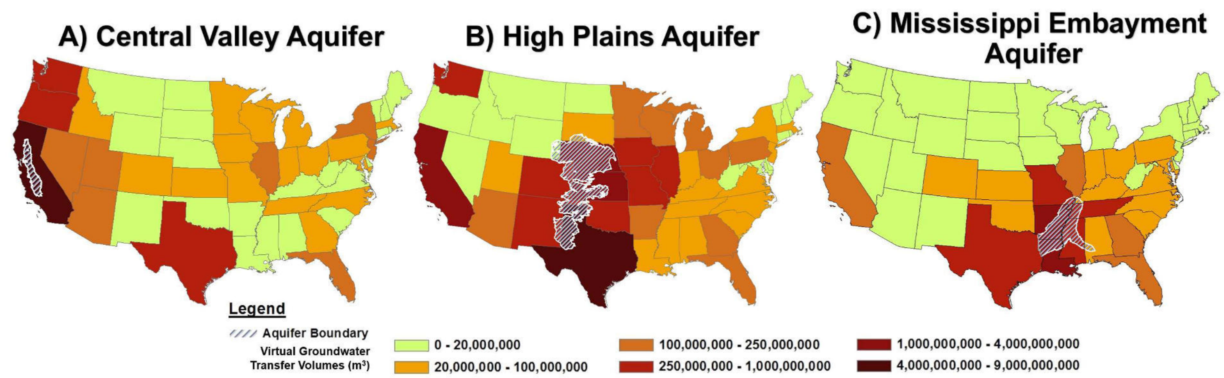

Figure 2.5: Maps of overexploited aquifers in the U.S.: (A) Central Valley, (B) High Plains, and (C) Mississippi Embayment. The areal extent of the aquifers are shown with white highlighting. The U.S. states are shaded to indicate the volume of virtual groundwater transferred from each overexploited aquifer. Darker shades highlight more virtual groundwater transfers into the U.S. State.

Much is understood about local food production and groundwater use in the HP, CV, and ME aquifer systems. It is now imperative to begin to evaluate the consumption side of the story and determine where these resources are being demanded if we are to better understand opportunities to slow their overexploitation (*Zhao et al.*, 2015). To this end, we comprehensively quantify and trace virtual groundwater transfers from these aquifers to their destination of final use. To our knowledge, this is the first time this has been done and represents an important first step in the evaluation of consumption flows of critical groundwater resources. In this paper, we utilize high-resolution empirical data on domestic food transfers within the United States in 2007 (*US Census Bureau*, 2014a; *Lin et al.*, 2014) and link this with port-level data on international exports (*US Census Bureau*, 2014b). Additionally, we employ national statistics on agricultural production (*USDA*, 2014) and irrigation (*Kenny et al.*, 2009) and modeled estimates of virtual water content (*Mekonnen and Hoekstra*, 2011b; *Gerbens-Leenes et al.*, 2011) to quantify virtual transfers of critical groundwater resources (refer to Methods). This approach enables us to identify the locations that are most responsible for – and currently most reliant upon – depletion of the HP, CV, and ME aquifers.

Nearly 9% of domestic VGT are to ports. This highlights the importance of food production for international export, with some states (*e.g.,* Louisiana and Washington) exporting over half of all incoming VGTs. California, Texas, and Arkansas have the largest VGTs; the large transfer volumes of Arkansas are primarily due to large intrastate transfers of rice, which has an extremely high groundwater footprint and is widely produced in Arkansas. Moreover, 74% of all VGTs from the ME aquifer system originate from Arkansas and 41% of Arkansas transfers (3.21 km^3) are transported within the state for consumption, processing, and/or storage. Not coincidentally, Arkansas has seen much greater groundwater level declines than other states overlying the ME (*Clark et al.*, 2011).

Virtual groundwater transfers from Texas and Kansas are of particular interest since they are the origin of the greatest overexploitation of groundwater resources in the High Plains. Transfer volumes from Kansas were 5.66 km^3 (28.8% of HP transfers), while Texas had VGTs of 4.38 km^3 (22.3% of HP transfers). The groundwater embodied within the trade of cereals, meat, and prepared foodstuffs make up the vast majority of the groundwater transfers from these overexploited areas. Groundwater embodied within traded corn makes up the largest fraction of cereal VGT (96% KS, 71% TX), beef makes up the largest fraction of meat VGT (98% KS, 71% TX), and dairy products make up the largest fraction of prepared foodstuffs (57% KS, 85% TX).

The Central Valley is often referred to as the 'fruit and vegetable basket of the world' because it grows and exports an abundance of fruit and vegetables. Interestingly, we find that a relatively small fraction (*i.e.,* 4%, refer to Fig. 2.4) of VGT from the CV aquifer is due to trade of fresh produce. This can be partially explained by three factors: First, amongst the Standard Classification of Transported Goods (SCTG) (*US Census Bureau*, 2007) seven food commodity categories, the one containing vegetables, fruits, and nuts (SCTG 03, refer to Table 2.1) has the smallest groundwater footprint. For example, the VGC of the vegetable commodity group is only 3% as much as the meat commodity group (SCTG 05). Secondly, although the CV area grows more vegetables, fruits, and nuts than other areas, it also produces and trades much of the other food commodities as well (*e.g.,* six and ten million more tons of SCTG 04 and SCTG 07 are traded than SCTG 03, respectively). Third, some produce and nuts are not represented in SCTG 03 because they are processed and included within the prepared foodstuffs trade category (SCTG 07), which, along with dairy products, make up the bulk of this category and account for 39% of total VGT from the aquifer.

Meat products comprise 4%, 10%, and 13% of traded agricultural tonnage from the CV, HP, and ME, respectively (Fig. 2.4); however, 38% and 31% of the total VGTs from the CV and HP are derived from meat products, respectively, while the ME only has 1% derived from meat products. The high VGC of feed and high proportion of cattle in the CV and HP (beef is 97% and 77% of meat VGT, respectively) are the primary drivers of the variances between the tonnage fraction of meat and the virtual groundwater fraction of meat. Only 30% of VGT associated with meat from the ME are from beef, with the majority of the VGT of meat comprised of poultry and fowl, which requires approximately 11 times less water for production than beef; furthermore, the VGC of the feed is significantly smaller in the ME region than it is in the CV and HP.

Domestic food security in the United States is heavily reliant upon the unsustainable extraction of groundwater in the HP, CV, and ME aquifer systems: the cereals produced by these aquifers amounts to 18.5% of the domestic cereal supply (refer to Table 2.2). These groundwater resources are especially critical for agricultural production during times of drought, particularly in California (*Scanlon et al.*, 2012). This buffer (*i.e.,* option) value of groundwater (*Qureshi et al.*, 2012) is not currently accounted for, but will be increasingly important under a more variable future climate (*Taylor et al.*, 2013) with increased irrigation demands (*Zhang and Anadon*, 2013). Additionally, these aquifers provide a trade advantage to the U.S., accounting for 8.6% of U.S. cereal exports (see Table 2.2). To protect U.S. food security and trade interests, domestic policy makers may want to consider slowing groundwater depletion by implementing policies that recognize the full value of groundwater, such as private water markets (*Qureshi et al.*, 2012), which may encourage technology adoption.

Table 2.2: Countries that are most reliant on cereals produced with groundwater from the Central Valley, High Plains, and Mississippi Embayment aquifers. Domestic cereal supply and total cereal import data [tons] were collected from the FAO (*Food and Agriculture Organization*, 2014). Countries are ranked in descending order by the fraction of their domestic cereal supply that originates from these three aquifers (column 4). Island nations that import more cereal than is required for their domestic supply have been excluded (*i.e.,* St Vincent and the Grenadines, Trinidad and Tobago, Grenada, Jamaica, and Barbados).

(1000 tons)	**Domestic Cereal Supply**	**Total Cereal Imports**	**Aquifer Fraction of Domestic Cereal Supply**	**Aquifer Fraction of Cereal Imports**	**Aquifer Fraction of Cereal Imports from USA**
United States	303,016	-	18.5%	-	-
Taiwan	7,268	6,336	10.0%	11.5%	13.4%
Japan	32,961	26,425	9.2%	11.4%	15.4%
Panama	863	649	9.1%	12.1%	13.4%
Costa Rica	1,153	1,133	7.9%	8.0%	8.9%
Dominican Republic	2,180	1,714	5.8%	7.4%	8.3%
Colombia	8,396	5,206	4.5%	7.2%	9.6%
South Korea	16,401	12,660	4.4%	5.7%	12.0%
Honduras	1,388	663	3.9%	8.2%	8.2%
Israel	3,436	3,007	3.4%	3.9%	8.9%
Ecuador	3,205	1,188	3.0%	8.2%	15.1%
Syria	6,746	1,845	2.9%	10.8%	12.0%
World Total	2,120,603	345,753	2.7%	2.5%	8.6%

2.3.3 International virtual groundwater transfers

Unsustainable 'blue water' (*i.e.,* fresh surface water and groundwater *Falkenmark and Molden* 2008) sources are estimated to comprise between 32% and 52% (estimated from *Hanasaki et al.* 2010; *Vorosmarty et al.* 2005; *Rost et al.* 2008) of the global blue water footprint of agricultural production. However, the overexploited U.S. aquifers comprise 35% of the U.S. blue water footprint for agricultural production. Two-fifths of global virtual blue water exports from agriculture is attributed to unsustainable water sources (calculated from *Hanasaki et al.* 2010). However, only 13% of U.S. virtual blue water agriculture exports are from the HP, CV, and ME aquifers. Therefore, the U.S. relies less on nonrenewable water sources for agriculture production and international export of agricultural goods than the rest of the world as a whole.

Fig. 2.6 shows the major international transfers of virtual groundwater from the HP, CV, and ME aquifers. Asia is the top importer of virtual groundwater from all three aquifers. Of all international VGT, approximately half goes to Asia. This finding parallels previous studies that show Asia as the principal importer of U.S. agricultural goods and virtual water (*Gerbens-Leenes et al.*, 2011; *Mekonnen and Hoekstra*, 2011b; *Konar et al.*, 2011). Reduced agricultural production due to aquifer depletion or policies restraining groundwater withdrawals should be of particular interest to Taiwan, Japan, Panama, and Syria: they all depend on the three aquifers of this study for over 10% of their total cereal imports (refer to Table 2.2).

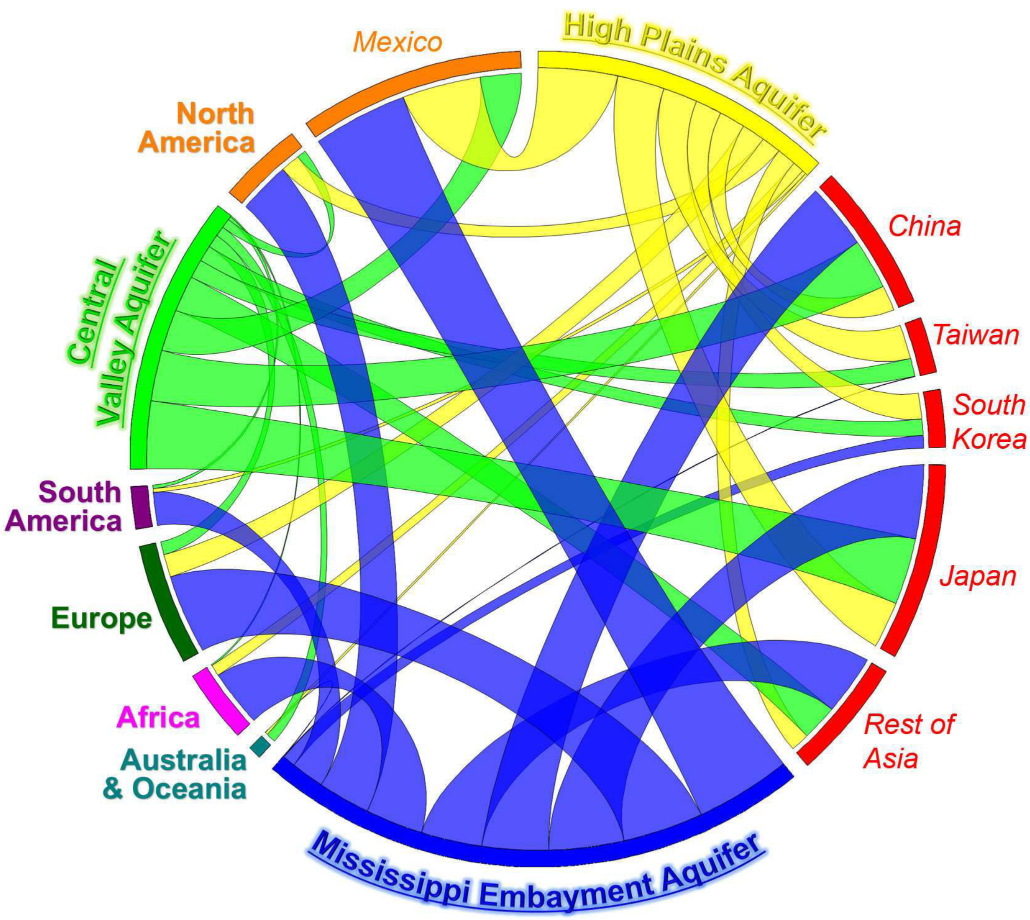

Figure 2.6: International virtual groundwater transfers from overexploited aquifers in the United States. The size of the outer bar indicates the total virtual groundwater export volume for the Mississippi Embayment aquifer (blue), High Plains aquifer (yellow), and Central Valley aquifer (green). Aquifer origin volume is indicated with links emanating from the outer bar of the same color. Export destination volume is indicated with a white area separating the outer bar from links of a different color. The countries and regions that import the most virtual groundwater are provided. The links are scaled relative to the volume of virtual groundwater exported. This figure was created with network visualization software available at http://circos.ca, developed by (*Krzywinski et al.*, 2009).

Table 2.2 presents the countries that are currently most reliant upon cereals – critical for food security (*e.g., Foley et al.* 2011) – grown in the HP, CV, and ME aquifers. Countries are ranked in order of the fraction contribution of cereals produced in these aquifers to total domestic cereal supply (column 4). Besides the U.S. itself, Taiwan is the country that is most reliant upon these aquifers: 10.0% of its domestic cereal supply is produced by these aquifers. These aquifers produce 9.2% of Japan's cereal supply, and 9.1% of the cereal supply of Panama. Consumers in these countries would be impacted by rising world prices if agricultural production in the HP, CV, and ME aquifer systems were to slow or halt. However, the willingness and ability of consumers to pay increased commodity prices differs across countries, such that some countries would be more impacted than others.

Panama, Costa Rica, and the Dominican Republic are the third, fourth, and fifth most dependent on cereal imports from U.S. aquifers to meet their domestic cereal requirements. However, these countries do not import large volumes of virtual groundwater because cereals have low VGC. Small, developing island nations, such as Cook Islands and Samoa, as well as some arid Asian countries, such as Mongolia, import relatively modest quantities of food from the U.S. aquifers, but they disproportionately import virtual groundwater resources, since they import mostly meat and processed food commodities, which have the highest VGC. Agricultural land availability, soil nutrients, and industrial capabilities – rather than water – are likely restricting local production of these commodities within the island nations. Comparative advantage across a wide suite of factors leads to complexities in the teleconnected food trade system, with unanticipated outcomes, such as non-local aquifer depletion.

This analysis highlights international consumers that are most vulnerable to eventual reductions in agricultural production from unsustainably managed reserves of groundwater in the United States. Countries that are heavily reliant on these aquifers can use this information to evaluate how their domestic food security will be impacted when agricultural production from these aquifers is eventually slowed or halted altogether. If future consumer welfare is at risk, then policy makers in those countries may want to consider diversifying the sources of their food supply, with implications for the global food trade system. Additionally, some consumers – currently not receiving a price signal of resource scarcity – may be willing to pay a premium now to store groundwater supplies for future food security. In principle, this could operate as a payment for ecosystem services (*Naeem et al.*, 2015). However, implementing the payment of such a premium may prove challenging in practice, because food commodities are available cheaply on international markets, counteracting such an exchange; instead encouraging tragedy of the commons behavior amongst consumers.

2.4 Concluding Remarks

It is imperative to understand the teleconnections and demand forces that are contributing to the unsustainable use of aquifers in the United States if we are to effectively slow their depletion. In this paper we quantified and traced virtual transfers of critical groundwater resources from the High Plains, Central Valley, and Mississippi Embayment aquifers. This is the first study to track virtual groundwater transfers to the final destination using high-resolution empirical data on food commodity transfers. This is an important first step towards empowering producers, consumers, water-planners, and decision-makers, by linking understanding of local production withdrawals with new knowledge on the virtual transfers of groundwater resources.

The vast majority (91%) of virtual groundwater transfers remains within the United States. Cereal production using groundwater from the High Plains, Central Valley, and Mississippi Embayment aquifers contributes to 18.5% of U.S. cereal supply and 8.6% of U.S. cereal exports. Since these aquifers are critical to domestic food security and trade interests, policy makers in the U.S. may want to consider implementing policies that properly value these groundwater reserves, particularly since they may represent a strategic domestic water source in the future. Decision makers may want to reconsider current measures that exacerbate common pool aquifer depletion, and, instead, explore opportunities to value these aquifers for their risk mitigation potential under an uncertain future. A relatively small fraction of the virtual groundwater transfers are international; however, cereals produced by these aquifers comprise a significant fraction of the cereal supply of some recipient countries, such as Taiwan, Japan, and Panama. Countries that are reliant upon these aquifers can determine their potential vulnerability to global price increases associated with eventually slowing groundwater extraction in productive locations. Policy makers in these countries may consider diversifying the sources of their food supply to mitigate supply chain risk.

One unintended consequence of the current landscape of economic and trade policies has been the overexploitation of groundwater reserves in the United States. Under an uncertain climate future, in which rainfed agriculture is likely to experience more droughts and extreme climate events, groundwater resources may become more valuable. This buffer value of groundwater – along with other non-extractive values that promote ecosystem services – is not currently incorporated into the calculation of the costs and benefits of groundwater extraction. To better determine the welfare tradeoffs and guide policy, the costs and benefits accrued along the entire value chain of this teleconnected system need to be taken into account. This includes the value of groundwater resources – both now and in the future – as a food security buffer to variable surface water supplies. Such an analysis must recognize that there are competing goals and multiple objectives related to water resources use, and that decision makers often work at vastly different spatial and temporal scales to those necessary to address global sustainability challenges.

Chapter 3

DROUGHT IMPACTS TO WATER FOOTPRINTS AND VIRTUAL WATER TRANSFERS OF THE CENTRAL VALLEY OF CALIFORNIA

3.1 Introduction [1]

California is one of the most productive agricultural areas in the world and is commonly referred to as the 'fruit and vegetable basket' of the United States, responsible for nearly half of U.S. grown fruits, vegetables, and nuts. California's agricultural industry is made possible by a complex and vast water system that relies on precipitation, surface water, and groundwater. From 2012–2014, California experienced its worst drought in over a millennium (*Griffin and Anchukaitis*, 2014). Although local impacts have been examined (*Howitt et al.*, 2014; *Cooley et al.*, 2015; *Swain*, 2015), it is not yet well understood how the drought has impacted distant consumers of California agricultural commodities through the global food system. In this paper we examine drought impacts to water footprints of agricultural production and food and virtual water transfers from the Central Valley of California, including tracing these flows to their final destination of consumption. Broadly, this study elucidates how local climate shocks reverberate through the global food system and highlights the critical role of groundwater aquifers.

Drought is not an uncommon occurrence in California, but the 2012–2014 drought was exceptional. For only the second time in its history, California proclaimed a State of Emergency due to drought on January 17, 2014. In 2015, Governor Brown introduced unprecedented mandatory water use restrictions on urban users, requiring them to reduce usage by 25%. In California, where irrigation is responsible for 74% of water withdrawals (*Maupin et al.*, 2014), drought is particularly impactful to agriculture, which is crucial to the identity, culture, and economy of California. In 2014 alone, drought led 173,200 additional hectares of irrigated cropland to be fallowed, $2.2 billion in economic cost, and the loss of 17,100 jobs (*Howitt et al.*, 2014).

The impacts of the drought to agriculture would have been much worse if not for Cali-

[1] This chapter is published as an article in Water Resources Research, 2017 (*Marston and Konar*, 2017)

fornia's conjunctive water use system, which permits farmers to rely more on groundwater during times of surface water deficits. However, farmers are currently extracting much more groundwater from the Central Valley Aquifer (boundaries shown in Fig 3.1) than is being recharged, leading to an annual average depletion of 1.85 x 10^9 m^3 since 1960 (*Faunt*, 2009) and nearly double that rate during the current drought (*Faunt and Sneed*, 2015). During average climate conditions, 40% of irrigation in the Central Valley comes from groundwater, but during drought groundwater provides closer to 70% of irrigation supplies, with more reliance on groundwater in the arid Tulare and San Joaquin Basins, and less groundwater use in the more humid Sacramento Basin (*Faunt and Sneed*, 2015; *Jones*, 2015).

The Central Valley aquifer system is likely to be under even greater pressure in the future. Anthropogenic warming is expected to increase the frequency of dry, warm years in California, thereby increasing the likelihood of severe droughts (*Diffenbaugh et al.*, 2015). Additionally, demand for water is increasing amongst environmental, urban, and agricultural uses (*Faunt*, 2009), while continued groundwater depletion (*Famiglietti et al.*, 2011), salinization of the deeper aquifers (*Schoups et al.*, 2005), and new legislation restricting future groundwater withdrawals (e.g. the 2014 California Sustainable Groundwater Management Act) will reduce groundwater availability. This not only has implications for Californians who depend on the aquifer for agricultural and urban uses, but also the millions of people globally who consume groundwater dependent agricultural products grown in the Central Valley (*Marston et al.*, 2015).

Much is understood about local impacts of drought to agricultural production (*Howitt et al.*, 2014; *Cooley et al.*, 2015; *Faunt and Sneed*, 2015). However, the food system is global in nature, such that agricultural commodities are part of a complex supply chain and typically consumed far from their location of production, in an example of a telecoupled system (*Liu et al.*, 2013, 2015). The trade of water-intensive food commodities is referred to as 'virtual water trade' (*Allan*, 1998; *Hoekstra and Hung*, 2005) and links distant consumption of water-intensive goods to local water use and impacts. Increasingly, it is critical to understand the non-local impacts of drought. Does the global food system amplify or dampen the impacts of local droughts shocks? On one hand, global food supply chains may propagate drought risk to distant consumers through the disruption of complex supply chains (*D'Odorico et al.*, 2010; *Suweis et al.*, 2015). On the other hand, the impacts of local climate shocks, such as droughts, may be mitigated if a country imports the same agricultural commodity from multiple producers, all of which experience spatially and temporally uncorrelated climate shocks.

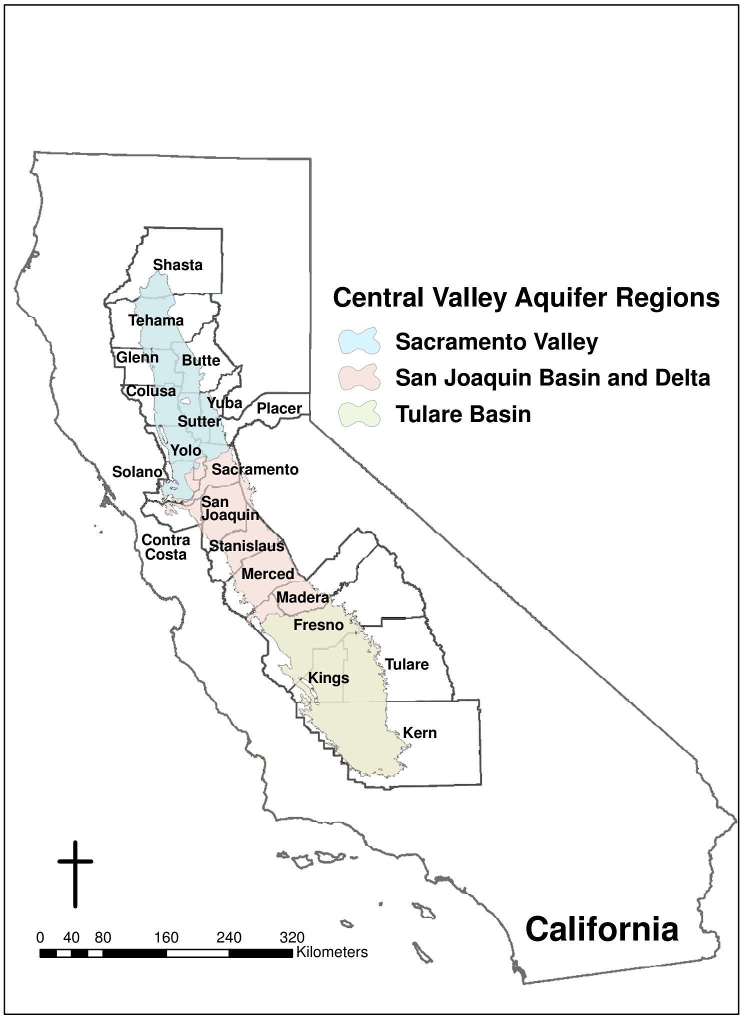

Figure 3.1: Map of the Central Valley Aquifer of California. The major basins of the Central Valley Aquifer are the Sacramento Valley (blue), San Joaquin Basin and Delta (red), and Tulare Basin (green). The 20 counties overlying the Central Valley Aquifer are provided.

In this study, we evaluate the impact of drought to agricultural water footprints and virtual water transfers from the Central Valley of California. Our work builds on recent high resolution studies of water footprints and food and virtual water flows in the United States. The water footprint of crops and derived crop products has been established for all states in the U.S. (*Mekonnen and Hoekstra*, 2011b; *Mubako and Lant*, 2013), with additional work on California (*Fulton et al.*, 2012, 2014; *Mubako et al.*, 2013). These studies, however, do not distinguish between surface water and groundwater sources and do not account for interannual variability in water footprints. High resolution intra-national food transfer data has been used to evaluate food and virtual water flows within the United States (*Lin et al.*, 2014; *Dang et al.*, 2015). *Marston et al.* (2015) determine virtual groundwater transfers from overexploited aquifers of the United States, as well as the major U.S. cities, U.S. states, and international export destinations that are most reliant upon agricultural production from these aquifers. These recent studies refined our understanding of the spatial variability in water footprints and virtual water transfers under average climatic conditions. However, climatic variability and extremes, such as drought, significantly impacts agricultural production, trade, and embedded water resources (*Dalin and Conway*, 2016; *Zhuo et al.*, 2016), making it essential to better resolve food and virtual water flows in time, which is a major novelty of this study.

We integrate high resolution databases and models to quantify the water footprints of agricultural production and virtual water transfers from California's Central Valley from 2011 (baseline, no drought) through three years of consecutive, exceptional drought (2012–2014). A major novelty of our methodology is that we distinguish precipitation, surface water, and groundwater contributions to the total water footprint of agricultural production. Our study describes (i) how local water footprints have evolved over the course of the drought, (ii) how local drought shocks propagate to distant consumers of water-intensive goods, and (iii) how distant consumption of virtual water resources is linked with local water impacts. In this way, we aim to address the following questions: (i) How do agricultural production water footprints in California evolve with drought? ii) How does drought impact food and virtual water transfers from California? and iii) How is global demand for California agriculture contributing to local water resources impacts? The paper is organized as follows. We describe our methods in Section 3.2. Our results are detailed and discussed in Section 3.3. We conclude and highlight implications of our work and future research needs in Section 3.4.

3.2 Methods

In this section, we describe how we quantify the water footprints and virtual water transfers from the 20 counties overlying the Central Valley of California (map provided in Fig 3.1). We calculate the total water footprint of agricultural production, which is comprised of contributions from precipitation (i.e. 'green water') and irrigation supplies (i.e. 'blue water'). A major novelty of our approach is that we further distinguish the irrigation component into surface and groundwater sources. We also explain how we calculate virtual water transfers from the Central Valley. Note that we use the term 'transfers' because we examine sub-national and international flows of agricultural commodities and embodied water. We reserve the standard trade terms (i.e. 'trade', 'import', and 'export') solely for international exchanges of goods.

Drought impacts to agricultural production and transfers are highly local and time-dependent, which necessitates the use of high-resolution spatial and temporal data. To determine the impact of the California drought, we pair empirical databases with modeled estimates of crop evapotranspiration. Refer to Table 3.1 for key data sources and models used in this study. We quantify the water footprints of crop production at the annual temporal scale, county spatial scale, and for each source of water (i.e. rainfall, irrigation from surface water sources, and irrigation from groundwater sources). We also quantify virtual water transfers at the annual and county scale for each source of water. The counties of the Central Valley are shown in Fig 3.1.

First, we detail how we calculate the virtual water content of agricultural products. Second, we describe the agricultural production and transfer datasets. Then, we explain how we quantify water footprints of agricultural production. Lastly, we describe how virtual water contents and food transfer data are brought together to quantify virtual water transfers.

3.2.1 Virtual water content estimates

The virtual water content (VWC) of a crop is defined as $VWC = ET/Y$, where ET is the total crop evapotranspiration [$m^3_{water}\ area^{-1}$] and Y is the crop yield [$ton_{crop}\ area^{-1}$]. VWC is equivalent to the water footprint of crops (*Hoekstra and Chapagain*, 2011) and indicates the amount of water embodied in crop production over the entire growing season. VWC values were calculated for each crop, county, and year (2011–2014) combination.

The 67 crops included in this study represent 98.5% of the harvested crop tonnage of California reported in the 2012 USDA Census of Agriculture. Importantly, we quantify the fractional contribution of each major water source to total crop ET. In other words, we segment the contribution of green (i.e., effective precipitation) and blue water (i.e. irrigation) to total crop ET. Additionally, we further segment blue water into irrigation from surface and groundwater sources. In this way, we estimate VWC from green, surface, and groundwater

Table 3.1: Primary data sources and models used in this analysis. A brief description of each item is provided along with its spatial and temporal resolution.

Model or Data Source	Description	Spatial Resolution	Temporal Resolution
California Department of Food and Agriculture (2017)	crop production & yield data	county & state	annual
CDWR (2015)	climate data	point	daily
CUP+ Model (*Orang et al.*, 2011)	crop evapotranspiration model	county	daily
CDWR (2013)	irrigation water source	county	annual
US Drought Monitor (2016)	drought index	county	weekly
FAF4 (2015)	commodity transfers	FAF Zone	annual

sources (i.e. VWC_{green}, $VWC_{surface}$, and VWC_{ground}, respectively).

Crop evapotranspiration

The ET of each crop was calculated using the *Consumptive Use Program Plus* (CUP+) model. The CUP+ model is a dynamic soil water balance model developed by the California Department of Water Resources (CDWR) and the University of California, Davis to help water agencies and growers determine crop water requirements in California. The CUP+ model computes reference evapotranspiration (ET_O) using the daily Penman-Monteith equation. Daily weather data, including solar radiation, maximum and minimum temperature, dew point temperature, wind speed, and precipitation, were inputs to the model and came from *CDWR* (2015). Planting and harvest dates, maximum soil depth, and available water holding capacity were provided within the model databases. The maximum rooting depth for each crop was taken from *USDA SCS* (1983).

Using the CUP+ model, we determine crop specific daily evapotranspiration (ET_c). To do this, we follow a similar methodology as *Doorenbos and Pruitt* (1977), in which different crop coefficients (K_c) are applied during the growing season to represent how plant water requirements vary during different growth periods. Each day, K_c is determined and multiplied by ET_O to arrive at daily ET_c. Total crop ET is determined by the sum of all daily ET_c values during the cropping season. Total crop ET estimates crop evapotranspiration from all water sources (i.e. rainfall, surface, and groundwater sources).

Importantly, the CUP+ model distinguishes between ET from rainfall and ET from irrigation supplies (ET_i). In CUP+, irrigation occurs when the soil water content in the effective root zone is less than half of capacity. This assumption is in accordance with the management allowable depletion historically used by most irrigators (*USDA SCS*, 1993; *Ozdogan et al.*, 2010). Irrigation water is applied until the soil water content returns to field capacity. The cumulative ET for the entire crop season attributed to irrigation is given by:

$$ET_i = CD_{sw} - \Delta WC = (CET_c - CE_{spg} - CE_r) - \Delta WC = \sum_{i=1}^{n} NA_i \tag{3.1}$$

where ET_i is ET of applied water (i.e., irrigation), CD_{sw} is the cumulative daily change in soil water content, ΔWC is the difference between initial and final soil water content, and NA_i is the net water application. CET_c, CE_{spg}, and CE_r are the seasonal cumulative crop evapotranspiration, cumulative effective seepage, and cumulative effective rainfall contribution, respectively (*Orang et al.*, 2011). CUP+ does not distinguish between E and T and does not incorporate capillary rise, unlike other ET models used to determine high-resolution water footprints (*Chukalla et al.*, 2015; *Zhuo et al.*, 2016). However, depths to the water table are considerable in the Central Valley, so capillary rise is negligible for crop growth in this region. An important feature of our approach is that we use daily climate data to force CUP+, whereas other studies use monthly climate variables (*Chukalla et al.*, 2015; *Zhuo et al.*, 2016). Importantly, CUP+ simulations encompassed several months before the growing season to appropriately capture antecedent soil moisture conditions.

Note that although the CUP+ model distinguishes between rainwater and irrigation supplies, it does not break irrigation water down between surface and groundwater sources. Thus, the CUP+ model is used to calculate county-level and crop-specific ET_{green} and ET_{blue} for each year of the drought. We explain how we separate the surface and groundwater contribution to irrigation in the next section.

Surface and groundwater contributions

As described in the previous section, we use the CUP+ model to distinguish precipitation and irrigation contributions to total ET for each crop and county. Here, we explain how we additionally segment irrigation into surface and groundwater contributions for each county overlying the Central Valley Aquifer for the years 2011–2014. First, we obtained data from the California Department of Water Resources on surface and groundwater irrigation volumes for all available water years, which were 2002–2010 (*CDWR*, 2013). Note that irrigation data from CDWR is provided annually (for the years 2002–2010) at the county spatial resolution, but is not crop specific. Data on irrigation by crop and for the years of this study (i.e. 2011–2014) would improve our estimates, but this data does not exist, unfortunately.

Next, we used this data to quantify the fraction of total irrigation coming from groundwater (i.e. the groundwater fraction, GF) for 2002–2010. Then, we determined the relationship between GF and a county-level drought index (DI) for 2002–2010. DI data was obtained from the US Drought Monitor (*US Drought Monitor*, 2016). The relationship between GF and DI varies across counties in the Central Valley, likely due to differing surface water rights and availability, as well as differences in agricultural production practices. Counties located in the wetter northern region had a relatively consistent reliance on groundwater, irrespective of drought conditions. In other words, GF does not vary with DI in these counties, so an average GF value was calculated based on their historic groundwater use and used to approximate GF from 2011–2014. Table 3.2 shows the GF between 2011–2014 for counties overlying the Central Valley aquifer system that do not exhibit a relationship between drought and groundwater irrigation.

Table 3.2: Groundwater fraction for the 10 counties overlying the Central Valley Aquifer that do not correlate with drought conditions. For these counties, GF is the average of 2002–2010 values.

County	Groundwater Fraction
Butte	0.36
Contra Costa	0.01
Placer	0.06
San Joaquin	0.19
Shasta	0.14
Solano	0.4
Sutter	0.24
Tehama	0.73
Yolo	0.41
Yuba	0.18

We constructed county-specific relationships between the groundwater fraction and drought intensity for counties located in the central and southern portions of the Central Valley (see Fig 3.2). Regressions in Fig 3.2 show the location-specific relationship between GF and DI. The relationships plateau as we might expect, since individual locations eventually reach a limit to groundwater pumping. This limit varies by location and is constrained by location-specific groundwater availability (well yields). The most senior water right holders will continue to have priority to any surface water supplies that are available during drought. Note that the blue points in Fig 3.2 present data from 2002–2010.

We fit a logarithmic trend line to the observed data, because this functional form best fits the data and enables us to employ a conservative approach when projecting GF. The logarithmic functional relationship means that GF levels off with increasing DI, thereby capturing location-specific surface water rights and groundwater pumping limits. The other extreme would be if the groundwater wells had all been pumped dry during the drought, such that we are overestimating the groundwater fractions available during the drought. However, it is important to note that we restrict our analysis with production data (see next section), meaning that water resources must have been available and used to meet crop demands.

To estimate GF for 2011–2014, we use DI data for 2011–2014 in conjunction with the regression relationships. The red points on Fig 3.2 illustrate estimated values of GF from

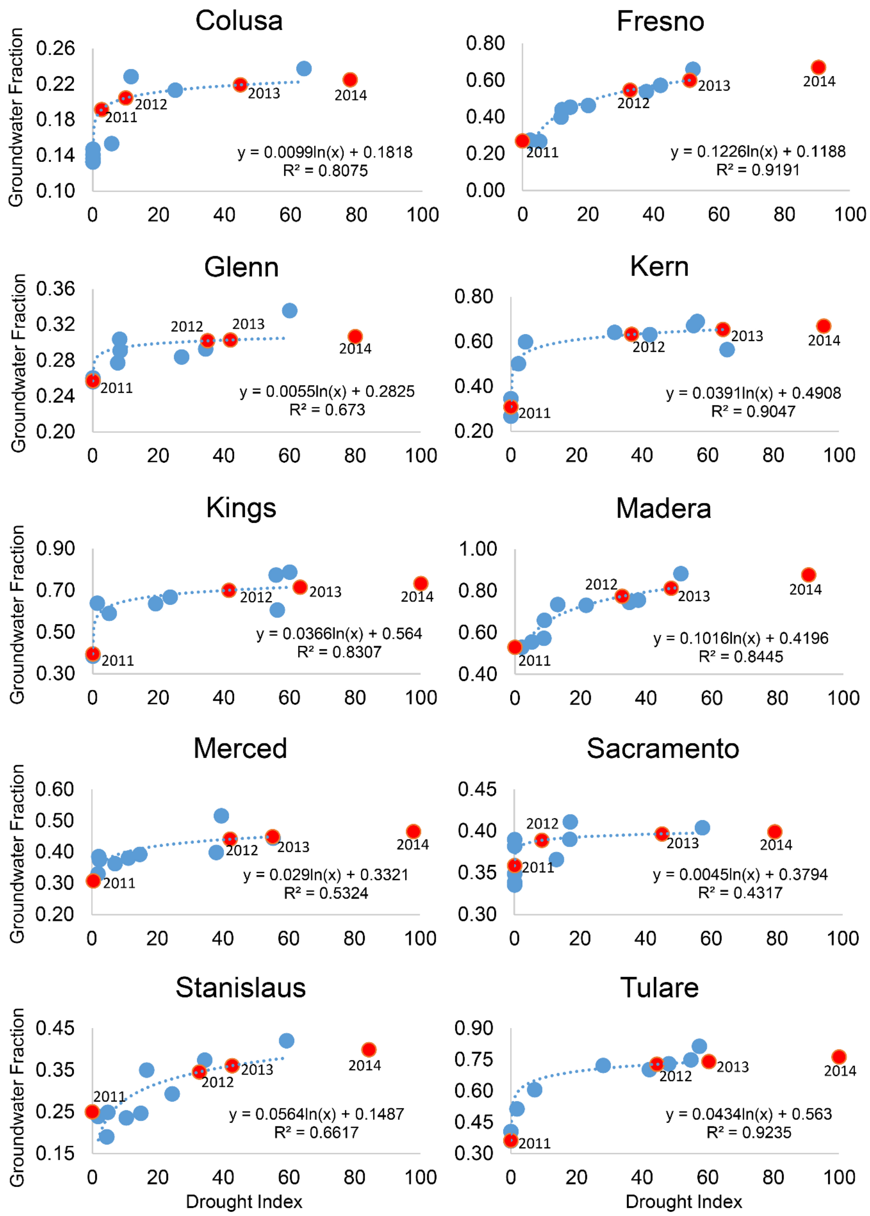

Figure 3.2: Regression equations relating the fraction of irrigation from groundwater (GF) to the drought index (DI). DI ranges from 0-100, with 100 being the most severe drought. The trend line is shown, along with its equation and its coefficient of determination. Empirical values of GF and DI from 2002–2010 are represented by blue markers. Estimated values of GF for 2011–2014 are shown with red markers and are labeled by year.

2011–2014. Since the 2012–2014 drought is the worst on record, we had to extrapolate beyond the x-axis bounds of the regression equation in some instances (typically just for the year 2014). However, extrapolation was often minimal (e.g. Colusa, Glenn, and Sacramento counties) since California experienced the seventh driest three-year period between 2007–2009 in terms of state-wide precipitation and it was the only other time a state-wide proclamation of emergency was declared due to drought. Importantly, note that most projected values of GF (i.e. red points) fall within the GF bounds of the observed past, despite the fact that DI for the year 2014 falls outside the bounds of historic DI observations. This means that estimated values of GF have been observed before, making them feasible and conservative. Only one estimate of GF during the drought exceeds observed values: Fresno's 2014 GF was 1.69% greater than the historical maximum. In 2014, river runoff in the Tulare Basin was 27% of average, while state and federal water project deliveries reached record lows; final allocations from the California State Water Project were 5% of assigned allocations (*McEwan et al.*, 2016). This unprecedented reduction in surface water availability likely dramatically increased the amount of groundwater used to meet irrigation demands beyond historical observations, giving us confidence that our GF estimate is likely conservative in this region.

Irrigation supplies from surface water sources are estimated by the difference between the total irrigation requirement and the groundwater contribution. Then, we obtained crop yield values for each county, crop, and year from the California Department of Food and Agriculture (CADFA) (*California Department of Food and Agriculture*, 2017) (see next section). We used the water source specific ET values in conjunction with Y values to determine VWC by water source. Note that our approach to segmenting irrigation into surface and groundwater sources is more refined in space than it is by crop. For this reason, crop-specific groundwater and surface water footprints should be used with caution.

3.2.2 Agricultural production and transfers

Here, we explain the data sources used to evaluate agricultural production and commodity transfers in California. We also explain how transfer data were interpolated across spatial and temporal scales.

Agricultural production

We use annual, county-level data on agricultural production, harvested area, and yields from the California Department of Food and Agriculture (CADFA) (*California Department of Food and Agriculture*, 2017). The 67 crops used in this study are presented in Table 3.3. Together, these crops represent 98.5% of all harvested crop tonnage in the Central Valley according to the 2012 Census of Agriculture (*USDA*, 2016).

Table 3.3: Alphabetical listing of the 67 crops represented in this study.

1	Almonds	18	Corn, Grain	35	Melons, Honeydew	52	Raspberries
2	Apples	19	Corn, Silage	36	Melons, Watermelon	53	Rice
3	Apricots	20	Cotton	37	Nectarines	54	Safflower
4	Artichokes	21	Cucumbers	38	Oats	55	Spinach, Fresh
5	Asparagus	22	Dates	39	Olives	56	Spinach, Processing
6	Avocados	23	Garlic	40	Onions	57	Squash
7	Barley	24	Grapefruit	41	Oranges	58	Strawberries
8	Beans, Dry Edible	25	Grapes, Raisin Type	42	Peaches	59	Sugarbeets
9	Beans, Snap	26	Grapes, Table Type	43	Pears	60	Sunflower
10	Blackberries	27	Grapes, Wine Type	44	Pecans	61	Sweet Corn
11	Blueberries	28	Hay & Haylage	45	Peppers, Bell	62	Sweet Potatoes
12	Broccoli	29	Kiwifruit	46	Peppers, Chile	63	Tangerines
13	Cabbage	30	Lemons	47	Pistachios	64	Tomatoes, Fresh
14	Carrots	31	Lettuce, Head	48	Plums	65	Tomatoes, Processing
15	Cauliflower	32	Lettuce, Leaf	49	Potatoes	66	Walnuts
16	Celery	33	Lettuce, Romaine	50	Prunes	67	Wheat
17	Cherries	34	Melons, Cantaloupe	51	Pumpkins		

Virtual water contents (VWC), water footprints, and virtual water transfers were calculated for each crop. Below, we explain how we combine this information with crop VWC to obtain the water footprints of agricultural production. We also use crop production data from from *USDA* (2016). Data from *USDA* (2016) is primarily used to estimate rainfed production in the Central Valley, which is a relatively minor component of total production, since the vast majority of agriculture in the Central Valley is irrigated.

Agricultural transfers

We estimate annual and county-level agricultural transfers from the Central Valley of California. To do this, we use version four of the Freight Analysis Framework (FAF4) (*FAF4*, 2015). The FAF4 dataset relies on several sources to reconstruct domestic and international commodity transfers but its foundation is the Commodity Flow Survey (CFS) and international trade data is from the U.S. Census Bureau. The CFS is a quarterly survey administered every five years (years ending in '2' and '7') that samples over 100,000 establishments on their shipment activity, including a description of the transported good and its commodity code, the good's origin and final destination, weight, value, and mode of transportation. The survey data sample is used to estimate the total value and weight of goods shipped in each industry (*US Census Bureau*, 2014a).

The movement of goods is traced from the point of production to the place of final consumption (this includes using the product as an input to value-added agriculture). Domestic origin and destination locations are represented by 132 FAF Zones, which are comprised of

84 U.S. metropolitan areas and 48 state or sub-state areas. International shipment destinations are represented by eight countries or regions: Africa, Canada, Eastern Asia, Europe, Mexico, Rest of Americas, Southeast Asia and Oceania, and Southwest and Central Asia.

Transfers of individual goods are not reported. Instead, commodities are aggregated according to the Standard Classification of Transported Goods (SCTG) coding system. The FAF4 dataset, along with the CFS dataset that it is based upon, tracks transfers from every sector of the economy. However, in this study we only use transfers of agricultural products. Our analysis uses SCTG 02 (Cereal Grains), SCTG 03 (Agricultural Products Except for Animal Feed), and SCTG 04 (Animal Feed and Products of Animal Origin). The individual crops that comprise each of these categories can be found in from (*US Census Bureau*, 2012).

The FAF4 dataset reports commodity transfers for the years 2012, 2013, and 2014. To determine transfer volumes for 2011, we scale 2012 food transfers by agricultural production data (*California Department of Food and Agriculture*, 2017). For instance, if a FAF Zone harvested 5% more cereal grains in 2011 than in 2012, then 2011 transfers would be 5% greater than 2012. This assumption captures potential changes in trade volume but presumes that relative trade patterns do not significantly change between 2011 and 2012. We spatially disaggregated the transfer data from the FAF scale to the county scale. This was achieved by multiplying a county's crop production by the fraction of the total production that is transferred out of the corresponding FAF Zone. So, if a county produced 50% of a FAF Zone's animal feed, for example, it is assumed that 50% of the FAF Zone's transfers can be attributed to that county. Disaggregating the origin of commodity flows based upon production data is a similar approach employed by (*Hoekstra and Mekonnen*, 2016). However, our empirical information on commodity transfers is provided at the sub-national scale, compared with the international trade data disaggregated in *Hoekstra and Mekonnen* (2016).

The final step was to disaggregate the transfer volumes of SCTG categories to transfers of individual crops. Together, all these processes can be simplified into one equation:

$$T_{c,a,n} = P_{c,a,n} \cdot \frac{T_{SCTG,FAF,a}}{P_{SCTG,FAF,a}} \tag{3.2}$$

Here, the transferred tonnage (T) of an individual crop (c) was calculated for each year (a) and county (n) by multiplying a crop's harvested tonnage (P) by the fraction of production that was transferred out of the associated FAF Zone. It was assumed that the difference between a FAF Zone's total crop production and the tonnage transferred out of the FAF Zone is what remained in the origin FAF Zone. In this way, mass balance was achieved.

The agricultural tonnage remaining in the FAF Zone of production can be attributed to either post harvest loss, food storage, internal consumption, or further processing into other products (e.g., corn into high fructose corn syrup).

3.2.3 Water footprints of agricultural production

The water footprint of agricultural production (WF) for each crop-county-year was calculated as:

$$WF_{c,a,n,w} = P_{c,a,n} \cdot VWC_{c,a,n,w} \tag{3.3}$$

where P indicates agricultural production, VWC is virtual water content, and the subscripts c, a, n, and w denote crop, year, county, and water source, respectively. Thus, water footprints are sensitive to changes in farmer decisions (e.g., crop production patterns and irrigation source), climate (e.g., effective precipitation and temperature), and crop response (e.g., ET and yield).

3.2.4 Virtual water transfers

Virtual water transfers (VWT) for each crop-county-year were calculated as:

$$VWT_{c,a,n \to FAF,w} = T_{c,a,n \to FAF} \cdot VWC_{c,a,n,w} \tag{3.4}$$

where T indicates commodity transfers, VWC is virtual water content, and the subscripts c, a, n, and w denote crop, year, county, and water source, respectively. The subscript $n \to FAF$ indicates the transfers from the county of origin (n) to a specific FAF Zone. VWT are traced from each county overlying the Central Valley Aquifer to 132 domestic destinations and 8 world regions. There are over two million virtual water transfers quantified in this study. When reporting our findings, we aggregate our results along particular spatial scales, water sources, or commodity resolutions of interest in order to make the results more clear.

Due to a lack of supply chain data, we did not trace virtual water flows associated with processed agricultural goods, livestock, or meat products. Available data does not indicate how the drought impacted where these products sourced their primary agricultural inputs from and what water source was used. Estimates of the total virtual water transfers leaving a FAF Zone are conservative since a portion of the agricultural products remaining in the

FAF Zone of origin (and the water embedded within them) will be processed or consumed by livestock and these secondary products will eventually be transferred and consumed outside the region.

3.3 Results and Discussion

Here we quantify the impact of drought in California to agricultural production and yields, virtual water contents, the water footprint of agricultural production, and food and virtual water transfers from the Central Valley. Our results present one example of how local climate shocks propagate through the global food system.

3.3.1 Drought impacts to agricultural area, yields, and production

Fig 3.3 presents the relative change [%] in harvested area, yields, and production over the course of the drought. From Fig 3.3 it is clear that harvested area decreased over the course of the drought, while yields actually increased and production remained relatively constant. Table 3.4 shows the values of harvested area, yields, and production by commodity category for each year of the study. Harvested area changed from 3,441,708 hectares in 2011 to 3,029,297 hectares in 2014, a 12% decline. This fallowing of less productive agricultural area helps to explain the yield gains (refer to yellow line on Fig 3.3). Nearly half of all crops saw 2014 yields maintain or exceed pre-drought yields in 2011. Crop production in the Central Valley, which represents approximately 75% of crop production in California by mass, only saw a 2% decline from 2011 to 2014. The vast majority of the decline in production occurred amongst cereal crops and animal feed crops, which saw a 28% and 10% decline, respectively (refer to Table 3.4). Other agriculture crop categories (namely fruits, nuts, and vegetables), actually saw a 7% increase in production during the drought, despite a 9% decline in harvested area (refer to Table 3.4).

3.3.2 Drought impacts to virtual water content

The drought had two distinct impacts on the VWC of crops. First, VWC values were generally larger during the drought (2012–2014) than in 2011 (pre-drought) (refer to Fig 3.4). This is because high temperatures during the drought increased plant water requirements

Table 3.4: California Central Valley harvested area [hectares], yields [tons hectare^{-1}], and production [tons] pre-drought (2011) and during the drought (2012–2014). Cereal grains (SCTG 2), other agricultural products (SCTG 3), and animal feed (SCTG 4) are provided. Yield gains in cereal grains and other agricultural crops over the course of the drought can be explained by the fallowing of less productive agricultural lands and changes in crop mix.

Harvested Area	2011	2012	2013	2014
Cereal grains	566,728	549,601	500,391	373,390
Other agricultural	2,165,484	2,150,788	2,036,481	1,995,848
Animal feed	709,496	740,716	709,636	660,059
Yield	2011	2012	2013	2014
Cereal grains	8.57	8.80	9.27	9.26
Other agricultural	15.90	16.85	18.10	18.68
Animal feed	28.19	28.94	29.52	26.47
Production	2011	2012	2013	2014
Cereal grains	4,339,420	4,234,356	4,209,809	3,121,597
Other agricultural	27,019,786	28,339,488	28,494,048	29,013,069
Animal feed	13,044,531	13,823,431	13,605,543	11,509,284

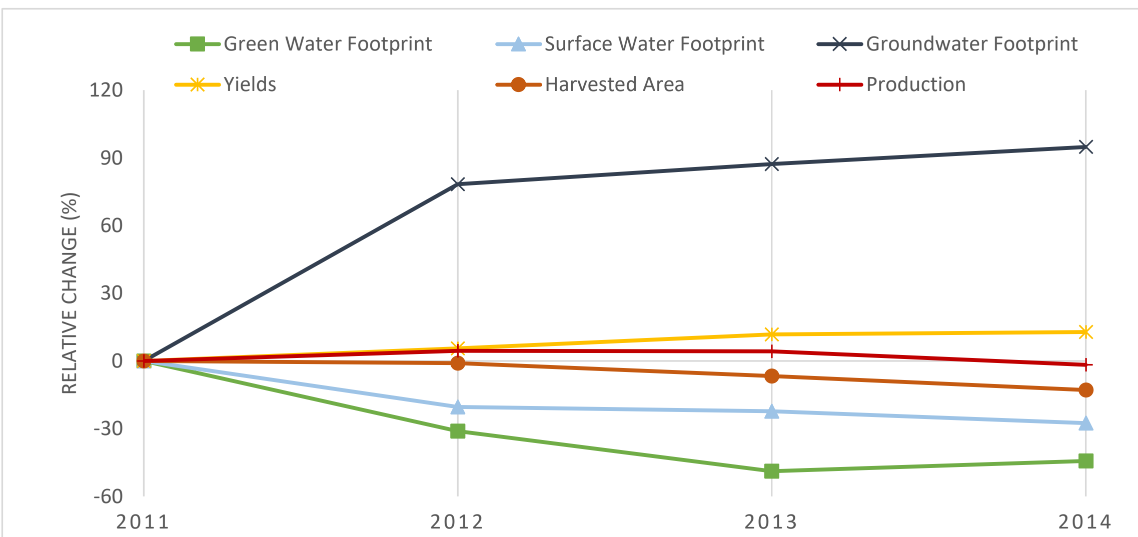

Figure 3.3: Relative change [%] in study variables over the course of the drought.

(numerator of VWC) while, in some instances, also reduced crop yields (denominator of VWC). Second, different water sources were used for crop irrigation during the drought. Since there was less rainfall available to meet crop water requirements during the drought, farmers increasingly relied on irrigation. Additionally, as the drought progressed, the water used for irrigation was increasingly obtained from groundwater sources. This is reflected in steady increases in the groundwater components of VWC shown in Fig 3.4, particularly in the Tulare Basin.

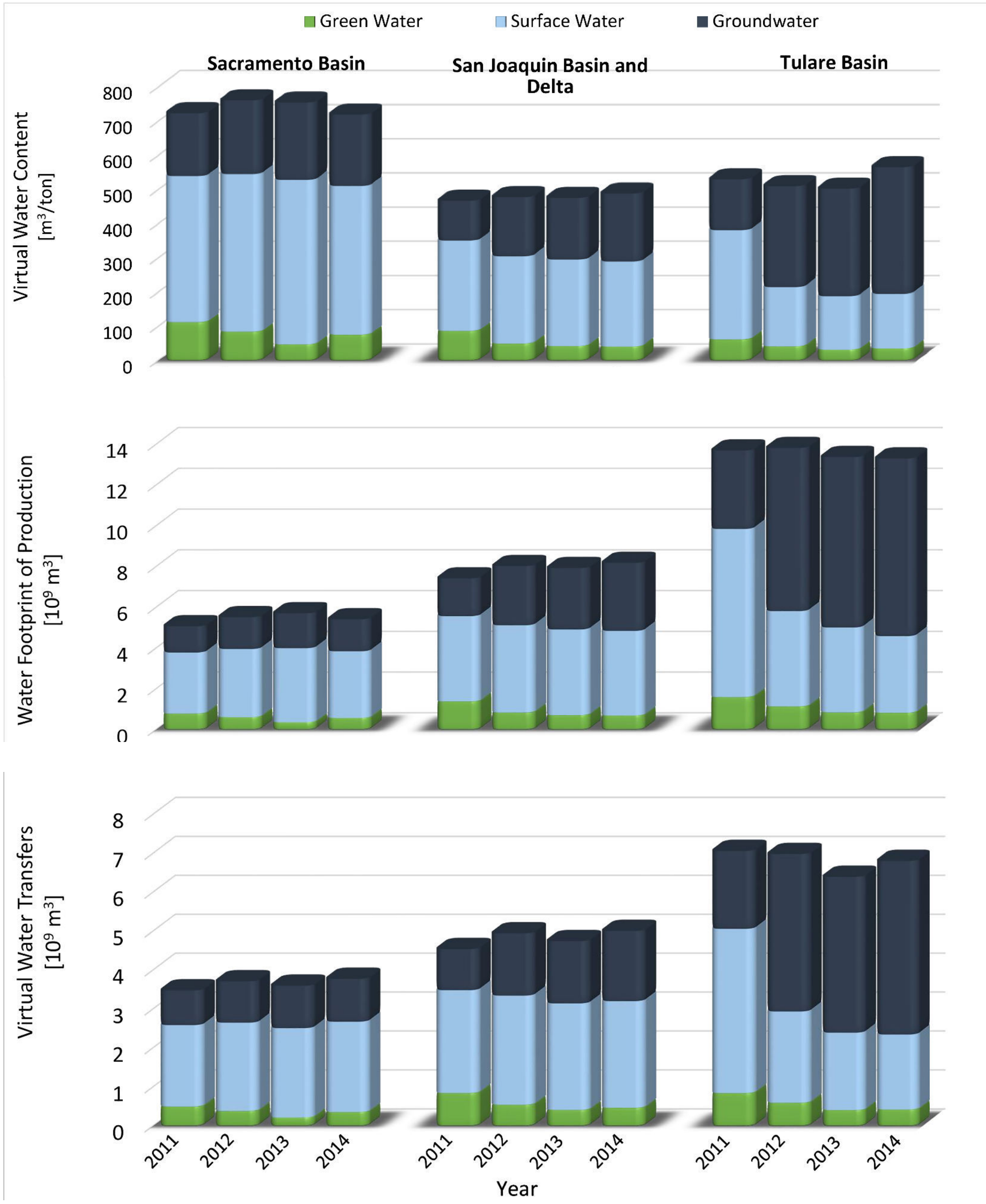

Figure 3.4: Virtual water content [m^3 ton^{-1}], water footprint of agricultural production [m^3], and virtual water transfers [m^3] pre-drought (2011) and during the drought (2012–2014) by water source are shown for the three basins of the Central Valley Aquifer: Sacramento Valley, San Joaquin Basin and Delta, and Tulare Basin. Note the increased contribution of groundwater over time, particularly in the Tulare Basin.

The average VWC was between 27% and 59% higher in the Sacramento Basin than in the San Joaquin and Tulare Basins. Although ET requirements for the same crop are lower in the cooler Sacramento Basin, average yields are also much lower. Nearly 40% of the Sacramento Basin's crop production is attributed to cereal grains (SCTG 2), which has average yields between 30-50% that of crops classified as SCTG 3 and SCTG 4 that are more widely grown in the San Joaquin and Tulare Basins (see Table 3.4). In comparison, only 1–5% of crop production in the San Joaquin and Tulare Basin is classified as cereal grains. Thus, the Sacramento Basin's overall crop yield is roughly two-thirds of the other two basins due to differences in cropping patterns. The lower yields lead to higher VWC in the Sacramento Basin, despite lower ET values.

Seventeen of twenty Central Valley counties saw average VWC_{ground} increase during the drought, some by over two-fold. VWC_{ground} within the Tulare Basin increased by 149% from 2011 to 2014 on average, reflecting increased dependency on groundwater in this basin. The San Joaquin and Sacramento Basins experienced average VWC_{ground} increases of 71% and 14%, respectively. In 2014, average VWC_{ground} of the Tulare Basin was 370.91 m^3 ton^{-1}. The average VWC_{ground} was 199.22 m^3 ton^{-1} in the San Joaquin Basin and Delta and 208.77 m^3 ton^{-1} in the Sacramento Valley. Permanent crops have VWC_{ground} values 4.5 times greater than average VWC_{ground} during the worst year of drought, with crops like almonds consuming 11.5 times more groundwater than average.

3.3.3 Drought impacts to agricultural production water footprints

The total WF of crop production in the Central Valley peaked at 27.38 x 10^9 m^3 in 2012. In 2011, before the onset of the drought, the total WF was 26.21 x 10^9 m^3. On average, for every 1 m^3 reduction in the green WF during the drought there was a 1.42 m^3 increase in the blue WF. This is due to increased crop ET during drought years, which is related to higher temperatures (between one and three degrees Celsius across the Central Valley) and a shift to more water-intensive orchard and vine crops.

We validate our results against two sources. First, we compare our estimates with the California Water Plan Update 2013 (*CDWR*, 2013). According to the California Department of Water Resources (CDWR), irrigation withdrawals for the 20 counties in this study were 28.07 x $10^9 m^3$ in 2010. We use CDWR irrigation efficiency parameters to convert our consumptive water use estimates into withdrawal values. We estimate irrigation withdrawals in 2011 as 31.20 x 10^9 m^3. Our 2011 value is roughly 11% higher than the 2010 CDWR value. This is reasonable given there was a 6% increase in harvested crop area between 2010 and 2011.

Second, we validate our numbers against *Howitt et al.* (2014). *Howitt et al.* (2014) estimate California's 2010 irrigation groundwater usage as 9.87 x 10^9 m^3, based on CDWR water use records. In 2010, CDWR records show that irrigators in Central Valley counties were responsible for 81% of groundwater use within California. Thus, based on these numbers, groundwater use in the Central Valley was roughly 8.01 x 10^9 m^3 in 2010. *Howitt et al.* (2014) projected an increase of 6.17 x 10^9 m^3 in groundwater irrigation across the Central Valley from 2010 to 2014. So, groundwater use in the Central Valley would be 14.18 x 10^9 m^3 in 2014, according to *Howitt et al.* (2014) estimates. We estimate 2014 groundwater use to be 13.63 x 10^9 m^3. So, our estimate compares favorably with *Howitt et al.* (2014) and is only 3.9% less.

Nearly all of the increase in the blue WF of production is due to increased groundwater consumption. In fact, 7 of the 20 Central Valley counties had a double-digit percentage decrease in $WF_{surface}$ (refer to Fig 3.5). The WF_{ground} of agricultural production in the Central Valley increased from 7.00 x 10^9 m^3 in 2011 to 13.63 x 10^9 m^3 in 2014. The increase in groundwater footprints coincided with a reduction in surface water and green water footprints, shown in Fig 3.4. The Tulare Basin experienced the greatest increase in both absolute and relative terms in groundwater footprints during the drought. In the Tulare Basin, total crop groundwater consumption increased from 3.85 x 10^9 m^3 in 2011 to 8.72 x 10^9 m^3 in 2014 (see Fig 3.4).

Depletion of the Central Valley Aquifer is not spatially uniform since groundwater withdrawals and recharge vary across the aquifer. Our study estimates volumes of groundwater consumption; however, not all consumptive groundwater use is unsustainable. Previous studies (*Faunt*, 2009; *Famiglietti et al.*, 2011) show that unsustainable groundwater use primarily occurs in the Tulare Basin, manifested by declining groundwater tables, subsidence, and reduced baseflow. We find that the Tulare Basin consumed 3.81 x 10^9 m^3 more groundwater in 2014 than the Sacramento and San Joaquin Basins combined. Furthermore, we find that locations with the largest groundwater footprint of agricultural production experienced the highest levels of land subsidence during the drought, as illustrated by Fig 3.6. Fig 3.6B shows the maximum recorded subsidence from 2011–2014 derived from USGS extensometers and continuous GPS measurements. Remote sensing studies have shown land subsidence up to 330 mm in just eight months in 2014 (*Farr et al.*, 2015), meaning that USGS in-situ measurements likely do not capture some of the more extreme instances of subsidence in the region. Subsidence alters land-surface slopes and has caused costly operational, maintenance, and construction-design issues related to water-delivery and flood-control canals and

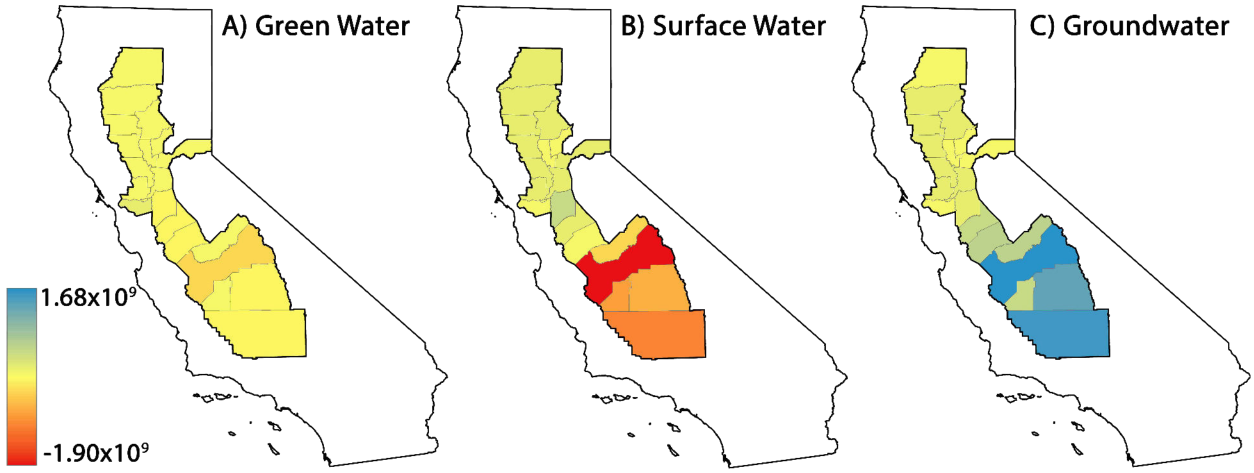

Figure 3.5: Drought impacts to the water footprint of agricultural production in the California Central Valley. Panels indicate volumetric changes [m^3] from pre-drought (2011) to drought conditions (2014) for green water footprints (A), surface water footprints (B), and groundwater footprints (C). Note that green and surface water footprints predominantly decrease, while groundwater footprints increase, particularly in the Tulare Basin.

other infrastructure (*Faunt and Sneed*, 2015).

From 2011 to 2014, the total irrigation water consumed in the production of cereal grains decreased by 16%, or 0.42 x 10^9 m^3. However, the blue water footprint of other agriculture products, such as fruits, vegetables, and nuts, increased by 6% (2.80 x 10^9 m^3) over this time period (see Fig 3.7). This reflects a reallocation of limited water supplies from low-value to higher-value crops during the drought.

Basin-wide, all crop categories increased their dependency on groundwater during the drought. However, cereals continued to meet the majority (∼70%) of their irrigation requirement from surface water sources. Fruits, nuts, vegetables, and animal feed crops went from groundwater supplying 32% of their irrigation requirement pre-drought to relying on groundwater to meet 57% of their irrigation needs in 2014. We estimate a smaller fraction of irrigation comes from groundwater sources than other studies, e.g., *Faunt and Sneed* (2015); *Jones* (2015). This is likely due to the conservative nature of our approach to estimating the contribution of groundwater to irrigation, as well as because our study encompasses all counties overlying the Central Valley aquifer system. This includes the area of counties that is not directly over the aquifer, while other studies just evaluate the land directly over the aquifer.

Changes in the water footprint of crop production during the drought occur for four rea-

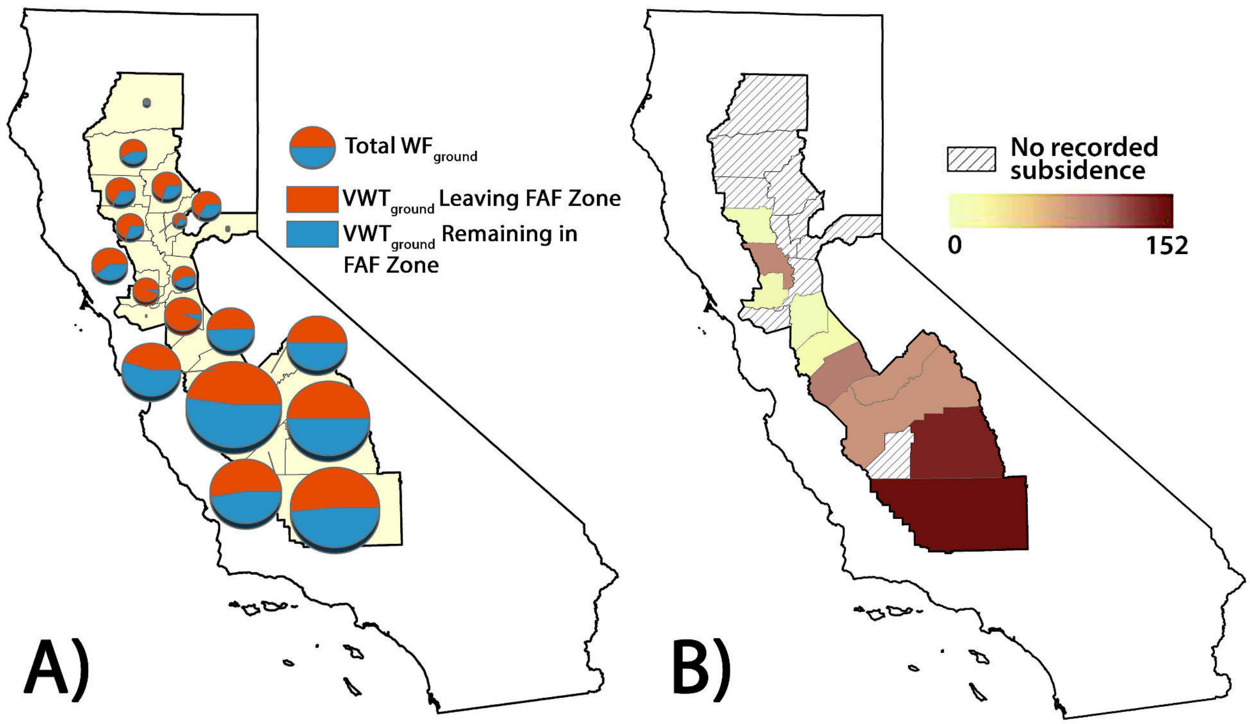

Figure 3.6: Map of groundwater footprints (A) and map of subsidence (B) for counties in the Central Valley of California. Panel A) Each circle is scaled according to the total groundwater footprint of crop production of each county aggregated from 2011–2014. The fraction of virtual groundwater transfers that leave the FAF Zone of production is shown in orange, while the fraction of virtual groundwater transfers that remain within the FAF Zone is blue. Panel B) Land subsidence recorded by USGS extensometers and continuous GPS measurements between 2011–2014 is mapped. Dark red indicates greater subsidence. Counties with no subsidence measurements are hatched. Note that the Tulare Basin counties have the largest groundwater footprint of agricultural production and the most subsidence.

sons: 1) a change in crop yield and harvested area; 2) an increase in crop evapotranspiration during drought years; 3) an increase in irrigation to compensate for rainfall deficits; and 4) an increase in groundwater irrigation due to reductions in surface water availability. There was an increase in the harvested area of permanent crops (vineyards and orchards) during the drought and a corresponding decrease in the harvested area of non-permanent crops (field crops and vegetables), shown in Fig 3.8. Between 2011 and 2014, 490,254 less hectares of non-permanent crops were harvested (29% reduction) in the counties overlying the Central Valley Aquifer. At the same time, 146,592 more hectares of permanent crops were harvested (15% expansion).

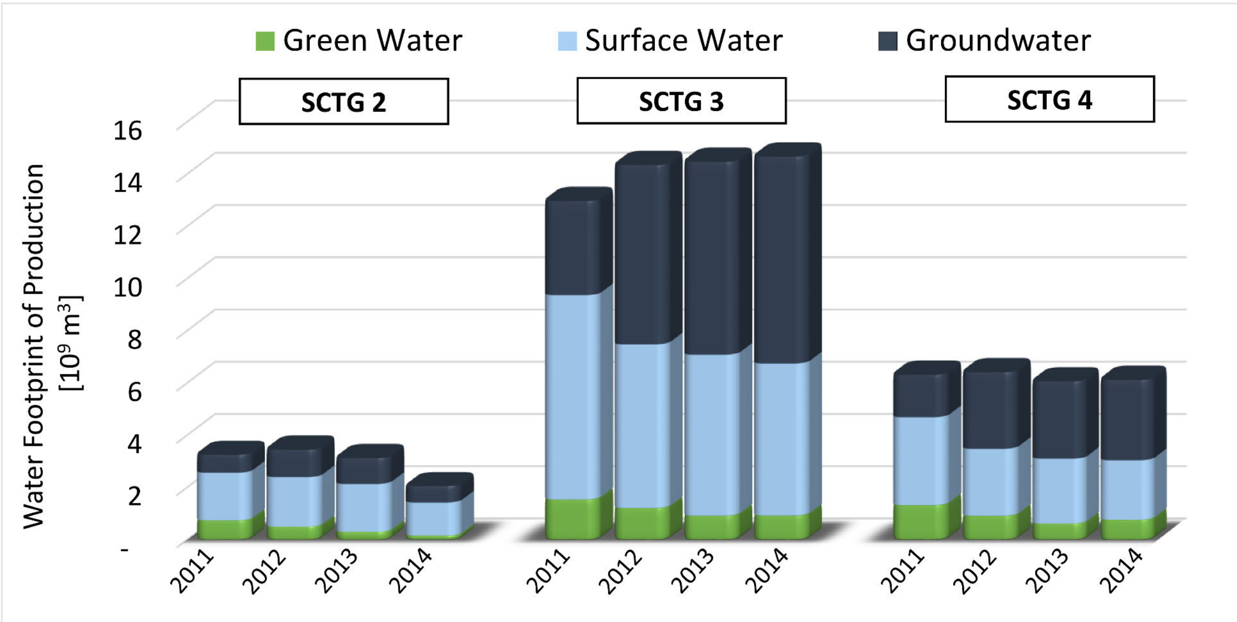

Figure 3.7: Green, surface, and groundwater footprints of agricultural production in the Central Valley from pre-drought (i.e. 2011) through the third year of drought (i.e. 2012–2014). Water footprints for cereal grains (SCTG 2), fruits, nuts, and vegetables (SCTG 3), and animal feed (SCTG 4) are shown.

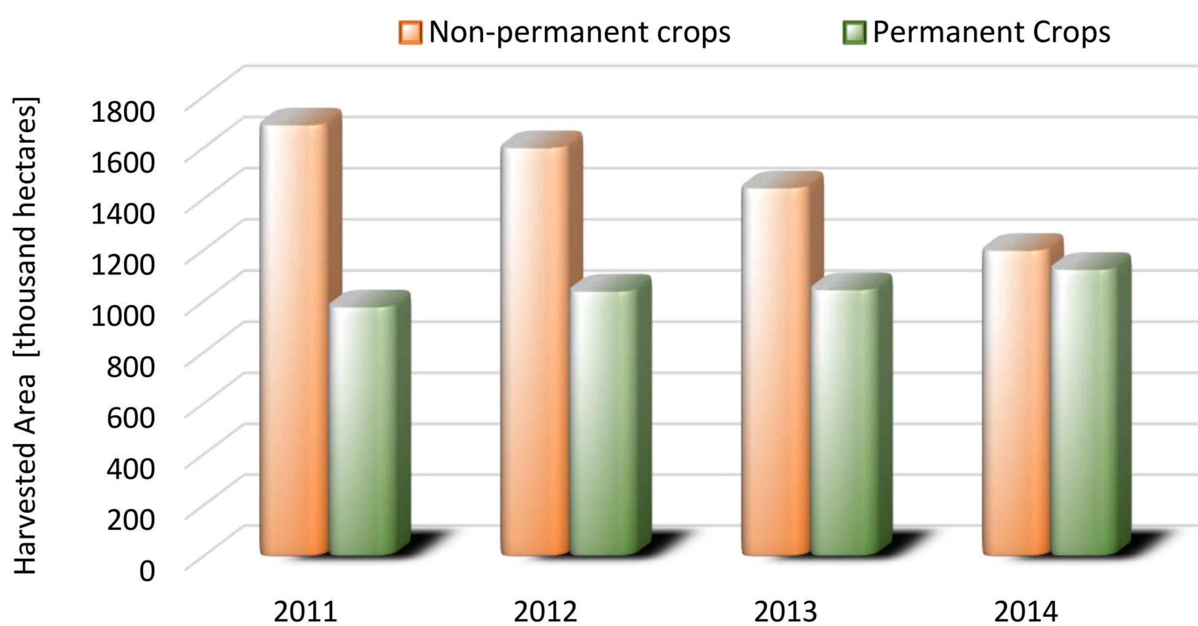

Figure 3.8: Harvested area [hectares] of permanent and non-permanent crops pre-drought (i.e. 2011) and during the drought (i.e. 2012–2014) in the California Central Valley.

On average, the water requirement per hectare of permanent crops was approximately 91% higher than non-permanent crops (see Fig 3.9). Thus, each additional hectare of permanent crops harvested during the drought would require nearly two hectares of non-permanent crops to be fallowed to maintain the same level of water consumption. A key exception is hay (including alfalfa), which, on a per hectare basis, consumes a considerable volume of water since the irrigated plot is harvested multiple times throughout the year. Permanent crops are also more likely to be insured than forage and field crops grown in California. Insured irrigated cropland requires farmers to maintain a certain level of water application to maintain insurance coverage, further reinforcing the water use implications associated with the transition from non-permanent to permanent crops (*Deryugina and Konar*, 2017).

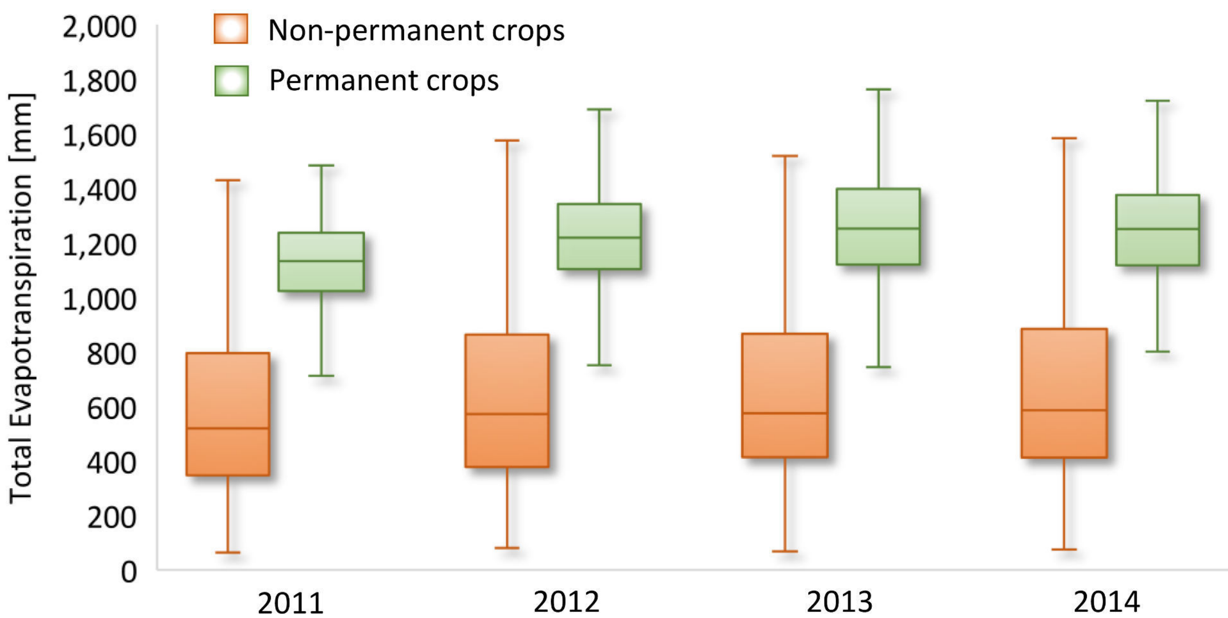

Figure 3.9: Total evapotranspiration of permanent and non-permanent crops pre-drought (i.e. 2011) and during the drought (i.e. 2012–2014) in the California Central Valley.

Fig 3.10 shows the blue ET requirement and the average revenue per hectare generated for permanent crops that saw the largest increase in harvested area and the non-permanent crops that experienced the largest decline in harvested area from 2011–2014. In the first year of the drought, these permanent tree and vine crops saw a sharp price increase, while the most widely grown non-permanent field crop prices remained relatively stable. Growing global demand for tree nuts is primarily responsible for the rise in prices and is likely to have caused the shift to more water-intensive tree nut crops. From an economic perspective, the reallocation of water to higher value uses is encouraged (*Zilberman et al.*, 2002; *Marston* *and Cai*, 2016) – this is foundational to California's water market. However, from a drought management perspective, changing cropping patterns from easily fallowed field crops to tree and vine crops reduces flexibility in the water system.

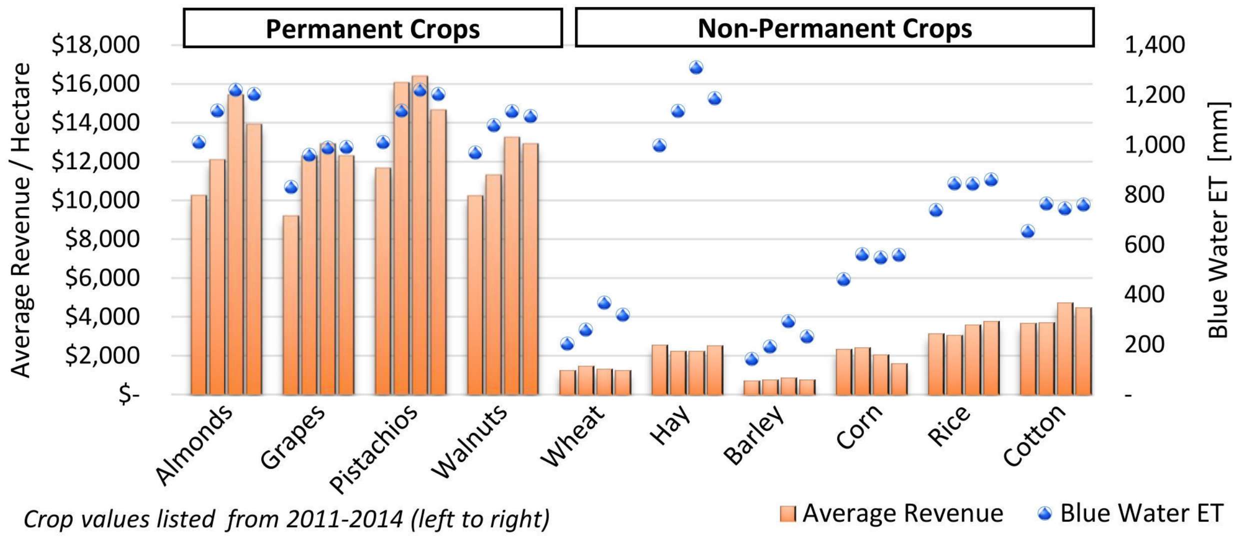

Figure 3.10: California Central Valley crop revenue and irrigation requirement from 2011 to 2014. Bars indicate the average revenue per hectare of crop production. Blue circles show the average crop evapotranspiration from irrigation. Permanent (tree and vine) crops with the greatest increase in harvested area during the drought are compared with non-permanent (field and vegetables) crops with the greatest decrease in harvested area. With the exception of hay, which is often harvested (and irrigated) multiple times per year, permanent crops have significantly higher ET requirements per hectare than the most fallowed non-permanent crops.

3.3.4 Drought impacts to food and virtual water transfers

Overall, food transfers from the Central Valley decreased by 1% from 2011 to 2014 (refer to Table 3.5). A decrease in food transfers was seen across 40% of all trade links, including 8 of the 10 largest trade links by tonnage. Transfers of cereal grains (SCTG 2) declined by 28%, with 95% of trade linkages facing a decline. Transfers of other agricultural products (SCTG 3) experienced a more modest decline of 2%, with tonnage falling across 27% of trade paths. Although production of animal feed (SCTG 4) decreased by 14% from 2011 to 2014, transfers increased by 15%. Nonetheless, 56% of animal feed was consumed, stored, or further processed in the FAF Zone of production. Note that there was large variation in food transfers during the drought depending on the transfer destination and food category.

From 2011 to 2014, total VWT from the Central Valley increased by 3% (0.51 x 10^9 m^3) (see Fig 3.4). During the same period, there was a 3% increase (0.71 x 10^9 m^3) in total WF. This can be explained by higher drought temperatures increasing crop evaporative demands and farmers switching to more water-intensive crops. These changes led to larger total water footprints and virtual water transfers, despite declines in total agricultural production and transfers. The driest year of the drought was 2013 (*Jones*, 2015), which is when VWT_{green} and WF_{green} reached their lowest values of 1.04 x 10^9 m^3 and 1.93 x 10^9 m^3, respectively. However, it was not until 2014 that reservoirs reached their lowest levels, leading to record-low distributions from federal and state water projects (*Jones*, 2015). In 2014, $VWT_{surface}$ and $WF_{surface}$ reached its lowest value of 6.98 x 10^9 m^3 and 11.19 x 10^9 m^3, respectively.

The increase of total VWT over the course of the drought can almost entirely be attributed to the 3.42 x 10^9 m^3 of additional VWT_{ground} during that same period. The increase in virtual groundwater transfers offsets the 0.94 x 10^9 m^3 reduction in VWT_{green} and the 1.96 x 10^9 m^3 decrease in $VWT_{surface}$. The Tulare Basin in particular was responsible for 59% of all VWT_{ground} from the Central Valley. Fig 3.6, shows that areas reporting greater levels of subsidence transferred 3.7 times more virtual groundwater than areas with no recorded subsidence by USGS.

Urban areas of California are major indirect consumers of Central Valley water resources. From 2011 to 2014, five major urban areas (i.e. Fresno, Los Angeles, Sacramento, San Francisco, and San Diego) utilized 12.41 ($\pm$ 0.35) x 10^9 m^3 $year^{-1}$ of virtual water from the Central Valley. In comparison, Los Angeles physical water demand and aqueduct de-

Table 3.5: Percent change [%] in agricultural transfers from the California Central Valley to major destinations from 2011 to 2014.

Destination	Cereal grains (SCTG 2)	Other agriculture (SCTG 3)	Animal feed (SCTG 4)	Total
Africa	-52%	-4%	462%	-38%
Canada	-34%	21%	71%	19%
Eastern Asia	-52%	-16%	79%	-1%
Europe	-61%	29%	26%	24%
Mexico	39%	13%	62%	32%
Rest of Americas	-13%	22%	38%	20%
SE Asia & Oceania	-63%	-37%	-32%	-40%
SW & Central Asia	-41%	8%	-44%	-25%
United States	-25%	-2%	12%	-2%
World Total	-28%	-2%	15%	-1%

liveries have averaged around 0.75 x 10^9 m^3 and 0.25 x 10^9 m^3 per year, respectively, since 1990 (*LADWP*, 2013). However, the portion attributed to each water source varied significantly between years (see Fig 3.11). In 2014, approximately 45% of surface water consumed by Central Valley crops was eaten or further processed (supporting jobs and local economies) by these five cities. Together, these urban areas experienced a 34% reduction in $VWT_{surface}$ between 2011 and 2014, reflecting a decrease in agriculture production and a switch in dependency from renewable surface water to the Central Valley Aquifer during the drought (there was an 89% increase in VWT_{ground}).

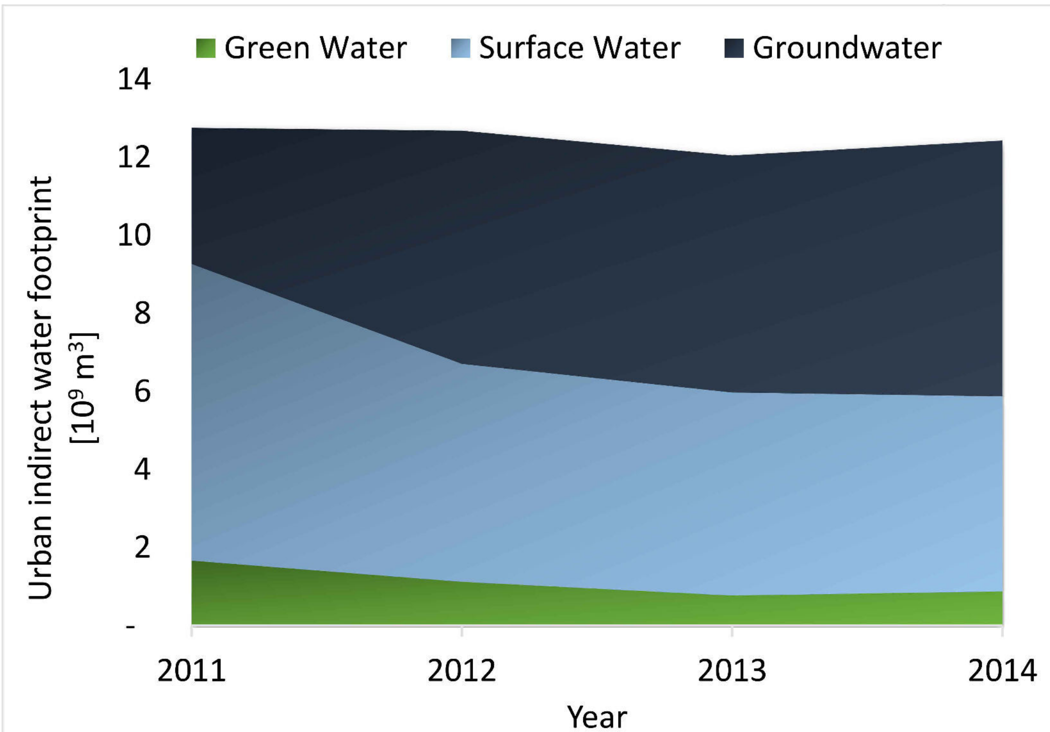

Figure 3.11: Indirect water footprint from the Central Valley of California cities: Fresno, Los Angeles, Sacramento, San Francisco, and San Diego by water source and year.

Overall, the State of California utilized 20.73 (± 1.06) x 10^9 m^3 of virtual blue water from the Central Valley region each year between 2011 and 2014. In 2011, roughly 69% of the state's virtual blue water use could be attributed to surface water sources but the fraction of surface water shrank to 43% by 2014. To put the virtual water volumes in context, in 2013, 14.60 x 10^9 m^3 of potable water was supplied for residential and non-residential users by the over 400 urban water suppliers across the state (*California EPA*, 2016). During the drought, urban water users in California were mandated to reduce their water use by 25%. California residential water users paid approximately \$1,630 per 1,000 m^3 of water in 2013 (*Gaur et al.*, 2013), while irrigators spent \$22.19 in pumping cost per 1,000 m^3 of on-farm water (surface water and groundwater) and paid \$36.96 per 1,000 m^3 for off-farm water supplies (*USDA*, 2014a). This highlights the high opportunity cost of water in agriculture in California, due to its heavy reliance on irrigation and proximity to urban areas.

Dependencies of US cities and states on the Central Valley's water resources changed significantly during the drought. Reliance on the Central Valley Aquifer more than doubled for 69 FAF Zones from 2011–2014 (refer to Fig 3.12). At the same time, 31 FAF Zones increased their utilization of both the Central Valley's surface water and groundwater during the drought. No areas saw an increase in VWT_{green} during the drought. Rural Arizona saw a significant reduction (85%) in cereal grain receipts from the Central Valley, making it the only US FAF Zone to experience decreased dependency on the Central Valley Aquifer as the drought intensified.

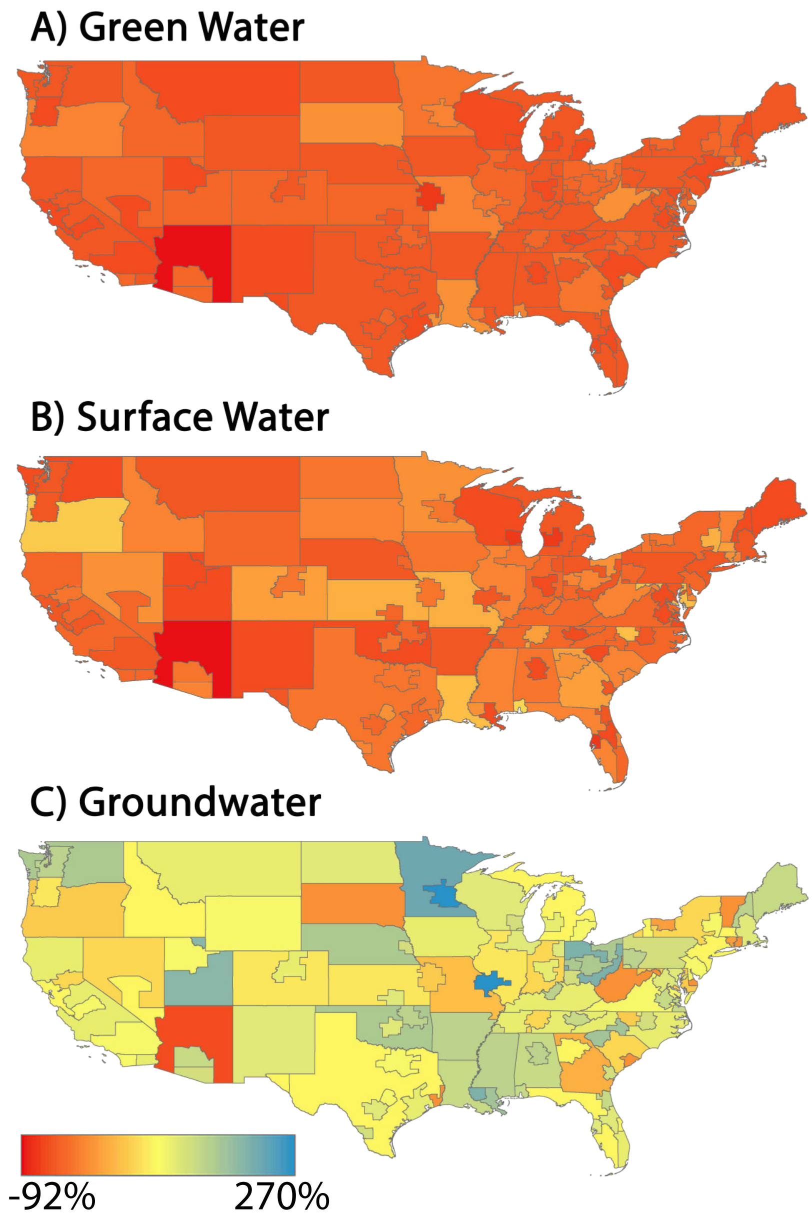

Figure 3.12: Percent change [%] in virtual water transfers from the California Central Valley to other areas of the United States between 2011 and 2014. Panels indicate green (A), surface (B), and groundwater (C) virtual water transfers within the United States. Note that green and surface virtual water transfers predominantly decrease, while groundwater transfers mostly increase.

VWT to international destinations increased by 4% during the drought. Fig 3.13 maps changes in virtual water exports from the Central Valley to international destinations from 2011 to 2014. All regions experience a decrease in VWT_{green} (see Fig 3.13), while five of eight world regions receive more $VWT_{surface}$ from the Central Valley over the course of the drought (note the predominantly blue shading in Fig 3.13B). Conversely, all areas experience an increase in VWT_{ground} during the drought, except for Africa (see Fig 3.13C). Africa's decrease in VWT_{ground} is due to a significant reduction in cereal grain exports from the Central Valley during the drought (Africa disproportionately imports more of these goods than other regions). Thus, during the California drought, global consumers relied more heavily on the overexploited Central Valley Aquifer. This demonstrates how local changes in production, such as greater reliance on groundwater during drought, can propagate through the global food system and create complex patterns of dependencies on scarce resources by distant consumers.

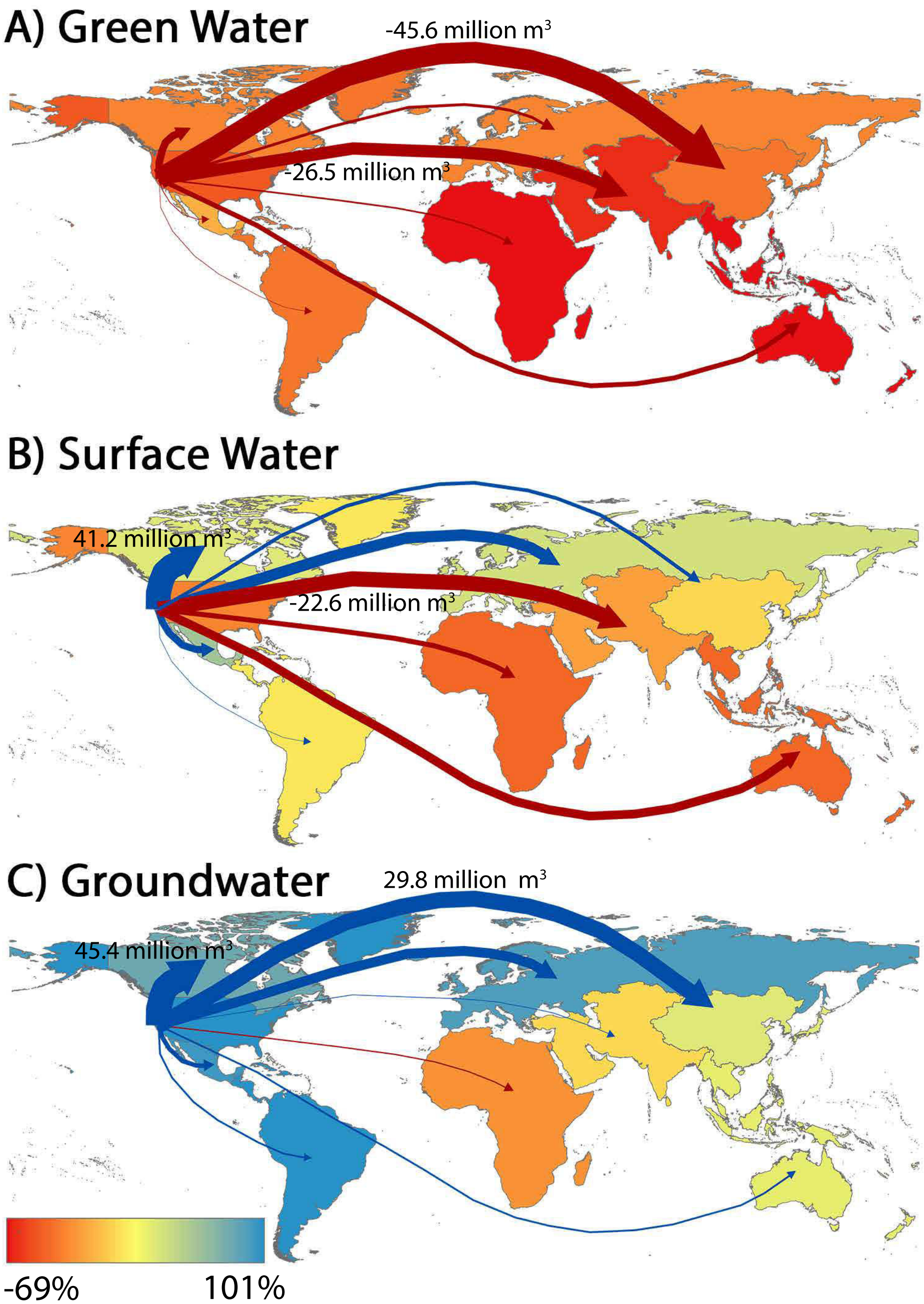

Figure 3.13: Percent change [%] in virtual water transfers from the California Central Valley to international destinations between 2011 and 2014. Panels indicate the percent change in green (A), surface (B), and groundwater (C) virtual water transfers. Arrows show the change in the volume of virtual water transfers [m^3] and are scaled relative to size. Volumes are provided for the largest links. Red arrows indicate a reduction in virtual water transfers; blue arrows signify an increase.

3.4 Concluding Remarks

In an increasingly globalized economy, it is critical that we understand how local production shocks propagate through and interact with the global food trade system. In this paper we quantified how severe drought impacted agricultural production, water footprints, and virtual water transfers of the Central Valley of California. We paired high resolution data of food commodity transfers and production with modeled estimates of water footprints by county, year, and water source to better understand the ramifications of drought for the coupled water-food-trade system.

We showed that there was a 3% increase (0.71 x 10^9 m^3) in the total WF of agricultural production over the course of the drought, due to increased crop water requirements and shifts in production patterns. In particular, the groundwater WF increased from 7.00 x 10^9 m^3 in 2011 to 13.63 x 10^9 m^3 in 2014, predominantly in the Tulare Basin. Similarly, we found that food transfers decreased by 1% (0.32 x 10^6 tons) during the drought, yet VWT increased by 3% (0.51 x 10^9 m^3). From 2011 to 2014, nonlocal groundwater VWT increased by 3.42 x 10^9 m^3, offsetting reductions in green and surface VWT (0.94 x 10^9 m^3 and 1.96 x 10^9 m^3, respectively). These findings demonstrate non-obvious patterns that emerge between drought, farmers' decisions on crop mixes and water use, and global commodity markets.

This study highlights the critical importance of existing national databases in the United States, which this study relied upon. Through this analysis, we were able to identify opportunities to improve national data collection efforts as well. In particular, the scientific and policy communities would dramatically benefit from high temporal resolution and metered water use data by source. It is important to note that we presented expected values only and do not quantify the uncertainty surrounding our results due to shortcomings in the input data. Quantifying the sensitivity and uncertainty of water footprint estimates is an area of active research (*Zhuo et al.*, 2014; *Tuninetti et al.*, 2015) and future research is needed to evaluate the additional uncertainties that are involved when commodity transfers are also considered.

Over the course of the drought, local and global consumers doubled their reliance on the Central Valley Aquifer (95% or 6.63 x 10^9 m^3). It is critical that groundwater resources are recharged following drought, so that they are there to draw upon during the next drought. Water pricing, water markets, property rights, managed aquifer recharge, and groundwater policy are critical to conserving our groundwater resources for use during future drought events (*Zilberman et al.*, 2002). Local solutions will continue to be essential to ensuring sustainability of the Central Valley Aquifer. In addition, our work enables consumers around the country and world to realize that they benefit from agricultural production that relies upon the Central Valley Aquifer. This information is critically important to help non-local Americans realize that they are connected to their distant, national resources and benefit from non-local infrastructure.

Chapter 4

HIGH RESOLUTION PRODUCTION WATER FOOTPRINTS OF THE UNITED STATES

4.1 Introduction

The United States is the largest producer and consumer of goods and services in the world. This economic activity relies on the nation's rainfall, streams, lakes, and aquifers as a fundamental input in economic production. Despite water's importance in the U.S. economy, it is often undervalued and overexploited. The first step towards sustainable, equitable, and economically efficient water use is to understand how water is currently being used throughout the nation's economy. Previous studies quantifying water use in the U.S. lack either the spatial resolution or the sectoral detail needed to get a meaningful understanding of how the country's economy utilizes and depends on its water resources. In this paper, we calculate the water footprint for over 500 unique industries and goods produced within the U.S. at an unparalleled spatial resolution. This study is the most detailed, comprehensive water footprint analysis of any country to date.

Future water availability within the United States is subject to dramatic changes in the coming decades. Growing and shifting populations, economic growth, expansion of the energy sector, as well as warming temperatures, shifting rainfall patterns, and shrinking snowpack due to climate change, will alter water supplies and demand. Furthermore, increasing water allocations to meet environmental requirements and the adjudication of Native American water-rights could further strain existing water uses (*Marston and Cai*, 2016). These issues are particularly concerning in the American Southwest, which is already water stressed and expected to face greater water scarcity in the coming decades (*Schewe et al.*, 2014). Unless we first understand the current linkages between the nation's economic production and its water resources, we will be challenged in predicting and managing future human and natural changes impact on water availability, water demand, and economic activity.

Across the U.S., water use is extremely heterogeneous, decentralized, and politically contentious, making it challenging to meter and report water consumption (also termed 'use' here) of different users. This is especially true of the agriculture industry, the largest water consumer, which has 213,621 irrigators scattered across the country (*USDA*, 2014b). Differences in state policy and water management further challenge water data availability and the development of a uniform methodology to estimate water use across state boundaries. In the absence of metered water use data collected from each water user, many turn to water use coefficients to approximate water use of an industry. Water use coefficients allow for the estimation of water use based off a known variable, such as the number of employees, the amount of energy produced, or the area of cropland, when water use is not strictly known. However, water use coefficients are often for a very narrow geographic area and industry or they are too broadly defined at the national and sector level. This makes comparison of water use infeasible across different areas and industries.

To overcome this deficiency, we calculate subnational water footprints of production (WFP) and product water footprints (PWF) for over 500 goods and services produced within the United States. In the creation of this comprehensive database, we use a variety of methods to leverage existing but disparate datasets on water use and economic production. Here, we define an industry or product's WFP as the volume of freshwater directly consumed during the present stage of production (*i.e.,* only includes water incorporated into a product or evaporated and does not include water indirectly used through the supply chain). PWF is analogous to a water use coefficient and is defined as the volume of freshwater consumed during the present stage of production, normalized by production output (in this case, U.S. dollars). We further delineate WFP and PWF by distinguishing the contribution of green water, surface water, and groundwater to meet the water requirements of the given industry or product. Surface water and groundwater can be more broadly classified as blue water, while green water corresponds to soil moisture in the root zone made available by rainfall.

Traditionally, water footprint assessments have primarily focused on agriculture, as it is the largest user of water globally (*Hoekstra and Mekonnen*, 2012). However, there is a growing body of literature that recognizes the important role other sectors of the economy play on water resources, especially at the local level (*Paterson et al.*, 2015). This follows a broader trend of moving to finer spatial resolution (*Mayer et al.*, 2016; *Rushforth and Ruddell*, 2015, 2016) and industry specific (*Blackhurst et al.*, 2010; *Wang and Zimmerman*, 2016) estimates of water use by water source — including distinguishing between groundwater and surface water (*Dalin et al.*, 2017; *Marston et al.*, 2015; *Marston and Konar*, 2017; *Schyns et al.*, 2015). *Blackhurst et al.* (2010) noted the need for subnational estimates of water use by water source for specific industries in environmental impact assessments. However, the current literature masks the significant variability in water use between regions and/or industries by providing findings at coarse geospatial or sectoral detail. Our work is a first step in resolving this shortcoming in the literature.

The purpose of this paper is to gain a spatially explicit understanding of U.S. industries dependency on different water resources to produce the goods and services demanded by society. When accurate, site-specific data is unavailable, this work provides a rich and high-resolution dataset of human-mediated water use in the U.S. that can be useful for water management and modeling, environmental life cycle assessment, water footprint assessments, demand forecasting and planning, and other US-based water studies. This study aims to answer the following questions: (i) How much surface water, groundwater, and green water is used to support production of different industries and products across the United States? (ii) How much variance in water use exist across the country within each industry? (iii) Do industries depend more on water directly or indirectly through their supply chains?

The rest of the paper is organized as follows. We highlight our primary dat a sources in Section 4.2 and describe our methodology in Section 4.3. Our results are detailed in Section 4.4. In Section 4.5, we discuss and highlight implications, as well as limitations, of our work.

4.2 Data

We leverage existing datasets on agriculture, aquaculture, mining, thermoelectric energy, hydropower, commercial, and industrial production and water use to build a comprehensive compilation of U.S. WFP and PWF. We do not include residential water use in our analysis since it is not considered an economically productive water use. Additionally, we do not include water used for golf courses and other recreational purposes (such as duck hunting or reservoirs purposed for recreation) due to incomplete data. Broadly, the data we do use can be classified as relating to either water use and supply or economic production. Data availability varied by sector and spatial scale. As much as possible, we use data within our study period (˜2010–2012). We present the finest spatial resolution that the data allows.

Table 4.2 list the major data products utilized in this study by water use category. Water use categories roughly follow those used by the U.S. Geological Survey (USGS) in their 5-year water use report (*Maupin et al.*, 2014), a foundational dataset used in this study. USGS estimates water withdrawals for each U.S. county for eight general water use categories: public supply, industrial self-supplied, domestic self-supplied, mining, irrigation, livestock, aquaculture, and thermoelectric power. We do not include the domestic self-supplied category since residential water use is outside the scope of this study. USGS public supply and industrial self-supplied water use categories fall under the 'Commercial, Industrial, and Institutional' water use category in Table 4.2. Furthermore, we more broadly define USGS's Irrigation category as 'Crops' since we include water use estimates of both irrigated and rainfed crops. Additional details relating to each data product are found throughout the methods section.

Table 4.1: Primary data sources.

Water Use Category	Data Product Purpose/Description	Source	Data Type	Finest Spatial Resolution	Data Year
Crops	Crop production (irrigated and rainfed); Crop prices	*USDA* 2014a	Production	County	2012
Crops	Crop blue and green water requirements	*Mekonnen and Hoekstra* 2011a	Water	5' x 5'	1996–2005
Crops	Groundwater and surface water irrigation fractions	*Maupin et al.* 2014	Water	County	2010
Aquaculture	Groundwater and surface water utilization; Aquaculture sales and production method	*USDA* 2014c	Water; Production	County	2012, 2013
Aquaculture	Average annual open water surface evaporation	*Farnsworth et al.* 1982; *Farnsworth and Thompson* 1982	Water	Point / isohyetal	1919–1979
Mining	Water withdrawals and consumption coefficients	*Maupin et al.* 2014	Water	County	1995, 2010
Mining	Water use coefficients	*Meldrum et al.* 2013; *University of Tennessee Center for Clean Products* 2008; *Spang et al.* 2014; *Norgate and Lovel* 2004; *Mudd* 2008, 2010; *Norgate and Haque* 2010	Water	Point	Varies
Mining	Non-fuel mineral prices	*Survey* 2017	Production	State	2012
Mining	Fuel prices	*EIA* 2017a,b; *EIA* 2013a,b	Production	State	2012
Thermoelectric	Water withdrawals and consumption	*Diehl and Harris* 2014	Water	Plant	2010
Thermoelectric	Plant fuel type	*EIA* 2017c	Production	Plant	2010
Thermoelectric	Electricity prices	*EIA* 2013c	Production	State	2012
Hydropower	Water consumption	*Grubert* 2016	Water	Region	2010–2014
Livestock	Water withdrawals	*Maupin et al.* 2014	Water	County	2010
Livestock	Livestock production and prices	*USDA* 2014a	Production	County	2012
Livestock	Water use coefficients	*Lovelace* 2009a; *Buchwald* 2009; *Carter and Neitzert* 2008; *Pugh and Holland* 2015; *Sargent* 2011; *Longworth et al.* 2013	Water	State	Varies
Commercial, Industrial, and Institutional	Water withdrawals	*Maupin et al.* 2014	Water	County	2010
Commercial, Industrial, and Institutional	Direct water requirement coefficients	*US Bureau of Economic Analysis* 2017; *Statistics Canada* 2017a,b	Water	Nation	2007, 2011, 2013

Table 4.1: Primary data sources.

Water Use Category	Data Product Purpose/Description	Source	Data Type	Finest Spatial Resolution	Data Year
Commercial, Industrial, and Institutional	Water Transfers	Numerous	Water	Point	Varies
Commercial, Industrial, and Institutional	Consumption coefficients	*US Census Bureau* 1986	Water	Nation	1982
Commercial, Industrial, and Institutional	Non-revenue water fraction	Chini et al., forthcoming	Water	City	Varies
Commercial, Industrial, and Institutional	Industry revenue and employment	*US Census Bureau* 2017	Production	County	2012

Agriculture (*USDA*, 2016) and business production data (*US Bureau of Economic Analysis*, 2017) used in this study are sometimes suppressed by government collection agencies when it may reveal information about specific companies or individuals. Instances of data suppression are flagged within the dataset, indicating that there are limited producers in that geographical area. Data suppression is more prevalent at smaller spatial scales (*e.g.,* counties) and among specialty producers. For instance, production data of the only almond farmer in Greeley County, Nebraska is flagged since reporting this data would reveal information related to that specific farmer. Like *Isserman and Westervelt* (2006), we take advantage of the hierarchical structure of the data by industry/product and geography to approximate suppressed values when encountered. Nonetheless, our study would benefit from a complete original dataset that doesn't necessitate estimates of suppressed records, which may introduce errors, especially for smaller industries and estimates at finer spatial scales.

4.3 Methods

A novel dimension of this research is the utilization of high resolution, spatially explicit, and empirically-based datasets of economic production and water use. We fuse together these disparate data sources to build a comprehensive understanding of how the U.S. economy utilizes and depends on its water resources. To date, no study has matched both the sectoral resolution and spatial detail found in this national study. Furthermore, we distinguish between surface water and groundwater footprints for each industry, highlighting each sector's dependencies on these different water resources. Indeed, we estimate state or sub-state scale surface and ground WFP and PWF for over 500 products and/or industries, including green WFP and PWF for around 140 crops. We provide PWF for every sector in units of $m^3/\$$ but also provide water use coefficients in other units when possible (*e.g.,* crops in m^3/ha; livestock in m^3 per head). Table 4.3 outlines the major contributions of our work and relates it to the current state-of-the-art with regards to spatial and sectoral resolution and water source delineation.

Table 4.2: Comparison of this study to the state-of-the-art with regards to spatial resolution, water source delineation, and product/industry specification for each water use category.

		Spatial Resolution	Water Source Delineation	Product/Industry Specification
Crops	State-of-the-art	5 x 5 arc minutes WFP and state PWF (*Mekonnen and Hoekstra*, 2011b)	Renewable and non-renewable groundwater (*Dalin et al.*, 2017) Streamflow, reservoirs, aquifers/nonlocal water sources (*Hanasaki et al.*, 2010; *Wada et al.*, 2014) Natural lakes, surface water tributaries, shallow groundwater, deep groundwater (*Mayer et al.*, 2016)	146 crops (*Mekonnen and Hoekstra*, 2011b)
	This study	Improved on *Mekonnen and Hoekstra* (2011b) estimates by using higher-resolution empirical data on agriculture production and irrigation patterns.	Green water, surface water, groundwater	141 crop WFP were estimated.
Livestock	State-of-the-art	5 x 5 arc minutes WFP (downscaled from national level); national PWF. (*Mekonnen and Hoekstra*, 2012)	Reservoirs, streamflow, non-renewable and nonlocal blue water/groundwater (*Hanasaki et al.*, 2010; *Wada et al.*, 2014)	8 livestock animals (*Mekonnen and Hoekstra*, 2012)
	This study	County level WFP and state-level PWF.	Fresh surface water and groundwater	WFP (PWF) for 9 (8) livestock animals.
Thermoelectric	State-of-the-art	Plant/reservoir level water withdrawals and consumption (*EIA*, 2017c; *Diehl and Harris*, 2014)	Specific water body (*e.g.,* Chena River) (*EIA*, 2017c; *Diehl and Harris*, 2014)	15 different generator technologies/fuel sources and 9 circulation categories. (*EIA*, 2017c; *Diehl and Harris*, 2014)
	This study	Same as state-of-the-art	Fresh surface water and groundwater	5 fuel types and 2 circulation categories.
Mining	State-of-the-art	County level water withdrawals (*Maupin et al.*, 2014)	Fresh and saline surface water and groundwater (*Maupin et al.*, 2014)	11 mining sectors (*Blackhurst et al.*, 2010)

Table 4.2: Comparison of this study to the state-of-the-art with regards to spatial resolution, water source delineation, and product/industry specification for each water use category.

		Spatial Resolution	Water Source Delineation	Product/Industry Specification
	This study	WFP at county scale.	Fresh surface water and groundwater	PWF for 15 mining sectors. WFP represent aggregate value across all mining operations.
Aquaculture	State-of-the-art	County level water withdrawals (*Maupin et al.*, 2014)	Fresh and saline surface water and groundwater (*USDA*, 2014c)	39 aquaculture species (*Pahlow et al.* 2015; indirect WF only)
	This study	State level WFP	Fresh surface water and groundwater	Represented as one sector
CII	State-of-the-art	Point withdrawals, aggregated to the county scale (*Mayer et al.*, 2016)	Lakes, surface water tributaries, shallow groundwater, deep groundwater (*Mayer et al.*, 2016)	∼380 industries (*Blackhurst et al.*, 2010)
	This study	117 state and sub-state areas	Fresh surface water and groundwater. Account for non-revenue water losses and inter-region water transfers.	378 industries.

Besides for improved sectoral resolution, a key difference between this work and water use statistics reported by U.S. government agencies is that we calculate consumptive water use, whereas their estimates are of water withdrawals (*e.g., Maupin et al.* 2014) or applied water (*e.g., USDA* 2014b). We are concerned with consumptive water use since reporting water withdrawals may overstate water scarcity and the amount of water necessitated for economic production since return flows are often reused numerous times. For instance, water withdrawals in the Colorado River Basin exceed renewable annual supply due to substantial reuse of return flows (*Richter*, 2014).

In accordance with the water footprint definition, our PWF and WFP estimates only include fresh water use. Some industries and locations may use treated or untreated saline water, which are not represented in our analysis. Saline water is not widely used in most production processes, but in a few sectors, such as mining, it may represent a significant portion of the industry's water use. *Maupin et al.* (2014) estimated that in 2010 saline water comprised 13% of the nation's total water withdrawals, with 90% of that attributed to thermoelectric power generation.

We reemphasize that this work estimates *direct* PWF and WFP (*i.e.,* water consumed in the immediate production process). Therefore, the results presented here cannot be directly compared to other studies that estimate total (*i.e.,* direct and indirect) PWF and WFP. However, our direct PWF estimates can be used within environmental extended input-output (EEIO) models to calculate direct and indirect water use, as we demonstrate later.

In the following subsections, we detail our methodology for determining WFP and PWF each broad economic sector.

4.3.1 Crops WFP and PWF

Growing crops for food, feed, biofuel, or other purposes requires a significant amount of blue and green water. Water is a critical input for crop production and in many areas the largest water user. We present county-level WFP (m^3) and water intensities (m^3/ha) of 141 crops grown in the United States based on modeled long-term estimates of blue and green crop water requirements from *Mekonnen and Hoekstra* (2011b). Additionally, we provide state-level green, blue, surface, and ground PWF (m^3/\$) for 133 crops (data limitations prohibited inclusion of some minor crops included in WFP assessment). Crop blue and green water requirements are extremely variable in many regions due to inter-annual fluctuations in precipitation and ET. Here, we present average water requirements based on 10-year climate averages. Although the production data we utilize is for 2012, the PWF and WFP we present reflect water requirements of production under average climatic conditions.

The 5 x 5 arc minute gridded global dataset from *Mekonnen and Hoekstra* (2011b) provides estimates of 146 crops' blue and green water requirements per hectare. We exclude 5 crops that are not grown in the United States or that have insufficient data coverage. Crop water requirements are derived from a dynamic water equilbruim model that computes a daily soil water balance and calculates crop water requirements and actual crop water use, attributing the proper allocation to green water and blue water sources. Water stress and non-optimal crop growth conditions are considered in the model. The model only accounts for the evapotranspiration (ET) requirement of each crop, not other potential consumptive water uses such as frost protection, field preparation, chemical application, and leaching. Although these water uses are generally small in relation to ET requirements, their exclusion means that our estimates are conservative.

We only use values corresponding to the U.S., though the model provides global output. The final water footprints of crop production as shown in *Mekonnen and Hoekstra* (2011b) rely on a global dataset of agriculture production and irrigation coverage. Here, we use more localized datasets on crop production and irrigated area from the U.S. Department of Agriculture 2012 Census of Agriculture (*USDA*, 2014a). We created a crosswalk table that matches 141 USDA reported crops to their corresponding FAO crop name used by *Mekonnen and Hoekstra* (2011b).

We average estimates of crop blue and green water requirements (m^3/ha) to the county scale to match USDA's crop production and irrigation data. For many berry, orchard, and vegetable crops, USDA does not specify whether the crop was rainfed or irrigated. In this case, we estimate county-level irrigated and rainfed area of each crop by taking the product of the crop's total harvested area and the irrigated or rainfed area fraction of the broader crop category to which it belongs (*e.g.,* blackberries use berries irrigated fraction, apples use orchard, celery use vegetables). Once we have the irrigated and rainfed crop area for each crop type for every U.S. county, we multiply it by the corresponding water requirements (blue and green for irrigated areas and green for rainfed areas).

Next, we attribute a portion of each crop's blue water footprint to surface water and groundwater sources. We use irrigation withdrawals from *Maupin et al.* (2014) to determine the fraction of total crop irrigation from surface water and groundwater sources within each county. This gives county-level, crop-specific estimates of surface water, groundwater, and green water footprints of production for 141 irrigated and rainfed crops. County-level, crop-specific data on groundwater and surface water utilization would further improve our estimates but this data is not available. Thus, we must assume that farmers do not preferentially irrigate some crops with either surface water or groundwater. Furthermore, some farmers, particularly those in the southern portion of California's Central Valley (*Marston and Konar*, 2017), employ a conjunctive use irrigation system. This means that utilization of groundwater or surface water for these areas may differ from USGS estimates during exceptionally dry or wet years.

We provide PWF estimates in two forms: the first normalized by hectares and the other normalized by dollars (see *Mekonnen and Hoekstra* 2011b for PWF in units of m^3/ton). Crop water requirements were scaled up from a grid of 5 x 5 arc minute areas to align with county boundaries. Consumptive water use per hectare is presented for each county, crop, and by irrigation practice (*i.e.,* rainfed or irrigated) for 141 crops. Additionally, we allocate total fresh water consumption to green water (irrigated and rainfed), and blue water sources, with the latter further apportioned between groundwater and surface water. The distinction between groundwater and surface water was determined as described previously. The values we present denote the average amount of water per hectare required to grow a specific rainfed or irrigated crop within a given county. This does not mean, however, that the crop was grown there. The coefficients we present would need to be paired with the crop's harvested area to calculate the WFP.

We also calculate state-level PWF (m^3/\$) for 133 crops. First, we divide the WFP for each crop by that crop's production (ton), which gives water consumption per ton of production. Next, we divide the previous quotient by crop prices (\$/ton), ending with a crop PWF in m^3/\$. Crop prices and production come from USDA 2012 Census of Agriculture (*USDA*, 2014a). National crop prices were used when state prices were not available. Like before, all PWF specify water source (*i.e.,* green water, blue water, groundwater, and surface water).

4.3.2 Aquaculture WFP and PWF

Aquaculture is the farming of aquatic organisms in a controlled water system for at least part of the year. Aquatic organisms harvested from non-controlled water (*i.e.,* wild caught) are not classified as aquaculture and thus not included in this analysis. Aquaculture products are sold for food or distributed for restoration, conservation, or sport and include different fish species, mollusks, crustaceans, and other products, such as alligators and turtles.

The water footprint of aquaculture farming is largely dependent on the production method. In 2013, there were 4,129 aquaculture operations in the United States. The primary production methods were ponds (36%), tanks (17%), raceways (9%), cages and pens (7%), and other methods (30%). Each of these methods require varying levels of water withdrawal and consumption. Only water evaporated from ponds was considered consumptive use and was directly attributable to aquaculture's WFP. The consumption of water used in aquaculture production associated with pass-through methods (such as raceways) or instream harvesting (as sometimes the case with cages and pens) is negligible. Furthermore, water withdrawn for oxygenation, waste discharge, temperature control, seasonal restocking, or seepage losses are often replacing water returned to local surface water or groundwater sources, and thus not considered a consumptive water use.

The WFP and PWF of aquaculture were calculated from data collected from USDA's 2013 Census of Aquaculture (*USDA*, 2014c) and 2012 Census of Agriculture (*USDA*, 2014a). Data limitations prohibit county level estimates, instead requiring us to only provide state estimates of WFP and PWF. Moreover, nine states aquaculture sales and distributions (representing 2.2% of the US total) were suppressed, as were freshwater surface area used in aquaculture production (representing 1.7% of the US total). Our estimates are conservative since state values corresponding to these missing records are not represented in our analysis.

Aquaculture blue WFP were estimated for each state by multiplying the total surface area of water in production by the state's long-term average open water evaporation rates. We determined minimum, maximum, and representative open water evaporation rates (mm/year) for each state based on long-term records from the National Oceanic and Atmospheric Administration (NOAA; *Farnsworth et al.* 1982; *Farnsworth and Thompson* 1982). Since evaporation rates can vary within a state, evaporation rates were collocated with the state's largest aquaculture producers and weighted more heavily in the state average. For instance, nearly all of Texas's aquaculture production occurs around the Gulf Coast where open water evaporation rates are considerably lower than in the western part of the state. Therefore, the representative open water evaporation rate was lower than a simple state average. Since annual evaporative demand varies across time and under different local conditions, minimum and maximum estimates of consumptive water use are also provided based on the range of evaporation rates commonly seen within the state. In addition, we estimate surface water and groundwater footprints by partitioning blue water use among the two sources. Following the approach of *Lovelace* (2009b), we use the number of aquaculture operations reporting groundwater and surface water use within each state to divide the blue water footprint among these different water sources.

Surface and ground PWF are given for each state in units of $\text{m}^3/\$$. These were calculated by dividing state WFP by aquaculture revenues from *USDA* (2014c).

4.3.3 Livestock WFP and PWF

Livestock directly use water for drinking, sanitation, cooling, waste disposal, onsite feed mixing, and other service activities related to animal husbandry. We estimate WFP for dairy cows, beef and other cattle, hogs and pigs, laying hens, broilers and other chickens, turkeys, sheep and lambs, goats, and equine (including horses, ponies, mules, burros, and donkeys). Estimates of water usage by specialty animals, such as alpacas and ostriches, are not included in this study. Specialty livestock represent less than 1% of all livestock inventory in the United States (*USDA*, 2014a). We assume that all water utilized in livestock production is consumed (*i.e.,* no return flows).

The WFP of livestock was calculated by multiplying county livestock population data from the 2012 Census of Agriculture (*USDA*, 2014a) by water-use coefficients from national (*Lovelace*, 2009a) and state reports (*Buchwald*, 2009; *Carter and Neitzert*, 2008; *Pugh and Holland*, 2015; *Sargent*, 2011; *Longworth et al.*, 2013). Water-use coefficients are required to estimate each animal's water footprint since most livestock water use is not metered or reported (*Lovelace*, 2009a). Livestock water use varies between States depending on farming practices and production methods, climatic conditions, and water availability. Some states have developed water-use coefficients for internal use but do not publish them to protect the privacy of livestock producers. Other states do not estimate state-specific coefficients, instead choosing to use the median water-use coefficient of each animal, as reported in *Lovelace* (2009a), when reporting livestock water use for the national USGS water use report.

We use publicly available state livestock water-use coefficients when available; however, when state water-use coefficients are not available, we estimate them. First, to estimate state coefficients we start by using the national median water-use coefficients (*Lovelace*, 2009a). For each state, the national coefficients are scaled so that the product of these coefficients and the state's animal inventory (*USDA*, 2016) equal the state's total water use as estimated in the *Maupin et al.* (2014). This approach allows us to estimate water-use coefficients for each state by replicating the approach used by USGS (*Lovelace*, 2009a) but in reverse (*i.e.,* we calculate water use coefficients for each animal, not total water use). The water-use coefficients are then checked to ensure they fall within the range of potential values reported by *Lovelace* (2009a).

Water use coefficients were converted from water use per head to groundwater and surface water use per U.S. dollar (*i.e.,* PWF). First, we calculate each animal's direct water use over its lifetime. The average animal's lifespan is calculated as the inverse of the inventory turnover rate. For example, the U.S. hog inventory in 2012 was 66,026,785 and the national hog slaughter total for 2012 was 113,246,600. This means that inventory turnover rate is 1.72 hogs per year (113,246,600 / 66,026,785) and thus, the average hog lifespan is 0.58 years (1 / 1.72). The animal's lifetime direct water use is the product of its daily water use coefficient and the animal's lifespan. The animal's lifetime direct water use is then divided by the total value of the animal and its derived products (*e.g.,* the total output value of a dairy cow is both the milk it produces and its slaughter price). This represents each animal's PWF in $m^3/\$$. Estimates of PWF, as well as WFP, were further partitioned into surface water and groundwater using the fraction of total livestock withdrawals from groundwater and surface water sources from *Maupin et al.* (2014).

Water use for each animal type was estimated for each U.S. county for the year 2012 (the year of the most recent agriculture census). We multiplied state specific water-use coefficients for each animal by each county's animal population within the state. Here, we assume that livestock inventories remain relatively stable throughout the year (*i.e.,* livestock sold for slaughter or that die are replaced by a new animal). We provide county WFP in terms of groundwater, surface water, and total blue water for each animal.

4.3.4 Commercial, industrial, and institutional WFP and PWF

Commercial, industrial, and institutional (CII) water use represent the water required to manufacture or process goods and provide services. Industrial water use is primarily used for heating and cooling (heat transfer), processing, fabricating, washing, diluting, or is incorporated into a product (*e.g.,* beverage manufacturing). Institutional and commercial water use is water used by motels/hotels, restaurants, hospitals, retail and grocery stores, office buildings, warehouses, schools, government, and other commercial facilities to serve the requirements of customers, employees, members, visitors, and/or students, as well as to maintain the premises (including heating, cooling, cleaning, and landscape irrigation).

We determined the WFP and PWF of 378 different CII enterprises across 117 different geographical areas within the United States. The boundaries of these 117 areas align with the Census Bureau and Department of Transportation Commodity Flow Survey (CFS) boundaries (though we merge a few together). Henceforth, we refer to these 117 areas as CFS Areas. These CFS Areas represent the largest U.S. cities (amounting to 78% of the nation's CII economic activity), with remaining land area classified along state or rest of state boundaries. Fig 4.6 shows the boundaries of our analysis. For each CFS Area, we also report PWF of every industry in m^3 of water consumption per dollar of revenue. We employ a methodology similar to *Blackhurst et al.* (2010) but we improve upon these methods in the following ways: i) we report subnational values, whereas they only report nation statistics; ii) we report values in terms of water consumption; iii) we account for non-revenue water losses; iv) our results offer a range of potential values capturing some of the uncertainty in these estimates.

Like *Blackhurst et al.* (2010), to estimate CII water use we begin with estimates of water withdrawals from the USGS national water use report (*Maupin et al.*, 2014). Whereas *Blackhurst et al.* (2010) used water use data from 2000, we employ the 2010 dataset. CII water users retrieve their water from public supply and/or self-supplied surface water and groundwater sources. Public supply systems are defined as public or private water providers having a minimum of 15 service connections or serving at least of 25 people. Public supply water is often locally sourced but can be conveyed across county or even state boundaries. Industries that supply their own water typically locate near a water body due to the nature of some water rights (*e.g.,* riparian water rights require adjacency to the water body) and the high cost of transferring water. We utilize different methods to disaggregate water use to each industry depending on the water supply source (public supply or self-supplied). Each industry's total water use is the sum of its self-supplied and publicly-supplied water.

We begin by disaggregating public supply water withdrawals and delivers to specific industries. As noted previously, the location public supply facilities withdraw water may not be the place where the water is used. Thus, the first step is to determine the net water usage within each CFS Area, after accounting for water imports and exports. CFS Areas stretch beyond traditional metropolitan boundaries, typically fully encompassing all major water distribution systems. Thus, in most instances water withdrawals and water use are contained within the same geospatial boundary. However, there are several instances of large water transfers across CFS Area boundaries. Journal searches, review of city, state, and national reports, quarrying online databases, and personal communications with personnel at state and federal agencies were employed to identify and quantify water transfers. Water transfers were subtracted from the exporting area and added to the importing area. It is likely that some small public supply systems straddle two or more CFS Areas and transfer water across borders. It is infeasible to capture all of these instances across the nation (there are roughly 156,000 water distribution systems in the US; *EPA* 2008), especially since most states do not collect this type of information. Here, we assume that small inter-CFS Area transfers cancel one another out and/or are negligible.

Once the total public water supply utilized in the area was determined, we calculate how much of that water was delivered to CII users. First, non-revenue water (NRW) was subtracted from total public water supplies. NRW is water that is lost through real losses (*e.g.,* leaks), apparent losses (*e.g.,* theft or faulty metering), or authorized but unbilled water use (*e.g.,* street cleaning or fire-fighting). This water is withdrawn and enters the distribution system but is not delivered to a paying customer. Web queries of major metropolitan water districts, governmental and American Water Works Association (AWWA) reports, and personal communication with water district staff were used to estimate NRW as a percent of total produced water for each CFS Area. When a recent NRW value was not available for a CFS Area, the median value of 15% was used, as recommended by *Solley et al.* (1998). The NRW percentage was multiplied by total produced water to determine the amount of water lost and that delivered to paying end-users.

Next, water deliveries to domestic users was deducted. Domestic water deliveries were estimated for each county in *Maupin et al.* (2014). These deliveries represent the water that reaches and is used by residential consumers. This is typically estimated using surveys, meter and billing data, and/or water use coefficients. We aggregate domestic water use from the county to the CFS Area and then subtract this from the total water sold within the CFS Area. We assume that the remaining water is all sold to CII users. The last time it was recorded (*Solley et al.*, 1998), only 0.3% of public water supplies went to thermoelectric power generation. It was therefore considered negligible. The following equation summarizes how we calculate total CII water deliveries.

$$(PS_{CFS} \pm NWT_{CFS}) \cdot (1 - NRW_{CFS}) - DOD_{CFS} = CIID_{CFS} \tag{4.1}$$

where PS is public supply withdrawals, NWT is net water transfers, NRW is the fraction of produced water that is considered non–revenue water, DOD is water deliveries to domestic users and $CIID$ is water deliveries to commercial, industrial, and institutional users within a given CFS Area.

Total water deliveries to CII users within a CFS Area were allocated to each industry according to their purchases from the 'water, sewage, and other systems sector' (*US Bureau* *of Economic Analysis*, 2017). This approach assumes a uniform pricing structure across all users and that the price of water relative to sewage is also constant across all industries within a CFS Area. Water purchases per unit of production (*i.e.,* direct requirement coefficients) were taken from the US Bureau of Economic Analysis (*US Bureau of Economic Analysis*, 2017) 2007 input-output direct requirement table. The product of an industry's direct requirement coefficient and it's reported revenue (*US Census Bureau*, 2017) yields the total water purchased by that industry within a CFS Area. These water purchases were normalized by dividing them by the sum of all CII water sales in the CFS Area. The fraction of total water purchases allocated to an industry is then multiplied by the CII water deliveries. This gives an industry-specific estimate of publicly supplied water deliveries ($S_{i,CFS}$) for each CFS Area, as depicted in the following equation:

$$\frac{WP_{i,CFS}}{\sum_{i \in CII} WP_{i,CFS}} \cdot CIID_{CFS} = S_{i,CFS} \tag{4.2}$$

where WP are water purchases of a given sector i within a particular CFS Area.

Industrial water use is often self-supplied, not purchased. Estimates of industrial self-supplied water withdrawals from (*Maupin et al.*, 2014) were allocated to industries manufacturing and processing food and beverage products, textiles, wood and paper, metals, minerals, petroleum, plastics, machinery, electronics, and other goods. Following the approach of *Blackhurst et al.* (2010), industrial water withdrawals per employee were taken from recent Canadian water use and employment surveys (*Statistics Canada*, 2017a,b). Although U.S. based estimates of water use per employee exist (*e.g., Davis et al.* 1987), the more recent estimates from (*Statistics Canada*, 2017a) better capture the potential changes in water use due to the significant changes and automation in the manufacturing sector over the last few decades. For each CFS Area, coefficients of water use per employee were multiplied by the number of employees (*US Bureau of Economic Analysis*, 2017) within the corresponding industry. Water withdrawal estimates were scaled to match self-supplied industrial water withdrawals reported by (*Maupin et al.*, 2014). The industrial self-supplied water allocation procedure is summarized as follows:

$$\frac{WC_{i,CFS} \cdot E_{i,CFS}}{\sum_{i \in I} WC_{i,CFS} \cdot E_{i,CFS}} \cdot TIW_{CFS} = IW_{i,CFS} \tag{4.3}$$

where WC is the coefficient of water withdrawal per employee for industry i, E is the number of employees, TIW is the total industrial self-supplied water withdrawals within a CFS Area, and IW is the water withdrawals of a specific industry.

Finally, we add self-supplied water use and water deliveries from public supply to derive each industry's total water use within each CFS Area. We assume that each industry within a CFS Area uses the same fraction of surface water and groundwater, although their total consumptive volume from each water source may differ. Industry-specific consumption coefficients (*US Census Bureau*, 1986) are applied to determine the consumptive water use, or WFP, of each industry. An industry's surface water and groundwater footprints are divided by the industry's revenue to calculate the corresponding PWF. In the end, we estimate 378 industry-specific WFP and PWF for 117 distinct locations within the U.S.

4.3.5 Thermoelectric and hydropower generation WFP and PWF

Collectively, thermoelectric power facilities withdrawal the largest volume of water in the United States. Thermoelectric power plants convert water to steam to turn turbines, which produces electricity. The largest use of water by thermoelectric facilities, however, is cooling the steam. Water withdrawals and consumption differ by fuel type utilized by the power plant (*i.e.,* fossil fuels, nuclear fission, or geothermal energy) but the major factor behind differences in water withdrawals and consumption is whether the plant uses open-loop or closed-loop cooling systems (*Macknick et al.*, 2012). Open-loop cooling systems (found predominately among older plants) withdrawal 96% more water than the same plant with a closed-loop cooling system (*DeNooyer et al.*, 2016). However, closed-loop systems, which recirculate water through the system many times, consume around 60% more water than an open-loop cooling system.

The U.S. Energy Information Administration (EIA) is the chief U.S. agency responsible for reporting water withdrawals and water consumption associated with thermoelectric power production. However, EIA thermoelectric water use estimates have been criticized due to data inconsistencies, incompleteness, and data quality issues (*Averyt et al.*, 2013; *Diehl et al.*, 2013). Moreover, a report by the USGS (*Diehl and Harris*, 2014) shows that most of the reported EIA values of both withdrawal and consumption were not thermodynamically plausible. Withdrawals reported by EIA were 24 percent higher than modeled estimates, while reported consumption was 8 percent lower *Diehl and Harris* (2014). Given the noted shortcomings of the EIA dataset, we use plant-level modeled estimates of consumptive water use by water source (fresh groundwater or surface water) and cooling system from *Diehl and Harris* (2014). These estimates are constrained and validated against collected data and heat and water budgets, which consider electricity production and fuel use. Fresh water consumption is summed across all thermoelectric plants within a county to arrive at county level WFP estimates.

Next, we calculate PWF for each plant, as well as aggregate state values. At each spatial scale, PWF are presented by cooling system, fuel type, and fresh water source. Water use is normalized by net energy generation (TJ) and, at the state level, also by revenue ($). State electricity prices (*EIA*, 2013c) and plant net energy generation (*Diehl and Harris*, 2014; *EIA*, 2017c) where collected and used in the calculation. We assume that electricity generated is sold at the average electricity price of the state where the electricity is produced.

A national estimate of consumptive water use attributable to hydroelectricity generation was taken directly from *Grubert* (2016). *Grubert* (2016) estimates net and gross evaporation associated with each US reservoir that is purposed with hydroelectricity generation. Furthermore, the author provides PWF (m^3/GJ) for 20 different regions within the United States. There are several critical assumptions made in this study, namely the allocation of storage space and evaporation amongst multi-purpose reservoirs, that lead to significant uncertainty, especially when evaluating individual reservoirs. Therefore, we only report the national WFP and do not detail other values here.

4.3.6 Mining WFP and PWF

Water is used to quarry minerals and ores, as well as in other mining operations such as crushing, screening, washing, floatation, and dust suppression. Water is also injected into the ground to extract crude petroleum and natural gas, which we also included here as a mining water use. In addition to withdrawing and consuming water, mining operations can also 'produce' water through the dewatering of mines or as a by-product of oil and gas extraction. This water is not included in our analysis unless it is used for a beneficial purpose in the mining operation, such as re-injection or dust suppression. Furthermore, the water used to transport or process raw minerals, ores, petroleum, or natural gas is not included here but as an industrial water use.

We estimate surface and ground WFP associated with the mining industry based off withdrawals from *Maupin et al.* (2014) and consumption coefficients from *Solley et al.* (1998). Due to limited extraction data for many of the materials, we are unable to attribute the WFP to specific minerals, ores, natural gas, or oil. Instead, we multiply consumptive water use coefficients by total mining withdrawals from *Maupin et al.* (2014) to estimate the WFP of the entire mining industry within each county. There is uncertainty in estimates of mining water withdrawals and consumption since very few mining operations track and report their water usage. Moreover, mining water usage can be highly variable depending on mining technique, climate, and available water supplies. Therefore, these estimates of mining WFP, much like mining water withdrawals reported by USGS, should be viewed as a first-order approximation. Although these estimates provide a more complete picture of the U.S. WFP, additional data on material- and site-specific water consumption would improve our understanding of how water is used within the industry in a spatially explicit way.

Localized data on mine and well productivity and water use limit mineral specific WFP estimates. Yet, through a review of current literature, we provide PWF estimates for 15 mined products (see Table 4.2 for list of references). We give preference to studies based in the United States but, due to limited research in mining water use for many materials, we also report PWF from international studies as well (mostly in Australia). We provide representative or median PWF values for all products in m^3 of consumed water per \$ of sales, as well as m^3 of consumed water per ton of production (or m^3 of production for liquids and gases). For energy materials, such as coal, natural gas, crude oil, and uranium, we also estimate PWF in m^3 of consumed water per TJ of energy potential. The 2012 prices of all minerals and ores came from USGS (*Survey*, 2017), while energy products came from EIA (*EIA*, 2013a,b; *EIA*, 2017a,b). When possible, we give the minimum, maximum, 1st quartile, and 3rd quartile PWF estimates for each material.

4.4 Results

4.4.1 Water footprint of US production

Water is used to produce the goods and services demanded by society. In the United States, we find that roughly 7.30 x 10^{11} m^3 of water is consumed annually to maintain the nation's economic production. Of this, 6.03 x 10^{11} m^3 is from green water sources used to grow crops for food, feed, and fuel. Surface water (6.68 x 10^{10} m^3) and groundwater (6.11 x 10^{10} m^3) are valuable inputs in the production of irrigated agriculture but also every other economic sector. For perspective, productive blue water use is roughly 15 times less than the U.S. total average annual runoff between 2010–2014 (1.96 x 10^{12} m^3; *USGS* 2017) and 1.5 times less than the water supply and irrigation storage capacity (1.94 x 10^{10} m^3; *US Army Corps* *of Engineers* 2017). Furthermore, blue water consumption is roughly one-third of all water withdrawals. This seems to suggest an abundance of water but national statistics such as these mask localized water shortage and scarcity.

The clear majority (95.4%) of the U.S. WFP is attributable to crop production. Green water comprises 86.5% of all consumptive crop water use, while surface water and groundwater makeup 5.9% and 7.6%, respectively. Roughly 84% of U.S. harvested crop area is strictly rainfed, with most of this cropland dedicated to corn, soybeans, wheat, hay and haylage grown in the Midwest and High Plains. Corn grain and silage, hay and haylage, rice, wheat, soybeans, cotton, and almonds are among the largest surface water and groundwater users. Together, these seven crops are responsible for 75% of the nation's total groundwater consumption and 47% of its surface water consumption. Fig 4.1 shows the crops with the largest WFP by water source, while Fig 4.2 illustrates the spatial distribution of U.S. surface, ground, green, and total crop water consumption.

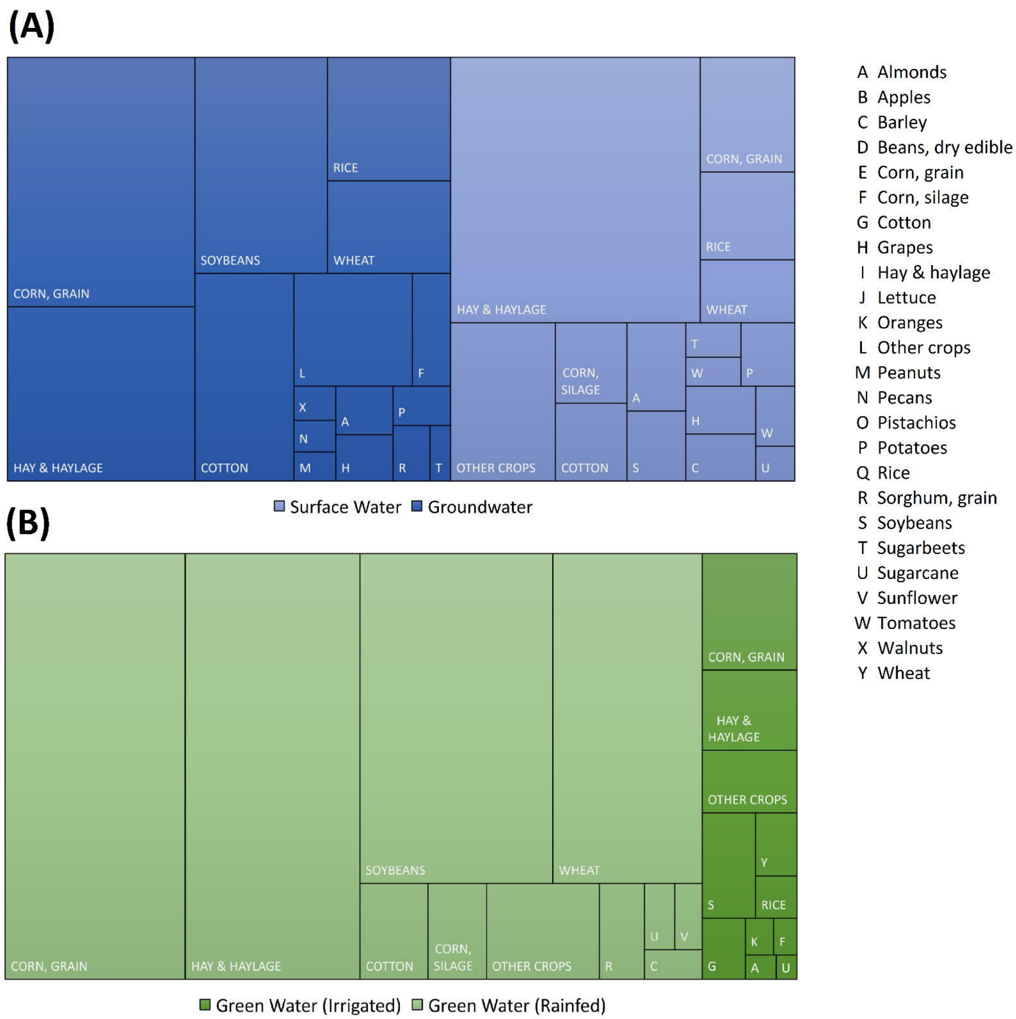

Figure 4.1: Water footprints of U.S. crop production by crop: (A) surface water and groundwater consumed by irrigated crops, and (B) green water consumed by irrigated and rainfed crops. Boxes in each panel are scaled relative to size; however, sizes cannot be compared between panel A and B, as green water consumption is substantially larger than blue water consumption.

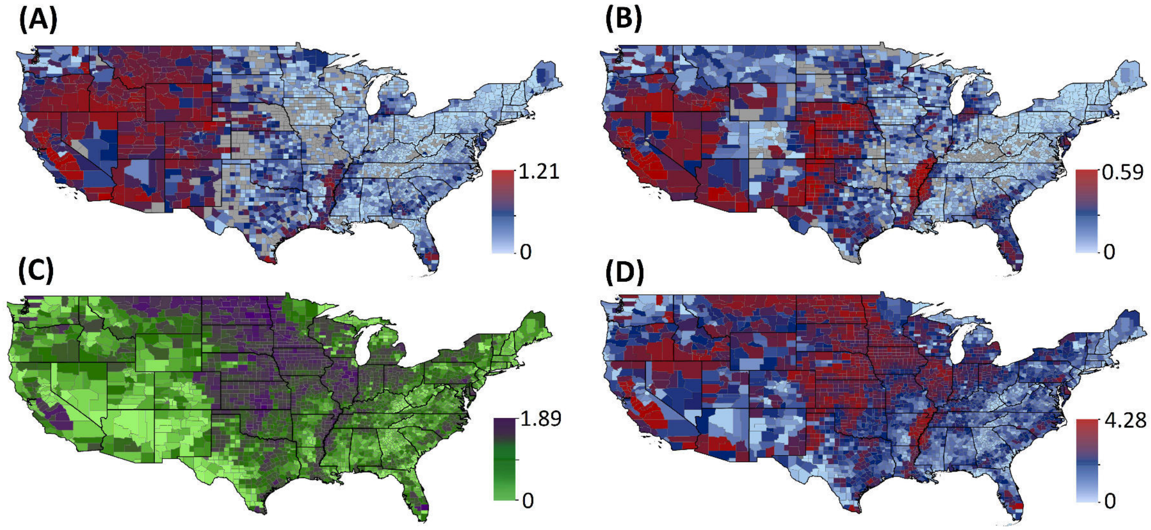

Figure 4.2: Crop surface (A), ground (B), green (C), and total (D) water footprint of production [km^3] for each county in the conterminous United States.

Non-crop industries contribute 6.4% and 19.5% of the U.S. groundwater and surface WFP, respectively. Fig 4.3 indicates that most (1.59 x 10^{10} m^3, or 61.7%) of the surface WFP is due to evaporative losses from hydropower reservoirs and non-revenue water losses in municipal distribution systems. Thermoelectric power generation, though the sector with the largest water withdrawals, consumes the third most surface water and seventh most groundwater amongst non-crop sectors. This is because 97.5% of thermoelectric freshwater withdrawals correspond to power plants that employ once-through cooling systems, which typically consume only 1-3% of withdrawals. Fig 4.4 illustrates the blue WFP for thermoelectric power generation for each U.S. county. Animal husbandry requires 2.59 x 10^9 m^3 blue water annually, with nearly half of the sector's water use occurring in the Great Plain States and California (see Fig 4.5 for county level map of livestock blue WFP). Beef cattle make up 56.0% of livestock water consumption nationally.

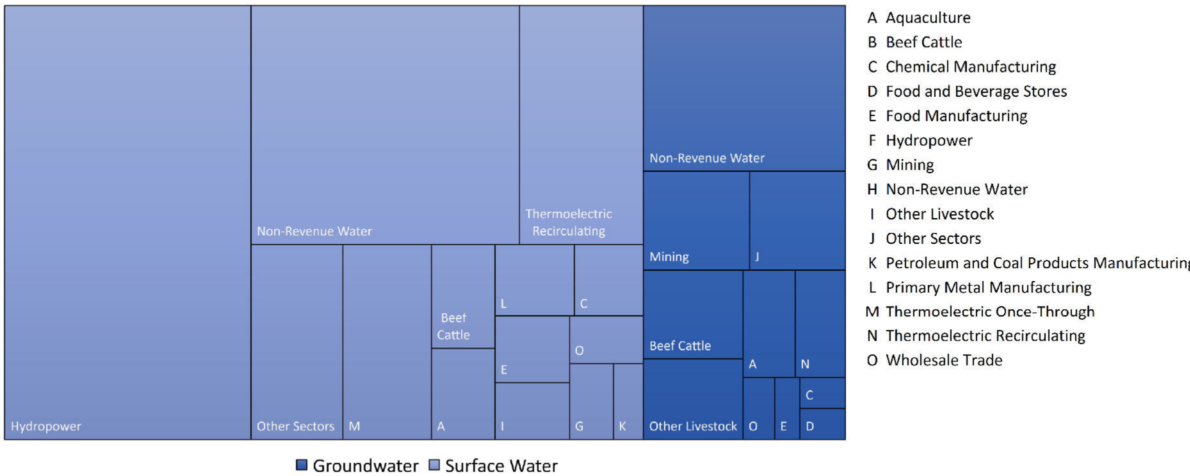

Figure 4.3: Groundwater and surface water footprint of production of non-crop sectors in the United States. Tiles are scaled relative to size.

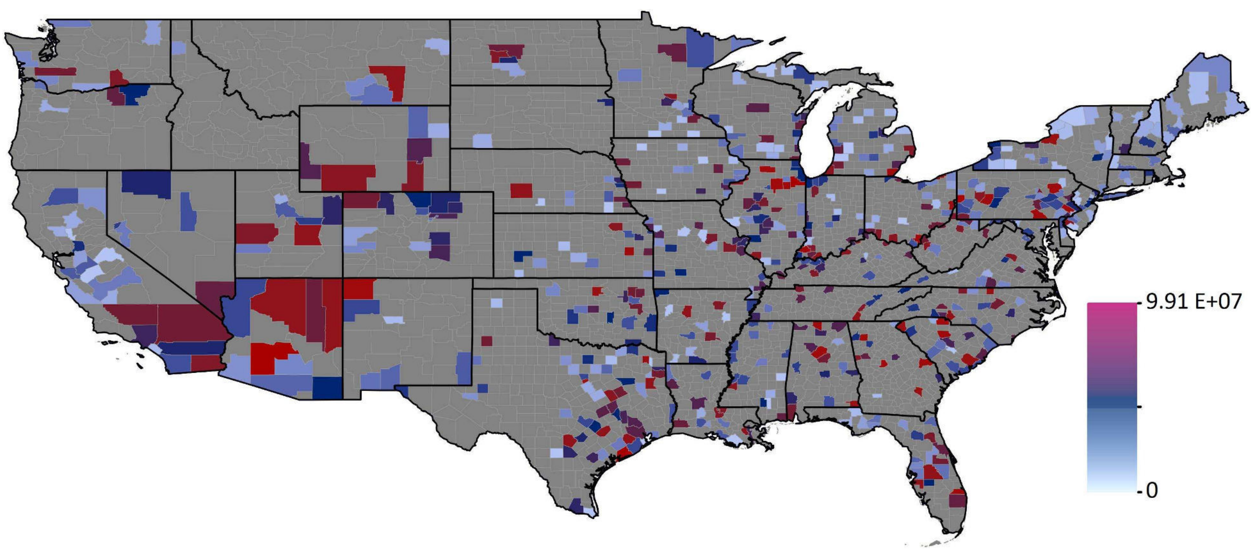

Figure 4.4: County-level blue water footprints of thermoelectric power generation. [m^3]

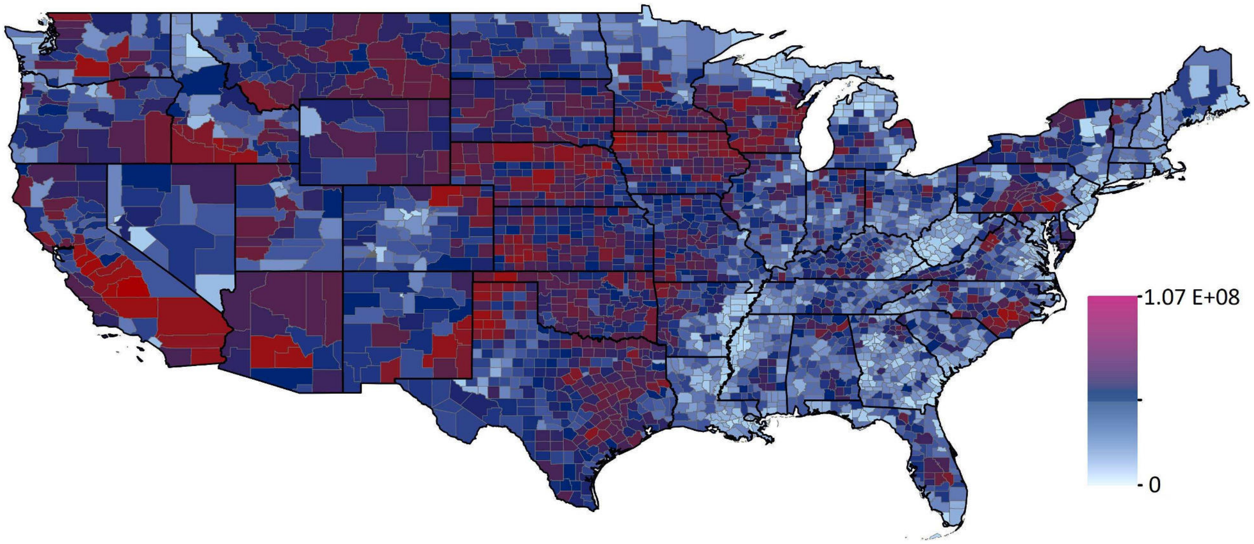

Figure 4.5: County-level blue water footprints of animal husbandry. [m^3]

The manufacturing and service sectors consumed 2.82 x 10^9 m^3 and 2.32 x 10^9 m^3 of blue water per year, respectively. The manufacturing sector relies less on groundwater than service industries, with only 21.6% of its WFP coming from groundwater sources, compared to 37.2% from the service sector. The top five water consumers are primary metal manufacturing, food manufacturing, chemical manufacturing, wholesale trade, and food and beverage stores. Whereas the first three manufacturing sectors can attribute their large WFP primarily to their high PWF, the two service sectors have modest PWF but their large WFP is a product of the sheer size of their economic production within the U.S. economy.

Fig 4.6 shows the spatial variation in blue PWF and WFP of the combined manufacturing and service sectors. Generally, states with larger manufacturing economies (primarily in the Rust Belt, the Southeast, Texas, and California) have the largest WFP. The cities with the largest CII blue water footprint are Chicago, Los Angeles, and New Orleans. Laredo, Texas has the smallest CII WFP of any city, largely since much of its publicly supplied water is from brackish groundwater that is not included in WFP estimates. New York City and Boston have the smallest blue PWF due to their focus on high value service industries, while New Orleans and West Virginia have the two largest PWF since their economies rely more on water-intensive heavy industries.

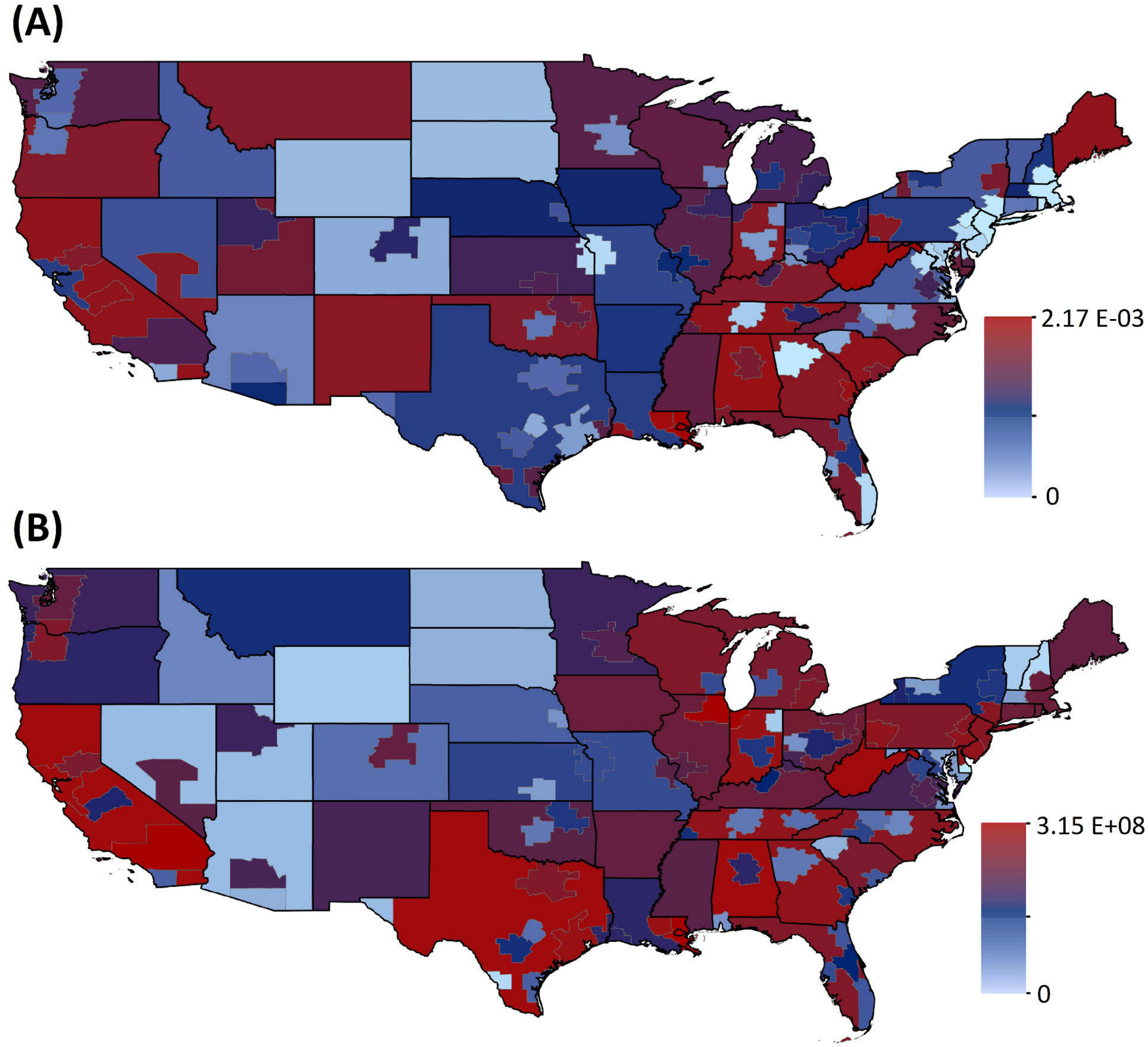

Figure 4.6: Average PWF (A) and combined WFP (B) of the manufacturing and service sectors for each CFS Area in units of [m^3\$$^{-1}$] and [m^3], respectively.

There is significant spatial variability in both total water consumption and sectoral water dependencies across the U.S. Fig 4.7 reveals the sector with the largest blue WFP in each county. Eight of the ten counties with the largest WFP are in California, with irrigated agriculture as the leading water user in each. Ten percent of U.S. counties are responsible for seventy-four percent of the nation's blue WFP. Clustering is evident in many places, such as cereal farming in the Midwest and High Plains, dairy farming in Wisconsin, New

York, and Pennsylvania, fruit and nut farming in California and Florida, and other livestock production (namely pigs) in Iowa and North Carolina.

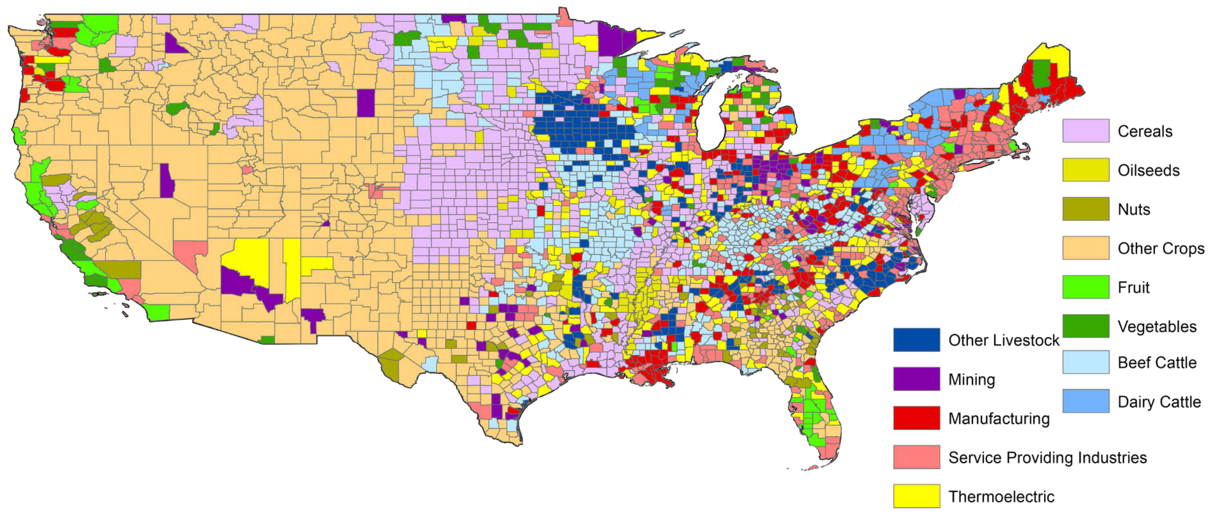

Figure 4.7: Sector with largest blue WFP in each U.S. county. Agriculture related industries are the largest water user in 2,164 of the 3,143 U.S. counties and county equivalents. Service providing industries (354), thermoelectric power generation (289), manufacturing (234), and mining (102) are the dominant water users in other counties around the nation. Hydropower, aquaculture, and non-revenue water uses are not included since county level data is not available for these water uses.

4.4.2 Water for food and energy production

Water is a critical input in the production of food and energy across the United States. Yet, water is not utilized uniformly across the nation to produce either food or energy, nor does water use perfectly align with population (see Fig 4.8). In fact, there is a strong demarcation around the 97th meridian, which, not coincidently, is where precipitation (P) equals potential evapotranspiration (PET). Nearly 80% of blue water consumed in food production (both crop and livestock) and 90% of irrigation dam storage is west of the P=PET line. Alternatively, 80% of water use for thermoelectric energy generation occurs east of the 97th meridian. Since food and energy products are ultimately for human consumption, this spatial mismatch between the FEW system and population demonstrate how people are supported by production, water consumption, and infrastructure located in distant 'elsewhere'.

The impact of groundwater in agriculture production is also evident from Fig 4.8. Groundwater makes crop production possible in many regions and can serve as a buffer against drought and changing climates. The rate of blue water consumption sharply increases as irrigators unsustainably pump large volumes of water from the Central Valley, High Plains, and Mississippi Embayment aquifer systems to grow crops. Over two-thirds of aquifer depletion in the United States between 1900-2008 has occurred in these three aquifers (*Konikow*, 2013), which are critical to local economies, as well as domestic and international food supplies (*Marston et al.*, 2015).

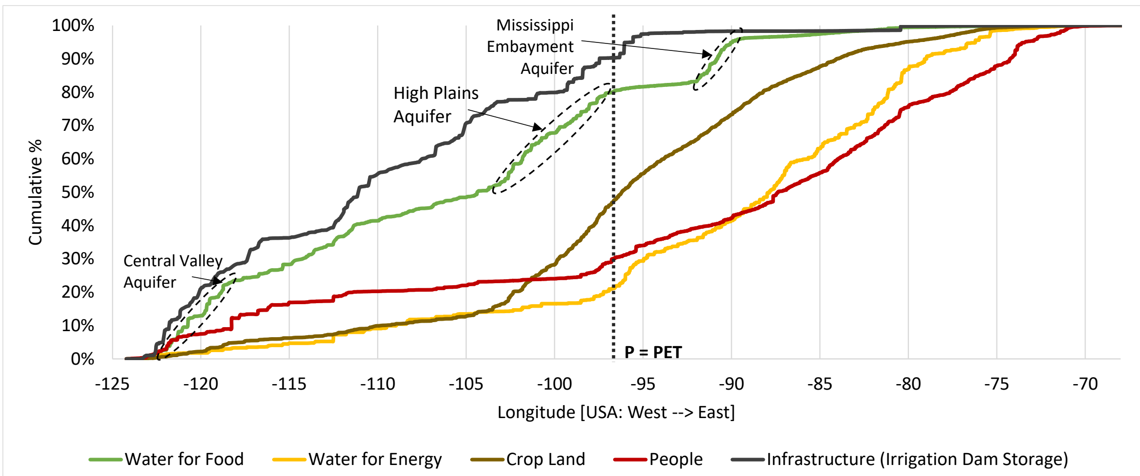

Figure 4.8: Cumulative blue water footprint of U.S. food and thermoelectric energy production moving from the Western U.S. to the Eastern U.S. The P = PET line represents the approximate location where precipitation equals potential evapotranspiration.

4.4.3 Product water footprints

We calculated the PWF for 133 crops, 8 livestock animals, 378 commercial, industrial, and institutional sectors, 15 mined resources, thermoelectric power generation (by 5 fuel types and 2 circulation types), and aquaculture. Together, this makes for the most comprehensive grouping of PWF in the literature to date. Moreover, most of the PWF values were calculated at the state or sub-state level. This allows for spatial differentiation and provides a range of potential PWF values, as opposed to one value for the entire nation, which doesn't capture the spatial variability or diversity in water users.

Over 50,000 unique PWF were calculated for different industries and products across the nation. In Table 4.3 we provide statistics on blue PWFs after grouping them into broader sectoral categories. There is significant variability within each sectoral category due to production differences between individual products or industries within a sector, spatial differences in water utilization intensities, and different production efficiencies within the same industry.

Table 4.3: Statistics of blue PWF (m^3/$1000) by broad sectoral categories.

Sector	5th Percentile	25th Percentile	Median	75th Percentile	95th Percentile
Nuts	0.00E+00	7.85E+00	7.55E+01	3.03E+02	1.94E+03
Fruit	2.27E+00	1.75E+01	4.81E+01	1.56E+02	8.69E+02
Vegetables	0.00E+00	2.44E+01	6.21E+01	1.64E+02	7.77E+02
Cereals	0.00E+00	0.00E+00	1.96E+01	5.15E+02	2.90E+03
Oilseeds	0.00E+00	0.00E+00	9.94E+00	4.70E+02	2.05E+03
Other Crops	0.00E+00	0.00E+00	2.34E+01	6.98E+02	3.45E+03
Beef Cattle	2.07E+01	2.42E+01	3.12E+01	3.76E+01	4.39E+01
Dairy Cattle	3.86E+00	5.11E+00	6.08E+00	7.35E+00	8.79E+00
Other Lifestock	1.93E+00	3.87E+00	1.03E+01	2.86E+01	5.29E+01
Aquaculture	3.24E+00	2.52E+01	1.09E+02	4.35E+02	7.42E+03
Manufacturing	1.34E-02	4.73E-02	1.24E-01	4.67E-01	4.03E+00
Service Providing Industries	5.66E-03	2.44E-02	6.17E-02	1.52E-01	5.81E-01
Mining	4.78E-04	4.58E-01	1.53E+00	5.26E+00	1.96E+01
Thermoelectric	1.99E+00	8.32E+00	1.63E+01	2.03E+01	2.29E+01

The greatest variability in PWF is within crop farming sectors. This variability is due to local climate differences, management decisions (*e.g.,* planting and harvesting dates, irrigation practices), and differences in water requirements by crop type. Although not represented in Table 4.3, crop farming sectors also utilize green water to meet its direct water requirement. In some locations and for certain crops, irrigation water (*i.e.,* blue water) is not applied, as the crop is strictly rainfed. This is represented by blue PWF values of zero in Table 4.3.

4.4.4 Direct and indirect water use

An economic input-output (IO) matrix can be paired with our estimates of direct PWF to calculate the direct and indirect water footprint of each sector. An IO table represents interdependencies between industries by showing intersectoral input purchases required to produce output in each sector. For instance, leather manufacturing requires purchases from the cattle industry, which requires feed purchases from the grain industry, who in turn purchases from the fertilizer industry and so on. The total direct and indirect water requirements throughout a product's supply chain can be calculated using an environmentally extended version of the Leontief IO model:

$$w = k(I - A)^{-1}y \tag{4.4}$$

Table 4.4: The direct and indirect blue water footprint of the 'sugar and confectionery product manufacturing' sector. ($1.07 / 5 lb.; *Index Mundi* 2017)

	$m^3/\$1000$			*gallons / 5 lb. bag*		
Sector Name	**Low**	**National Average**	**High**	**Low**	**National Average**	**High**
Direct Water Use						
Sugar and confectionery product manufacturing	0.61	1.44	3.25	0.17	0.41	0.92
Indirect Water Uses						
Sugar cane and sugar beets farming	0.30	37.35	412.99	0.09	10.55	116.61
Oilseed farming	0.02	3.75	61.45	0.00	1.06	17.35
Grain farming	0.00	3.56	22.12	0.00	1.01	6.25
Electric power generation, transmission, and distribution	0.01	0.29	0.51	0.00	0.08	0.14
Paperboard container manufacturing	0.02	0.07	0.14	0.00	0.02	0.04
Fertilizer manufacturing	0.01	0.03	0.03	0.00	0.01	0.01
All other indirect uses	0.59	11.43	79.01	0.17	3.23	22.31
Total, all sectors	1.55	57.91	579.51	0.44	16.35	163.62

where w is a vector of industry direct and indirect water consumption (m^3) due to the final demand of y goods ($). Direct water consumption is water either purchased or self-supplied by sector j to produce its goods, while indirect water consumption is water used in sector i production whose products are input to sector j output. k (m^3/$) is a row vector of direct water footprints per dollar of output for each industry (*i.e., PWF*). In the IO literature, the k vector of PWF values is considered an environmental multiplier. Finally, I is the identity matrix and A is the direct requirement matrix, which represents how much input from sector i is needed to produce one unit of output in sector j. Total requirements $[(I - A)^{-1}]$, which represent total industry inputs (direct and indirect) to deliver one dollar of industry output to final users, were taken from (*US Bureau of Economic Analysis*, 2017).

Table 4.4 provides an example of the direct and indirect blue water footprint of $1,000 in 'sugar and confectionery product manufacturing' production. Additionally, Table 4.4 shows the direct and indirect WF for a 5 lb (2.27 kg) bag of sugar. This example mirrors that of *Blackhurst et al.* (2010), but here we demonstrate how national average water use coefficients (as used by *Blackhurst et al.* 2010) mask significant variability in water use across the country. We compare model outputs using the national average, high, and low blue PWF estimates. The high and low PWF vector assume each sector sources its inputs from the most inefficient and efficient blue water users, respectively.

The example in Table 4.4 shows that total water consumption associated with the national average blue PWF is roughly 37 times larger than if the sugar manufacturer sourced its inputs from the most efficient blue water users. Alternatively, if the sugar manufacturer sourced from the most inefficient water users, it would consume 10 times as much water through its supply chain as the national average value. Direct water use by the sugar manufacturing industry is a small contributor to the overall blue PWF. The largest determinant of the sector's PWF is the amount of blue water used to grow agricultural products, namely sugar cane and sugar beets, that are utilized throughout the industry's supply chain. Whether these crops are primarily rainfed (as the case in the 'Low' PWF case) or are heavily irrigated ('High' PWF) can have dramatic implications on the sugar industry's total blue PWF. In fact, agricultural products contribute only 35% of the sugar industry's total blue PWF in the 'Low' scenario but are responsible for over 99% of total water consumption in the 'High' scenario, whose total blue PWF is over three orders of magnitude larger than the 'Low' scenario. Water use attributed to other sectors can also differ by one to two orders of magnitude but the relative effect of these changes on the overall PWF is small.

Sugar refining, like most industries, consume more water indirectly than directly. This point was made by *Blackhurst et al.* (2010), who found that indirect water use exceeded direct water use for 93% of U.S. sectors. We too find that, on average, 93% of sectors use more water through their supply chains than they consume directly. However, these statistics provide a national average value and therefore disguise significant water use variability within each sector and water use through an industry's supply chain. For instance, if all sectors sourced their inputs from the most efficient water users, only 72% would use more water indirectly than directly. Alternatively, if all sectors were supplied by the most inefficient water users, 99% would depend more on indirect than direct sources of water (see Fig 4.9).

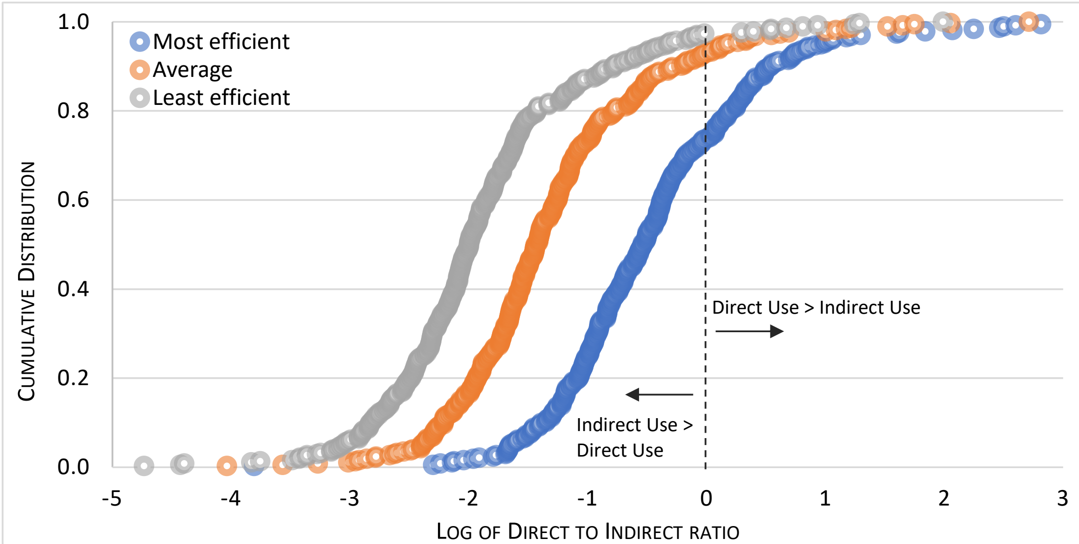

Figure 4.9: Cumulative distribution of direct to indirect water consumption ratio for U.S. economic sectors assuming inputs to production are sourced from i) the most efficient water users, ii) water users matching the national average efficiency, and iii) the least efficient water users within each sector.

Together, the previous examples demonstrate the importance of further spatial refinement in water footprint and life cycle impact assessments. Otherwise, one may significantly over- or under-estimate water use, as well as the impacts associated with its use.

4.4.5 Validation and uncertainty

Water footprint assessments, including this one, are subject to considerable uncertainty. To date, only *Zhuo et al.* (2014) has conducted a rigorous uncertainty analysis on water footprint estimates. Others (*e.g.*, *Wang and Zimmerman* 2016; *Mayer et al.* 2016; *Grubert* 2016 have performed a basic sensitivity analysis on highly uncertain parameters to demonstrate how this uncertainty impacts their findings. A lack of water metering and insufficient data availability make it challenging to validate findings at the sector and spatial scale found in our study. Nonetheless, we compare our results to other studies, albeit at coarser spatial or sectoral resolutions, to determine the reasonableness of our results. We would expect our findings to be similar but not exactly that of others since study periods and methods (*e.g.*, some report withdrawals, not consumption, or classify sectors differently) do not perfectly align. In addition to this comparison, we perform a sensitivity analysis by isolating and varying key variables to assess their impact on WFP and PWF values.

Table 4.4.5 shows that our results compare favorably with previous water footprint studies and government reports on water use within the United States. At the national level, our estimates of crop blue, green, surface and groundwater footprints fall between estimates within the literature and government reports. Water withdrawals, when possible, are converted to consumption values for more direct comparison. For instance, *Maupin et al.* (2014) and *USDA* (2014b) only report water withdrawals and applied water, respectively. Irrigation efficiencies and conveyance losses were taken from *Stewart and Howell* (2003) and *Brouwer et al.* (1989), respectively, to convert estimate what portion of water use was consumptive. An average consumptive coefficient was calculated for each area by taking the weighted average irrigation efficiency, with the fraction of cropland employing each irrigation technology acting as the weight. The crop area utilizing each irrigation method comes from *Maupin et al.* (2014) and *USDA* (2014b).

Table 4.5: Comparison of estimates of United States freshwater use by sector (km^3y^{-1}).

	This study	*Hoekstra and Mekonnen* (2012)	*Wang and Zimmerman* (2016) *	*USDA* (2014b)	*Mubako* (2011)	*FAO* (2017)	*Maupin et al.* (2014)	*EIA* (2017c)
Study Period	~2010–2012	1996–2005	2007	2013	2008	2008–2010	2010	2010
Crops (Green)	603.1	612.0			239			
Crops (Blue)	B: 94.0	95.9	79.7	*Applied water* *	93	108.5	*Withdrawal* *	
	GW: 52.9			B: 109.2			B: 157.8	
	SW: 41.1			GW: 59.8			GW: 67.5	
				SW: 49.3			SW: 90.3	
				WFP†			*WFP*†	
				B: 82.0			B: 101.3	
				GW: 45.1			GW: 50.1	
				SW: 36.9			SW: 51.2	
Livestock (Blue)	2.6	3.4	2.6			2.8 *	2.8 *	
Aquaculture	1.1					13.0 *	13.0 *	
Mining	1.3		3.2			7.4 *	3.1 *	
Thermoelectric	4.7		3.0				161.6 *	3.3
Industrial	2.8	11.0 ‡	3.8			22.1 *	20.7 *	

Table 4.5: Comparison of estimates of United States freshwater use by sector ($km^3 y^{-1}$).

	This study	*Hoekstra and Mekonnen* (2012)	*Wang and Zimmerman* (2016) ⋆	*USDA* (2014b)	*Mubako* (2011)	*FAO* (2017)	*Maupin et al.* (2014)	*EIA* (2017c)
Study Period	~2010–2012	1996–2005	2007	2013	2008	2008–2010	2010	2010
Commercial/ Services	2.3		2.1					

⋆ Values not provided in manuscript. Authors provided us values for the United States.

† Values represent estimates of consumptive volumes. B = Blue Water; GW = Groundwater; SW = Surface Water

‡ Includes mining, industrial, and thermoelectric water uses.

* Values represent water withdrawals or applied water and are included to give context. They are not directly comparable to our estimates.

At the state-level, there is greater variability between our results and those reported by *Maupin et al.* (2014) and *USDA* (2014b) (see Fig 4.10). However, discrepancies between *Maupin et al.* (2014) and *USDA* (2014b) estimates are greater than between our study and each of these studies. Variances in water use estimates are likely due to differences in study years and methodology. For example, we use long-term average crop water requirements, whereas *USDA* (2014b) estimates of applied crop water correspond to 2013 farmer surveys, and *Maupin et al.* (2014) uses a variety of techniques to estimate water withdrawals which differ by state and may utilize key data sources from various years (*Dickens et al.*, 2011).

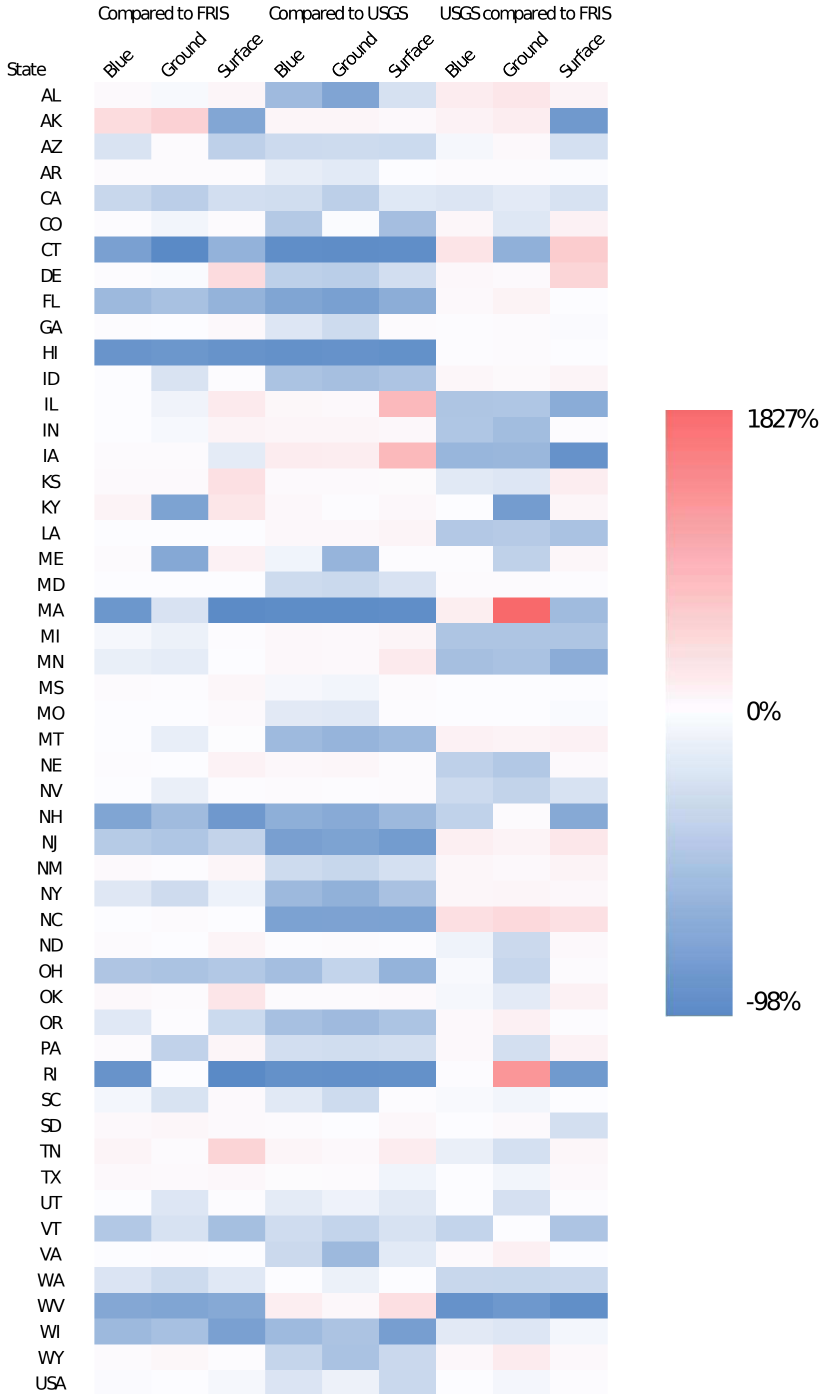

Figure 4.10: Comparison of state crop blue water, surface water, and groundwater consumption estimates between this study, *USDA* (2014b), and *Maupin et al.* (2014). There is larger variability between studies at the state-scale than the national scale, especially among states with smaller irrigation requirements. The largest discrepancies are when comparing *USDA* (2014b) and *Maupin et al.* (2014).

In addition to comparing our results to existing estimates in Table 4.4.5, we perform a sensitivity analysis on key parameters to capture some of the uncertainty within our results. A recent study by *Zhuo et al.* (2014) allows us to approximate the uncertainty in our total crop WFP estimates. Using the same crop water model as we utilize in this study, *Zhuo et al.* (2014) found that the average uncertainty in crop water footprints was $\pm30\%$ (at 95% confidence level). Two-thirds of the uncertainty in crop WF was due to uncertainty in precipitation and reference evapotranspiration estimates. It is important to note, however, that their study did not have the range of crop coverage as our study, nor was it based in the United States. The primary focus of our sensitivity analysis, though, is non-crop sectors since these sectors have fewer points of comparison in the literature to validate our findings, as demonstrated by Table 4.4.5.

Fig 4.11 illustrates the sensitivity of non-crop sectors blue WFP to critical parameters and assumptions in our analysis. The bar chart represents the expected WFP, while the error bars depict the WFP range under varying assumptions. High and low WFP estimates of mining, manufacturing, and service providing industries were calculated by varying each industry's consumption coefficient. Our results are particularly sensitive to consumptive use coefficients, which exhibit a great deal of variability and uncertainty. Following *Mayer et al.* (2016), we use the consumptive coefficients representing the first and third quartile of coefficient values as our upper and lower bounds. In addition to adjusting consumption coefficients, we also varied non-revenue water fractions by $\pm20\%$ based on average inter-annual variability seen in cities where we collected records for multiple years. The high and low values shown for manufacturing and service providing industries are a combination of reduced/increased consumption coefficients and increased/reduced non-revenue water percentage. The former determines how much supplied water is consumed, while the latter determines how much water is supplied from municipal sources. The combination of decreased (increased) consumption coefficients and increased (decreased) non-revenue fraction yield conservative bounds on our estimates, assuming each extreme scenario is independent and unlikely to actually occur simultaneously.

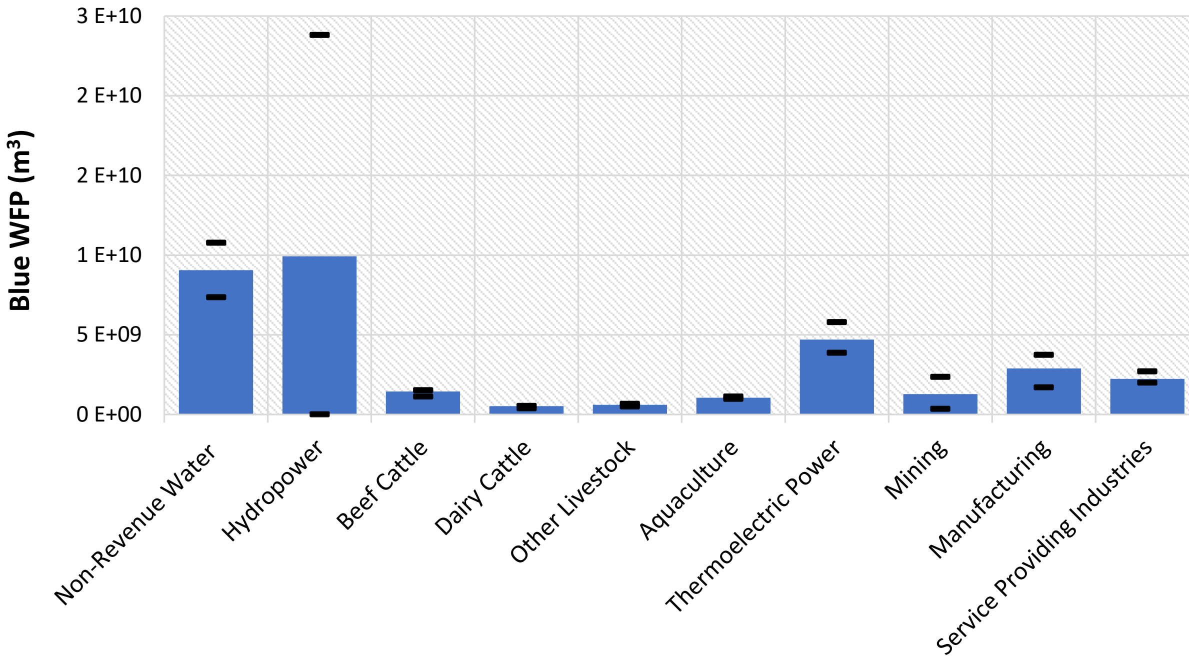

Figure 4.11: Blue water footprint of production [m^3] for non-crop sectors in the U.S. with error bars representing WFP sensitivity to changes in key parameters or assumptions.

For all livestock sectors, we calculate low and high WFP estimates by adjusting each state's animal water-use coefficients to match coefficients representing the first and third quartile coefficients seen across all 50 states. Since evaporative demand is highly variable and uncertain, aquaculture WFP is bounded by high and low estimates of open water evaporative demand seen within each state of production (*Farnsworth et al.*, 1982; *Farnsworth and Thompson*, 1982).

There is a great deal of uncertainty surrounding WFP in the energy sector, particularly hydropower. Hydropower WFP estimates are most sensitive to how water consumption is allocated among multipurpose reservoirs different users. The low and high estimates in Fig 4.11 represent two common assumptions found in the literature on how to allocate reservoir evaporation among multiple users: i) no water is allocated to hydropower or ii) all water consumption is allocated to hydropower. Other methods based on economic valuation or equal weighting of all dam purposes provide estimates between the ranges we present. The bar graph value is the representative value used in this study and follows the assumption of *Grubert* (2016), who only assigns evaporative demand to hydropower if it is the dam's primary purpose. High and low estimates of thermoelectric WFP come directly from (*Diehl and Harris*, 2014). These upper and lower bounds are determined by adjusting model parameters and inputs within plausible ranges.

4.5 Discussion

Water is a critical input in U.S. economic production. The evolving human and natural pressures placed on the country's water resources necessitate a better understanding of how the nation utilizes its water resources within its economy. This will enable us to better predict how future changes will impact water availability, water demand, and economic activity. In this paper, we quantify WFP and PWF of food, energy, services, manufacturing, and mining products produced within the U.S. and, in doing so, create the most detailed, comprehensive water footprint assessment of any country to date. Understanding the range of potential water use within an industry allows us to explore opportunities for water savings and benchmarking along the supply-chain, as well as assess direct and indirect dependencies on water.

We find that the U.S. economy directly utilizes 7.30 x 10^{11} m^3 of the nation's water resources annually to produce goods and services demanded by domestic and international consumers. This amounts to over one and a half times the volume of Lake Erie, America's fifth largest fresh water lake and the seventeenth largest in the world. The majority (83%) of the country's water use is attributed to green water used to support crop farming. Although much attention has been given to blue water in the literature and in practice, the sheer volume of green water required to sustain economic production and livelihoods, both directly through support of farming occupations and indirectly through the supply chain, calls for it to be given more consideration in water management. Green water is often disregarded because it has little opportunity cost given that it cannot be readily used to meet water requirements of most water users. However, optimal use of green water through enhanced crop genetics, better farm management, and strategic regional incentives and coordination can lead to greater production and free up blue water resources for more economically productive uses (*Marston and Cai*, 2016; *Davis et al.*, 2017). In areas of water scarcity, blue water conservation and reallocation to more valuable uses can be encouraged through institutional measures, such as water right banking, option markets, and aquifer and river basin caps.

Our results show clustering of some sectoral water users, especially in the agricultural industry. This economic clustering can leverage local economies of scale, natural resources, and comparative advantages; however, it can also make industries and their supply chains more exposed to water-related shocks, such as drought and floods, which are expected to increase under climate change (*IPCC*, 2014). Policies and subsidies have encouraged increased agricultural productivity at the expense of agricultural diversity. Yet, recent research (*Swenson*, 2010) has shown the value of regionally diversified crop production, including growing fruits and vegetables along the Corn Belt. As water becomes scarcer due to increasing demands and more variable supplies, policy makers and planners should consider both economic productivity and resilience to local and nonlocal water-related shocks, which can propagate through supply chains.

This paper demonstrates the significant variability in total water consumption and water consumption per dollar of output between sectors and locations. Even within a relatively narrow sector, such as manufacturing or oilseed farming, WFP and PWF can differ by several orders of magnitude depending on local conditions and individual industry processes. For this reason, continued spatial and industrial refinement in water use estimates is needed.

Our results do not replace user-specific water use data, which better capture nuances of individual users not reflected in our study. There is a need for annual surveys of water use, much like is done in the agriculture industry by USDA. Surveys could be bolstered by strategic water metering and remote sensing applications to validate survey results and fill in coverage gaps. Furthermore, disparate local, state, and national water agencies should coordinate their efforts and develop a common platform for collecting water use data. Data should be reported along meaningful geopolitical and hydrologic boundaries and at more refined sectoral resolution so that it can be useful for both local and regional planning and management. We hope demonstrating the usefulness of these data and identifying current data limitations will bring increased attention and resources needed to improve water use metering, data collection, and reporting.

Chapter 5

AN OVERVIEW OF WATER REALLOCATION AND THE BARRIERS TO ITS IMPLEMENTATION

5.1 Introduction [1]

Traditional water supply and demand management techniques have faced challenges in many regions throughout the world trying to adapt to the rapid changes in overall water availability and demands. Over 2.3 billion people (approximately one-third of the global population) live in areas with chronic water shortage (*Rockström et al.*, 2014), despite that many of these regions already have water supply and demand mechanism in place to better utilize their water resources. Globally, numerous basins are closing or are already closed, that is, all or most of the renewable water is already allocated or committed for some purpose (*Gleick and Palaniappan*, 2010). Water management institutions, infrastructure, and water laws that were put in place decades or even centuries ago are not equipped to handle these rapidly evolving conditions. Water reallocation can inject antiquated water management systems with the flexibility they need to meet the most pressing demands within regions facing water scarcity.

Water reallocation is the transfer of water among users who are committed formally or informally to a certain amount of water, for example, by water right (also known as a water entitlement), water use permit, or agreement, when the existing allocation is physically impossible, economically inefficient, or socially unacceptable. Some authors (*Vaux*, 2012) consider the initial development of a water resource as reallocating water from environmental to human uses; however, we contend that for water to be reallocated it must first be formally or informally committed to a water use (*i.e.,* allocated). We follow the precedent in the literature and use the terms 'reallocation' and 'transfer' interchangeably; however, the term transfer has a broader meaning in that it is not necessary that the water be initially allocated to a specific purpose.

[1] This chapter is published as an article in Wiley Interdisciplinary Reviews: Water, 2016 (*Marston and Cai*, 2016)

Water reallocation can act as an important piece component in a water supply and demand management portfolio and has been shown to improve the costs, adaptability, and performance of a water resources system (*Kasprzyk et al.*, 2009; *Zhu et al.*, 2015). Water supply management primarily involves structural measures to increase the availability, reliability, and quality of water resources for productive uses, while demand management intends to reduce the use of water through increased efficiency from source to disposal. Reallocation, while considered by some as strictly a demand management strategy (*Palomo-Hierro et al.*, 2015; *Ghimire and Griffin*, 2014; *Chong and Sunding*, 2006), may not directly yield overall water savings but it can greatly increase the benefits received from water use. Reallocation can, however, prompt water conservation since it incentivizes an increase in water productivity and allows any superfluous water to be transferred to more efficient users (*i.e.,* more benefits derived per unit of water), essentially providing additional water supplies for new or growing demands, especially those with higher marginal values of water use. In this way, reallocation acts as a two-way bridge connecting the purposes of both water supply and demand management. Where traditional engineering supply measures and conservation efforts fall short, reallocation will prove to be critical in ensuring the benefits derived from a region's water resources are optimized and water is used in a more sustainable way.

Researchers (*Cai et al.*, 2015; *Molle and Berkoff*, 2006; *Hadjigeorgalis*, 2009), practitioners (*Johnson et al.*, 1990), and politicians (*Committee on Western Water Management*, 2012) have cited the benefits of water reallocation, yet its realization is far behind what is needed or expected and when implemented it is not as effective as theorized (*Eden et al.*, 2008; *Giannoccaro et al.*, 2013).As more regions search for ways to deal with growing water scarcity, this paper provides a timely overview of water reallocation based on a comprehensive search of English language journal articles and reports, including major works and important recent contributions. The purpose of this review is to draw a consensus amongst the current literature as to the primary ways water reallocation has taken place around the world and the major obstructions to its wider implementation. In addition to overviewing the current body of literature, which primarily focuses on the institutional, economic, and social obstructions to water reallocation, we also demonstrate how greater input from the natural sciences and engineering fields can lead to more holistic understanding and solutions to water reallocation impediments. We discuss reallocation cases from six continents to demonstrate the ubiquitousness of water reallocation and give specific context to major issues and/or concepts; nevertheless, our paper reflects the literature, which primarily focuses on countries with higher occurrences of water reallocation (*e.g.,* United States, Australia, Spain, and

China). Although each area faces unique challenges when reallocating water, many of the issues this paper highlights (*e.g.,* third-party effects, lack of information support, transaction cost) are evident in most cases of reallocation, regardless of a specific region's governing or socioeconomic system.

This paper begins by explaining the impetuses for water reallocation, which are broadly categorized as water supply forces and water demand forces. Next, the various forms of reallocation are outlined, including voluntary versus non-voluntary reallocation, intra-sectoral versus inter-sectoral reallocation, temporary versus permanent reallocation, and local versus non-local transfers. Major obstacles to wider implementation of water reallocation are discussed, as well as measures taken in practice or proposed in the literature to resolve hindrances to effective water reallocation. The paper concludes by calling for researchers to work within a broader interdisciplinary framework for water reallocation so that it can become a more viable tool for water planners and managers around the world.

5.2 Impetus for Water Reallocation

Changes in societal preferences toward how water is distributed across all users, coupled with evolving water demands and limited water supplies, prompt governments to establish the requisite framework for water reallocation. In some regions, water supply development and demand management will continue to be sufficient in managing water resources; yet a growing number of regions around the world are facing inadequate water supplies to fulfill unsustainable demands, with some regions already past "peak water" and moving toward even less water availability. When a river basin becomes fully allocated (*i.e.,* "closed"), establishing a means for reallocation to occur is critical to allow for new water-dependent development to take place; otherwise water allocations, and hence development, will be relatively fixed. Fig 5.1 depicts the typical development of water resources, starting at a point (i) where renewable water supplies (blue outer circle) can easily meet all water demands (red inner circle). Water supplies expand through engineering measures as demands for water grow (ii). As water demands approach available water supplies, water is typically allocated or committed to specific users or purposes (division of the demand circle amongst different users or sectors, represented by different colors). Next, water demands begin to outpace renewable supplies and traditional supply and demand management strategies are unable to fully reconcile the difference between water supplies and demands; thus, the water source is overcommitted, leading to water scarcity and resource overexploitation in many places (iii).

Water reallocation allows for the redistribution of water to new, growing, more productive, and/or more needed uses from a social perspective and also provides a mechanism to bring water use back within sustainable limits (iv).

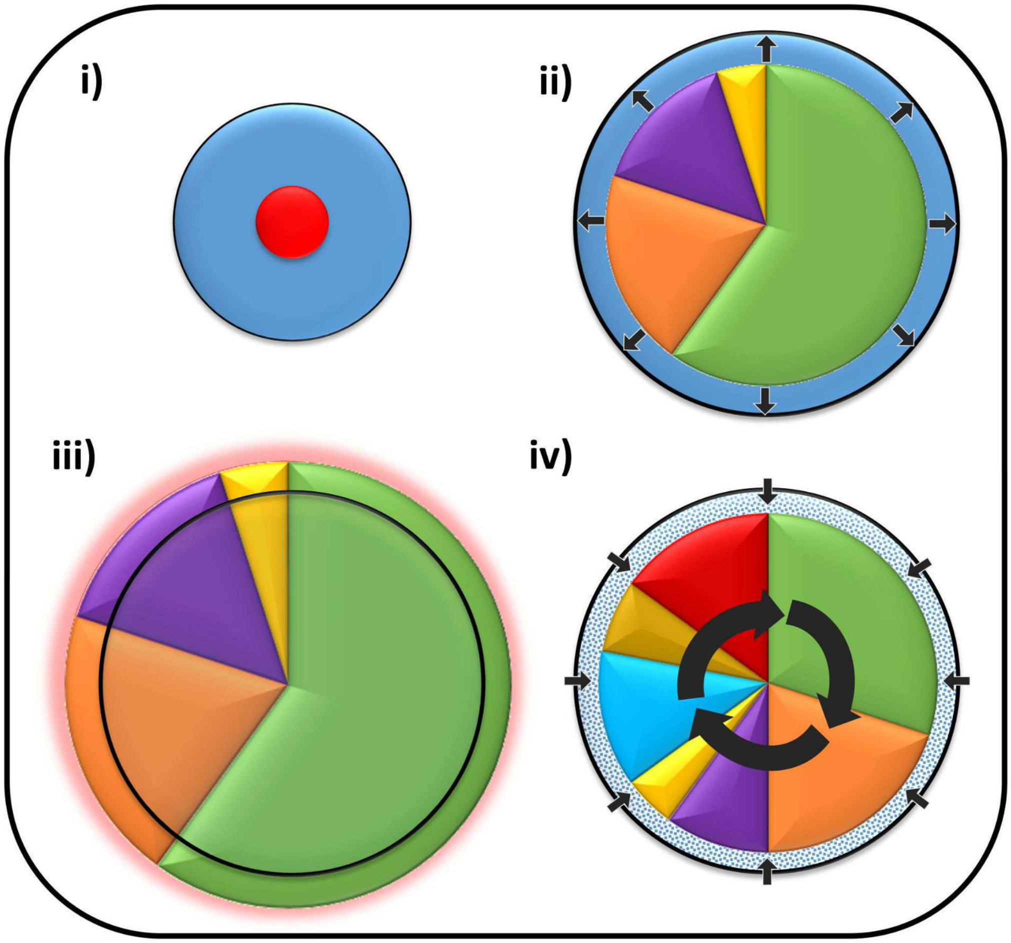

Figure 5.1: (i) Initially, water demands (red inner-circle) can be easily met by renewable water supplies, which are much greater in size (blue outer-circle). (ii) Over time, water demands grow and supplies are expanded through engineering measures, as is illustrated by the expansion of both the inner circle and outer circle, representing increasing water demands and supplies, respectively. As water demands approach available supplies, water is allocated amongst each of the current users, as represented by the division and coloring of the inner demand circle. (iii) Water demands eventually outpace available supplies (inner demand circle now larger than water supply circle), leading to over-allocation, water scarcity, and environmental degradation. Under this condition, water allocations are fixed and no new uses can occur due to a lack of unappropriated waters. (iv) Water reallocation remedies this common occurrence by allowing water use to return within sustainable limits through allocations to the environment (decrease in the size of the inner water demand circle) and permitting water rights to be transferred between new and existing users (new users represented by additional fragmentation of demand circle, while the size of each slice — *i.e.,* the amount allocated to each user — can dynamically change as well).

In areas with fully allocated waters (*i.e.,* the basin is closed), reallocation can be driven by new or increasing water demands caused by technological advances, socioeconomic development, changes in societal understanding or values, and/or population growth. For instance, the expansion of water-intensive crops for biofuel production, along with water guzzling oil and gas recovery technologies, have already been shown to be restricted in some areas due to water availability (*De Fraiture et al.*, 2008; *Nicot and Scanlon*, 2012; *Scanlon et al.*, 2014), thus making reallocation an attractive solution in meeting these new demands. Growing scientific insights regarding the value of environmental sustainability and ecosystem services have lead some societies to demand a change in the current allocation of water (*Colby et al.*, 1991), which often neglects environmental uses in favor of human needs. Recognition of previously neglected water rights of people groups or nations further necessitates water reallocation as a means of adjusting to shifting values or governmental positions (*e.g.,* reallocation of water in South Africa to increase the equitable distribution of water after apartheid (*Dinar et al.*, 1997) or the US government calling for reallocation of nearly ten percent of Arizona's total developed water supply to meet formally unrecognized Native American water claims (*Bark*, 2009).Water reallocation has been shown to resolve conflict among, and balance the needs of, multiple water users while improving local and regional economic robustness (*Zhu et al.*, 2015; *Rosegrant and Binswanger*, 1994).

Water reallocation can also be necessitated due to limited water supplies. Augmenting water supplies through traditional measures, such as dam storage, is becoming more challenging due to increasing economic cost, a better understanding of the associated environmental consequences, and the physical scarcity of unappropriated water. At the same time, current water supplies are being reduced by the deterioration of existing infrastructure, a growing issue in industrialized countries where many dams and other structures are near or have surpassed the end of their design life and are filling with sediment. Water availability is expected to reduce significantly in some basins already facing water scarcity due to climate change, further straining the current water resource system (*Bates et al.*, 2008; *Molle and Berkoff*, 2009). Moreover, groundwater depletion and the degradation of water by point and non-point source pollution have reduced viable water supplies in many regions, especially developing countries (*Molle and Berkoff*, 2006). When current supplies are inadequate and further source development is infeasible, reallocation becomes the most cost effective means of supplying water to the highest priority users (*Bathia et al.*, 1995; *Gomez et al.*, 2004) and, in some cases, can reduce water shortage vulnerabilities by diversifying users' water sources (*Kasprzyk et al.* 2009; see the case of Manila in Table 5.1 which gives examples of different forces driving water reallocation). For example, *Firoozi and Merrifield* (2003) used a theoretical model to demonstrate how a water portfolio including water reallocation can delay the construction of costly reservoirs.

5.3 Comparison of Water Reallocation Forms

Water reallocation can take many forms that vary in duration, spatial scale, complexity, and required institutional structure. The transferred water can be used to serve numerous purposes, such as improving water quality and ecosystems, directly meeting water demands, enhancing system flexibility and reliability, and decreasing water supply cost (*Lund and Israel*, 1995). For all forms of water reallocation, it is important to have diverse water users with different water requirements and productivity levels since transfers are generally prompted by differences in users' marginal value of water. The applicability of different forms of reallocation depends on various conditions, particularly, the development of water rights (*i.e.,* do property rights exist for market transfers?), institutional development (*i.e.,* do organizations and policies exist to implement reallocation?), and infrastructural development (*i.e.,* is appropriate engineering, information, and transaction infrastructure available to facilitate reallocation?). In practice, a combination of water reallocation approaches is used.

Meinzen-dick and Ringler (2008) categorized three forms of formal water reallocation: administrative reallocation, collective negotiations, and market-driven reallocation. Administrative reallocation is a mandatory (non-voluntary) measure taken by a centralized public or quasi-public entity (*e.g.,* river basin authority) to redistribute existing water entitlements. Collective negotiations and water markets are voluntary and decentralized reallocation methods which permit users to sell their water rights to other users, which can include a government entity. Other, informal means of reallocation can sometimes be found in practice that rely on force, surreptitiousness, or illegal means to reallocate water to other purposes (*Meinzen-dick and Ringler*, 2008). Table 5.2 provides some examples of how different water reallocation forms have been used around the world to transfer water between different users and places. The following sections describe the typical forms of reallocation and the relative benefits and drawbacks of each form.

Table 5.1: Reallocation is typically driven by limited and/or changing water supplies and evolving water demands. Examples of some of the more common forces necessitating water reallocation are described below.

Reallocation Driven by	Driving Forces	Cases
Limited water supplies	1. Law/policy change	**1. Arizona, USA** — A change in law made treated effluent transferable water, distinguishing it from other surface and groundwater sources. Longstanding downstream users of treated municipal water have seen water availability decrease as it is sold to other users (*National Research Council*, 1992).
	2. Economic feasibility	2. **Maipo River Basin, Chile** — Burgeoning cities determined that buying water rights from farmers through the area's water market was five times less costly than building a new dam (*Bathia et al.*, 1995).
	3. Climate change	3. **Lima, Peru** — Pronounced recession of glaciers in the Andes, a major water source, may lead to long-term declines in available water (*Raup et al.*, 2007) in an area already facing water-related tensions amongst sectors. Water has already been transferred from agriculture to urban uses, with more reallocation potentially forthcoming.
	4. Reliability of water supplies	4. **Manila, Philippines** — The city of Manila receives 97% of its water supply from a single source, subjecting it to risk during times of drought (*Molle and Berkoff*, 2009). Reallocation, among other methods, have been used to bring about greater water security and reliability.
	5. Infrastructure degradation	5. **Kansas, USA** — John Redmond Reservoir filled with sediment more rapidly than projected, thereby reducing available water supplies for M&I and cooling operations at a nuclear power plant. Instead of developing more storage, a portion of the flood pool was reallocated to conservation storage (*Johnson et al.*, 1990).
	6. Infrastructure development	6. **Delhi, India** — The City of Delhi reduce water losses by lining irrigation canals that transferred its municipal supplies. Farmers contend that this reduced recharge to their groundwater wells, thus indirectly reallocating water from their historic irrigation uses (*Molle and Berkoff*, 2009).
	7. Insufficient infrastructure	7. **Sana'a, Yemen** — Unreliable municipal water infrastructure has led to the pervasive use of tanker trucks to reallocate water from agricultural wells to domestic users (*Molle and Berkoff*, 2009).
Evolving water demands	1. Recognition of previously neglected rights	1. **Nevada, USA** — Judicial courts partially recognized previously unacknowledged water rights of Native Americans. This prompted further reallocation of water through market transfers and negotiations between the city of Reno and Indian tribes (*National Research Council*, 1992).
	2. Recognition of environmental uses	2. **California, USA** — The California Supreme Court ruled that the city of Los Angles reallocate water back to the Mono Basin because the ecological and environmental harm the transfers caused were in opposition to the public interest (*National Audubon Socy v. Superior Ct. 33 Cal.*, 1983).
	3. Growing / Emerging Demands	3. **Coimbatore, India** — Growing urban demands, including an emerging water-intensive textile industry, led to numerous formal and informal water transfers from rural agriculture to urban uses (*Meinzen-dick and Ringler*, 2008).
	4. Population Growth	4. **Amman, Jordan** — A rapidly increasing population due in part to an influx of refugees of war has caused domestic water demands to outpace supplies. Agricultural water has been reallocated via tanker trucks to urban dwellers (*Molle and Berkoff*, 2009).
	5. Economic development	5. **Lesotho & South Africa** — Water is transferred from Lesotho to South Africa to meet the needs of South Africa's evolving economic hub (*Patrick*, 2014).

Examples of some the more common forces necessitating water reallocation are described in this table.

Table 5.2: Broad categorization of the most common types of water reallocation between different uses and/or places. Examples from around the world are given for each reallocation form.

Reallocated Between	**Example Cases**		
	Location	**Form**	**Case Description**
Human and Environment	1. China	1. Administrative	1. The Chinese government ordered a portion of irrigation water within the Hei River Basin to be reallocated to maintain greater instream flows in an effort to restore the ecosystem of the region. Detrimental impacts to farmers due to a decrease in irrigation water were partially offset by a significant crop pattern change and financial support from the central government for more efficient irrigation technology (*Liu et al.*, 2005).
	2. Australia	2. Market	2. Water markets have been established in Victoria, Australia to allow greater allocations of water to environmental purposes. A cap on water extractions has ceased further overexploitation and the government has purchased billions of dollars' worth of water for the environment but there is debate to how effective these efforts have been (*Ladson and Finlayson*, 2002).
Agricultural and Non-Agricultural	1. Wyoming, USA	1. Negotiations	1. The city of Casper, Wyoming helped the Alcova Irrigation District with irrigation improvements, such as canal lining, under the agreement that water savings (several 1,000 AF/y) would be transferred to the city for municipal water supply (*Colby*, 2011).
	2. Indonesia	2. Administrative, Negotiations, and non-voluntary (illegal means)	2. Textile factories in West Java have placed new demands on available water resources, prompting transfers from agricultural users through government reallocations, deals with farmers, and/or unpermitted withdrawals (*Kurnia et al.*, 2000).
Traditional and New Uses	1. New Mexico, USA	1. Negotiations	1. In the upper Rio Grande basin, water was voluntarily reallocated from agriculture to support a ski resort. The transfer was protested in the courts by local interest but eventually the transfer was allowed (*National Research Council*, 1992).
	2. Texas, USA	2. Negotiations	2. During drought conditions in Texas, energy producers bought water from nearby farmers and municipalities to use for hydraulic fracturing (*Cooley and Donnelly*, 2013).
	3. Iran	3. Administrative	3. The rapid growth of Isfahan, Iran has been supported by developing industries, such as steel production and tourism, which required reallocation of water to maintain, especially during dry years. For instance, during drought conditions all agriculture water was reallocated to support urban uses (*Molle and Berkoff*, 2009).
Upstream and Downstream	1. Tunisia	1. Administrative	1. The downstream and water-rich areas of northern Tunisia, redistribute water to water-poor regions upstream. In this way, much of the water is reallocated from its original purpose (irrigation) to cities/tourist resorts (*Molle and Berkoff*, 2009).
	2. Mexico	2. Non-voluntary (stealth)	2. Monterrey, the largest city in the state of Nuevo Len, continues to claim more water from the El Cuchillo reservoir, insidiously reducing water supplies allocated to farmers in the downstream state of Tamaulipas (*Molle and Berkoff*, 2009).

Examples from around the world are given for each reallocation form.

5.3.1 Non-voluntary reallocation

Administrative reallocation

Administrative reallocation involves the transferring of water by the national, provincial/state, or basin entity from one user to another, usually under the premise that it is for the benefit of society as a whole. Administrative reallocation has been used to meet environmental flow requirements (*Liu et al.*, 2005) and social equity concerns (*Dinar et al.*, 1997) since these needs often cannot be directly met by these stakeholders; therefore, the task falls to the government to fulfill their water needs under the public trust doctrine. Administrative reallocation may be more suitable in many areas because it has fewer institutional and investment requirements than voluntary reallocation, especially when water rights are not well defined.

Common forms of administrative reallocation are through the redistribution of the storage volume in publicly-owned reservoirs, revoking water rights (through forfeiture or abandonment provisions), eminent domain, legal action, or construction of large-scale water projects. Administrative water reallocation in developed countries like the United States often considers input from other stakeholders but direct stakeholder consultation is much rarer in developing countries (*Meinzen-dick and Ringler*, 2008). For instance, reservoir storage originally dedicated to irrigation in the northeastern Hubei Province of China was unilaterally reprioritized to recreation and tourism purposes during a drought event, which had negative ramifications on the livelihoods of low-income farmers who depended on the water for drinking (*Cai*, 2008). In some cases, indirect or direct compensation may be provided but payments often do not fully recompense the former users (*Cai*, 2008).

Other means of non-voluntary reallocation

Non-voluntary water reallocation can occur by other means which do not depend on law or custom for justification, but instead on force or stealth. Inherently, these types of transfers do not involve compensation and are done unilaterally by those seeking to acquire water. Common examples include stealing water from agricultural canals or municipal lines or infringing on others' groundwater pumping by over-extracting from a nearby well. Forcible water reallocation can be motivated or justified along ethnic, racial, and/or class lines, as has been shown in the reallocation of water from Palestine to Israel during the formation of Israel last century (*Frederiksen*, 2003) and between white settlers in South Africa and black South Africans. In the latter case, white South Africans first took claim of 91% of the land and then installed riparian water rights, which necessitates land ownership to acquire a water allotment, thus implicitly reallocating water from black natives to new white land owners (*Molle and Berkoff*, 2009).

Non-revenue water (*i.e.,* water intended for sale for a specific purpose but reallocated to unauthorized users by theft or to the environment and/or aquifers through distribution leakage), is a form of non-voluntary water reallocation and a major issue in many developing countries (*Kingdom et al.*, 2006). In several underdeveloped regions, a vibrant tanker truck industry has formed where parties with water access (sometimes illegally) transport and then sell their water to domestic users with limited or no access to fresh water. In places like Karachi, Pakistan, this form of reallocation accounts for roughly one-fifth of the population's domestic water supply (*Kjellén and McGranahan*, 2006).

Non-voluntary water reallocation can also be seen within transnational river basins. For example, Turkey acted unilaterally in diverting and storing water within the headwaters of the Tigris and Euphrates Rivers to expand irrigation within the country, thereby implicitly reallocating water from downstream users (Syria and Iraq) to upstream uses. This has in turn forced Iraq and Syria to overexploit their groundwater reserves (*Voss et al.*, 2013).

5.3.2 Voluntary reallocation

Collective negotiations

Collective negotiations can create innovative solutions for water reallocation, which provide mutually agreeable solutions between existing water users, old and new users, or users and the government (*Meinzen-dick and Ringler*, 2008). Informal voluntary negotiations are seen widely around the world but are most prevalent in Asian and Southeast Asian countries, where water scarcity exists but no formal water markets are in place (*Bjornlund*, 2003). As seen in Spain, this type of reallocation lays the foundation for further institutional reform and the formation of a formal market (*Palomo-Hierro et al.*, 2015).

Negotiations allow for reallocation of water at all scales: from small cities and nearby farmers to neighbouring nations. For instance, a small city in Utah struck a deal with a nearby irrigator to have the option to purchase his senior water right during times of drought for a onetime payment of $25,000, along with 300 tons of hay and $1,000 for any year that the city exercised its right to the water (*Shupe et al.*, 1989). Before the demise of the Soviet Union, the regions that are now Kyrgyzstan and Uzbekistan agreed for reservoir storage in a Kyrgyzstan dam, which was designated for hydropower generation during the harsh winters, to be reallocated to irrigation supplies for Uzbekistan during the summer growing season. In return, energy rich Uzbekistan would provide cheap energy supplies in the winter months to offset Kyrgyzstan's loss of hydropower (*Cai et al.*, 2003).

Water reallocation by collective negotiations is typically beneficial to all stakeholders involved in the agreement, yet one of the key concerns with this reallocation method is that it sometimes acts to the determinant of those not involved in negotiations (*i.e.,* it tends to have negative externalities). Appropriate governance is required to guide and manage negotiated water transfers so to protect third-party users, especially environmental uses, which may be "invisible" and ignored during negotiations (*Meinzen-dick and Ringler*, 2008).

Market-based reallocation

Water markets provide a means for current water users to sell their existing water rights, either temporarily or permanently, to new water users or existing users seeking greater water availability. A prerequisite for well-functioning water markets is fully specified, exclusive, transferable, and enforceable water rights (*Coase*, 1960); this requirement implies a robust legal, institutional, and regulatory framework that is able to monitor, enforce, and provide the necessary infrastructure for transfers. A sufficient number of willing market participants with different opportunity cost of water improves the economic efficiency of markets. Empirical studies largely support that markets move water from low-value uses to higher value uses as theorized (*Chong and Sunding*, 2006; *Leidner et al.*, 2011); however, prices are generally not equalized across all users, as would be expected in a perfect market, because water markets often exhibit high transaction cost, administrative regulation, and participants who invoke their market power to distort the market (*Ansink and Houba*, 2012). Markets are necessitated by water scarcity and can take form gradually, as in the Murray-Darling Basin of Australia, or relatively rapidly, driven by a catalyzing event such as a lawsuit to protect endangered species, as was the case in Texas (*Debaere et al.*, 2014). Formal and informal water markets are active in at least nine countries, with varying levels of sophistication and success (*Hadjigeorgalis*, 2009).

Market-based exchanges of water rights have demonstrated significant welfare gains to buyers, sellers, and the economy on whole (although there can be losers, as discussed later). The water markets of the Murray-Darling Basin Australia were first established in the 1980s and are among the largest, most advanced, and active in the world (*Grafton and Horne*, 2014; *Grafton et al.*, 2012) and offer a glimpse of the potential benefits of water markets. Australia's National Water Commission reported (*Commission*, 2010; *Wheeler et al.*, 2013) that water markets increased the gross regional product of the southern Murray-Darling Basin by $370 million, lessened the effect of the Millennium Drought on economic output by $4.3 billion, and had a net positive effect on the environment because transfers to downstream users also benefited riverine ecosystems.

5.3.3 Other distinctions amongst reallocation forms

Intra-sectoral versus inter-sectoral reallocation: Traditionally, reallocation has almost exclusively occurred within the agriculture industry (*Levine et al.*, 2007). Often water reallocation within the agriculture sector occurs within a ditch or mutual irrigation company, whose existing canal network makes reallocation amongst users feasible (*Bjornlund*, 2003). In these cases, water rights or shares in the irrigation company (which entail a certain amount of water) can be traded with other company shareholders on a temporary or permanent basis.

There is now increasing interest in inter-sector reallocation due to changing economics, recognition of environmental water needs, and policies regarding water transfers (*Levine et al.*, 2007). Inter-sectoral water transfers are more complex and regulated, thus increasing transaction cost and decreasing their occurrence. In the Western US and, to a lesser degree, the Murray–Darling Basin of Australia, the price of water sales and leases between agriculture users can be 10 times less than between agriculture and urban users, even after considering transaction, delivery, and treatment cost (*Grafton et al.*, 2012; *Chong and Sunding*, 2006). This indicates the potential benefits of out-of-sector water transfers, yet several obstacles remain, as discussed later.

Temporary versus permanent reallocation: Permanent water reallocation is the transfer of water entitlements from one entity to another for perpetuity. The permanent reallocation of water acts similarly to supply expansion and, in some cases, delays costly supply and demand management efforts to increase water availability and/or reliability (*Lund and Israel*, 1995; *Firoozi and Merrifield*, 2003). Temporary water reallocation typically occurs over one-year, although longer leases can be between 2 to 100 years (leases from 25-40 years are most common, however; *Committee on Western Water Management* 2012). Typically, temporary transfers occur through spot markets, contingent transfers, dry-year options, water banks, and eminent domain. In most regions, temporary water reallocation is far more prevalent than permanent water reallocations.

Local versus non-local reallocation: Most water reallocations occur locally since these transfers exhibit lower transaction cost and fewer regulatory restrictions. Non-local reallocation of water requires significant differences in the opportunity cost of water between users to overcome the large social, administrative, and physical cost to transfer the water. Furthermore, non-local transfers frequently necessitate significant infrastructure investments that far exceed the capacity of the stakeholders. Therefore, governments nearly always play an integral role in large-scale, non-local reallocation of water. For example, the Chinese government lead non-local water reallocation efforts in the Hei River basin (*Liu et al.*, 2005), the Yellow River basin (*Cai*, 2008), as well as the South-North Water Transfer Project (*Cai and Rosegrant*, 2004).

5.4 Impediments to Effective Water Reallocation

Despite a voluminous collection of water reallocation research, successful examples of water reallocation are relatively sparse. Its implementation is hindered due to many social, institutional, economic, environmental, and physical barriers that have proven difficult to overcome. For these barriers to be abated, we contend that a more holistic approach should be employed that couples advancements in the natural sciences and engineering disciplines with current water reallocation scholarship, which is predominately rooted within the social sciences, especially the field of economics. The following sections overview the major difficulties faced in carrying out water reallocation and offers suggestions of what can be done to overcome some of these obstacles, including research needs.

5.4.1 Poorly defined water rights

The efficacy of voluntary water reallocation, specifically water markets, is often hindered by the lack of well-defined and enforceable water property rights, especially in developing countries (*Molle and Berkoff*, 2009). However, adjudicating all water rights can come at a great cost and take considerable time. For example, in an effort to better facilitate water markets, the State Engineer Office of New Mexico has sought to adjudicate state rights in the Middle Rio Grande but they have estimated it will cost $300 million and take over 50 years (*Pease*, 2010). Private water rights, and subsequent trading of these rights, are only effective if they are enforced and monitoring. As noted by *Leidner et al.* (2011), the costs of monitoring and enforcement may also act as a barrier to voluntary water reallocation. *Zhang* (2007) asserts that experimental water markets established in Zhangye City by China's Ministry of Water Resources were unsuccessful, in large part because there were few measures to stop water users from extracting water in excess of their entitlement, thus reducing their need to purchase water in the market.

Water is typically owned by the state, which grants usufructuary rights to private parties or local communities for use under specified conditions. Voluntary water reallocation requires fully specified, exclusive rights that are separated from land, as well as provisions for trade of the rights; however, these requirements are rarely met, especially in many developing countries (*Hadjigeorgalis*, 2009). How to effectively and equitably promote water reallocation by transitioning from land-based water rights (either formal or informal) to use-based rights is a pertinent question for researchers and policy-makers. In an effort to promote water reallocation, water licenses in Australia's Murray-Darling Basin were separated from land rights and converted to volumetric entitlements (*Grafton and Horne*, 2014) (typically by equating land area to a given allocation of water). Even when riparian rights are converted to use-based rights, it is difficult to break the linkage between land and water rights in users' minds, which can act as a barrier to trade, especially the permanent reallocation of water. For instance, *Giannoccaro et al.* (2013) found that a lack of permanent water transfers in Southern Spain's water market could be partly explained by irrigators' reluctance to separate water entitlements from their land.

When water users are small and fragmented it can prove challenging to establish individual water rights, especially in developing countries that lack the institutional capacity to overcome this difficulty. In these instances, water entitlements may be allocated to local communities or water user associations that can assign water entitlements internally, as the case in Mexico (*Easter et al.*, 1999). *Hadjigeorgalis* (2009), along with *Rosegrant and Binswanger* (1994), argue that developing countries may be able to assign tradeable water rights using a subsidiarity approach that includes stakeholder participation, instead of the centralized top-down method recommended by donor agencies. This approach is more likely to efficiently and equitably allocate water, while also being more socially acceptable.

Water right doctrines typically have a beneficial use provision, most of which require right holders to "use it or lose it". Although these provisions are meant to prevent speculation and non-beneficial water uses, they can also act as a deterrent to water reallocation due to fear that a right might be lost or reduced when temporarily transferred. Many Western US states have addressed this issue by including provisions in their water transfer programs that protect water right holders from forfeiture or abandonment of their water rights if they are transferred to other users. However, many water users are still unwilling to enter transfer agreements because they are fearful that the legislatures will eventually revoke their excess water rights.(*Committee on Western Water Management*, 2012).

The difficulty in properly assigning and enforcing tradeable water rights amongst human users has been given significant attention in the water reallocation literature. However, the daunting task of reallocating the appropriate amount of water to the environment may prove to be an even grander challenge. Growing understanding of the societal benefits offered by environmental flows (*i.e.,* water needed to maintain ecosystems, as well as the livelihoods and well-being of people that depend on these ecosystems) has led to efforts to formalize environmental water allocations; yet, reallocating water from productive human uses to environmental uses involves difficult trade-offs, which are exacerbated due to the uncertainty and challenges in quantifying environmental flow requirements in terms of volume, duration, timing, frequency, and quality.

The science required to determine the required flow regime for healthy riverine ecosystems has evolved considerably in the last two decades, with more than 200 different environmental flow assessments developed (*Tharme*, 2003; *Acreman and Dunbar*, 2004), including the index of hydrologic alteration (IHA) method (*Arthington et al.*, 1992; *Vogel et al.*, 2007; *Yang et al.*, 2008; *Richter et al.*, 1996, 1997; *Poff et al.*, 1997); numerous studies have also reported on water requirements for terrestrial ecosystems (*Baldocchi et al.*, 1996; *Gerten et al.*, 2004). Despite the seeming plethora of approaches for assessing environmental flows, debates remain regarding the applicability and merits of each method. Additional scientific support is needed to answer remaining critical questions that, left unanswered, will continue to restrain water reallocation's potential, especially for ecosystem restoration. For instance, how much water has already been over-depleted (allocated to human uses) from ecosystems? When reallocating limited water supplies, how are trade-offs made between different, and sometimes conflicting, flow requirements of various ecosystem services? What is the 'natural' state of the environment and to what degree can the system diverge from this state before hitting a critical (unsustainable) point? What flow regime and water quality are needed to restore ecosystems to a target level (and how do we determine the 'target' level)? How will reallocation to other human uses affect the environmental and ecological systems that have become dependent on the original human water use (*e.g.,* ecosystem associated with irrigation)? How will ecosystems and human systems co-evolve in the future and what implications does this have on water reallocation? Answers to these biophysical questions can be integrated into social science research on water reallocation, thereby linking ecohydrology and ecosystem services with stakeholder outcomes.

5.4.2 Third-party effects

Third-party effects, that is, impacts to those not directly involved in the transfer of water, are one of the greatest hurdles to water reallocation reaching its potential as a water management strategy (*Murphy et al.*, 2006; *Leidner et al.*, 2011). The legitimate potential for negative third-party effects can cripple reallocation opportunities that may create a net positive economic and/or social benefits at a regional level or different location, yet have detrimental effects at the area of origin. Most water policy dictates that water reallocation can only occur if "no injury" occurs to other users, but these rules vary greatly and are difficult to quantify. The incidental effects of water reallocation aren't typically identifiable immediately, making it difficult to quantify damages and compensate those negatively impacted. The reallocation of water can cause impacts to the quality and availability of water for other users, including the environment. Furthermore, water reallocation can have negative externalities that permeate throughout the area of origin and beyond, such as the deterioration of a community's values and culture, which are often tied to the livelihoods that the water once supported, a depression in land values, excessive weeds and dust from fallowed fields, reduction in the local tax base, and harm to supporting industries, such as agribusinesses. If proper reinvestments and policy measures are not in place when water is reallocated out of a region, the area exporting water will likely face socioeconomic and population decline (*Howe et al.*, 1990).

Rural communities within the American West have attempted, mostly unsuccessfully, to require the water rights purchaser to pay the county of origin a fee for the amount of property tax revenue lost as a consequence of water rights being transferred out of the region (*Brown*, 2006). Nebraska mandates that property taxes must be paid on the pre-transfer land value, which ensures that local government remains funded but the additional cost may deter some water reallocation (*Committee on Western Water Management*, 2012). Researchers have suggested a fee or tax paid on reallocated water to compensate third-parties (*Levine et al.*, 2007), as the case between the Southern California's Metropolitan Water District (importer) and the Palo Verde Irrigation District (exporter), which established a $6 million fund to provide grants for community projects, business loans, and vocational training to offset losses associated with the required fallowing of land (*Committee on Western Water Management*, 2012). *Murphy et al.* (2006) found a tax on transfers to compensate those negatively affected provides very economically efficient and socially acceptable outcomes, while an alternative approach of allowing third-party participation in market outcomes increased volatility, reduced economic efficiency, and prompted strategic behaviour.

Water reallocation is often viewed as a way to meet the neglected water requirements of the environment, as seen in the United States (*Brown*, 2006), Australia (*Garrick et al.*, 2009), and China (*Liu et al.*, 2005); ironically, water reallocation has also been opposed for environmental externalities it sometimes creates. Some areas have legislation prohibiting water transfers that would cause "unreasonable impact on fish, wildlife, or other instream uses" (*Rosegrant et al.*, 1994), with the most significant in the US being the Endangered Species Act. Reallocation of water can disrupt the temporal pattern of instream flows and reservoir releases, thus negatively impacting other uses — especially ecosystems that require a very particular flow regime. Furthermore, reallocation to upstream or out-of-basin diversion points can reduce the incidental environmental benefits achieved in the previous conveyance to downstream users. Environmental externalities can be exacerbated when irrigators replace transferred surface water with increased groundwater extractions because when surface water and groundwater are hydraulically linked this can lead to a further reduction in baseflow, which is critical for ecosystems during low-flow conditions (*Poff et al.*, 1997). Small wetlands and riparian ecosystems have formed along some irrigation conveyance systems, which would also be threated if water is reallocated from agriculture. *Llop and Ponce-Alifonso* (2012) have taken a first step toward establishing the trade-offs between economic benefits and the environment under different reallocation schemes using a computable general equilibrium (CGE) model with an ecological sector.

Most water transfer schemes are based on consumptive use, not the full diversion, so to ensure downstream users dependent on upstream users' return flows (non-consumptive portion of withdrawals) to fulfill their water right are not adversely effected (*Rosegrant and Binswanger*, 1994). This protection, however, can impede potential reallocation by adding greater uncertainty and cost due to the challenge in quantifying consumptive use and return flows. In a recent survey, one US State said determining the amount of water consumed by the original user was the most difficult challenge in assessing water transfers (*Committee on Western Water Management*, 2012).

Water conservation measures have been implemented to make "saved" water available for transfer to other uses (*Cai*, 2008; *Levine et al.*, 2007; *Ward and Pulido-Velazquez*, 2008), though in most cases, water is not truly saved but inadvertently reallocated from those that depend on the existing return flows to other users (*Ward and Pulido-Velazquez*, 2008; *Pfeiffer and Lin*, 2014; *Huffaker and Whittlesey*, 2003). Water conservation and efficient water use (profit per drop) are often goals of water agencies, yet reduced return flows to streams and aquifers have implications on other water users, including the environment, which are difficult to measure and typically happen with considerable delay. *Qureshi et al.* (2010), found empirical evidence that excess water produced from investments in more efficient irrigation and conveyance systems do not provide a justifiable increase in environmental flows; yet, management practices that reduce consumptive water use (such as land fallowing, deficit irrigation, and less water-intensive crops) allow for considerable opportunities to reallocate water to the environment.

The Colorado Big Thompson Project (CBT) in northern Colorado is the most active water market in the United States (*Grafton et al.*, 2012), largely because it has circumvented the return flow issue by assigning rights proportional to streamflow and retaining rights to return flows internally. Water users can access return flows but their availability is not guaranteed. Chile and Mexico have adopted a similar approach, yet the institutional barriers to transition to proportional water rights may make it infeasible for most regions (*Grafton et al.*, 2012). Other methods to sidestep the complexity of determining return flow rights have been taken by New Mexico and Wyoming, which determine the permissible water transfer volume using standard formulas applied for different conditions to avoid additional costs, time, and uncertainties that can act as barriers to trade (*Rosegrant et al.*, 1994). Similarly, assumptions regarding return flows, consumptive use, and third-party impacts can be established as a rebuttable presumption, thereby shifting the burden of proof to those with objections to the transfer. This could reduce the number of baseless claims, while allowing for greater consideration under special circumstances; however, the imprecision of these methods may limit some transfers.

The inadequacies of current methods necessitate the creation of new methods and technologies that provide economical, scalable, and reliable estimates of a user's consumptive water use and utilizable return flows (*Gates et al.*, 2012). Continual improvements of relatively new technology, namely remote sensing tools (*e.g.,* Landsat Thermal Infrared Sensor), can help quantify consumptive water use and thus provide quick and inexpensive information for reallocation decision making. However, determining utilizable return flow volumes will likely prove to be even more challenging, especially from irrigation systems, because hydrologic heterogeneity and the dearth of data have encumbered the development of inexpensive and widely acceptable methods for quantifying return flow. The usefulness of return flow depends on the path of the flow (*i.e.,* via natural systems such as aquifers and interflow, or man-made pathways such as drainage systems), the duration for it to become available for reuse, and its quality — all of which are very difficult to determine. For example, a useable quantity of return flow may not be truly useful due to extraordinary salinity from irrigation districts (*e.g.,* the Aral Sea case in Central Asia; *Cai et al.* 2003), increased temperature from power plants, and, more seriously, the various types of pollutants from industrial areas return flow. Therefore, new technologies and methods for determining return flows must consider the quantity and timing of their availability, but also the quality.

5.4.3 Lack of information support and limited stakeholder involvement

Benefits derived from voluntary water reallocation often do not meet expectations because participation is relatively limited — water transfers in many markets are around 2-5% of total water demand (*Zhang*, 2007; *Brown*, 2006; *Bjornlund et al.*, 2014). *Giannoccaro et al.* (2015), along with *Tisdell and Ward* (2003), argue that the overestimation of benefits by theoretical water market models is because stakeholder's perceptions and values are not considered. Many water users, especially farmers (which hold the majority of water entitlements), exhibit a general aversion to the commoditization of water through market mechanisms (*Easter et al.*, 1999), instead preferring that a public entity maintain control of water allocations (*Bjornlund et al.*, 2013). The distrust of water markets have been shown to stem from beliefs that markets will disadvantage low-income farmers (*Easter et al.*, 1999) perceptions water and land rights should not be separated (*Bjornlund*, 2003), and views that water should not be treated as a commercial good (*Giannoccaro et al.*, 2015). However, *Bjornlund* (2003), and *Giannoccaro et al.* (2015), have shown that these negative perceptions of water transfers are somewhat abated over time and drought induced water scarcity acts as a catalyst to market involvement (*Wheeler et al.*, 2013). Early adopters to water markets in Victoria, Australia were typically newer farmers with a farm plan, more educated, higher earners, and female (*Wheeler et al.*, 2009).

Improving the transparency of water reallocation decisions and increasing the availability of information regarding the timing, location, volume, price, and purpose of water transfers is critical to improve market performance, achieve societal acceptance of water reallocation (*Easter et al.*, 1999) (both voluntary and non-voluntary), and enable new research. Governments can increase acceptance and awareness of water reallocation by launching outreach programs to educate citizens and gain trust. They can also make potential buyers and sellers feel more comfortable trading water rights by sponsoring mock transfer programs or engaging in "seed" trades as a basis for larger and/or longer future agreements.

The government is best suited to collect data on water supply and facilitate data exchange, while users are able to better determine their own demand. With the exception of Australia, where the state and commercial water brokers make transfer records available, information on water transfers is fragmented, incomplete, informal, or non-existent (*Grafton et al.*, 2010). Transfer data can be made available through a real-time, searchable online geographic interface system, which would allow potential reallocation participants to gauge transfer activity, inform prospective transfers, and gain insights into the spatial and temporal trends in the basin. In addition, governments can provide seasonal or long-term projections of water supplies so that water users can assess their needs and potentially free-up water (*e.g.,* fallowing land or shifting to less water-intensive crops) for transfer.

Comprehensive, accurate, timely, and accessible information is needed to support water reallocation and research advancements. The growing capabilities of technologies such as remote sensing, geographic information systems (GIS), low-cost sensors, mobile phone applications, and cyber-physical infrastructure can provide critical data concerning water availability, quality, and demands. These data can lead to more informed reallocation decisions with increased efficiencies, decreased cost, and reduced uncertainty. New data mining techniques and Big Data predictive analytical tools can harness the voluminous amounts of data these technologies can provide to bring about new insights and solutions to complex water reallocation issues (*Cai et al.*, 2015). However, before these tools are used, some fundamental questions must be addressed: How can the water use of heterogeneous and fragmented users be monitored and measured and how can this information be included in the decision-making process? What other tools and new technologies are needed to account for and allocate the ever-changing water (full hydrologic cycle) within the basin? How can information describing human processes, institutions, and stakeholder behaviour be collected and paired with physical and ecosystem data in a meaningful way?

5.4.4 Transaction and transition cost

Transaction cost are the collective cost of acquiring information, identifying transfer opportunities, negotiating or administratively determining transfer details, conveyance (including water loss and infrastructure cost), mitigating third-party effects, and monitoring and enforcing transfers (*Rosegrant and Binswanger*, 1994). More broadly, transition costs are the institutional costs to shift the institutional structure to one more favourable to water reallocation. This may involve reorganizing water agencies jurisdiction to match hydrologic boundaries, strengthening water rights, loosening policies restricting water reallocation, reforming water-user associations, building new transfer infrastructure, and the general facilitation of reallocation (*Garrick et al.*, 2013).

The water reallocation literature primarily focuses on transaction cost associated with markets; however, transaction costs are prevalent in all reallocation forms and have not been proven as a larger deterrent in markets than they are for other transfer types. The key difference between transaction costs in voluntary transfers versus administrative reallocation is that the water authority absorbs the cost of administrative reallocation, whereas the individual participants bear much of the cost in voluntary reallocation (*Hadjigeorgalis*, 2009). There are particular factors that have been shown to either increase transaction costs or make transaction cost a greater deterrent in both voluntary and non-voluntary transfers. First, transaction cost increase with larger geographic dispersion of stakeholders and greater number of small fragmented users due to the infrastructure and coordination required to facilitate transfers (*Rosegrant and Binswanger*, 1994). Second, high transaction cost (in terms of time and money) can deter small or short-term water transfers because the monetary cost may exceed the actual value of the water, while the time to gather information, gain approval, and complete the transfer may go beyond the timeframe of when the water is required (*Committee on Western Water Management*, 2012). Third, the uncertainty associated with determining environmental requirements can add greater cost and make reallocating water to the environment potentially more challenging than reallocating water to other water users (*Ladson and Finlayson*, 2002). Finally, transaction cost have been shown to be a greater barrier for some water users (namely agriculture) than others (industry) due to the latter's greater economic gains from utilizing water and capacity to absorb additional cost (*Garrick et al.*, 2013; *Wang*, 2012).

Insufficient water infrastructure often constrains potential transfers by not offering a means to store and then convey water to the new user; yet, the cost to build, maintain, and operate infrastructure projects can make many transfer schemes economically infeasible. *Ansink and Houba* (2012) contend that a lack of water transfer infrastructure leads to narrow markets and bilateral oligopolies, which creates a non-Pareto optimal water allocation because of an imbalance in market power between potential buyers and sellers. Private water vendors have capitalized on the dearth of water infrastructure in many developing countries by creating an informal non-pipe water distribution system to serve unmet water demands. However, these water transfers are sometimes illegal and have been shown to cost up to 12 times more than a formal utility, which can amount to a significant portion of the income of poor consumers who often rely on such a system (*Kariuki and Schwartz*, 2005). One solution is to readapt existing infrastructure for the purpose of water reallocation (*i.e.,* infrastructure sharing). Infrastructure sharing has seen some success (see *Committee on Western Water Management* 2012 for examples in the US) but requires considerable planning and coordination of operations amongst treatment facilities, canals, pumps, and storage systems (*Lund and Israel*, 1995). These systems are often inefficient because they are being utilized for reasons beyond their originally designed purpose. This can increase transaction cost considerably and can possibly strain infrastructure in meeting its original purpose or limit reallocation potential. Further advances in smart infrastructure (*i.e.,* the nexus of physical and cyber infrastructure) have the potential to revolutionize water management and make reallocation a more physically and economically feasible option (*Hoult et al.*, 2009). However, any engineering solution will also necessitate collaboration by agencies controlling major components of a region's water infrastructure (*Lund and Israel*, 1995).

The uncertainty and lack of information regarding water availability further increases transaction costs. It is difficult to determine the potential for water reallocation when basic questions still need to be answered, such as how much water is available, of what quality, at what time, and at what location. The traditional assumption of stationary hydro-climatic conditions has been deemed inappropriate with our growing understanding of the impacts of climate change and human interference (*Milly et al.*, 2015), yet no methods exist that incorporate the complex, uncertain, and rapidly changing effects of climate and human change on water resources. It is difficult to reallocate water if we are uncertain how much water is even available and how this availability changes over time and under different conditions. Key research questions need to be addressed, including the following: What impact does climate change and socioeconomic growth have on water availability of a particular economic sector? Given non-stationary hydro-climatic conditions, what is the reliability of the water supply in meeting all water demands? To what degree can water reallocation mitigate the negative impacts associated with climate-change related fluctuations in water availability?

5.4.5 Unsuitable institutional structure and operation

The efficacy of water reallocation is largely dependent on water governance institutions, including how they coordinate and cooperate amongst themselves (*Moore*, 2015). As Table

5.2 previously demonstrated, reallocation can occur under vastly different government or institutional settings around the world. Yet, water transfers are more efficient, equitable, and/or sustainable under certain legal and institutional frameworks than others. *Grafton et al.* (2010) provides a comprehensive and integrated method to benchmark how well an institutional structure promotes water reallocation (namely through markets) over time and against other institutional types.

Overcomplicated regulations or restrictive rules that limit water transfers to other uses or places can dissuade water reallocation, keeping water locked in economically inefficient uses. For instance, Spanish law classifies priority levels to different water uses and prohibits the transfer of water from higher priority uses to lower priority uses. This legal constraint therefore inhibits potentially beneficial reallocation between agricultural and recreational uses or some industrial uses (*Palomo-Hierro et al.*, 2015). In addition, policies not directly aimed at water resources, such as China's and India's policy promoting food self-sufficiency, can have large implications on how water is allocated throughout the country, constraining space for water reallocation.

Since the agriculture sector controls the majority of water rights, many researchers have focused on how the institutional structure of irrigation organizations (including irrigation districts) has impeded water reallocation (*Ghimire and Griffin*, 2014, 2015). Irrigation organizations (IO) are entities that hold communal water rights and supply irrigation water to its members. They are widely found in various forms in both developed (*e.g., Ghimire and Griffin* 2014) and developing (*e.g., Rosegrant and Binswanger* 1994) countries. The literature shows that IOs are less likely to transfer water rights than irrigators not in IOs primarily because of difficulties in 1) collectively deciding on transfer prices, 2) determining the distribution of water transfer gains amongst members, 3) providing equitable compensation methods for individual water conservation efforts, and 4) quantifying the incidental district impacts due to seepage reduction (which replenishes aquifers and can be used again) and the increased internal water conveyance cost (*Moore*, 2015; *Ghimire and Griffin*, 2015). *Ghimire and Griffin* (2014) found IOs with larger water entitlements are more likely to participate in inter-sectoral water reallocation, which implies that merging IOs with smaller water right holdings could act as a catalyst for reallocation. In addition, reallocation can be fostered by reshaping IOs ownership structure, such that members own a share of the organization's water right (opposed to exclusive possession by the IO), voting privileges are weighted by irrigated acreage to prevent small-holders from exerting disproportional influence on transfer decisions, exit or termination fees assessed on IO members who sell their rights are waived or reduced, and water is priced according to its opportunity cost (*Griffin*, 2012; *Garrick et al.*, 2009)

The present operation of most water storage systems locks in current water uses and therefore must be adjusted to facilitate water reallocation. Many basins have undergone extensive natural and human induced changes, yet the operation of the basin's dams still abide by the water control plan from the time of its design, thus implicitly assuming the same hydrologic conditions, basin development (*e.g.,* land use patterns, other reservoirs), and water withdrawal patterns have remained constant throughout the years (*McMahon and Farmer*, 2004). The assumed stationarity of human and natural systems leads to a miscalculation of water availability and a misappropriation of water to demands that may now hold less economic or societal value than when first allocated.

There have been recent calls for water to be reallocated to meet environmental and ecological objectives by changing reservoir operation procedures (*Suen and Eheart*, 2006; *Suen et al.*, 2009; *Yang and Cai*, 2010). However, adjusting reservoir operation rules to meet such objectives may jeopardize the original purposes of the reservoirs. Thus, research is needed to determine the tradeoffs between the new (ecological) and original (economic) objectives and how to balance these in the reallocation of reservoir storage (*California Deptartment of Water Resources*, 2005). Great reallocation potential also exists through the joint-operation of multiple reservoirs within a river basin. Institutions can coordinate reservoir operating procedures to reallocate water to meet basin-wide objectives, not just water uses adjacent to the reservoir. Basin-wide mechanisms for information sharing, coordination, and regulation are not in place in many regions and technical difficulties in developing optimal operations of multiple reservoirs still represent research challenges (*Labadie*, 2004).

Federal multi-purpose reservoirs in the U.S. provide an example of reallocation of reservoir supply in response to changing conditions but also highlight the need for continued progress. The Water Supply Act of 1958 has expedited reallocation of federally operated dams and allowed the Corps to reallocate 640,000 acre-feet of storage across 44 dams to municipal and industrial water supply since its inception (*Carter* 2010; refer to Fig 5.2); however, this represents only had a small percentage (0.3%) of the 216 million acre-feet of storage across its portfolio of dams. Further reallocation will be required (and in some cases is already needed) to obtain greater economic or social benefits from U.S. water resources (*McMahon and Farmer*, 2004). In particular, storage space allocated to flood control may hold promise for reallocation because the hydrologic record upon which flood storage was based has been extended since many dams were designed, thereby changing risk profiles and flood storage requirements.

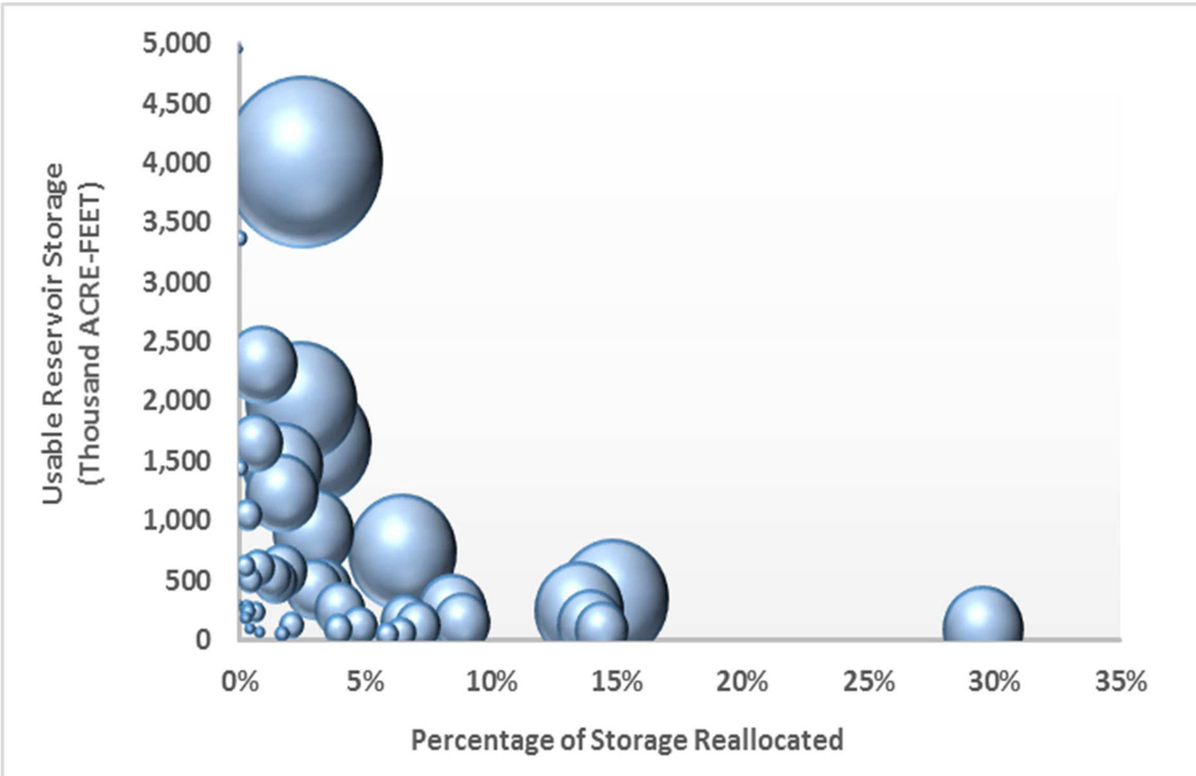

Figure 5.2: Percentage of total reservoir storage reallocated under the Water Supply Act of 1958 (abscissa) for 44 different sized (ordinate) U.S. Army Corps of Engineers' reservoirs. The size of each bubble represents the relative volume of reallocated storage. The majority of dams have had less than 3% of their storage reallocated. Data from (*Carter*, 2010).

Overcoming the obstacles to water reallocation will require an integrated systems approach and collective action amongst researchers of different disciplinary backgrounds, practitioners, and policy makers. In particular, researchers need to move towards more interdisciplinary water reallocation research by consolidating advances in natural sciences, engineering and social sciences. We propose an initial path for the unification of the social sciences, natural sciences, and engineering fields into water reallocation solutions in 5.3. A summary of the major water reallocation barriers overviewed in this paper is provided in 5.3, where we also show how contributions from the social sciences, natural sciences, and engineering can be integrated into holistic approaches to overcome water reallocation obstacles. Economic studies of water reallocation (chiefly related to water markets) should be extended to include a broader social science context to deal with equity and community resilience associated with water reallocation. Meanwhile, natural science and engineering studies should be extended from traditional focuses on water availability and water supply to more effective realization of water reallocation through novel infrastructure development, system operations and information support (*Cai et al.*, 2015).

5.5 Concluding Remarks and Suggestions for Future Research

Water reallocation exhibits great promise as an adaptive water management tool which can reduce the economic, social, and environmental harm caused by water scarcity. Despite its promise, this paper notes the numerous and diverse barriers that hinder water reallocation from reaching its potential. Reforms in water institutions and policies are required to bring about water reallocation at a greater scale; however, these changes must be guided by improved scientific understanding and new tools to assess the trade-offs and interdependencies involved in any water reallocation decision. Past research offering only policy, economic, institutional, or engineering approaches to water reallocation have not been widely implemented because it only addresses one part of a grander problem. The complex economic, social, technical, environmental, and institutional underpinnings of water reallocation must be all integrated into proposed solutions, not simply ignored or assumed away. Decision making must be based on a comprehensive scientific understanding of the complex human-water system, informed through new data collection and interpretation tools, and be actionable through advances in technology and smart infrastructure. This seems to follow the calls from Integrated Water Resources Management (*Rahaman and Varis*, 2005; *Grigg*, 2008) (IWRM), but research for water reallocation is expected to provide a specific and realizable context to implement IWRM to solve real world water management problems, as outlined by Biswas (*Biswas*, 2004) who reviewed the implementation obstacles of IWRM. In short, a broader interdisciplinary framework is needed to guide water reallocation decisions and remove its barriers. Although suitable choices of particular reallocation forms vary by regions depending on the various governance, legal/regulatory, and social systems, shared issues (*e.g.,* poor governance, lack of information support, transaction cost, and third-party effects) exist among all water reallocation occurrences; thus, the proposed integrated research framework can be utilized around the world as a means of overcoming these common barriers.

Until researchers resolve some of the critical barriers to water reallocation discussed throughout this paper, water reallocation will continue to be challenging or infeasible in

Table 5.3: The removal of water reallocation barriers requires an interdisciplinary approach that integrates the social sciences and the natural sciences and engineering into a holistic framework.

Major Barrier to Reallocation	Social Science Focus	Natural Sciences and Engineering Focus	Holistic Approach	Anticipated Outcome
Poorly defined water rights	Establish water rights and improve policies to facilitate proper reallocation	Quantify environmental flow requirements	Link environmental water requirements to stakeholder outcomes and create a systems approach to evaluate trade-offs between environmental and human water uses	Balanced water allocation to human-nature needs
Third-party effects	Assess economic and non-economic third-party impacts, as well as methods for compensation	Estimate consumptive water use and return flows more accurately and develop more effective monitoring methods. Assess impacts of climate and societal change on water.	Co-optimize water benefits based on physical and socioeconomic connectedness throughout the system	Reduced/enhanced negative/positive externalities associated with reallocation
Lack of information support and limited stakeholder involvement	Increase transparency regarding transfers and clarify stakeholders' values and beliefs	Increase reliability of water availability and use data, improve information accessibility, and monitor environmental effects. Provide insights into the human-water system through advanced information technology, especially Big Data tools.	Incorporate hydrologic data and human responses and values into a coupled human-water framework	Reduced uncertainty, lower transaction cost, and enhanced stakeholder support
Transaction and transition cost	Comprehensive identification and mitigation of social and economic factors that lead to high transaction cost	Advance physical and cyber infrastructure, novel operation schemes, reliable forecast, and robust methods to deal with uncertainty	Manage transaction cost through an integration of institutional, policy, scientific, and technological advances	Reduced transaction cost and better informed decision-making
Unsuitable institutional structure and operation	Identify institutional structures and policies that inhibit proper reallocation	Improve system operation schemes, provide more reliable hydrologic information, and facilitate communication among stakeholders by novel technologies	Establish adaptive institution based on scientific and engineering information support and agency collaboration	Improved institutional support and mitigated institutional barriers

many areas. Non-market means of reallocation will remain the primary avenue of reallocating water in the foreseeable future (*Turner and Hildebrandt*, 2005). Water markets can take considerable time and resources to establish, even if the prerequisite institutions, policies, and infrastructure required to facilitate a water market are already in place, which typically is not the case (especially for inter-sectoral markets). Although much of the literature supports water markets over other means of reallocation because of its potential economic efficiency and decentralized approach, pragmatically, widespread application of water markets may not be appropriate or even feasible in many circumstances, especially cases that require immediate solutions. Thus, research is needed to support non-market based water reallocation (*i.e.,* administrative and collective negotiations), which often have fewer prerequisites and can be implemented more readily. Specific to administrative reallocation, continued research regarding how to optimally reallocate storage in existing multipurpose reservoirs offers a promising means of managing scarce water resources at a very low cost. Regardless of what method of reallocation is employed, it is evident that water reallocation will play a critical role in dealing with growing water scarcity and in some closed basins it is the only way to meet future demands (*Committee on Western Water Management*, 2012).

Chapter 6

CONCLUSIONS

6.1 Concluding Remarks

This dissertation focuses on characterizing human appropriations of fresh water, both directly and indirectly through the global trade network. The set of methods, concepts, and applications developed and explored here allow us to better understand the drivers of unsustainable water use and water dependencies, which stretch across global supply-chains. Connecting nonlocal water dependencies with impacts to specific water resources allows for exploration of feedbacks and unexpected outcomes that can only be explained in the context of telecoupled human-nature systems (*e.g.,* California farmers switching to more water-intensive crops during the worst drought on record).

6.1.1 Groundwater overexploitation is not just a local issue

As globalization expands, groundwater resources act as a global commons shaped by distant actors and exchanges (*Sanderson and Frey*, 2015). The systemic causes of groundwater overexploitation are not fully understood because there is a mismatch in scale between water use impacts and water demands, which are both local and global in nature (*Vörösmarty et al.*, 2015). In this dissertation, we are the first to address this scale mismatch with regards to groundwater overexploitation, focusing on three critical US aquifer systems: the Central Valley, High Plains, and Mississippi Embayment. Irrigated agriculture is contributing to the depletion of these three aquifers. We show that agricultural production within these aquifer regions comprise a significant portion of domestic and international food supply; thus, potential food security implications arise if production significantly decreases to bring groundwater withdrawals within sustainable limits. For the first time, this study tracks and quantifies the food and embodied groundwater resources from these aquifer systems to the final destination and determines the major US cities, US states, and countries that are currently most reliant upon them. Tracking virtual groundwater transfers highlights the role of distant demands on local groundwater sustainability and the fact that aquifer depletion must be considered within its global context.

After establishing nonlocal dependencies on critical aquifers in the United States, we demonstrate how these global dependencies and local water use and decision-making co-evolve under drought shocks. For the first time, we explore the hidden, but important, connections between drought, food, water, and trade. Using the recent drought in California's Central Valley as a case study, we find that drought can have important implications on local water supplies, particularly aquifer depletion, and global consumers' reliance on groundwater-intensive agricultural commodities. To this point, we find that during California's 2012–2014 drought, local and global consumers nearly doubled their reliance on the Central Valley Aquifer, which is being rapidly depleted. Furthermore, our work suggests that global food markets shape farmer's cropping and irrigation decisions, which can in turn lead to greater groundwater depletion, subsidence, and reliance on unsustainable water use by global consumers. This work is critically important in informing consumers around the US and world of their role in local resource use and reliance on nonlocal infrastructure and production.

6.1.2 Water is a critical input in U.S. economic production

The United States economy utilizes water to produce goods and services demanded by society. Despite its importance to economic production, there is little information on what economic sectors are most dependent on America's water resources and where these water dependencies are located. The evolving human and natural pressures placed on the country's water resources necessitate a better understanding of how the nation utilizes its water resources within its economy. In this dissertation, we quantified water footprints of production of U.S. food, energy, services, manufacturing, and mining industries. In doing so, we create the most detailed, comprehensive water footprint assessment of any country to date and offer a database of product water footprints (*i.e.,* VWC) of over 500 industries/products for different state and sub-state areas across the country. We find that 93% of industries are more dependent on water resources used indirectly through their supply chains than they use directly as an input in production. However, sourcing production inputs from more water efficient or inefficient suppliers can change an industry's total indirect water requirements by several orders of magnitude. Additionally, we find that just seven crops are responsible for 75% of the nation's groundwater footprint and 47% of its surface water footprint. Our results also identify geographic clustering of water use by industry, potentially signaling reduced resilience to water-related shocks in the supply chain. This work enables direct comparison of water uses across economic sectors and cities. It will provide a valuable resource for water management and modeling, environmental life cycle assessments, water footprint assessments, benchmarking water use, and demand forecasting and planning.

6.1.3 Water reallocation requires an interdisciplinary approach

The use of water reallocation (*i.e.,* physical water transfers) as an adaptive water management strategy is expected to increase with growing water scarcity. Yet, we do not fully understand the complex feedbacks it invokes between society and nature, which hinders its wider and more effective use. As part of this dissertation, we provide the first survey of the water reallocation literature with the specific purpose of integrating findings across multiple disciplines to understand the major barriers impeding water reallocation. We contribute an interdisciplinary framework to make water reallocation a more viable, effective, and equitable water management strategy.

6.2 Limitations of Current Work

This dissertation highlights the critical importance of existing national databases in the United States, which this work depended upon. The key data sources used throughout this work provide greater spatial resolution, production detail, and site-specific knowledge than data sources commonly used throughout the literature. Although the key data sources we utilize have gone through extensive quality control and assurance measures by the US government agencies that release them, they do not specify the degree of error or uncertainty associated with their estimates. Thus, any unspecified uncertainty in these data products is reproduced within our results. Water footprint sensitivity and uncertainty analysis is an area of active research (*Zhuo et al.*, 2014; *Tuninetti et al.*, 2015) and future research is needed to evaluate the additional uncertainties that are involved when commodity transfers are also considered.

Through this dissertation, we were able to identify opportunities to improve national data collection efforts. The policy and scientific communities would greatly benefit from higher temporal resolution and metered water use data by source. Ideally, every water user in

America would provide sub-annual reports of metered water withdrawals and return flows by source, economic activity, production processes, and the origin and destination of all supplies and sales. This would dramatically improve our understanding of how water is used throughout the economy and the implications of water use, as well as identify those directly and indirectly dependent on the water and the economic activity it supports. Given that much of this data is not available at the level of detail desired, assumptions were necessary to fuse disparate data sources together across different spatial and temporal scales. Though the data sources utilized in this study are of higher quality than those used in most WF studies, there is still room for progress in data collection, which would further reduce the uncertainty surrounding our WF and VW trade estimates.

Due to the data quality and availability issues just discussed, we checked the validity of our main findings using three different approaches. First, we used alternative datasets representing key variable of interests to test the sensitivity of our results when possible. For instance, the California Department of Food and Agriculture (CDFA) and USDA both collect county-level crop production data — a foundational data input. In Chapter 3 we calculated WF and VW trade patterns associated with each dataset. Most of our results only varied by 1-10%, while our main conclusions remained unchanged. Second, we tested our findings against other published values of water use. Using Chapter 3 as an example again, we found that our estimates of groundwater irrigation were within 3.9% of that estimated by *Howitt et al.* (2014). These tests demonstrate the robustness of our approach and key findings and gives us confidence in the general patterns and trends that we present. Finally, we performed a sensitivity analysis by isolating and varying key variables to assess their impact on our major results. For example, in Chapter 4 we inspect the impact of variability in consumption coefficients, water-use coefficients, model parameters, non-revenue water losses, and other assumptions had on our key findings.

6.3 Future Research Extensions

The methods and datasets created as part of this dissertation open several natural avenues for future research in water resources and beyond, such as:

- Exploring the relationship between urban areas and the nonlocal river basins they are indirectly reliant on. The methodological framework and WF datasets developed in this dissertation can address outstanding questions, such as: i) Which watersheds are most critical in meeting US cities direct and indirect water requirements? (ii) Does virtual water trade allow for 'infrastructure sharing' across basins? and (iii) Do the physical and virtual flows of water mitigate or intensify water stress in US river basins?

- Comprehensively quantify the hydro-economic interdependencies and exposures for all urban areas of the United States using metrics developed by *Rushforth and Ruddell* (2015) and *Rushforth and Ruddell* (2016). The methodology we employ in this dissertation could be used to calculated embedded energy as well, yielding interdependent FEW relationships.

- Results from this study could help set water footprint benchmarks per product or industry. Additionally, river basin water footprint caps could be determined to put sustainable limits on water extraction, as suggested by *Hoekstra* (2014).

- The decoupling of consumption from the local resource base is not only an issue pertinent to water resources, but all natural resources. The full social and environmental cost associated with the use of natural resources in the global production and trade system are rarely embodied in the price paid by consumers — a misevaluation that can cause inefficient consumption decisions and unsustainable resource use. The framework used in this dissertation can illuminate the hidden linkages between nonlocal consumption patterns and local impacts related to production, use, or extraction of other resources.

In conclusion, the work presented here provides an important first step toward empowering producers, consumers, water planners, and decision makers to manage water resources more holistically and at the appropriate scale by linking more detailed understanding of local water used in production processes with new knowledge of virtual water transfers. As water becomes scarcer and globalization increasingly connects consumers to distant natural resources, this work is imperative to understand the demand forces that are contributing to the unsustainable use and indirect dependencies of these water resources.

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