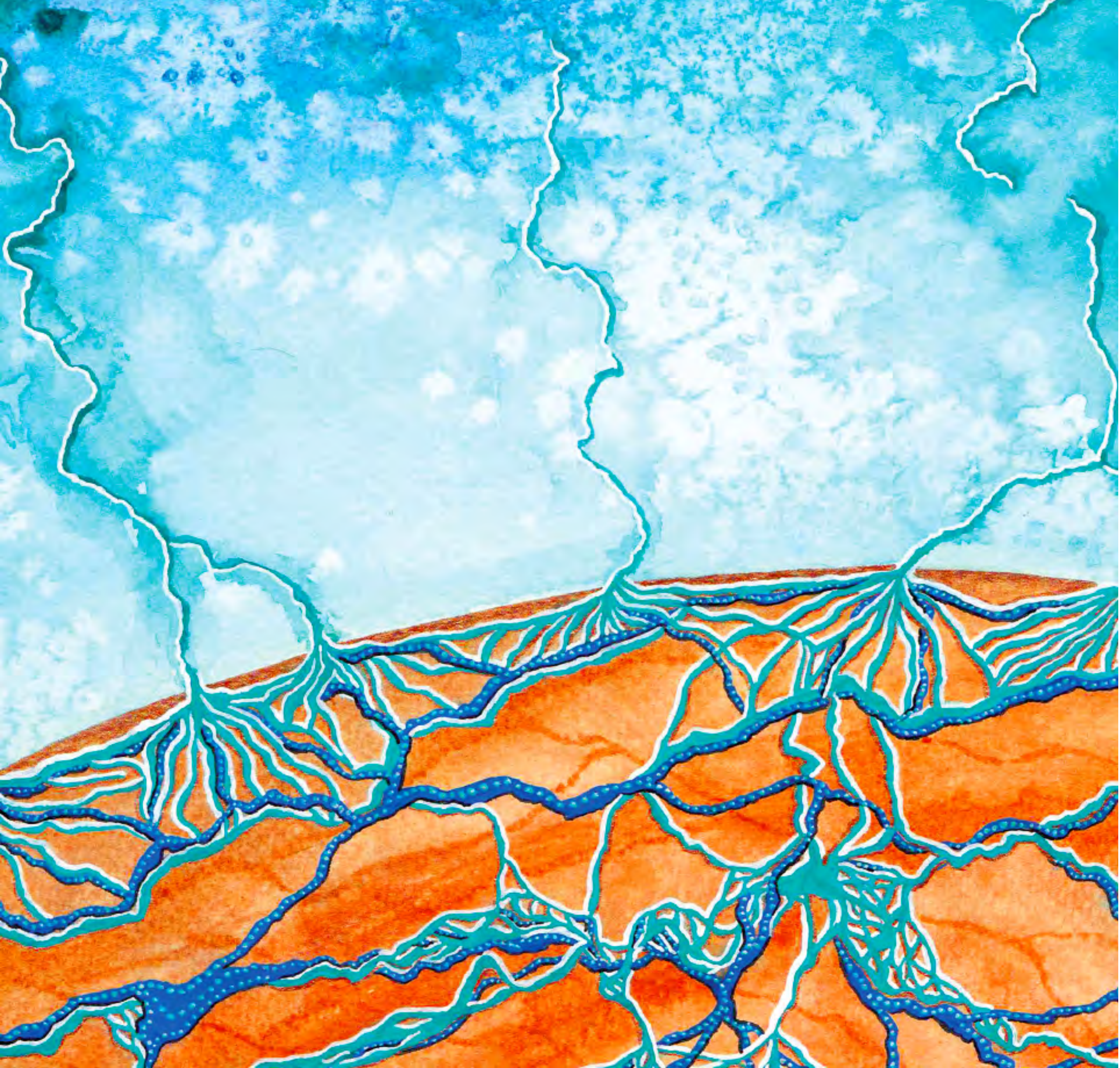
SUSTAINABLE AND EFFICIENT WATER USE

from water footprint accounting to setting targets

RICK HOGEBOOM



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FROM WATER FOOTPRINT ACCOUNTING TO SETTING TARGETS

Rick J. Hogeboom

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SUSTAINABLE AND EFFICIENT WATER USE

FROM WATER FOOTPRINT ACCOUNTING TO SETTING TARGETS

DISSERTATION

to obtain

the degree of doctor at the University of Twente,

on the authority of the rector magnificus,

prof. dr. T.T.M. Palstra,

on the account of the decision of the doctorate board,

to be publicly defended

on Thursday 27 June 2019 at 14:45

by

Hendrik Jan Hogeboom

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This research was supported by the Netherlands Organisation for Scientific Research - Earth and Life Sciences (NWO-ALW), project 869.15.007.

This research was conducted under the auspices of the Graduate School for Socio-Economic and Natural Sciences of the Environment (SENSE).

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A tree has been planted for every copy of this thesis.

ISBN: 978-90-365-4790-1

DOI: 10.3990/1.9789036547901

URL: https://doi.org/10.3990/1.9789036547901

Table of Contents

Acknowledgements

Embarking on a PhD trajectory is an exciting and challenging step in your career. It is a long and rewarding journey, that alternates between traversing rocky roads and smooth sailing. You encounter fellow PhD-pilgrims, dangerous reviewers, supportive colleagues, ambitious students and critical editors. It is also a period of formation and self-reflection, with ample time to battle your inner crises on whether this is the right journey for you or to find out what you want to do when you grow up. I am happy I reached the destination shore, sound and well, and with most crises resolved. And with a finished thesis. I cannot deny I'm a bit proud of the result you are holding in your hands right now. A result that I could not have accomplished without the help and support of many people, and I would like to use this space to thank some of them.

First and foremost I would like to thank Arjen and Maarten, who supported and guided me through my PhD journey. Maarten, if it wasn't for your encouraging instigation during my master's final project that I should compete for the TGS Award 2013 (that would potentially grant me funding for a PhD trajectory), I would never even have considered pursuing a PhD. I had no academic aspirations at the time and was not sure if it would fit me. But your enthusing confidence persuaded me to give it a try. I am very happy that I did, as I grew to enjoy the PhD journey way beyond my initial expectations. You greatly contributed to this enjoyment, with your ever positive words and incitement, your unabated willingness to chew over any deliberation I threw at you, and your admirable collegiality. Thank you so much for everything.

Arjen, it was a privilege to have you as advisor. I benefitted a lot from your vision and wisdom, as you showed me what it is like to be a star researcher and what it takes to be successful in academia. While some PhD candidates hardly ever speak to their advisors, we had ample discussions over the past six years – mostly over lunch or in the hallways rather than in a formal meeting. Thankfully you are not the micro-managing type, allowing me to find my own course in research. From the onset of my PhD, you granted me many opportunities to build my skills and broaden my experience. I'll never forget stepping in for you, for keynote speech at an expert workshop in Berlin – just a few months into my PhD. New and nervous among a group of solely senior professors, an ambition was ignited in me to aspire for greater

things. Fast-forward to today, leading Water Footprint Network and staying on in your research group as Assistant Professor testify to that ambition materializing, and I know full well that these accomplishments would not have been possible without your continued support and trust in me.

Joke, a special word of thanks for you. Together with Anke you are the backbone of the department, making our lives so much easier by making sure everything work-related was functioning, paid or present. Also thank you, Joep and Lara, for standing beside me during my defense as my paranymphs.

Over the past years, I worked, talked and laughed with many students and colleagues – both in and beyond the WEM department – whom I all would like to thank for the good times and fun. Joep, thanks for the many conversations, our shared struggles and infinite number of times we postponed the deadline for our final model simulations. How often have we said to one another: 'Thís week we will finalize the reference run of our *Aqua21* model', just to spend yet another few months debugging the code. When at some point we were both being interviewed and the reporter commented we sounded like a married couple – "You are finishing each other's sentences!" – I realized just how deep into this effort we both were. And that it was time to finally get this model going.

Luckily, I received some assistance from a few outstanding master students whom I got to supervise with their final projects. Thanks Luuk, Davey, Ilja and Kees, for your hard work and contribution to this thesis.

Occasionally, I needed to escape the desk-study grind. The futsal indoor soccer league provided a welcome weekly distraction. Thanks, all fellow South-WEMton players. Another (quite substantial) distraction I found in Water Footprint Network, in which Joke, Joep, Lara and Arjen were and are my valued partners in crime. Thank you for all your efforts in trying to make WFN thrive. Yet another activity that kept me distracted from research, was Wetskills. A big thank you to all Wetskills team members with whom I was fortunate enough to experience memorable events all around the world: in South Africa Nelspruit, South Africa Cape Town, The Netherlands, India, Israel and Oman.

I'd finally like to thank my friends and family for the many fun moments away from research; our church De Schuilplaats for all support we received over the past years, also in times of hardship; Anneke and Aletta for your steady compassion and great help during these years. I'm so sorry, Aletta, you could not live to share this moment with us, as I'm sure you would have enjoyed every moment of it; a big thanks to my parents for giving me the opportunity to go to college, the love and encouragement. Mom, I know you are happy we decided to stay in Enschede rather than move abroad – even though you, Dad, seem to be traveling more nowadays than I do and it was the both of you who taught us to go out and explore the world; my dearest Elma, love of my life, for your unfailing and unconditional love. What you had to go through these years makes any PhD path resemble kindergarten. Your strength and endurance are a true inspiration; God, Who carried us and is always for us: "Taste and see that the LORD is good; how blessed is the one who takes refuge to Him." (Psalm 34:8)

Summary

Humans consume freshwater in all sectors of the economy and across all layers of society. Growing populations, economic development and climate change add to already large pressures on freshwater resources in many places around the world. The challenge for authorities, planners – and also businesses, farmers and investors – is to strike an acceptable and sensible balance in allocating limited freshwater resources to the various demanding and often competing uses without compromising nature.

Two policy instruments that have emanated from the field of Water Footprint Assessment (WFA) are particularly promising to help transition to sustainable and efficient use of freshwater worldwide. The first is setting water footprint (WF) caps at the river basin level, aimed at preventing overshoot of limited natural endowments and to reconcile human freshwater appropriation with conservation. The second is formulating water footprint benchmarks per water-using activity, aimed at identifying reference levels of 'reasonable' WFs for specific water-using activities. This research investigates these two instruments in five different studies reported in Chapters 2-6.

Chapter 2 quantifies maximum sustainable WF levels for all river basins in the world, using multiple state-of-the-art Global Hydrology models and Environmental Flow methods. The study proposes various WF caps that set an upper limit to aggregate WFs in a basin and their implications, thereby effectively quantifying humanity's safe operating space in terms of freshwater consumption.

Chapters 3-5 relate to efficient use of water. Chapter 3 quantifies WF benchmarks of global crop production, using a newly developed model, *Aqua21*, that calculates WFs of major crops on a high spatiotemporal resolution at a global scale. The study reveals that large water savings can be made if producers of crops would reduce their WFs to benchmark levels. Moreover, the analysis shows that much of the blue water savings thus made can be achieved in severely water-scarce regions, thereby potentially alleviating blue water scarcity.

Chapter 4 estimates WFs of manmade reservoirs worldwide and attributes that WF to a the various purposes of the reservoirs: hydroelectricity generation, irrigation, residential and industrial water supply, flood protection, fishing and recreation. The

study found that reservoirs are large water consumers that add substantially to humanity's blue WF.

Chapter 5 moves from a global to a local perspective and zooms in on a case study on silk production in Malawi. The research analyzed the WF of mulberry shrubs (the leaves of which are fed to the silk worms) under various farm management systems, and compared these to crops usually grown in the area. The study thus informs farmers on water considerations when deciding which crops to grow.

Establishing and maintaining sustainable and efficient water consumption is a shared responsibility, where a role is to be played by each of the actors involved. Therefore, the last study in Chapter 6 focused on an under-emphasized yet influential actor, namely institutional investors. In this study, a framework is developed and applied to assess to what extent investors incorporate water sustainability targets in their investment decisions. Although the results show that concerns over widespread water scarcity and inefficient water use are largely invisible to them, the study provides investors with systematic handles to give substance to their improvement efforts.

Research on setting WF caps at the river basin level and formulating WF benchmarks for water-using activities is in its infancy. There are few practical examples of uptake in policy, despite a sensed urgency to act by many actors. This research presents various new studies on WF caps and benchmarks as well as several rich datasets that can be used in future research. It thereby significantly advances the discourse on how to transition to sustainable and efficient use of freshwater resources worldwide.

Samenvatting

Mensen gebruiken zoetwater in alle sectoren van de economie en in alle lagen van de maatschappij. Op veel plekken op aarde drukt deze watervraag ondertussen flink op de lokale zoetwatersystemen. Deze druk wordt daarbij verhoogd door bevolkingsgroei, economische ontwikkeling en klimaatverandering. De uitdaging waar overheden, ontwikkelaars, bedrijven, boeren en investeerders daarom gezamenlijk voor staan is het vinden van een acceptabele balans. Een balans in hoe het beperkt beschikbare water wordt verdeeld over de verschillende gebruikers, zonder dat de natuur daarbij vergeten of geschaad wordt.

Uit het onderzoeksveld Water Footprint Assessment zijn twee veelbelovende beleidsinstrumenten naar voren gekomen die kunnen helpen de transitie naar duurzaam en efficiënt watergebruik wereldwijd mogelijk te maken. Het eerste is het instellen van een plafond aan de watervoetafdruk in een stroomgebied. Het doel van deze maatregel is te voorkomen dat méér water wordt gealloceerd dan beschikbaar is, terwijl er ook expliciet water voor de natuur wordt gereserveerd. Het tweede is het ontwikkelen van referentieniveaus voor de watervoetafdruk voor iedere activiteit die water gebruikt. Het doel van deze maatregel is een 'redelijke' watervoetafdruk te vinden voor specifieke activiteiten, om zo verspilling te voorkomen. Dit onderzoek richt zich op de doorontwikkeling van deze twee beleidsinstrumenten, middels vijf verschillende studies die zijn beschreven in Hoofdstukken 2 tot 6.

Hoofdstuk 2 kwantificeert de maximale watervoetafdruk die nog als duurzaam kan worden beschouwd. Dit is gedaan voor alle stroomgebieden op aarde. We hebben hiervoor een aantal gerenommeerde hydrologische modellen gebruikt, als ook enkele methoden die schatten hoeveel water de natuur nodig heeft. Deze studie presenteert een aantal watervoetafdrukplafonds om de bovengrens van het totale watergebruik in een stroomgebied aan te geven. Ook wordt een aantal mogelijke implicaties bij ieder plafond beschreven. De studie geeft daarmee een kwantitatief overzicht van hoeveel water de mensheid duurzaam kan gebruiken.

Hoofdstukken 3 tot 5 gaan over efficiënt gebruik van water. Hoofdstuk 3 kwantificeert referentieniveaus voor de watervoetafdruk van gewassen. Hiervoor is een nieuw model ontwikkeld, genaamd Aqua21, waarmee de watervoetafdrukken van de grootste gewassen wereldwijd in kaart zijn gebracht op een hoge resolutie

in tijd en ruimte. De studie laat zien dat grote hoeveelheden water bespaard zouden kunnen worden als boeren hun watergebruik verminderen tot de vastgestelde referentieniveaus. Bovendien geeft de analyse aan dat een groot deel van de blauwe waterbesparing die zo bereikt wordt, kan worden gerealiseerd in gebieden die nu al met ernstige schaarste te kampen hebben. Het verminderen van deze blauwe waterverspilling kan bijdragen aan het verlichten van blauwe waterschaarste.

Hoofdstuk 4 schat de watervoetafdruk in van stuwmeren wereldwijd, en schrijft deze toe aan de verschillende doelen waar de reservoirs voor worden gebruikt, zoals het opwekking van elektriciteit, aanvoer voor irrigatie, naar huishoudens en bedrijven, het beschermen tegen overstromingen, visserij en recreatie. De studie laat zien dat stuwmeren grote watergebruikers zijn die substantieel bijdragen aan de totale blauwe watervoetafdruk van de mensheid.

In Hoofdstuk 5 verschuift het perspectief van mondiaal naar lokaal, middels een casus over de productie van zijde in Malawi. Deze studie brengt de watervoetafdruk van verschillende teeltvormen van moerbeistruiken in kaart (moerbijblaadjes vormen het voedsel voor de zijderupsen). Deze watervoetafdruk van zijde is vergeleken met die van gewassen die nu in het gebied verbouwd worden. De analyse verschaft lokale boeren inzicht in water-gerelateerde overwegingen bij het kiezen van de te telen gewassen.

Het bereiken van duurzaam en eerlijk gebruik van water wereldwijd is een gedeelde verantwoordelijkheid, waar alle betrokken partijen een rol te spelen hebben. In de laatste studie in hoofdstuk 6 richten we ons op een onderbelichte maar zeer invloedrijke groep belanghebbenden, namelijk institutionele beleggers. In deze studie wordt een raamwerk gepresenteerd dat beoordeelt in hoeverre investeerders rekening houden met water-gerelateerde aspecten in het maken van investeringsbeslissingen. De resultaten laten zien dat de zorgen over wijdverbreide waterschaarste en inefficiënt watergebruik grotendeels aan investeerders voorbijgaan. Tegelijkertijd biedt de studie praktische handreikingen aan investeerders om hun verbetertrajecten systematisch vorm te geven.

Onderzoek naar het stellen van watergebruiksplafonds per stroomgebied of het ontwikkelen van referentieniveaus voor watergebruik per activiteit staat nog in de kinderschoenen. Ondanks de onderkende urgentie met betrekking tot waterproblematiek bij alle belanghebbenden, zijn er slechts weinig voorbeelden waar genoemde maatregelen daadwerkelijk in beleid zijn opgenomen. Dit onderzoek heeft door middel van zowel de beschreven nieuwe studies als door het delen van rijke datasets significant bijgedragen aan de wetenschappelijke discussie over hoe we de transitie naar duurzaam en efficiënt watergebruik wereldwijd kunnen gaan realiseren.

1

INTRODUCTION

1. Introduction

1.1. The importance of freshwater to humans and nature

Freshwater is a finite and vulnerable resource, essential to sustain life, development and the environment (ICWE, 1992). Starting as precipitation over land, water either directly evaporates from the land surface back into the atmosphere (known as green water) or it finds its way to rivers, lakes and aquifers (known as blue water).

Humans consume freshwater in all sectors of the economy and across all layers of society. It is used by firms, farms and families, to produce food, feed, fuel and fibers (UN-WWAP, 2019; Hoekstra, 2013). The lion's share of humanity's water consumption lies in agriculture, accounting for 92% of freshwater fluxes appropriated for human use, with the remainder being shared between domestic and industrial users at 4% each (Hoekstra & Mekonnen, 2012).

The environment needs water too, to support ecosystems, provide habitats for (aquatic) species and bolster biodiversity (Pastor et al., 2014; Vörösmarty et al., 2000; Rockström, 2004). Moreover, water-dependent ecosystem services provide life-supporting functions indispensable to human well-being (Costanza et al., 1997; Costanza et al., 2014).

1.2. Humanity is using too much water

Pressure on freshwater resources is grave and growing in many places around the world. Population growth, economic development, increased biomass demand for bio-energy and a shift towards more water-intensive diets all amplify human demand for water (Hejazi et al., 2014; Vanham et al., 2018; Mekonnen et al., 2016). While future projections of water related to food and energy security and environmental degradation sketch a dim outlook (Kummu et al., 2016; Greve et al., 2018; Ibarrola-Rivas et al., 2017), we do not need to wait to see detrimental consequences of overexploitation of water resources materialize. Already today, half a billion people live in regions that face severe water scarcity year-round (Mekonnen & Hoekstra, 2016); economic momentum is stalling because of lack of access to clean and sufficient freshwater (World Bank, 2016); habitats associated with 65% of continental discharge are being threatened by over-abstraction (Vörösmarty et al., 2010); and social and political conflicts over water are a reality in many places (Kundzewicz & Kowalczak, 2009; Munia et al., 2016). Failure to restrain humanity's growing and unsustainable water footprint make reaching UN's Sustainable Development Goals (SDG) a daunting task – not only dedicated SDG6 '*to ensure availability and* *sustainable management of water and sanitation for all*' – but also other SDGs for which water is foundational (Jägermeyr et al., 2017; Colglazier, 2015).

1.3. Sustainable and efficient water consumption

The challenge for authorities, planners – and also businesses, farmers and investors – is to strike an acceptable and sensible balance in allocating limited freshwater resources to the various demanding – and often competing – uses (Postel et al., 1996; Hoekstra, 2014a; Gleick, 2003). This balance can be operationalized along two dimensions: *sustainable scale and efficient use*. The former refers to the sustainability of aggregate consumption volumes given the water system's (environmental) carrying capacity, to avoid scarcity and overexploitation of available water resources, while the latter means unnecessary or unproductive use of scarce water resources is avoided (Hoekstra, 2014a; Hoekstra, 2013; Daly, 1992; Steffen et al., 2015).

1.3.1. Measuring water consumption

Planning and policy instruments that strive to achieve sustainable and efficient water consumption require indicators to assess and evaluate their effectiveness. The water footprint (WF) concept provides such an indicator, since a WF measures water consumption along the entire value chain of a product or activity (Hoekstra et al., 2011). Multi-dimensional by design, a WF can express, for example, the amount of green and blue water that is consumed to grow wheat in a certain place during a given period (in m^3 t^{-1} or in m^3 ha^{-1}) or to generate hydroelectricity from a reservoir (in GJ m^{-3}) or to manufacture any industrial product (in m^3 $unit^{-1}$).

Over the past decade, Water Footprint Assessment (WFA) developed as a new field of research and application (Hoekstra, 2017), with numerous studies being published that assessed WF accounts of various products (Ercin et al., 2011), sectors (Mekonnen & Hoekstra, 2011) and countries (Hoekstra & Mekonnen, 2012; van Oel et al., 2009).

1.4. Towards quantified water targets

Water footprints become more meaningful if assessed in terms of their sustainability and efficiency, and more actionable if employed to set water targets that help reduce WFs where necessary. Two policy instruments that have emanated from the field of WFA that are particularly promising as a basis to formulate quantifiable water targets are i) water footprint caps at the river basin level, and ii) water footprint benchmarks per water-using activity (Hoekstra, 2013).

A (blue) WF cap pertains to the sustainable scale of water consumption, and represents an upper ceiling to total water consumption from renewable groundwater and surface water, accounting for the fact that the natural replenishment rate is limited and that part of the water flow needs to be reserved for nature. A cap aims to prevent overshoot of limited natural endowments and to reconcile human freshwater appropriation with conservation, thereby forming a quantified target for the maximum level of the aggregate WF in a basin (Hoekstra, 2014a).

A water footprint benchmark relates to efficient use of water, and identifies a 'reasonable' WF for a specific water-using activity, such as producing a product or growing a crop. A WF benchmark therefore constitutes a quantified reference point for producers that they can possibly strive for or adhere to, e.g. by resorting to more water-efficient technologies and methods, and for governments to use as basis for issuing water consumption permits (Mekonnen & Hoekstra, 2014).

1.4.1. The effect of inter-annual variability

To date, several global-scale high-resolution WF assessments have been carried out that may serve as input for assessing water targets at the global scale, e.g. Hoekstra & Mekonnen (2012); Mekonnen & Hoekstra (2012a); Mekonnen & Hoekstra (2012b); Mekonnen & Hoekstra (2011). However, these studies provide WF estimates based on long-term-averaged climatic data and relatively coarse water-balance modelling. Since both water demand and availability vary in time, these time-averaged datasets and coarse models cannot reveal the effects of inter-annual variability to which both water demand and availability are subjected. This poses a challenge to our understanding of the temporal dynamics of setting quantified water targets. After all, water availability is lower in relatively dry months or years and higher in relatively wet periods. Crop water productivity will also be higher or lower from one growing season to the other. Quantifying water targets thus requires an improved granularity in existing WF accounts, particularly regarding the temporal resolution, in order to study effects of variability over time on formulating water targets.

1.4.2. A shared responsibility

Regarding sustainable and efficient water use policies, *governments* typically determine the rules of the game, meaning they can agree on allocating or pricing water consumption permits based on WF benchmarks, or on setting WF caps for river basins in their jurisdiction to ensure consumption stays within safe ecological boundaries (see e.g.

Grafton et al. (2014) for an application in the Murray-Darling Basin in Australia), or extend supply by building reservoirs. With agriculture as the main water user, *farmers* clearly have an important role to play too, in avoiding unproductive evaporation from their fields and choosing crops suited to local (climatic) conditions. *Companies* in the food, beverage and apparel sectors can encourage or require their supplying farmers to adopt water-efficient production systems. Towards the future, particularly *investors* wield substantial influence, since their decisions made today whether or not to invest in certain water-using activities profoundly affect the state and shape of water resources of tomorrow. It goes to show that achieving sustainable and efficient water use globally is a shared responsibility, where a role is to be played by each of the actors involved (ICWE, 1992; UN-WWAP, 2019; Hoekstra, 2013).

1.5. Research objectives

The overarching objective for this research is to feed the discourse on sustainable and efficient consumption of freshwater resources worldwide. More specifically, I aim to investigate how the field of Water Footprint Assessment can advance from water footprint accounting to formulating quantified water footprint reduction targets, particularly regarding two policy instruments, i.e. setting water footprint caps at the river basin scale and formulating water footprint benchmarks for water-using activities.

I distinguish two main research questions, which are centered around these two promising policy instruments:

<u>Q1.</u> *How can policies that promote water consumption at a sustainable scale be enriched by the notion of setting water footprint caps per river basin?*

<u>Q2.</u> *How can policies that promote efficient use of freshwater resources benefit from formulating water footprint benchmarks per water-using activity?*

1.6. Approach

Straightforward as it may sound, the idea of setting a WF cap is novel and still in its infancy stage, with only one study exploring its potential merits (Zhuo et al., 2019). This study, however, concerned just one basin in China and left many questions unanswered, e.g. regarding the role of temporal variability in availability and regarding uncertainties in estimating runoff and environmental flow requirements. For Question 1, therefore, I developed WF caps for all river basins in the world, based on i) a suit of state-of-the-art Global Hydrology Models that provide monthly and annual estimates of water availability, and ii) a suit of Environmental Flow Methods that estimate environmental flow requirements (**Chapter 2**). I explored effects of inter-annual and intra-annual variability on the resulting WF caps, for various scenarios, each of which was based on a different WF cap definition. While the main purpose was to explore the implications of setting WF caps as target at the basin level, this study extended into the discussion on a Planetary Boundary for freshwater use (viz. a global WF cap), to explore potential quantification pathways of such a global 'target' as well (Rockström et al., 2009; Steffen et al., 2015).

For Question 2, I carried out three independent studies, for different cases: two global studies (water consumption in crop production and water consumption from artificial reservoirs) and one local study (water consumption in silk production in Malawi).

First, I focused on farmers' water use in crop production. Food production is responsible for the lion's share of water use in the world, hence setting WF benchmarks for particular crops promises substantial potential water savings in agriculture – should producers strive to achieve these benchmarks (**Chapter 3**). Several previous studies probed the concept of formulating WF benchmarks for crop production, with the first global assessment by Mekonnen & Hoekstra (2014) showing – for 124 crops and averaged over the period 1996-2005 – WF benchmark levels based on the spatial variability of crop WFs. For winter wheat in China, Zhuo et al. (2016) dove deeper by exploring the temporal variability in WF benchmarks, and formulated WF benchmark per climate zone. I built on these studies by quantifying global climate-specific WF benchmarks for the 57 most important food crops, while addressing implications of inter-annual variability. Hereto, I estimated multi-year global WFs of these crops at a 5 × 5 arc minute spatial resolution, using FAO's flagship crop model AquaCrop (Steduto et al., 2009; Raes et al., 2012). Next, I sorted – within each climate zone – WFs from small to large, and selected various percentiles as potential benchmark levels (e.g. the 25th production percentile). In addition to developing these consumptive (green plus blue) WF benchmarks, this study is the first-ever to explore specific blue WF benchmarks that complement the overall consumptive WF benchmark. I estimated how much (blue) water can be saved annually and globally if all producers where to meet the WF benchmarks set by the 25th best-production percentile.

Second, I calculated WFs of over 2000 of the world's largest manmade reservoirs, for the period 1970-2005, which I subsequently attributed to various reservoir purposes, i.e. hydroelectricity generation, irrigation, residential and industrial water supply, flood protection, fishing and recreation (**Chapter 4**). Humans have resorted to building dams to increase, guarantee or stabilize supply for centuries, but while a reservoir may be an apt measure to increase water availability during a certain time of the year, they paradoxically only come at the cost of reducing total water availability over the whole year (Shiklomanov & Rodda, 2003; Di Baldassarre et al., 2018). The reason is they 'loose' water through evaporation from their surfaces. This blue WF of reservoirs has been ignored in global WF studies, even though it constitutes a substantial share of humanity's total blue WF (Hoekstra & Mekonnen, 2012). Although the emphasis in the current study lies more on WF accounting than on benchmarking reservoirs or reservoir purposes, it establishes a solid basis to do so for future research.

Third, I developed – in a case study for producing silk in Malawi – field-level WF benchmarks for specific crops, in an attempt to illustrate how WF benchmarks may aide farmers in choosing which crops to grow (Chapter 5). Alternative to the ranking-method employed by Mekonnen & Hoekstra (2014), Chukalla et al. (2015) tried an alternative method of formulating WF benchmarks that is based on identifying WFs associated with the use of best available technologies (e.g. drip irrigation, mulching, deficit schemes). I calculated, once more using AquaCrop, WFs of various crops grown (and planned to be grown, in this case mulberry shrubs for silk production) on several estates in Malawi, under differing farming systems, including various irrigation techniques and strategies. I estimated locally attainable WFs of the considered crops to quantify inefficiencies related to each of the evaluated farming systems.

In the final study of this thesis, I aimed to bring the two topics of sustainable and efficient water use together in a practically applicable framework for investors (Chapter 6). I focused on the actor group currently under-emphasized in the discourse on sustainable and efficient water management, namely institutional investors – banks, pension funds and insurance companies. Taking an earlier framework targeting multinational companies as inspiration (Linneman et al., 2015), I developed an assessment framework to investigate to what extent investors incorporate water sustainability targets in their investment-decisions. I scored and ranked several large Dutch investors based on their current policies, distinguishing leaders and frontrunners from followers and stragglers, in an attempt to incentivize investors to improve their business practice with regards to sustainable and efficient use of water resources.

Throughout this research, I used the Global WFA standard as reference for terminology and definitions (Hoekstra et al., 2011). A conceptual structure of this research is shown in Figure 1-1.

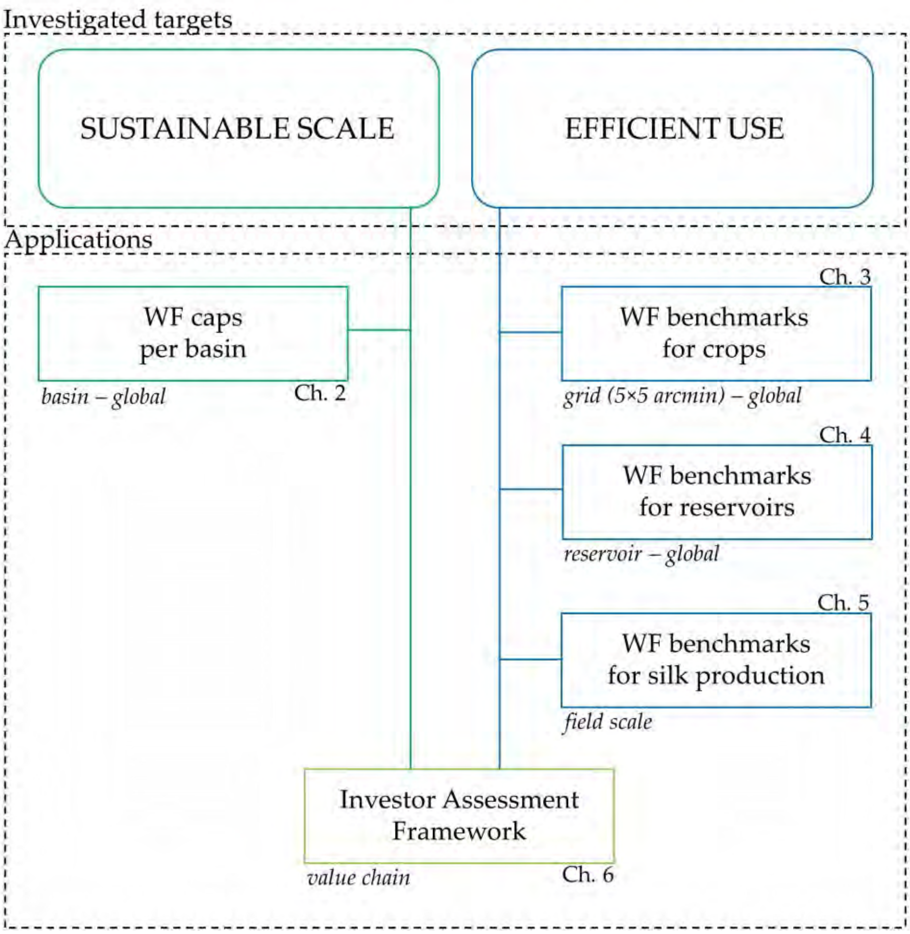


Figure 1-1. Conceptual diagram of this thesis.

2

CAPPING HUMAN WATER FOOTPRINTS IN THE WORLD'S RIVER BASINS

2. Capping Human Water Footprints in the World's River Basins

Abstract

Profligate water use and overexploitation of limited freshwater resources around the world cause widespread water scarcity, economic downturn and conflicts over water. A promising policy measure to bridle humanity's unsustainable water footprint (WF) is to set local and time-specific water footprint caps, to ensure water appropriation for human use remains within maximum sustainable limits. Here, we quantify – for the world's river basins – monthly allocable blue water flows for human consumption, while explicitly earmarking water for nature. Addressing the implications of temporal variability, we describe trade-offs between potentially violating environmental flow requirements versus underutilizing available flow. Capping water consumption, to support the transition towards sustainable freshwater use, is urgent in all river basins where water resources are already overexploited, which concerns about half the world's basins.

2.1. Introduction

Pressure on freshwater resources is grave and growing in many places around the world, with detrimental consequences. Already today, half a billion people are facing severe water scarcity year-round (Mekonnen & Hoekstra, 2016). Economic downturn because of lack of clean and sufficient freshwater is a new reality for many (World Bank, 2016). Overexploitation and mismanagement undermine biodiversity and resilience of aquatic ecosystems that provide life-supporting functions (Vörösmarty et al., 2010) and social and political conflicts over water are looming (Kundzewicz & Kowalczak, 2009).

Future projections of surging demand sketch a dim outlook (Kummu et al., 2016; Greve et al., 2018; Veldkamp et al., 2017). By 2050, nearly half the world population is estimated to live in places with insufficient land and blue water resources (i.e. surface and groundwater) to meet local demand for food production (Ibarrola-Rivas et al., 2017; Foley et al., 2011). Concerns have been raised as to whether enough freshwater is available to complete the energy transition under the pathways currently pursued by the International Energy Agency, particularly water availability limiting bio-energy production (Mekonnen et al., 2016). Moreover, failure to restrain humanity's growing and unsustainable water footprint (WF) makes reaching UN's Sustainable Development Goals (SDG) a daunting task – not only dedicated SDG6 'to ensure availability and sustainable management of water and sanitation for all' – but also other SDGs for which water is foundational (Jägermeyr et al., 2017; Colglazier, 2015).

The key issue concerning humanity's current water footprint is that it already exceeds sustainability thresholds in many places, indicating we are not living within our (local) means in terms of water use (Hall et al., 2014; Mekonnen & Hoekstra, 2016). An appealing and apparent policy measure to prevent overshoot of limited natural endowments and to reconcile human freshwater appropriation with conservation is to set a blue water footprint cap at the river basin scale (Hoekstra & Wiedmann, 2014; Hoekstra, 2013). A blue WF cap represents an upper ceiling to total water allocations from renewable groundwater and surface water, accounting for the fact that the natural replenishment rate is limited and that part of the water flows need to be reserved for nature. Straightforward as it may sound, the idea of setting a WF cap is novel and still in its infancy stage, with only one study exploring its merits (Zhuo et al., 2019). This study, however, concerned only one basin in China and left many questions unanswered, e.g. regarding the role of temporal variability in availability and uncertainties in estimating runoff and environmental flow requirements. Initial attempts to formalize a WF cap in policy were made in Australia's Murray-Darling basin, showing preliminary success as well as several difficulties, especially regarding how to deal with inherent temporal variability in water availability (Grafton et al., 2014).

This study aims to propel the discourse on curbing freshwater consumption by investigating potential merits and drawbacks that come with setting WF caps at the basin level. It is the first-ever study to quantify monthly blue water footprint caps for the world's river basins, while addressing implications of temporal variability in availability that emerge once a WF cap is set in policy.

We first estimated maximum sustainable levels of monthly blue water availability by subtracting environmental flow requirements (EFR) from pristine runoff, for each basin in the world and each month in the period 1970-2005. Monthly blue water runoff was taken from three state-of-the art Global Hydrology Models (Wada et al., 2016), of which we calculated an ensemble mean to account for model uncertainty (Haddeland et al., 2011). Likewise, three well-known methods for establishing EFR (Pastor et al., 2014; Richter et al., 2012; Smakhtin et al., 2004) and their ensemble mean provided us with monthly EFR to be set aside in each basin to guarantee proper aquatic ecosystem functioning (Oki & Kanae, 2006; Vörösmarty et al., 2010). The high spatiotemporal resolution of the models captures inter-annual and intra-annual variability in the resulting water availability levels, allowing the exploration of dynamics that setting a WF cap at a certain historic or average level will likely bring about. We hereto present three procedures for formulating a WF cap, set at varying levels of blue water availability, and address implications of temporal variability in terms of unutilized WF potential and implicitly allowed violations to environmental flow requirements.

This study presents a major advancement to the field of Water Footprint Assessment (Hoekstra, 2017), by providing a first exploration of how we could formulate WF caps, the uncertainties involved and the implications once adopted in policy. The study is highly relevant for developing well-informed policy at the basin level, by proposing a means to transition away from persistent overshoot towards sustainable use of a basin's limited freshwater resources. Moreover, we add to the contemporary discourse on a planetary boundary for freshwater use (Gerten et al., 2013; Steffen et al., 2015; Rockström et al., 2009) by postulating regionalized and time-specific upper ceilings that must underlie any global annual planetary boundary for water use (Heistermann, 2017).

2.2. Method and data

National reports on 'annual renewable water resources' as provided by AQUASTAT (FAO, 2016) prove inadequate if we are to capture – at the basin level – intra-annual and inter-annual dynamics of supply. Despite recent attempts to harmonize existing observations (Do et al., 2018), no comprehensive runoff observation systems are in place to provide monthly statistics on maximum sustainable levels of water availability (Syed et al., 2010). For our ambitious global assessment, Global Hydrology Models (GHMs) are therefore the best means to derive water availability estimates, at the high spatiotemporal resolution required (Shiklomanov, 2000; Gleeson et al., 2012).

2.2.1. Blue water availability

Monthly blue water availability, BWA ($m^3\ s^{-1}$), was estimated by subtracting monthly environmental flow requirements, EFR ($m^3\ s^{-1}$), from monthly blue water runoff, BWR ($m^3\ s^{-1}$), for all river basins in the world per month in the period 1970-2005.

2.2.2. Blue water runoff

BWR was derived from three state-of-the-art Global Hydrology Models (GHMs) that were included in IIASA's Water Futures and Solutions initiative, which aims to establish a consistent set of global water scenarios using similar forcing and input data, thereby facilitating model inter-comparison (Wada et al., 2016). The GHMs used here are H08 (Hanasaki et al., 2008), PCR-GLOBWB (Wada et al., 2014) and WaterGAP (Müller Schmied et al., 2016). Although their routines and algorithms vary, all GHMs operate on a 30×30 arc minute spatial resolution with global coverage (except Antarctica); incorporate the GRanD database on major reservoirs (Lehner et al., 2011); are forced with the same historic meteorological timeseries; and use the flow direction map DDM30 to delineate basins (Döll & Lehner, 2002). By selecting the end nodes of each basin in DDM30, we extracted 11,558 unique basins globally (with many one-cell coastal basins driving this high number). For each basin, monthly pristine runoff as computed by the three GHMs – i.e. the natural runoff without water consumption by humans – for each month in the period 1970-2005 was taken to represent BWR. Differences in model outcomes are known to be a major source of uncertainty (Haddeland et al., 2011), hence we took the ensemble mean of the three GHMs to obtain a best-guess BWR value.

2.2.3. Environmental Flow Requirements

An extensive list of environmental flow frameworks has emerged (Brewer et al., 2016), ranging from hydrological and hydraulic frameworks to habitat simulation and more holistic approaches. Required input data at the global level is only readily available for hydrological methods. We therefore selected three such methods to estimate EFR: Smakhtin (Smakhtin et al., 2004), Richter (Richter et al., 2012), and the Variable Mean Flow (VMF) method (Pastor et al., 2014). The Smakhtin method typically yields low EFR values and distinguishes between high and low flow conditions and allocates a base flow volume represented by the 90th percentile (Q90) of BWR to environmental needs, plus a percentage of remaining flow depending on the flow regime. Likewise, the VMF method sets apart EFR based on high, intermediate and low flow regimes, at which between 30% and 60% of BWR is allocated to the environment. The Richter method is the most precautionary method and takes EFR to be a constant percentage of 80% of BWR without distinguishing between flow regimes. Because monthly EFR estimates in both the Smakhtin and VMF method depend on (annual) basin hydrographs, it can occur that for a particular month EFR exceeds BWR, yielding a negative BWA after EFR is subtracted from BWR. Since this is physically impossible, BWA is set to zero for those particular months. In the Richter method this phenomenon does not occur. Given the known spread between and structural uncertainty of the methods, we took – analogous to the BWR estimates – the ensemble mean of the three environmental flow methods to obtain best-guess EFR values.

2.2.4. WF cap options and implications

We drafted three alternative WF cap options, which differ in the procedure followed to formulate WF caps based on the historical BWA statistics. In the first option, a monthly WF cap is set for each basin at the long-term average of the monthly average BWA over the period 1970-2005. This implies that when actual WFs will equal the level of the caps, WFs will as often exceed the WF cap (thereby violating environmental flows) as underrun it (thereby underutilizing available runoff for human appropriation). In the second option, the monthly WF cap is set at the 25th percentile (Q25) of the monthly average BWA (viz. BWA that is exceeded 25% of the time for a particular month of the year) over the period 1970-2005. In this option, WFs (when equal to the caps) will exceed the cap in fewer occasions than in the previous option, but at the cost of a larger unutilized WF potential. In the third option, the monthly WF cap is set at the minimum monthly BWA that occurred in a particular month during the period 1970-2005. This is the most precautionary definition of what is maximally allowed and implies that total WFs in the basin will always remain below the WF cap. Environmental flows will never be compromised, but the unutilized WF potential will be highest in this option. The implications of the three options are expressed in terms of a trade-off between potentially allowing environmental flows to be violated versus leaving an unutilized WF potential in the river basin.

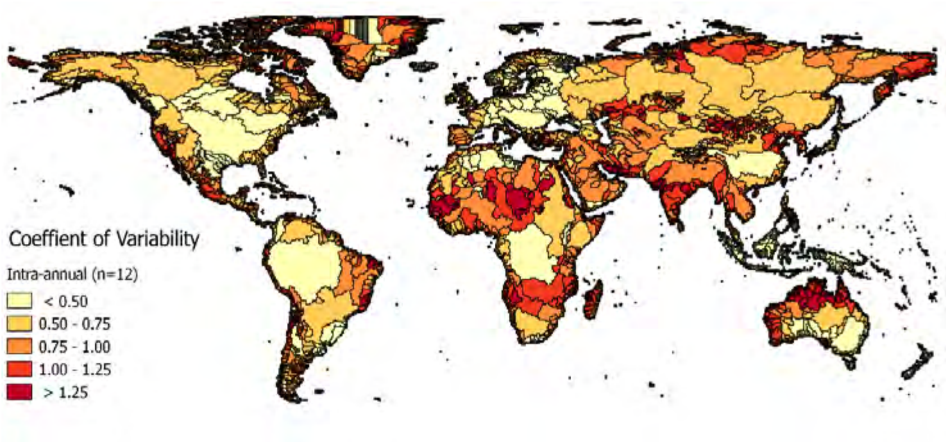
2.3. Results

2.3.1. Blue water availability varies in space and time

We estimated monthly blue water availability, BWA (m^3 s^{-1}) by subtracting environmental flow requirements, EFR (m^3 s^{-1}), from blue water runoff BWR (m^3 s^{-1}), for each month in the period 1970 – 2005, for all basins in the world. Figure 2-1 shows the coefficient of variation in BWA within and over the years. Clearly, BWA is unevenly distributed over time, with certain basins displaying a relatively constant level of availability while others exhibit more fickle behaviour. Figure S 1 shows the spatial variability of BWA, expressed in in mm per year and Figure S 2 the EFR that needs to be reserved from BWR, expressed as percentage of BWR.

Figure 2-2 zooms in from the global picture to three basins with varying hydrological regimes and characteristics – the rain and snowmelt-fed Rhine basin, the predominantly semi-arid Tigris/Euphrates basin with a pronounced wet and dry period, and the monsoonal Indus basin. Where the Rhine shows a relatively constant BWA both throughout the year and over the years, the Tigris/Euphrates and Indus basins display a much larger intra-annual and inter-annual variability. Particularly pronounced periods of extreme low flows, e.g. in the Tigris/Euphrates in the late 1980s, leave little room for human appropriation and either water shortages or overexploitation seem inevitable (Kavvas et al., 2011). The Rhine basin – and to a lesser extent even the Indus basin – experienced fewer extreme low flow months that can foil continuous allocation of water for human purposes under a capping policy arrangement. Already stressed basins will likely face bigger challenges in reducing WFs to cap values, especially in conjunction with high temporal variability in BWA.

a



b

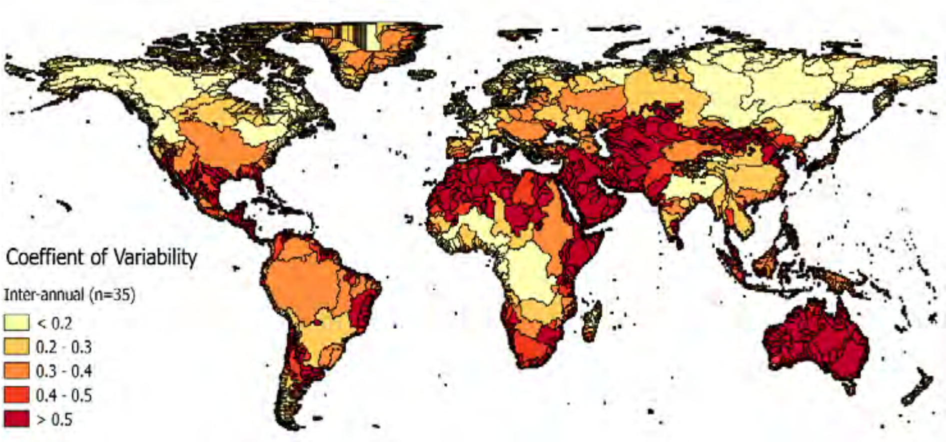


Figure 2-1. Coefficient of intra-annual variation in long-term average monthly blue water availability (BWA) over the period 1970-2005 (n=12) (a); inter-annual variation in annual BWA over the period 1970-2005 (n=35) (b). Darker red basins experience more pronounced variability in BWA within or over the years, respectively. The data refer to ensemble means of three Global Hydrology Models and three methods to estimate environmental flow requirements (see methods).

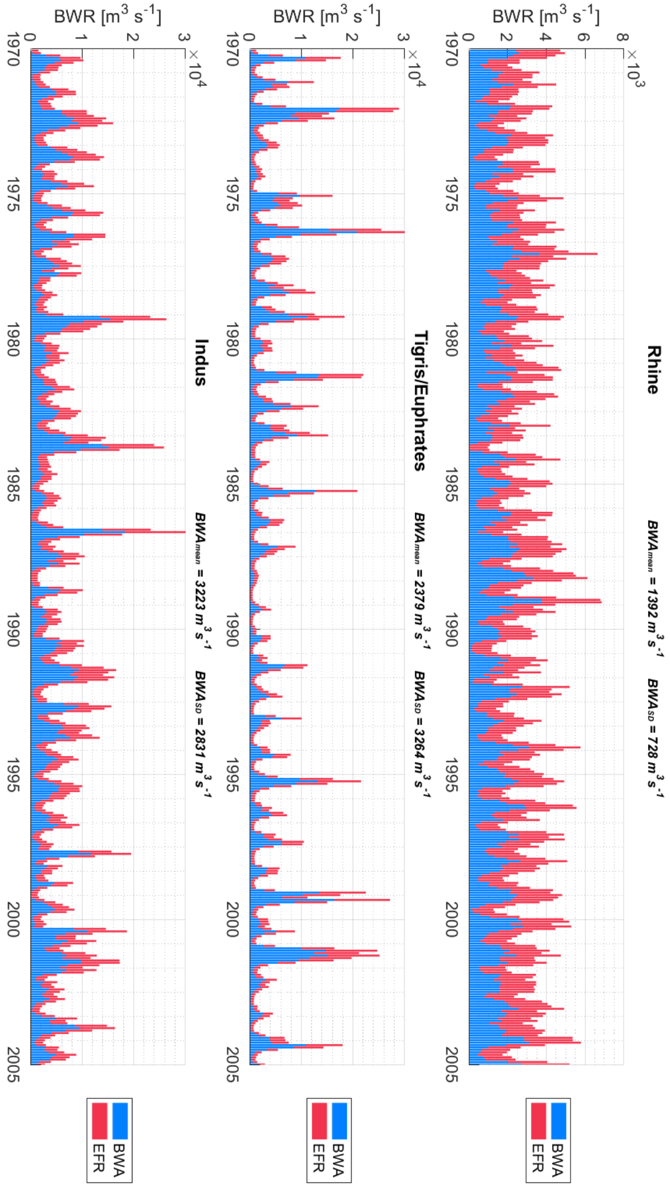


Figure 2-2. Monthly blue water runoff (BWR) partitioned into environmental flow requirements (EFR) and blue water availability (BWA), for three selected basins, with mean and standard deviation (SD) values of BWA. Monthly and annual averages for these basins are shown in Figure S 3. The data refer to ensemble means of three Global Hydrology Models and three EFR methods (see methods).

2.3.2. Variability has implications for setting WF caps

The premise of setting a monthly blue WF cap in a river basin is that the sum of consumption over all water using activities in that basin should not exceed the cap. Ideally, threshold values are to be formalized dynamically, so that they will be stricter in months that are relatively dry compared to the long-term average and less strict in relatively wet months, but herein lies the difficulty that long-term predictions of runoff (i.e. a seasonal lead at least) are difficult and surrounded by significant uncertainties. Therefore, a WF cap will have to be based on some historic or average measure of BWA in the basin. Here, we distinguish three WF cap options, each one following another procedure for formulating WF caps. In the three options, caps are set at: a) the long-term average of the monthly average BWA over the period 1970-2005, b) the 25th percentile of monthly average BWA, or c) the minimum monthly BWA that occurred in the period 1970-2005. For each of these cap settings and each river basin, we addressed – given the actual regime of fluctuating water availability – the implications of sticking to the caps on the violation of environmental flow requirements (in case of relatively dry periods) or on the underutilization of WF potential (in case of relatively wet periods).

Figure 2-3 illustrates the implications of setting a monthly WF cap according to the three procedures for the Rhine, Tigris/Euphrates and Indus basins. In order to test the consequence of using WF caps, it is assumed that in every river basin WFs are allocated and realised up to the governing cap. For all three WF cap options, it shows that if in a given month BWA is higher than the WF cap set for that month, environmental flows will not be violated (WFs in dark blue are fully within BWA). At the same time, these months leave an unutilized WF potential, since any BWA beyond the cap is not allocated to human use (light blue). In wet months in which BWA exceeds the cap, an unutilized WF potential is inevitable, but underutilizing available flow in dry months in which BWA still exceeds the cap can be undesirable. If BWA is lower than the WF cap set for a particular month, utilizing BWA to its full extent means that environmental flow requirements will be partly compromised (red stacks).

The balance between potentially violating environmental flow requirements and allowing an unutilized flow in the system tips towards the latter when shifting from a WF cap set at average BWA to a WF cap set more precautionary at the 25th percentile of BWA. The strictest cap, set at minimum BWA, prevents EFR from being compromised at all times. While included to illustrate potential dynamics of various cap regimes, setting the WF cap so low is arguably unrealistic, especially in basins that already face severe water scarcity.

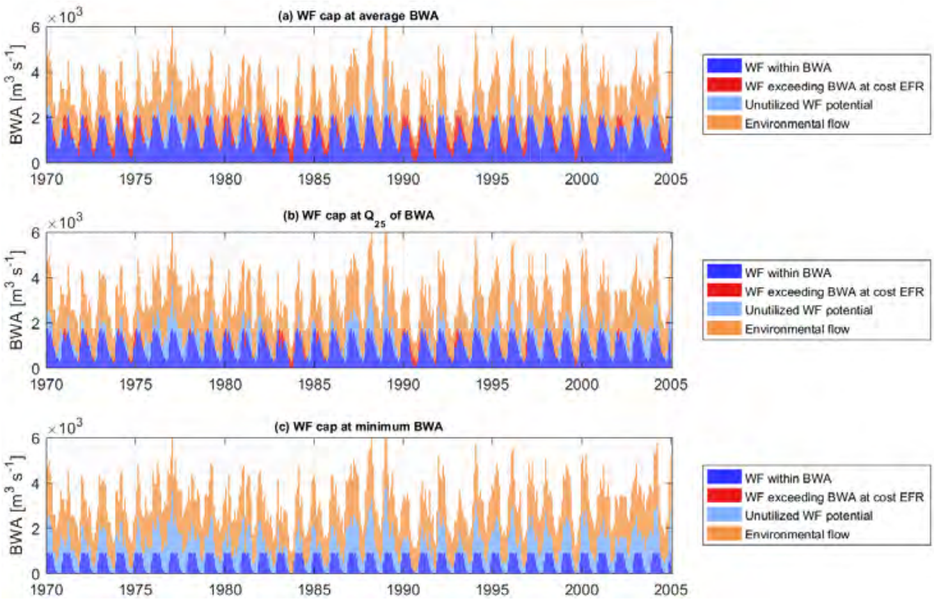
Table 2-1 quantifies the implications of each of the three procedures to set monthly WF caps for the three river basins discussed earlier. Particularly the Tigris/Euphrates and Indus basins would experience prolonged periods of consecutive EFR violations even in the middle case in which WF caps are set at the 25th percentile of BWA (4.4 and 3.3 consecutive months on average, respectively). In case of an average BWA-based cap, in the water-rich Rhine basin EFR is potentially violated regularly (6.4 months ever year), but there is always at least a part of EFR remaining (0 months of exceeding 90% of EFR). This is not the case for the Tigris/Euphrates and Indus basins, where, with an average BWA-based cap, over 90% of EFR is consumed for months at a time (3.8 and 2.7 consecutive months on average, respectively).

Table 2-1. Implications of three different procedures to setting a monthly WF caps, for three river basins with distinctive hydrological regimes. Long-term average annual EFR as % of BWR for the Rhine, Tigris/Euphrates and Indus are 56%, 46% and 50%, respectively.

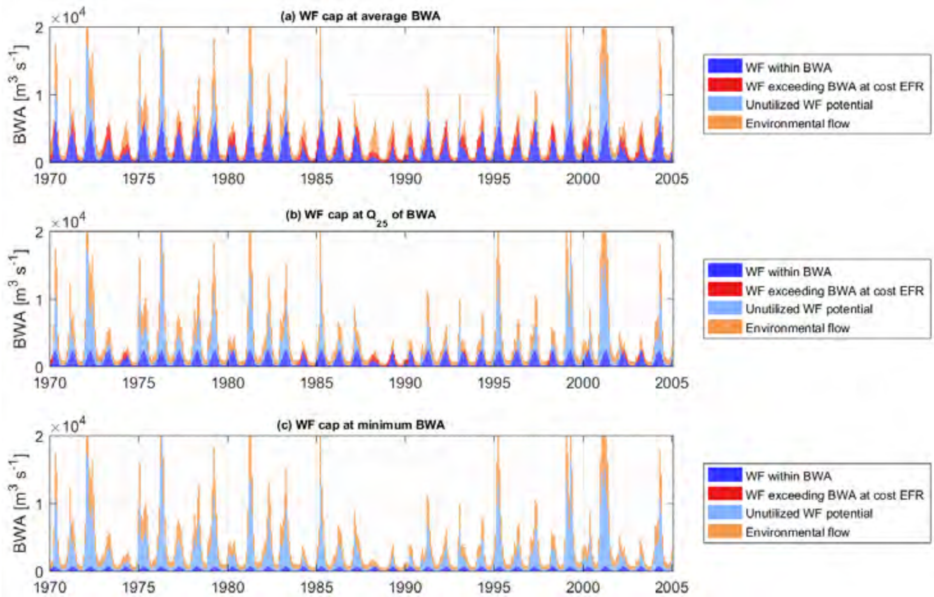
River basin	WF cap option	Unutilized WF potential		EFR violation[1]				
		10^9 m^3 y^{-1}	% BWA	10^9 m^3 y^{-1}	% BWA	#mo y^{-1}	#c mo	#90 mo
Rhine	avg BWA	72.5	14	72.5	11	6.4	3.6	0
	Q25 BWA	159	30	18.2	2.8	2.7	2.1	0
	min BWA	337	64	0	0	0	0	0
Tigris / Euphrates	avg BWA	283	31	283	37	7.6	7.6	3.8
	Q25 BWA	562	62	32.0	4.2	2.7	4.4	2.0
	min BWA	809	90	0	0	0	0	0
Indus	avg BWA	302	25	302	25	7.1	6.9	2.7
	Q25 BWA	618	51	53.0	4.4	2.7	3.3	2.3
	min BWA	1070	87	0	0	0	0	0

[1]EFR violations are expressed in cubic meter per year, as percentage of BWA, the average number (#) of months per year EFR is violated, the average number of consecutive (#c) months EFR is violated if a violation occurs and the average number of months 90% or more of EFR is violated (#90).

a


(a) WF cap at average BWA
(b) WF cap at Q_{25} of BWA
(c) WF cap at minimum BWA
BWA [m^3 s^{-1}]
$\times 10^3$
WF within BWA
WF exceeding BWA at cost EFR
Unutilized WF potential
Environmental flow

b


(a) WF cap at average BWA
(b) WF cap at Q_{25} of BWA
(c) WF cap at minimum BWA
BWA [m^3 s^{-1}]
$\times 10^4$
WF within BWA
WF exceeding BWA at cost EFR
Unutilized WF potential
Environmental flow

c

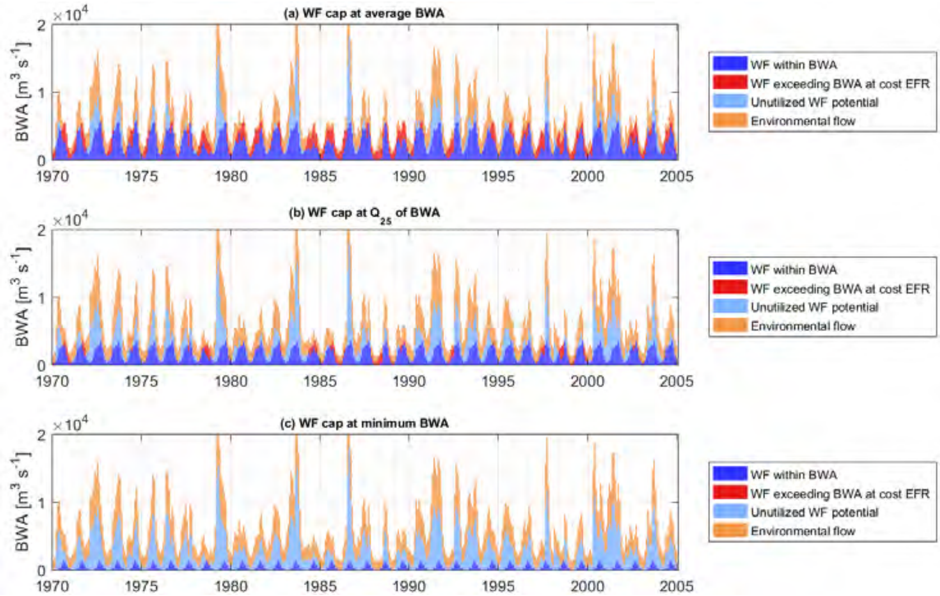


Figure 2-3. The implications of three WF cap options for the Rhine (upper), Indus (middle) and Tigris/Euphrates (lower) basin, where WF caps are set at average BWA (a); Q25 of BWA (b); and at minimum BWA (c).

2.4. Discussion

2.4.1. Model variability

We set out to quantify monthly blue WF caps for all basins in the world and address the implications of three different procedures to formulate WF caps. First, we estimated monthly blue water availability per basin by subtracting environmental flow requirements from blue water runoff for each month in the period 1970-2005. Ideally, monthly BWA had been based on in situ runoff observations and environmental needs tailored to local basin circumstances rather than modeling. However, until such observations become available with global coverage, modeling can provide a first indication of basin-level BWA. From the wide spectrum of Global Hydrology Models and EFR models available, three of each were selected in this study to provide estimates for monthly BWR and EFR. While taking ensemble means averages out model anomalies or divergencies, Figure S 4 shows a considerable spread in model outputs, indicating that both BWR and EFR estimates are prone to substantial uncertainties – in addition to inherent natural variability.

On the global level, long-term average BWR is found to vary between 41,300 - 67,200 $km^3\ y^{-1}$, with an average of 54,100 $km^3\ y^{-1}$. These estimates are comparable to the range of 42,000 - 66,000 $km^3\ y^{-1}$ found by a previous multi-model runoff assessment by Haddeland et al. (2011) – although they included anthropogenic water use. Our average is a bit higher than the pristine runoff of 49,300 $km^3\ y^{-1}$ reported by Oki & Kanae (2006) and substantially higher than the 41,700 $km^3\ y^{-1}$ given by Gerten et al. (2013). Globally aggregated EFR in this study is 26,700 $km^3\ y^{-1}$ on average (with a range of 8,800 - 53,900 $km^3\ y^{-1}$), or 49% (21 - 80%) of BWR. Gerten et al. (2013), who applied five different EFR-methods, estimated that globally on average 36% of runoff should be allocated to EFR, with the high end of the range at 57%. The inclusion of the presumptive standard of 80% by Richter et al. (2012) in our selection drives this study's EFR estimate up considerably. Aggregating BWA across basins, we obtain a global total blue water availability of 27,400 (8,300 - 53,700) $km^3\ y^{-1}$ or 187 (55 - 366) $mm\ y^{-1}$.

2.4.2. A basis for a planetary boundary on water

While our primary intention was to quantify WF caps at the basin level, the estimated monthly BWA figures for the world's river basins – when summed up over the year and all basins – can feed the discourse on planetary boundaries. Steffen et al. (2015) have proposed 4,000 $km^3\ y^{-1}$ (with an uncertainty range of 4,000 - 6,000 $km^3\ y^{-1}$) as planetary boundary (PB) for global blue freshwater consumption, while acknowledging the need for basin boundaries as well. Gerten et al. (2013) propose a lower PB, at 2,800 $km^3\ y^{-1}$, with a range of 1,100 - 4,500 $km^3\ y^{-1}$. Our globally aggregated BWA estimate is much higher. However, since much of BWA runs off during flood periods and/or in areas where not enough people live to use the water, appropriate reductions need to be applied to translate *total* BWA into a more comparable *exploitable* BWA (Postel et al., 1996). Here, we do not intend to propose a new PB for freshwater consumption, but rather we illustrate how different rationales will result in a different PB.

If we take, for each basin, the lower end of the uncertainty range of BWR and the higher end of the uncertainty range of EFR (thus yielding the lowest estimate for BWA), following the precautionary principle as proposed by Steffen et al. (2015), we have a global BWR of 41,300 $km^3\ y^{-1}$ and BWA of 8,300 $km^3\ y^{-1}$. One rationale to assess annual exploitable BWA in every river basin is to equate exploitable BWA in each month to BWA in the most critical month (in which long-term average BWA is lowest). Aggregating across basins, this yields a global exploitable BWA of 2,400 $km^3\ y^{-1}$. This can be interpreted as an estimate for the PB for blue water consumption. Alternatively and less strict, we can define exploitable BWA in each month as BWA in that month, but with the baseflow in a river basin – here taken as the 5th percentile of monthly BWR over the study period of 1970-2005 – as a maximum. Aggregating to the global level, this rationale results in a PB of 3,200 km^3 y^{-1}. As a third rationale, we take the previous rationale but apply an additional remote flow criterion following Steffen et al. (2015) and Postel et al. (1996). Subtracting all flows in a basin that exceed 1,000 m^3 cap^{-1} y^{-1} results in a PB of 1,200 km^3 y^{-1}, viz. the most precautionary value.

Further academic debate is necessary on the precise procedure and rationales to establish a PB from bottom-up estimates of BWA and assumptions regarding unexploitable flows. In the meantime, the wide spread in estimates underscores that caution is warranted in interpreting such aggregated global values. It is clear that regionalized boundaries are more meaningful in terms of assessing appropriable volumes and more actionable with regards to drafting effective water policies (Steffen et al., 2015; Heck et al., 2018; Heistermann, 2017).

2.4.3. Towards policy uptake

If WF caps are to become effective and practical concepts for policy arrangements, a number of limitations have still to be overcome. Explicitly splitting WF caps into a surface water and renewable groundwater component would be a valuable yet complicating refinement over this study's lumping of the two; where the current study represents groundwater inputs to runoff, separating WF caps in both components would call for an explicit consideration of groundwater extraction potential. Related and in addition, we suggest to incorporate the effect of reservoirs on WF caps, since reservoir storage and operations typically attenuate basin hydrographs and thereby redistribute BWA over time. A recent study by Zhuo et al. (2019) shows that reservoirs can increase monthly BWA in dry months at the cost of lowering BWA in wet months, occasionally even adding to 'scarcity in wet months' indicating environmental peak flow requirements were no longer met. Lastly, and particularly pertaining to larger catchments, the spatial distribution of BWA over the basin complicates arrangements in which a single cap is set for the entire basin, suggesting multiple caps may be needed at sub-basin scales.

The implications of formalizing a WF cap according to different alternative procedures have been expressed here in terms of balancing EFR violations (biodiversity interests) against unutilized WF potential (economic interests). Which implications and trade-offs are acceptable – and thereby which WF cap setting procedure – varies from one basin to another and emerge from policy choices. There may be additional implications or impacts deemed worthy to consider, such as e.g. equitable distribution of water over users or communities – especially when it concerns transboundary basins – or more specific ecological or economic indicators. Such elaborate assessments, however, require additional data and are therefore more readily done in a local study setting.

Our work clearly demonstrates the intrinsic difficulty of dealing with variability in any practical policy arrangement and successful showcases have yet to be developed (Grafton et al., 2014). One avenue for further research is to explore whether a dynamic WF cap regime can be developed whereby monthly caps partially depend on runoff forecasts. Basins with highly variable water availability that already experience severe water stress by overconsumption may have greater challenges in adopting a WF cap regime than moderately stressed basins with less variability. However, particularly variable and scarce basins should feel a strong imperative to keep a close eye on their water allocation policies. Still, while it is indispensable to curb WFs to cap levels in scarce regions, reduction efforts should not be constrained to severely stressed basins only. A strategic pathway to conserve limited blue water resources in scarce basins is to increase water productivity in basins that still have the potential for it. Setting WF benchmarks for water-using activities and using green water resources more productively, concurrently with setting WF caps at the river basin level, can be instrumental in achieving truly sustainable and efficient use of freshwater worldwide (Hoekstra, 2014a).

2.5. Conclusion

The world's limited blue water resources are shared by humans and nature. The continued growth in human water consumption has tremendous impacts on global biodiversity (Vörösmarty et al., 2010). Clearly, bridling humanity's unsustainable water footprint is one of the key environmental challenges of the 21st century. Capping water consumption, to support the transition towards sustainable freshwater use, is urgent in all river basins where water resources are already overexploited, which concerns about half of the world's basins (Hoekstra et al., 2012). Setting WF caps calls for seeking a compromise between underutilizing the potential of sustainable water use and implicitly accepting violations of environmental flow requirements – a trade-off that is particularly pronounced in basins with a high seasonal and inter-annual variability.

Despite identified uncertainties that have to be overcome, necessary refinements that have to be made and remaining knowledge gaps that have to be filled, e.g. regarding implementation pathways, we underscore the evident merit of the concept of capping water consumption in the first place and its invaluable contribution to the ongoing quest for sustainable freshwater use worldwide.

3

GLOBAL WATER SAVING POTENTIAL AND WATER SCARCITY ALLEVIATION BY REDUCING WATER FOOTPRINTS OF CROPS TO BENCHMARK LEVELS

3. Global Water Saving Potential and Water Scarcity Alleviation by Reducing Water Footprints of Crops to Benchmark Levels

Abstract

Widespread water scarcity indicates that too much of Earth's limitedly available freshwater is being appropriated for human use. Since agriculture assumes the largest share in global freshwater consumption, increasing water use efficiency in crop production may effectively reduce humanity's water footprint (WF). This study explores the potential merits of formulating WF benchmarks for the world's major crops – i.e. reference targets indicating reasonable amounts of water consumption per unit of output – towards saving valuable water resources and alleviating water scarcity.

Using an newly built model framework, *Aqua21*, we were able to identify both consumptive and specific blue WFs benchmarks, differentiated by climate zone. If current consumptive and blue WFs worldwide are reduced to benchmark levels associated with the best-25th percentile of production, global annual average total and blue water savings are 44% and 31%, respectively, compared to the reference consumption. The largest savings are possible in India, Brazil, China and USA, and in wheat, rice, maize and soybean production. Of the blue water saving potential, 89% can be achieved in water-scarce areas, of which 83% in regions classified as severely scarce. Policy measures striving to have producers meet WF benchmarks would thus boost the transition towards sustainable use of freshwater globally.

3.1. Introduction

The global food system is a major driver of depletion of freshwater resources (Springmann et al., 2018). Agriculture accounts for 92% of freshwater fluxes currently appropriated for human use (Hoekstra & Mekonnen, 2012). This water footprint is projected to grow due to population growth, shifts towards more water-intensive diets and climate change (Hejazi et al., 2014; Vanham et al., 2018; Hoekstra & Wiedmann, 2014). Overconsumption of limitedly available water resources profoundly impact ecosystems, societies and economies, with already today half a billion people living in regions that face sever water scarcity year-round (Mekonnen & Hoekstra, 2016); the majority of water-dependent habitats and ecosystems under threat (Vörösmarty et al., 2010); economic momentum stalling (World Bank, 2016); and social and political conflict over water on the rise (Kundzewicz & Kowalczak, 2009; Munia et al., 2016).

The large share of agriculture in humanity's water footprint allows large water savings to be made if water would be used more efficiently. Studies have shown that improved farm water management can both lower water consumption per unit of output (more crop per drop) and diminish the yield gap, both of which are indispensable to continue feeding growing populations within ecological boundaries of the world's water systems (Jägermeyr et al., 2016; Brauman et al., 2013; Giordano et al., 2017). Moreover, several global assessments revealed considerable spatial variability in water productivity in crop production, allowing identification of inefficient production locations at high spatial resolution (Mekonnen & Hoekstra, 2011; Zwart et al., 2010; Wada et al., 2014).

In search of what comprises efficient use of freshwater, Hoekstra (2013) proposed to formulate water footprint (WF) benchmarks for water-using activities or products. A benchmark identifies a 'reasonable' amount of water that can be consumed to produce a unit of output – in case of crop production in $m^3\ t^{-1}$. A WF benchmark, therefore, constitutes a quantified reference point for producers that they can try to meet, e.g. by resorting to better farming practices and more water-efficient irrigation technologies and strategies, and for governments to use as basis for issuing water consumption permits (Hoekstra, 2014a).

A new concept in the discourse on efficient water use, few studies to date probed the merits of benchmarking. In a pioneering global assessment of crop production, Mekonnen & Hoekstra (2014) presented WF benchmarks for 124 crops and found that if producers everywhere in the world would reduce their WF to the level of the best-25th percentile of current production, global water saving in crop production would be 39%

compared to the reference water consumption. However, they used WF estimates that were based on long-term-averaged climatic data and relatively course water-balance modelling. The effect of inter-annual variability on WF benchmark levels could thus not be assessed, despite the fact that crop water productivity will be higher or lower from one year to another. Moreover, Zhuo et al. (2016) showed for winter wheat in China that – all else being equal – WF benchmarks vary with environmental factors, particularly climate. Therefore, if WF benchmarks are to be achieved by producers across locations, they argue it is sensible to differentiate WF benchmarks per climate zone. Karandish et al. (2018) recently calculated climate-specific benchmarks for 26 crops in Iran to see how the incurred water savings would alleviate groundwater scarcity and pollution. The above studies exclusively estimated benchmarks for the total consumptive WF of crops, meaning that the benchmarks lumped consumption of green water (i.e. rainwater insofar it does not become runoff) and blue water (surface water and groundwater), and did not distinguish between them. Developing specific blue WF benchmarks, that complement the benchmark for the overall consumptive WF, would allow estimating potential blue water savings to alleviate blue water scarcity specifically.

This research advances the study of potential water saving in crop production by: i) developing WF benchmarks for the world's major crops, differentiated per climate zone, ii) identifying specific blue WFs in irrigated agriculture associated with these benchmarks, and iii) assessing total and blue water saving and scarcity reduction potential if producers were able to reduce WFs in crop production to benchmark levels set by the best-25th percentile of production per climate zone. We developed a new model framework, *Aqua21*, to estimate WFs of 57 crops with FAO's flagship crop growth engine AquaCrop (Steduto et al., 2009) at high spatiotemporal resolution on a global scale, for the period 1961-2015. These crops collectively constitute 91% of global food production (FAO, 2017) and 68% of global freshwater consumption (Hoekstra & Mekonnen, 2012).

3.2. Method and Data

We developed a model framework, *Aqua21*, to estimate WFs of crop production (m^3 t^{-1}) for 57 crops globally. *Aqua21* embeds FAO's crop growth model AquaCrop (Steduto et al., 2009; Raes et al., 2009) to calculate crop water use and yields over the growing season, on a 5 × 5 arc minute grid with a global coverage, for the period 1961-2015. For each year, the spatial pattern of simulated yields within a country is scaled to match reported national yield statistics from FAOSTAT (FAO, 2017). Because reported yields show a trend for many crops, we confined our WF benchmark analyses to the near-steady period 1996-2015. Keeping a shadow water balance as described by Chukalla et al. (2015), we distinguished between the green and blue component of resulting WFs, where the blue component consists of a blue irrigation water fraction and a capillary rise fraction.

Production-weighted WFs simulated over each growing season in the period 1996-2015 were sorted from small to large – per crop, climate zone and farming system (rainfed or irrigated) – after which we selected several production-weighted percentiles of this WF distribution as benchmark level (e.g. 10^{th}, 25^{th}). The 25^{th} production-weighted WF percentile value thus corresponds to the WF of the best-25^{th} percentile of production with smallest WFs.

Per crop and climate zone, we formulated specific blue WF benchmarks as a function of the crop's overall consumptive WF benchmark. For each crop, efficient producers were identified by selecting all growing seasons across production locations (i.e. grid cells) during the period 1996-2015 that had WFs below or at the benchmark level associated with the best-25^{th} percentile of production over the period 1996-2015 (WF_{p25}). Of this selection, we took the median irrigation water fraction in the consumptive WF (i.e. the share (blue) irrigation water takes in the total green-blue WF) as specific blue WF benchmark.

Total (green plus blue) water saving potential was estimated by calculating the difference between the total water consumption in the reference situation and total consumption if all producers reduced their WFs to WF_{p25}. Specific blue water savings were estimated in a similar way, as the difference between reference and reduced blue water consumption. Resulting blue savings were mapped against average annual blue water scarcity levels as estimated by Mekonnen & Hoekstra (2016), to discover how much of these blue water savings could be made in places with severe water scarcity.

The model was forced with monthly fields of CRU climate data (CRU, 2013) that were downscaled to daily fields using ERA reanalysis products (Dee et al., 2011) through the procedure by Van Beek et al. (2011). Climate zoning is based on UNEP's aridity index (Barrow, 1992) using average precipitation and potential evapotranspiration fields over the period 1961-2015. Soil hydraulic parameters for 253 unique soil classes for two soil layers were obtained from De Lannoy et al. (2014). We took steady shallow groundwater tables below 200 cm from Fan et al. (2013). Crop-specific harvested areas around the year 2000 were obtained from MIRCA2000 for 21 major crops (Portmann et al., 2008) and from Monfreda et al. (2008) for the remaining crops. For the latter, the percentage of the harvested area under irrigation is estimated based on the percentage under irrigation for the corresponding crop grouping in MIRCA2000. To account for large-scale temporal developments in crop harvested areas, we masked crop-specific harvested areas with historical cropland extent maps from HYDE 3.1 (Klein Goldewijk et al., 2011). A similar procedure was applied to capture large-scale developments in irrigated areas, using historical areas equipped for irrigation from HID (Siebert et al., 2015). If a crop is irrigated, we assumed it received full irrigation applied by a system that wets the full soil surface (comparable to sprinkler or furrow irrigation). Crop parameters not in AquaCrop's default library were derived from Chapagain & Hoekstra (2004) and Portmann et al. (2008). The resulting climate zones, production locations with access to a shallow groundwater table and the average percentage of harvested area under irrigation thus obtained are shown in Figure 3-1.

3.3. Results

3.3.1. Benchmarks for the overall consumptive WF of crops

We used the Aqua21 model framework to estimate WFs of 57 crops globally. Figure 3-2 shows the spatial variability of resulting consumptive WFs for wheat, rice, maize and sugarcane – the four largest crops in terms of global annual production (FAO, 2017). WFs are categorized in climate-specific WF production percentiles to distinguish water-efficient production locations from inefficient ones. WF levels associated with the best-25th percentile of production over the period 1996-2015 (WF_{p25}) are achieved on all continents, with water-efficient production (i.e. small WFs) predominantly found in Western Europe (wheat and maize) and China (wheat, maize, rice). Table 3-1 provides WF benchmarks and yields of these four selected crops, at various percentiles of global production, differentiated per climate zone and farming system. Increasing WFs and reducing yields from low to high production percentiles reflects the variability across production locations. In most cases, the 10th and 25th production percentile, i.e. the 10% or 25% of production with smallest WFs, show comparable WFs for rainfed and irrigated fields, but at the 90th percentile the WFs for rainfed fields substantially exceed the WFs for irrigated fields. Rice, maize and sugarcane have large WFs in hyper-arid climates that reduce in the direction of wetter climate zones, while wheat WFs peak in semi-arid zones. Differences across climate zones confirm the finding by Zhuo et al. (2016) that – if WF benchmarks become a reference target for producers across differing production locations and accepting that these crops should be grown in their current climate zones in the first place – it is sensible to formulate WF benchmarks as a function of climate. Corresponding values of the remaining crops are provided in Table S 1.

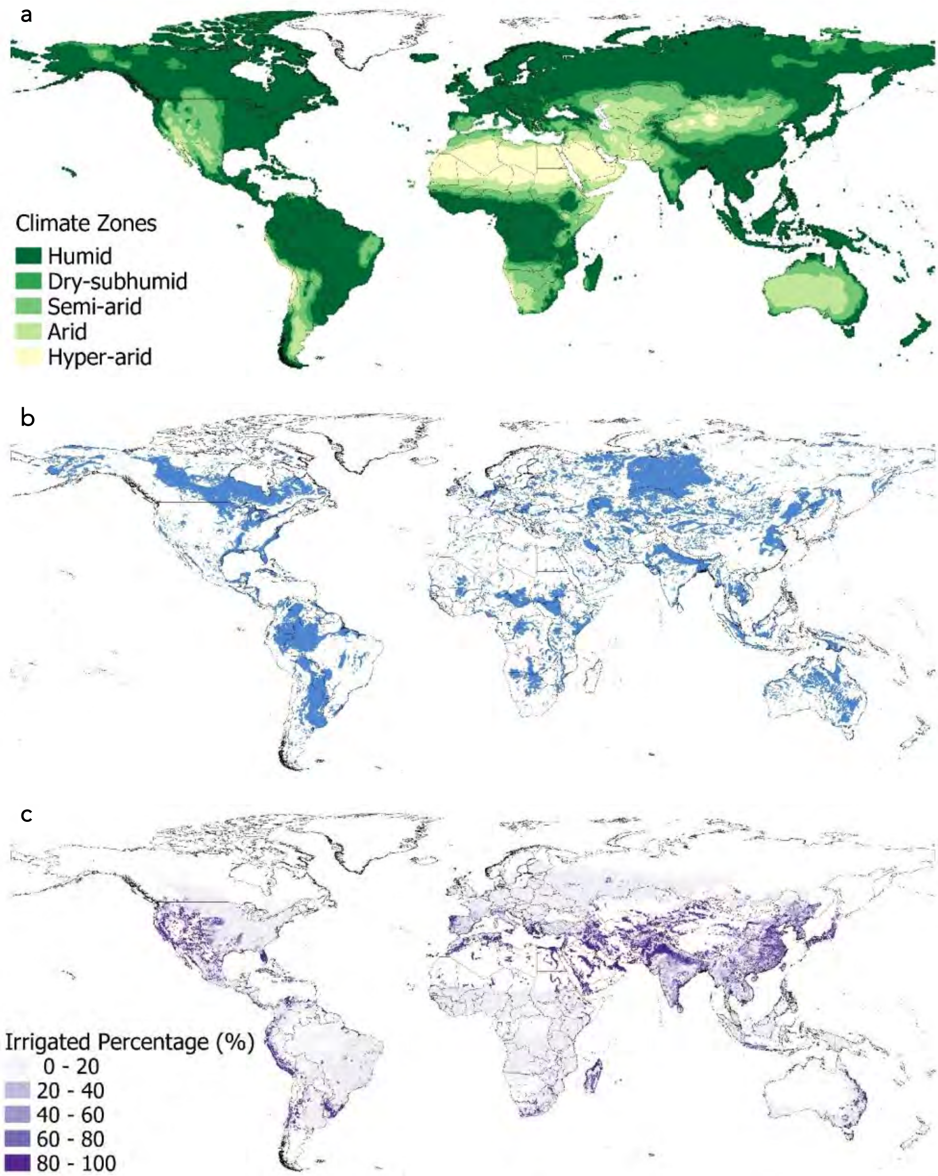
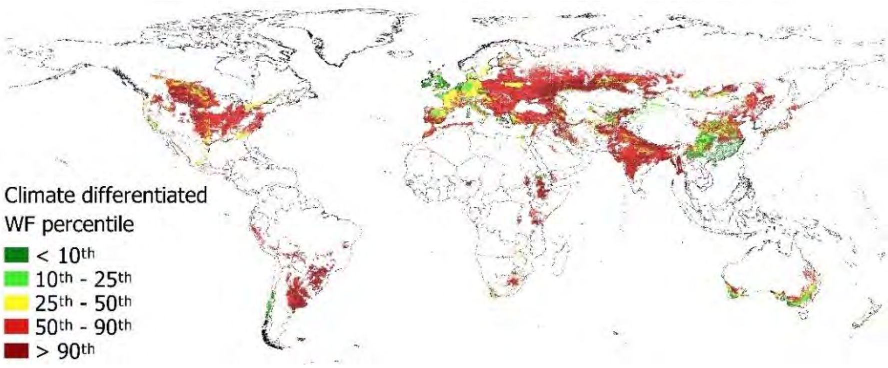
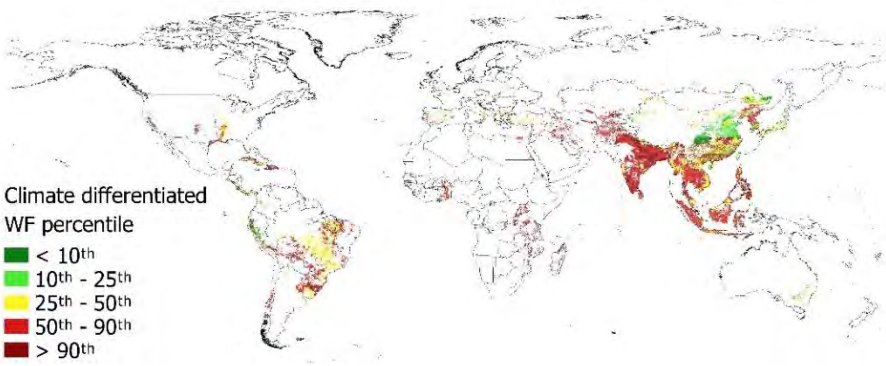


Figure 3-1. Climate zones used to differentiate WF benchmarks, based on Barrow (1992) (a); cells with access to a shallow groundwater table (<200 cm), based on Fan et al. (2013) (b); percentage of harvested area in grid cell that is being irrigated, on average, over the period 1961-2015.

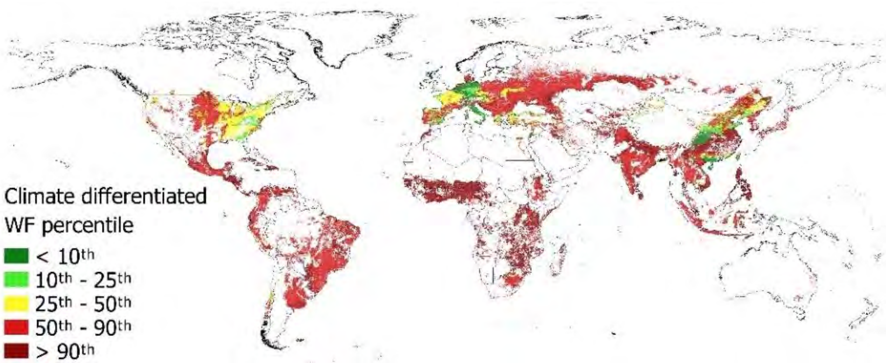
a) Wheat



b) Rice



c) Maize



d) Sugarcane

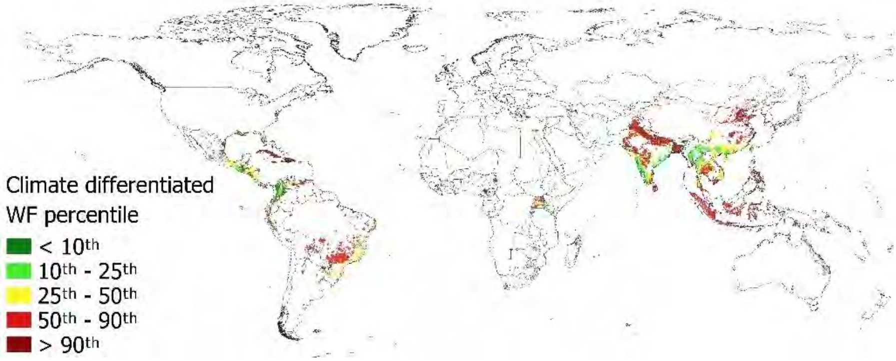


Figure 3-2. Production percentiles of wheat (a), rice (b), maize (c) and sugarcane (d) at spatial resolution of 5 × 5 arcmin. Per climate zone, production volumes over all grid cells in that zone a ranked based on the WF of the crop in each grid cell (from low to high), whereby the lowest production percentile thus represents the production with the smallest WFs (average WFs over the period 1996-2015).

Table 3-1. WF benchmarks (m^3 t^{-1}) and yield (Y, t ha^{-1}) for the four largest crops at various percentiles of global production, differentiated per climate zone (HA: Hyper-arid; A: Arid; SA: Semi-arid; DH: Dry-subhumid; H: Humid) and farming system (I: Irrigated; R: Rainfed), over the period 1996-2015. Corresponding values for remaining crops are provided in Table S 1.

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
Wheat	HA	I	882	6.8	976	6.9	1267	6.7	2757	5.7	*1703*	5.2
	A	I	646	8.4	880	7.4	1622	5.0	2663	3.2	*1723*	3.6
	SA	R	775	5.3	1090	3.6	1596	2.7	4460	1.4	*2604*	1.2
	SA	I	844	7.7	1253	5.5	1694	4.9	2227	4.8	*1779*	3.9
	DH	R	644	7.4	977	4.1	1372	3.7	3089	2.1	*1869*	2.2
	DH	I	767	9.2	1020	5.1	1582	4.9	2106	5.2	*1700*	3.9
	H	R	427	10.2	556	8.7	776	6.9	2020	3.9	*1146*	4.2
	H	I	290	14.0	387	10.9	569	10.3	1851	5.5	*950*	5.9
Rice	HA	I	716	13.1	781	10.3	911	9.5	1476	9.3	*1174*	8.6
	A	I	599	12.1	658	11.7	753	11.3	3253	4.6	*1540*	5.8
	SA	R	538	11.6	584	12.1	905	9.5	3168	3.0	*1588*	3.3
	SA	I	560	11.5	625	10.8	1245	5.7	2386	4.0	*1367*	4.5
	DH	R	514	11.6	554	12.3	872	11.2	2065	3.8	*1288*	4.2

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
	DH	I	509	12.1	570	12.0	1367	5.6	2123	4.3	*1425*	4.5
	H	R	497	11.4	909	6.2	1217	5.2	2535	3.5	*1614*	3.6
	H	I	432	12.2	512	11.4	1019	6.7	1955	4.7	*1142*	5.1
Maize	HA	I	801	7.9	900	8.1	1044	8.0	1443	8.1	*1121*	7.7
	A	I	562	11.5	611	10.7	688	9.5	1809	7.4	*973*	6.7
	SA	R	533	8.8	712	6.8	1513	3.6	3515	2.0	*1940*	2.0
	SA	I	500	12.9	569	11.2	691	10.9	1229	10.5	*821*	8.5
	DH	R	479	10.3	588	9.1	808	6.8	2374	2.9	*1242*	3.2
	DH	I	461	12.9	494	11.9	570	11.6	893	10.5	*670*	9.0
	H	R	426	11.8	495	11.6	587	10.7	1843	5.0	*962*	5.1
	H	I	379	14.4	438	11.9	502	11.7	1160	9.4	*651*	8.0
Sugar cane	HA	I	78	210.2	88	138.2	115	136.5	206	123.6	*125*	128.6
	A	I	87	181.6	146	104.9	164	84.4	189	79.1	*157*	85.9
	SA	R	69	168.1	108	80.5	151	59.7	329	21.5	*208*	25.4
	SA	I	99	134.4	113	117.0	126	112.1	154	101.7	*130*	100.9
	DH	R	89	98.9	116	75.7	150	63.1	292	31.0	*182*	32.6
	DH	I	89	152.7	105	118.3	118	110.9	142	104.0	*118*	106.5
	H	R	96	114.9	115	84.9	135	80.4	179	68.7	*146*	63.8
	H	I	85	150.6	102	112.6	121	99.7	172	86.7	*129*	90.6

3.3.2. Specific blue WF benchmarks

For each crop and climate in irrigated farming systems, we calculated the irrigation water fraction in the total consumptive WF for the subset of grid cells that meet WF_{p25} (i.e. for the 25% of most water-efficient production). For the four biggest crops, Figure 3-3 shows the distribution of this irrigation water fraction within each climate zone. A clear trend appears going from dryer to wetter climates, with production in hyper-arid zones relying almost exclusively on blue water, while humid production locations hardly require adding irrigation water at all. Irrigation water fractions for the remaining crops are provided in Table S 2.

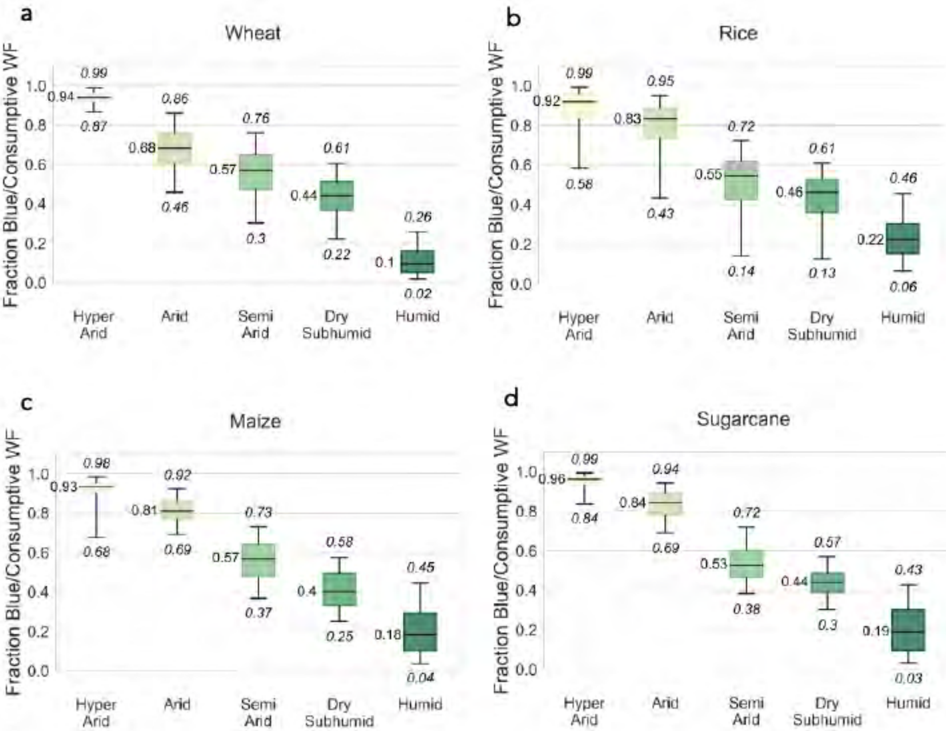


Figure 3-3. Irrigation water fractions in overall consumptive WFs of irrigated production locations where WF ≤ WF_{p25}, per climate zone, for wheat (a), rice (b), maize (c) and sugarcane (d). The boxplots indicated the range between the 25th and 75th percentile of the distributions; the whiskers the range between the 5th and 95th percentile of the distribution.

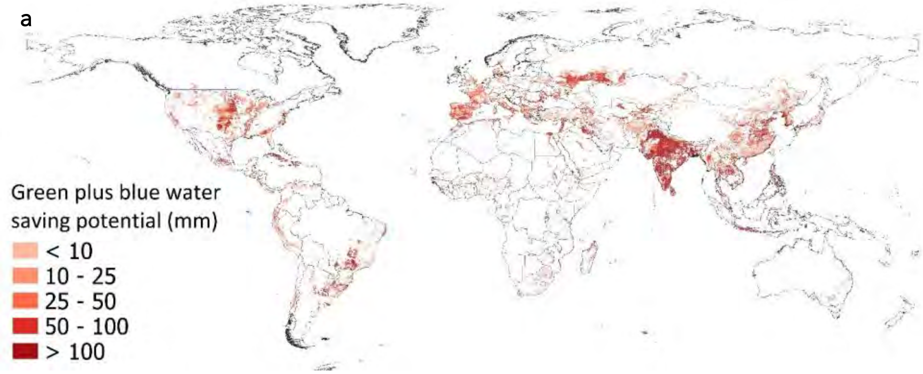
3.3.3. Potential water saving

Figure 3-4 shows the spatial distribution of the total and specific blue water saving if the WFs of all 57 crops investigated are reduced to the climate-specific benchmark level set by the best-25th percentile of production. Table 3-2 and Table 3-3 provide aggregations of potential water savings per country and crop, respectively. The countries with the largest WFs are: India (802 × 10^9 $m^3 y^{-1}$), Brazil (387 × 10^9 $m^3 y^{-1}$), China (545 × 10^9 $m^3 y^{-1}$) and the USA (551 × 10^9 $m^3 y^{-1}$). Brazil's WF varies most over time (standard deviation of 96 × 10^9 $m^3 y^{-1}$), which is probably due to their relatively large dependence on erratic rainfall. Typically, countries with large WFs also have the largest potential to save water. Where China can save 29% of its total consumptive WF, Bangladesh can save 60%.

Focusing on blue water specifically, Argentina has a blue water saving potential of 3% (of their blue WF of 34 × 10^9 m^3 y^{-1}), while Pakistan can save 45% (of their blue WF of 66 × 10^9 m^3 y^{-1}). In terms of crops, the highest total water savings can be achieved in the four big crops (wheat: 49% of 923 × 10^9 m^3 y^{-1}, rice: 51% of 824 × 10^9 m^3 y^{-1}, maize: 48% of 727 × 10^9 m^3 y^{-1} and sugarcane: 28% of 512 × 10^9 m^3 y^{-1}). The highest relative blue water savings can be achieved for barley (47%) and almonds (46%). Across countries and crops, lowering WFs to WF_{p25} can reduce the world's annual average consumptive WF in crop production by 44 (39-47)% and the blue WF by 31 (29-33) % compared to the reference WF.

3.3.4. Water scarcity reduction

Water savings incurred by meeting WF benchmarks can alleviate water scarcity, particularly if savings are achieved in already stressed areas. We aggregated blue saving potential of all production locations globally per blue water scarcity level as estimated by Mekonnen & Hoekstra (2016) in Table 3-4. We find that 89% of blue water savings, i.e. 228 × 10^9 m^3, can be achieved in areas facing moderate to severe water scarcity, of which most (204 × 10^9 m^3) in regions classified as severely scarce. We also discovered that rice contributes 37% to the total blue water saving potential in severely scarce areas, with wheat, cotton, maize and sugarcane completing the top-five. Boosting water use efficiency in the production of these crops thus seems a particularly effective pathway toward reducing blue water scarcity.



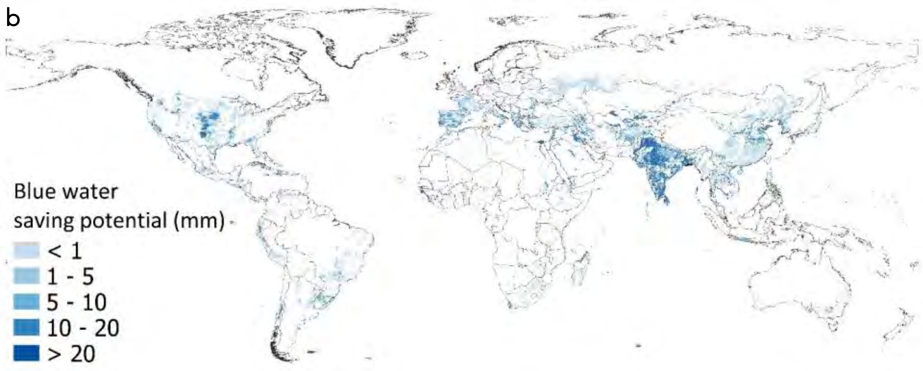


Figure 3-4. Annual average total (green plus blue) water saving (a) and blue water saving (b) (mm y^{-1}) if water footprints are reduced to climate-specific benchmarks levels set by the best-25th percentile of global production (WF_{p25}).

Table 3-2. Annual total (green plus blue) and blue water saving per country, over the period 1996-2015 if water footprints in all production locations would be reduced to benchmark levels associated with the best-25th production percentile. Numbers between brackets indicate standard deviation (SD) over time. Provided are the fifteen countries with the highest absolute water saving potential globally.

Country	Total WF 10^9 m^3 (SD)	Saving % (SD)	Blue WF 10^9 m^3 (SD)	Blue water saving % (SD)
India	802 (74)	51 (3)	236 (23)	42 (3)
Brazil	387 (96)	44 (6)	6 (2)	41 (5)
China	545 (44)	29 (3)	135 (18)	11 (1)
United States of America	551 (38)	28 (5)	85 (13)	24 (4)
Nigeria	193 (21)	56 (5)	30 (5)	39 (3)
Russian Federation	173 (18)	57 (7)	10 (1)	10 (2)
Indonesia	181 (36)	33 (4)	13 (4)	41 (8)
Argentina	157 (27)	38 (7)	34 (7)	3 (1)
Pakistan	108 (8)	50 (4)	66 (6)	45 (4)
Philippines	88 (11)	53 (3)	2 (1)	19 (5)
Ukraine	84 (11)	53 (12)	2 (1)	32 (8)
Bangladesh	72 (6)	60 (5)	17 (3)	20 (3)
Mexico	75 (6)	52 (3)	15 (2)	34 (3)
Thailand	88 (8)	43 (4)	13 (2)	29 (3)
Vietnam	67 (6)	43 (6)	11 (2)	5 (2)
World	*5247 (461)*	*44 (3)*	*938 (98)*	*31 (2)*

Table 3-3. Potential annual consumptive (green plus blue) and blue water saving potential per crop, over the period 1996-2015 if all production locations would meet the benchmark of the best-25th production percentile. Number between brackets indicates standard deviation (SD) over time.

Crop	Consumptive WF (10^9 m^3 y^{-1})	Saving % (SD)	Blue WF (10^9 m^3 y^{-1})	Blue water saving % (SD)
Wheat	923	49 (3)	214	34 (2)
Rice	824	51 (2)	269	43 (1)
Maize	727	48 (4)	86.0	21 (3)
Soybeans	512	28 (3)	36.8	8 (2)
Sorghum	197	56 (2)	28.4	20 (3)
Seed cotton	198	45 (7)	76.4	29 (6)
Barley	143	50 (3)	12.5	47 (3)
Groundnuts	119	45 (2)	16.9	22 (3)
Sugar cane	215	24 (3)	40.2	23 (4)
Millet	134	38 (4)	14.3	14 (3)
Cassava	134	38 (4)	2.66	0 (0)
Beans	80.8	56 (3)	6.67	22 (3)
Oil palm fruit	123	32 (5)	1.11	1 (0)
Sunflower seed	85.4	40 (8)	7.72	17 (3)
Rapeseed	81.7	40 (5)	13.3	13 (5)
Potatoes	86.4	36 (4)	15.3	27 (3)
Coconuts	104	28 (4)	3.10	15 (3)
Bananas	41.2	55 (5)	3.12	21 (4)
Sweet potatoes	28.3	66 (5)	2.98	18 (3)
Plantains	50.1	36 (2)	1.45	28 (4)
Grapes	42.0	43 (3)	9.67	24 (3)
Rye	29.1	56 (5)	0.94	29 (3)
Olives	36.9	44 (6)	9.06	34 (6)
Apples	28.9	49 (9)	6.46	30 (2)
Mangoes. mangosteens. guavas	36.6	31 (2)	7.22	18 (3)
Oats	28.9	38 (4)	1.92	29 (4)
Sugar beet	25.8	34 (9)	7.37	27 (5)
Cow peas	24.0	38 (10)	0.48	0 (0)
Onions	17.2	50 (4)	5.49	37 (6)
Peas	17.7	48 (2)	2.04	8 (1)
Oranges	23.1	35 (2)	4.31	27 (4)
Tomatoes	13.9	49 (5)	5.41	35 (3)
Almonds	9.41	66 (7)	3.92	46 (9)
Watermelons	10.7	50 (6)	3.62	28 (4)
Cucumbers and gherkins	6.40	63 (9)	1.68	29 (4)
Dates	12.1	31 (2)	10.0	22 (5)
Pears	8.60	43 (11)	2.01	21 (3)
Peaches and nectarines	8.90	41 (9)	2.12	27 (4)

Crop	Consumptive WF (10^9 m^3 y^{-1})	Saving % (SD)	Blue WF (10^9 m^3 y^{-1})	Blue water saving % (SD)
Yams	13.5	24 (7)	0.34	8 (1)
Cabbages and other brassicas	9.00	37 (6)	1.98	19 (3)
Green chillies and peppers	4.90	57 (5)	0.93	26 (3)
Garlic	4.90	48 (9)	1.26	25 (4)
Green peas	5.40	41 (3)	1.51	16 (3)
Green beans	3.20	61 (9)	0.66	20 (3)
Pumpkins. squash and gourds	4.30	44 (2)	0.69	21 (3)
Triticale	2.30	71 (6)	0.3	25 (4)
Pineapples	2.90	50 (3)	0.11	12 (2)
Carrots and turnips	3.10	42 (7)	0.84	24 (3)
Okra	1.90	58 (8)	0.35	14 (3)
Lettuce and chicory	2.20	47 (2)	0.69	28 (2)
Spinach	1.60	44 (12)	0.41	17 (4)
Lupins	1.20	41 (14)	0.04	18 (8)
Fonio	2.00	19 (8)	0.21	1 (1)
Raspberries	0.40	31 (5)	0.02	10 (6)

Table 3-4. Blue water saving per water scarcity level if everywhere water footprints of crops would be reduced to benchmarks levels associated with the best-25th production percentile, and the crops contributing most to this water saving.

Water scarcity level	Blue WF 10^9 m^3	Blue water saving 10^9 m^3	Blue water saving (%)	Contribution from top-5 contributing crops (%)
Non-scarce	129	28	22	Rice (46), Wheat (12), Maize (8), Seed cotton (7), Sugar cane (5)
Moderate	53	13	25	Rice (43), Maize (11), Wheat (11), Seed cotton (7), Sugar cane (3)
Significant	47	11	23	Rice (49), Wheat (15), Seed cotton (8), Maize (6), Sugar cane (3)
Severe	633	204	32	Rice (37), Wheat (27), Seed cotton (9), Maize (5), Sugar cane (3)

3.4. Discussion

The WFs of 57 crops assessed in this study were estimated with a comprehensive computational framework, *Aqua21*, with global coverage for the period 1961-2015. Although the model routines are more sophisticated than in the first global assessment by Mekonnen & Hoekstra (2014), the underlying global datasets still vary in quality and resolution, incurring uncertainties in the modelled output. For example, the model does not explicitly account for the effects of diseases, weeds, pests, crop genetic varieties and other managerial factors affecting yield. Since we scaled simulated yields with reported national yield statistics, these effects are collectively captured in the scaling factor applied. While a direct comparison is impossible because of the climate-differentiated values presented in this study, Table 3-5 shows that the estimated WFs at the 25th percentile of production as well as the global average WF according to Mekonnen & Hoekstra (2014) fall within the ranges found in this study.

Table 3-5. Comparison of the consumptive WF of selected crops at the 25th percentile of production and the global average with literature. Ranges for this study comprise smallest to largest WFs across climate zones.

Crop	Consumptive WF (m^3 t^{-1}) at 25th percentile of production		Global average consumptive WF (m^3 t^{-1})	
	Mekonnen & Hoekstra (2014)	This study	Mekonnen & Hoekstra (2014)	This study
Barley	546	257-748	1292	894-1936
Cotton	1898	1006-2913	3589	1689-5375
Maize	565	438-900	1028	670-1940
Millet	2905	984-3612	4363	2517-5972
Potatoes	154	125-268	224	197-399
Rice	952	512-781	1486	1142-1614
Sorghum	1122	467-3079	2960	1054-5309
Soybean	1620	1721-3221	2107	2025-4860
Sugarcane	128	88-146	197	118-208
Wheat	1069	387-1253	1620	950-2604

WFs of crops varied in time. Given this inter-annual variability, we tested whether the benchmark level at the best-25th percentile of production over the period 1996-2015 (WF_{p25}) constitutes a reasonable reference target that can be met in individual years as well, particularly in dry years. For any crop-climate combination, we found that some but not all of the 10% driest growing seasons were indeed able to attain WFs below WF_{p25}. Meeting WF_{p25} in dry growing seasons is thus more difficult than in average or wet years.

We followed recommendations by Zhuo et al. (2016) to differentiate WF benchmarks by climate, but other factors, including soil type or crop variety could additionally serve as determinants for differentiation. Moreover, within climate zones numerous rainfall and potential evaporation patterns can emerge over the various phenological stages in the growing season. Accounting for such sub-climate or in-season variability may increasingly refine WF benchmarks for particular production locations or growing seasons. Ideally, a WF benchmark becomes crop, time and location specific, tailored to account for all local factors affecting crop water consumption, and may even vary with seasonal weather forecasts.

The WF benchmarks presented in this study are statistics on the spatiotemporal variability of WFs in global crop production. Although this assessment reveals *that* certain WFs are realized by water-efficient producers, it does not explain *how* these proposed WF benchmarks can be achieved. Alternative to ranking WFs and selecting a certain best-production percentile, Chukalla et al. (2015) developed benchmarks based on the WF associated with the best available technology and farm management practices. This method, tested for only for a few crops in selected locations, generates more actionable pathways for farmers striving to meet resulting WF benchmarks. A useful follow-up study may thus recalculate WF benchmarks of crops based on their approach.

Should WF benchmarking become a (policy) target, trade-offs may occur between meeting either the total consumptive or specifically the blue WF benchmark. For instance, in attempting to lower blue WFs, producers may reduce blue water application to such a degree that it affects the yield, thereby increasing the total consumptive (green plus blue) WF per tonne of crop. While this study did not develop grey WF benchmarks as reference target to reduce pollution, a comparable situation can emerge when lowering grey WFs by providing less fertilizers, as illustrated by Chukalla et al. (2018).

We found that reducing WFs to benchmark levels can alleviate blue water scarcity in many (stressed) locations. The increased water use efficiency can therefore effectively counter pending water crises. However, this is only the case as long as water thus saved is returned to or left in the water system. If freed-up water is used to expand production, the scarcity reduction will soon be nullified by the new demand (Grafton et al., 2018).

3.5. Conclusion

With the global food system as major driver of freshwater consumption and the challenge of securing sufficient freshwater resources to feed growing populations, there is an increasingly strong imperative to reduce water consumption in agriculture. This study explored how water use efficiency in global crop production may be increased by formulating WF benchmarks. We found that 44% of the total water currently consumed in crop production (green plus blue) can be saved, if inefficient producers of the world's major crops would reduce their WFs to climate-differentiated benchmark levels as defined by the best 25th-percentile of current production within each climate zone. Furthering the discourse by developing specific blue WF benchmarks, we discovered that 31% of blue water resources currently consumed in irrigated agriculture can be saved, if producers would reduce WFs of crop to blue WF benchmark levels set by the best 25th-percentile of current production. Leaving water savings thus realized in the water system can alleviate water scarcity in many places, as 83% of the blue water saving is located in areas that face severe water scarcity (Mekonnen & Hoekstra, 2016).

Formulating WF benchmarks is primarily meant to provide a reference target, against which farmers – but also governments and other actors along the value chains of agricultural products – can measure water use efficiency, and be incentivized to implement WF reduction strategies (Hoekstra, 2013). The concept is therefore a helpful tool towards boosting water use efficiency in global crop production, instrumental in reducing humanity's water footprint and promoting the transition to sustainable use of freshwater worldwide.

4

THE BLUE WATER FOOTPRINT OF THE WORLD'S ARTIFICIAL RESERVOIRS FOR HYDROELECTRICITY, IRRIGATION, RESIDENTIAL AND INDUSTRIAL WATER SUPPLY, FLOOD PROTECTION, FISHING AND RECREATION

4. The Blue Water Footprint of the World's Artificial Reservoirs for Hydroelectricity, Irrigation, Residential and Industrial Water Supply, Flood Protection, Fishing and Recreation *

Abstract

For centuries, humans have resorted to building dams to gain control over freshwater available for hu- man consumption. Although dams and their reservoirs have made many important contributions to human development, they receive negative attention as well, because of the large amounts of water they can consume through evaporation. We estimate the blue water footprint of the world's artificial reservoirs and attribute it to the purposes hydroelectricity generation, irrigation water supply, residential and industrial water supply, flood protection, fishing and recreation, based on their economic value. We estimate that economic benefits from 2,235 reservoirs included in this study amount to 265×10^9 US$ a year, with residential and industrial water supply and hydroelectricity generation as major contributors. The water footprint associated with these benefits is the sum of the water footprint of dam construction (<1% contribution) and evaporation from the reservoir's surface area, and globally adds up to 66×10^9 m^3 y^{-1}. The largest share of this water footprint (57%) is located in non-water scarce basins and only 1% in year-round scarce basins. The primary purposes of a reservoir change with increasing water scarcity, from mainly hydroelectricity generation in non-scarce basins, to residential and industrial water supply, irrigation water supply and flood control in scarcer areas.

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* This chapter has been published as:

Hogeboom, R.J., Knook, L., & Hoekstra, A.Y. (2018). The blue water footprint of the world's artificial reservoirs for hydroelectricity, irrigation, residential and industrial water supply, flood protection, fishing and recreation. *Advances in Water Resources* 113: 285-294.

4.1. Introduction

Increasing the limited availability of freshwater to meet ever growing and competing demands is on many policy agendas (WEF, 2017). Drivers for the growing concern include a growing world population, increasing wealth, a transition from fossil-based to renewable energy sources and climate change (UN-WWAP, 2015). For centuries, humans have resorted to building dams to gain control over freshwater available for human consumption. Toward the middle of the 20th century, construction intensified. What started off mainly in the developed world, was soon followed by developing countries in the 1970-80s. When most suitable locations had been developed and most rivers regulated, construction slowed down. Today, new reservoirs are being built mainly for the purpose of hydroelectricity generation (Shiklomanov & Rodda, 2003; Liu et al., 2015; Timpe & Kaplan, 2017).

Dams and their reservoirs have made an important contribution to human development in many ways, such as storing, using and diverting water, for consumption, irrigation, cooling, transportation, construction, mills, power generation, fishing and recreation. Derived benefits have been – and continue to be – considerable (Gernaat et al., 2017; World Commission on Dams, 2000). Associated costs, both in (socio)economic and ecological terms, have been considerable as well (Pacca & Horvath, 2002; Gleick, 2003; Latrubesse et al., 2017). Moreover, since artificial reservoirs have become so prevalent in our modern world, it is increasingly acknowledged that reservoirs are not mere in-stream water users. They can be large water consumers, because of the water that evaporates from their surface. This consumptive term adds pressure on (regional) water resources (Shiklomanov, 2000; Hoekstra, 2013; Vanham, 2016; Liu et al., 2015).

A previous influential study to humanity's water footprint (WF), however, excluded the WF of reservoirs altogether (Hoekstra & Mekonnen, 2012). In addition, despite an addendum acknowledging the importance of water consumption by reservoirs through evaporation, AQUASTAT does not list them as water consumers in national statistics (FAO, 2016). About hydropower, the International Energy Agency does not even mention the term evaporation in their recent World Energy Outlook 2016 (IEA, 2016). By it, they ignore one of the most important balance terms of water for energy

Studies that try to account for water consumption by reservoirs typically employ one of two methods, the so called net approach and the gross approach (Bakken et al., 2013). The net approach reduces evaporation from the reservoir surface with evapotranspiration in the 'natural' state before dam development (e.g. Shiklomanov & Rodda (2003), Grubert

(2016), Scherer & Pfister (2016) and Strachan et al. (2016)). The gross approach, which most studies use, takes total evaporation from the reservoir as measure of reservoir consumption (e.g.Torcellini et al. (2003), Pasqualetti & Kelley (2008), Mekonnen & Hoekstra (2012a), Zhao & Liu (2015)). Although scholars debate on which approach to use, we postulate both approaches have their merit. Confusion arises because of misinterpretation of the intention for which one method is chosen over the other: the net approach is suitable for analyzing changes in hydrology on a catchment scale, while the gross approach is preferred for water footprint (WF) assessments, where the aim is to show the total volume of water appropriated to certain purposes and that is therefore not available for another purpose (Hoekstra, 2017; Hoekstra et al., 2011). This study intents the latter.

Following the global water footprint standard (Hoekstra et al., 2011), the WF related to reservoirs must include all steps of the supply chain. The total WF should afterwards be attributed to derived products and services, based on their economic value. A reservoir generally serves multiple purposes, the most common of which are hydroelectricity generation, supplying water for residential and industrial use, supplying irrigation water, regulating the flow of rivers to prevent flooding and enabling inland navigation (ICOLD, 2011). Reservoirs are rarely created for recreational and fishing purposes, but after a dam is built, these are important secondary purposes (Ward et al., 1996) and therefore share in the WF of reservoirs.

Previous studies attribute the total reservoir water footprint to purposes either partially or using simpler methods. Instead of using economic value, one purpose takes all, purposes receive an equal share, or some prioritization is set up (Scherer & Pfister, 2016; Mekonnen & Hoekstra, 2012a; Grubert, 2016; Bakken et al., 2016).

The aim of this study is to estimate the blue WF of the world's artificial reservoirs, and attribute it to the purposes hydroelectricity generation, residential and industrial water supply, irrigation water supply, flood protection, fishing and recreation, based on their economic value. The blue WF refers to consumption – which includes evaporation – of blue water resources (surface water and groundwater). For each purpose, the WF is expressed in terms of water consumption per unit (that is, m^3 GJ^{-1} for hydroelectricity generation, m^3 ha^{-1} for irrigation water supply, and so on). This unit WF is translated into water consumption per US dollar (in m^3 $US\$^{-1}$), and its inverse, economic water productivity (in US\$ m^{-3}). Productivities of hydroelectricity generation are compared with both productivities found in other studies and those of other types of electricity, thereby feeding discussions on energy scenarios. Although water consumed by reservoirs is no longer available for (downstream) use, the question is how worrisome this is. In water-scarce river basins, the opportunity cost of water consumption may be high and a large WF may worsen scarcity, whereas in more water-rich basins effects may be small. We therefore close with an investigation into the water scarcity levels in all river basins with reservoirs.

4.2. Method and data

The blue water footprint related to an artificial or man-made reservoir (WF_{res}, in $m^3\ y^{-1}$) includes both an operational and a supply-chain part. It thus comprises the WF related to evaporation from the reservoir surface (WF_{evap}, in $m^3\ y^{-1}$) and the WF related to reservoir construction (WF_{constr}, in $m^3\ y^{-1}$):

$$WF_{\mathrm{res}} = WF_{\mathrm{evap}} + WF_{\mathrm{constr}} \quad \text{(Eq. 3-1)}$$

WF_{evap} is determined by means of the so-called gross consumption approach:

$$WF_{\mathrm{evap}} = 10 E A \kappa \quad \text{(Eq. 3-2)}$$

where E (mm y^{-1}) is the depth of water that evaporates yearly from the reservoir surface, A (ha) the maximum reservoir area and κ an area correction factor of 0.5625 to correct for the fact that the reservoir surface at average filling conditions is smaller than the maximum area reported in the databases. This factor is derived from a volume-area relation that is based on the assumption that a reservoir is on average half-filled during the year (Kohli & Frenken, 2015) and is trapezoid-shaped. Multiplication by 10 adjusts the units. Since reservoir areas are considered constant in the databases (see section 2.1) and also κ is kept constant, we fail to capture anomalies in surface areas and hence its effect on WFs. We prudently quantified the resulting uncertainty range for two resulting indicators, namely the global total WF and global average WF of hydroelectricity generation. For these two indicators, we calculated two extreme scenarios, one in which we set κ to 0.2 (indicating all reservoirs evaporate from a surface area that roughly corresponds to the dead storage filling, resulting in the smallest possible WF_{evap}), and one in which we set κ to 1 (indicating all reservoirs evaporate from their maximum surface area, yielding the largest possible WF_{evap}). The resulting range should be interpreted as a preliminary estimate of uncertainty associated with fluctuating reservoir areas.

WF_{constr} depends mainly on the construction material of the dam. Earth and rock fill dams are usually constructed with materials found near the dam site, whereas gravity, buttress and arc dams are mostly made of reinforced concrete (Novak et al., 2007; Chen, 2015). For earth and rock fill dams we accounted only for water consumption related to the energy used to excavate and transport the rock or earth. We took average fuel use from Ahn et al. (2009) and applied to it the WF of diesel (1,058 $m^3 MJ^{-1}$) from Gerbens-Leenes et al. (2008) under the assumption that the material on average is sourced 20 km from the construction site. For gravity, arc and buttress dams we estimated only the WF of reinforced concrete, using the WF of cement and unalloyed steel from Gerbens-Leenes et al. (2018) and assuming a mixture of 1% steel, 29% cement and 70% aggregates. We used the WF for rock and earth as describe above also for aggregates. Water consumption related to clearing the construction site, equipment and installations either lack reliable data or were assumed negligible. We therefore did not accounted for these terms. Finally, the annual WF_{constr} is calculated by dividing the water footprint of construction by the assumed typical lifespan of a dam of 100 years.

WF_{res} is assigned to the different reservoir purposes i (WF_I, in m^3 per unit of production) through an allocation coefficient η_i that is based on the economic value V (US$) of each purpose i:

$$WF_i = \eta_i WF_{res} \quad \text{with} \quad \eta_i = V_i \Big/ \sum V_i \qquad \text{(Eq. 3-3)}$$

Lastly, we placed WF_{res} in the context of local water scarcity. We used monthly water scarcity levels per river basin, representative of and averaged over the period 1996-2005 as provided by Hoekstra et al. (2012), to examine the scarcity level of the basin in which the reservoir is located. A basin is considered water scarce if the total blue WF of all human activities combined exceeds water availability (runoff minus environmental flow requirements) in any given month. Because the study by Hoekstra et al. (2012) does not cover all basins, this scarcity analysis includes 71% of reservoirs in this study that are located in basins with data available; the remaining 29% were excluded.

4.2.1. Reservoir data

Dam and reservoir data are obtained by combining two databases: the World Register of Dams (WRD) from the International Commission on Large Dams (ICOLD, 2011) and the Global Dams and Reservoirs Database (GRanD) by Lehner et al. (2011). WRD contains over 37,000 reservoirs, including information on reservoir purpose, depth and area, dam height, type and body volume, location (latitude, longitude) and production data for hydroelectricity generation, irrigation water supply and other benefits. GRandD contains information on 6,854 reservoirs, including reservoir purpose, average depth and maximum area, dam height and elevation, and comes with a georeferenced vector map of reservoir-shaped polygons. Neither database reports temporal variations, either intra- or inter-annual, in any dam or reservoir variable.

WRD and GRanD are combined, since neither one is complete nor contains all information required. We linked the two databases based on the name and country of each dam. Also, we put in a manual effort to match as many entries as possible based on alternative or slightly different dam or country names. If an element, such as height or area, was present in both databases, GRanD data were selected because of its perceived higher quality. After excluding reservoirs with a reported natural origin, river and coastal barrages and entries with missing production data, our final database contained 2,235 reservoirs with full data availability (Figure 4-1). These 2,235 reservoirs cover a maximum surface area of 129,000 km^2 (~50% of total GRanD database surface area of manmade reservoirs).

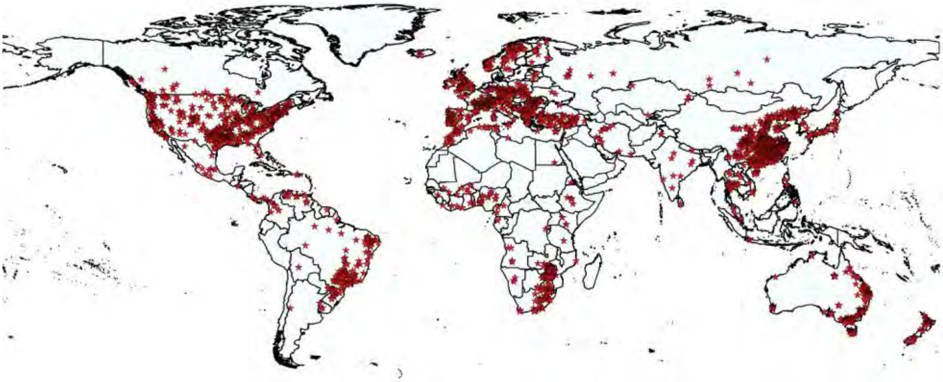

Figure 4-1. Combining the WRD and GRanD databases yields 2,235 reservoirs with full data availability.

4.2.2. Evaporation estimation

Many methods exist to calculate evaporation. To prevent bias toward any one method, we estimated open water evaporation from the 2,235 reservoirs as an ensemble mean of four different methods: the ones provided by Kohli & Frenken (2015), Jensen & Haise (1963), Hamon (1961) and a modified version of Penman (Kohler et al., 1955; Harwell, 2012).

The Kohli and Frenken (KF) method is straightforward:

$$E_{\mathrm{KF}} = \sum_{\mathrm{j=1}}^{365} k_{\mathrm{c}} ET_0 \qquad \text{(Eq. 3-4)}$$

where k_c is a crop coefficient, set to 1 for open water, and ET_0 (mm d^{-1}) the daily FAO reference evapotranspiration rate.

The Jensen and Haise (JH) method is an energy budget based method which has proven accurate under limited data availability (Rosenberry et al., 2007; Majidi et al., 2015):

$$E_{\mathrm{JH}} = \sum_{\mathrm{j=1}}^{365} 0.03523 R_{\mathrm{s}} (0.014 T_{\mathrm{a}} - 0.37) \qquad \text{(Eq. 3-5)}$$

where R_s (W m^{-2}) is incoming solar radiation and T_a (°F) mean daily temperature. If T_a becomes lower than 26.5 °F, evaporation becomes negative, in which case evaporation becomes zero.

The Hamon (H) method calculates evapotranspiration based on the relation between maximum incoming energy and the moisture capacity of the air. It assumes open water evaporation is equal to evapotranspiration. We took the modified version of this method as used by the US Army Corps of Engineers (Harwell, 2012):

$$E_{\mathrm{H}} = \sum_{\mathrm{j=1}}^{365} 13.97 \left(\frac{N}{12}\right)^2 \left(\frac{SVD}{100}\right) \qquad \text{(Eq. 3-6)}$$

where N is the maximum number of daylight hours and SVD (g m^{-3}) the saturation vapor density.

The modified Penman (P) method combines mass transfer and energy budget methods, which expels the need for the surface water temperature, to determine open water evaporation (Harwell, 2012; Majidi et al., 2015):

$$E_{\mathrm{P}} = \sum_{\mathrm{j=1}}^{365} \left(\frac{\Delta}{\Delta + \gamma} R_{\mathrm{n}} + \frac{\gamma}{\Delta + \gamma} E_{\mathrm{a}} \right) \qquad \text{(Eq. 3-7)}$$

where Δ is the gradient of saturated vapor pressure, γ the psychrometric constant, R_n (mm d^{-1}) the effective net radiation and R_a (mm d^{-1}) the evaporation from a Class A pan.

Climate data necessary to evaluate the four evaporation methods was taken from the ERA-Interim database (Dee et al., 2011), with a spatial resolution of 5 × 5 arc minutes, for the period 1981-2010. We aggregated sub-daily data to daily values, because not all variables were available on a daily same time step. For each variable, we calculated average daily values over the period 1981-2010 to yield evaporation estimates for one (climate-averaged) year. Evaporation was evaluated at the midpoint of each reservoir.

4.2.3. Economic value and attribution

The WRD database states the purposes of a reservoir and provides production information on hydroelectricity generation, irrigation water supply and flood control storage, if present. For some reservoirs, production information was conflicting with purpose data. As a rule, we recognized hydroelectricity generation, irrigation water supply and flood control storage as a purpose if production data was available – even if the database did not explicitly list it as a purpose. We excluded navigation as a reservoir purpose, because only a few reservoirs are reported to serve this purpose. Besides, no data are reported on the economic value of reservoirs for the specific purpose of navigation. We converted and discounted all prices to represent 2014-equivalent US dollars, using inflation correction factors and exchange rates from the World Bank (2015) and Williamson (2015).

4.2.3.1. Hydroelectricity generation

The economic value of hydroelectricity generation (US$ y^{-1}) is calculated by multiplying the mean annual electricity generation (GWh y^{-1}) with the economic value of electricity in the country where the reservoir is located (US$ GWh^{-1}). For some reservoirs (984 in total), WRD reports both mean annual electricity generation and production capacity. For reservoirs with only a production capacity reported (359 reservoirs), we assumed mean annual electricity generation as 34% of the production capacity. This percentage is based on the ratio between mean and capacity production for reservoirs which have both metrics stated. National electricity prices were taken from IEA (2012); RCREEE (2013); EUROSTAT (2015) or EUROSTAT (2015) or, if not available in these, based on prices found for comparable neighboring countries.

4.2.3.2. Irrigation water supply

The economic value of irrigation water supply (US$ y^{-1}) is calculated by multiplying the irrigated area serviced by the reservoir as provided by WRD (ha) with the average economic value of agricultural land in the country where the reservoir is located (US$ ha^{-1} y^{-1}). The latter is a proxy for the value of crops that are actually being irrigated with water from the reservoir, because the databases are limited to reporting general servicing area for agricultural purposes. The average economic value of agricultural land is estimated per country by dividing the value of agricultural production of all crops in the country (US\$ y^{-1}) by the production area of all crops in the country (ha). Agricultural value and harvested area per crop per country were taken from FAOSTAT (2015) for the year 2013.

4.2.3.3. Flood control storage

The economic value of flood control storage (US\$ y^{-1}) is calculated by multiplying the available flood storage volume as provided by WRD (m^3) with the economic value of flood storage (US\$ m^{-3} y^{-1}). The only study plainly stating economic value of flood storage to our knowledge is by Zhao & Liu (2015), who found a value of 0,16 US\$ m^{-3} y^{-1} for the Three Gorges reservoir in China. The US Army Corps of Engineers (USACE, 2016) reports on prevented flood damage since the year of completion for several of its projects, most noticeably 24 reservoirs in the North-East US. Translated to mean annual values, their study yielded estimates of 0,002 to 0,58 US\$ m^{-3} y^{-1}, with an average of 0,117 US\$ m^{-3} y^{-1}, which is similar to the aforementioned estimate by Zhao & Liu (2015). We used 0,117 US\$ m^{-3} y^{-1} as a proxy for all reservoirs globally that have flood control as a stated purpose.

4.2.3.4. Residential and industrial water supply

The economic value of residential and industrial water supply from reservoirs (US\$ y^{-1}) is calculated by multiplying the estimated yearly abstracted volume (m^3 y^{-1}) with the economic value of residential water in the country where the reservoir is located (US\$ m^{-3}). However, estimates of yearly abstracted volumes are not readily available. Based on data from 132 reservoirs in the United States (IWR, 2012) and 30 in Australia (Knook, 2016), the ratio between abstracted volume over reservoir volume was determined. These ratios showed a large variation, mainly because of size of the reservoir and climate: small reservoirs in humid climates typically have higher ratios than large ones in A regions. Based on the set of 162 reservoirs, we drew two rating curves – one for humid and one for (semi-) A regions. Depending on the climate zone of individual reservoirs, either curve prescribes the estimated abstracted volume. Note that in this procedure, other factors that might influence abstracted volumes are not considered. National water prices were taken from Danilenko et al. (2014), IWA (2012) and OECD (2010) or, if not available, based on prices found for comparable neighboring countries.

4.2.3.5. Recreation

The economic value of recreation (US$ y^{-1}) is calculated by multiplying the economic value of recreation (US$ ha^{-1} y^{-1}) with the reservoir surface area (ha). Although expressing values of recreation in terms of surface area is rare, Costanza et al. (1997) gave a value of 368 US$ ha^{-1} in 2014 net present value. For lack of better estimates, we applied this value to all reservoirs with recreation as stated purpose.

4.2.3.6. Commercial fishing

The economic value of recreation (US$ y^{-1}) is calculated by multiplying the average economic value of wild caught freshwater fish (US$ kg^{-1}) with the reservoir surface area (ha) and the fishing yield (kg ha^{-1} y^{-1}). Economic values were obtained from WFC (2008), Mitchell et al. (2012) and FAO (2015a). Fishing yields depend on multiple factors such as water body volume, food supply and climate (Marmulla, 2001), but country-average yields were the best metric we could find. Average national fishing yields were taken from Marmulla (2001), Van Zwieten et al. (2011) and Mitchell et al. (2012) or, if not available, based on prices found for comparable neighboring countries. For lack of reliable data, we excluded aquaculture in our analysis.

4.3. Results

4.3.1. The water footprint related to artificial reservoirs

The global water footprint of evaporation from the 2,235 reservoirs in this study, averaged over the four estimation methods, is 65.7×10^9 m^3 y^{-1}. Figure 4-2 shows the evaporation distribution for each method over all reservoirs. Table 4-1 gives the WF_{evap} aggregated to the continent level. We grouped the reservoirs by climate class following the Köppen-Geiger classification (Kottek et al., 2006) in Figure 4-3. The different methods vary in their resulting estimates of evaporation. Typically, the straightforward Kohli and Frenken method gives the highest evaporation estimates (especially in warm A climates). The Hamon method yields the lowest estimates, which was anticipated by previous studies by Harwell (2012) and Majidi et al. (2015). The Jensen-Haise method estimates higher evaporation rates in equatorial climates compared to other methods, possibly because the Jensen-Haise method was originally developed for more A regions (Jensen & Haise, 1963). Given that this study included only those reservoirs for which all data were available, the total, global water footprint of reservoirs must be substantially higher than the number presented here.

Table 4-1. Estimated evaporation volumes from reservoirs for each continent.

	WF_{evap} (10^9 m^3 y^{-1})			WF_{constr} (10^6 m^3 y^{-1})
	Minimum	Average	Maximum	
Africa	15.6	18.8	25.1	0.38
Asia	15.3	18.0	22.3	33.4
Europe	2.9	3.8	4.7	0.74
North America	2.9	3.5	4.3	4.62
Oceania	1.0	1.2	1.5	0.18
South America	16.4	20.5	25.6	0.35
Global	*54.1*	*65.7*	*83.6*	*39.6*

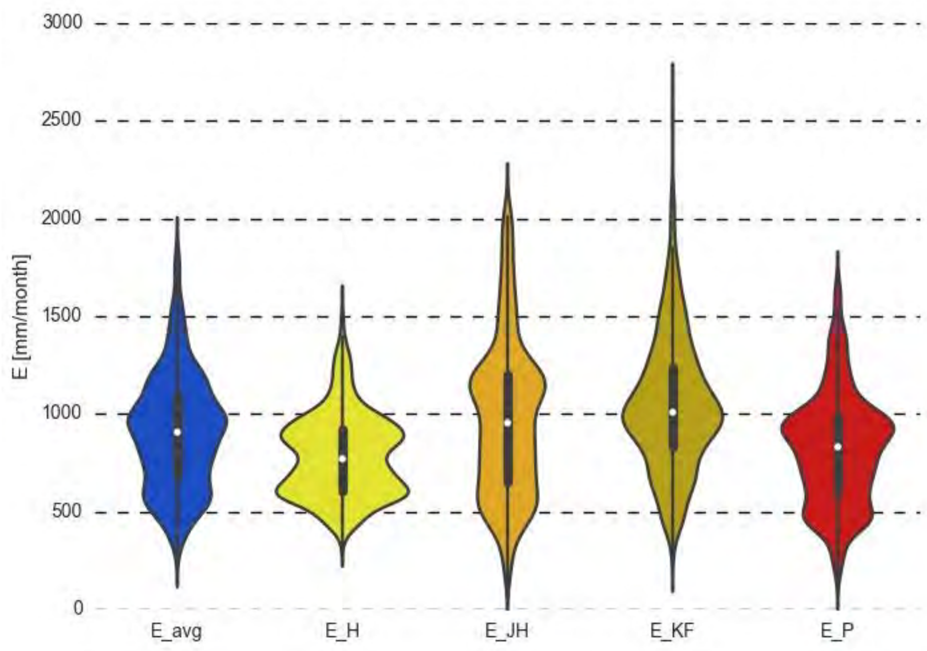

Figure 4-2. Comparison of estimated distribution functions and quartiles of different evaporation (E) methods (average (avg), Hamon (H), Jensen-Haise (JH), Kohli-Frenken (KF) and Penman (P)) over all reservoirs.

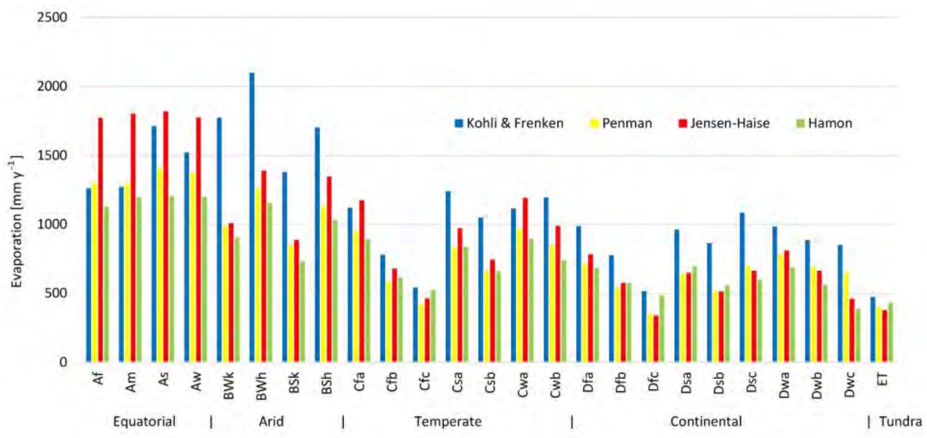

Figure 4-3. Average evaporation rate per climate class for the four evaporation methods.

The global water footprint of reservoir construction for the 2,235 reservoirs in this study is 39.6×10^6 m^3 y^{-1} (Table 4-1). This number represents 0.05% of WF_{evap} and thus hardly contributes to the total yearly WF of reservoirs. Each reservoir individually shows such insignificant WF_{contr} share of the total as well. Note, however, that WF_{constr} is only trifling because we discounted it over the dam's lifespan. Water consumption could be significant still during the period of actual dam construction.

4.3.2. Allocation of WF_{res} to purposes based on economic value

The total economic value of the reservoirs in this study, spawned by hydroelectricity generation, irrigation water supply, flood control, domestic and industrial water supply, recreation and fishing, is 265×10^9 US$ in 2014 dollars. Table 2 shows the total economic value and allocation coefficients for each continent. Hydroelectricity generation, irrigation water supply and residential and industrial water supply account for the largest part of reservoirs' economic value. These are also the most common reservoir purposes.

Table 4-2. Total annual economic value generated and allocation coefficients for each continent.

	Number of Reservoirs	Economic value (10^9 US\$ y^{-1})	Allocation coefficients (η) per purpose (%)					
			Hydroelectricity generation	Irrigation water supply	Flood control	Residential/ industrial supply	Recreation	Fishing
Africa	203	16.5	23	18	37	22	0.0	0.0
Asia	653	92.8	21	52	17	10	0.0	0.5
Europe	519	39.2	27	4	17	53	0.0	0.0
North America	549	20.5	30	0	0	70	0.4	0.0
Oceania	171	15.1	14	5	0	80	0.0	0.0
South America	140	80.8	84	0	1	15	0.1	0.0
Global	*2235*	*264.8*	*41*	*20*	*11*	*27*	*0.1*	*0.2*

With the water footprint and allocation coefficients of each reservoir, we calculated the WF per purpose per reservoir. Table 4-3 summarizes the results at the continental level. The global water footprint study by Hoekstra & Mekonnen (2012) – which ignored water losses from reservoirs by evaporation – estimated the blue water footprint of crop production at 899 × 10^9 m^3 y^{-1} and the blue water footprint of industrial and domestic water supply at 80 × 10^9 m^3 y^{-1} (averages over the period 1996-2005). To be complete, the evaporation from reservoirs assigned to irrigation water supply (4.45 × 10^9 m^3 y^{-1}) must be added to the WF of crop production. Likewise, the WF of reservoirs assigned to residential and industrial water supply (6.47 × 10^9 m^3 y^{-1}) and hydroelectricity generation (48.4 × 10^9 m^3 y^{-1}) must be added to the WF of industrial and domestic supply. If we do so, the global blue WF of crop production is roughly 0.5% higher than estimated by Hoekstra and Mekonnen, and the global blue WF of industrial and domestic water supply even 69% higher. Note that these are still conservative estimates, since our study includes only 30% of the world's reservoir area.

Table 4-3. Water footprint of reservoirs per reservoir purpose for each continent. Dashes imply reservoirs do not serve the specified purpose and/or lack sufficient data on the purpose according to the used datasets.

	Hydro-electricity generation (10^9 m^3 y^{-1})	Irrigation water supply (10^9 m^3 y^{-1})	Flood control (10^9 m^3 y^{-1})	Residential/ industrial supply (10^9 m^3 y^{-1})	Recreation (10^6 m^3 y^{-1})	Fishing (10^6 m^3 y^{-1})
Africa	12.3	1.95	4.01	0.39	53	31
Asia	12.7	2.09	1.21	1.96	3	66
Europe	2.54	0.06	0.06	1.09	3	0
North America	0.97	-	-	1.64	863	3
Oceania	0.38	0.32	0.01	0.52	3	-
South America	19.4	0.03	0.13	0.86	3	-
Global	*48.4*	*4.45*	*5.42*	*6.47*	*928*	*100*

Table 4-4. Global average WF per unit of production per reservoir purpose, as it varies across evaporation methods (columns 2-4) and across reservoirs (columns 5-8).

Reservoir purpose	Evaporation method			Reservoirs in 66% range		Reservoirs in 95% range	
	Minimum	Average	Maximum	Low	High	Low	High
Hydroelectricity generation (m^3 GJ^{-1})	12.1	14.6	18.3	0.3	10.0	0.1	207.1
Irrigation water supply (m^3 ha^{-1})	229	277	368	94	1634	21	10989
Flood control (m^3 m^{-3})	0.018	0.022	0.031	0.003	0.044	0.001	0.358
Residential/ industrial supply (m^3 m^{-3})	0.071	0.090	0.112	0.015	0.177	0.003	0.538
Recreation (m^3 ha^{-1})	2013	2321	2833	18	11360	2	41532
Fishing (m^3 t^{-1})	0.81	0.94	1.12	0.10	0.99	0.04	26.95

Table 4-4 shows the global average WF per unit of production for each purpose, using the lowest, the highest and the average evaporation estimate of the four evaluated evaporation methods. Note that results are not comparable among purposes, because the unit of production or the unit interpretation differ (for example, for flood control a cubic meter refers to a volume stored, while for residential and industrial water supply it refers to a volume delivered). The right-hand side of Table 4-4 shows that the WF per unit of production not only differs for each evaporation method, but also from reservoir to reservoir. The 66% range around the median – that is, 66% of the reservoirs in this study with the stated purpose have a WF per unit of production between the reported high and low value – demonstrates the large variability found among reservoirs. This variability is mainly owing to reservoir surface size in relation to each purpose's production size, rather than to climate (cf Mekonnen & Hoekstra, 2012a; Liu et al., 2015). Related is the variation in reservoir surface area itself, induced by seasonality or reservoir regulation, leading to uncertainty around our estimates. Table 4-5 shows the uncertainty associated with the global total WF and the global average WF of hydroelectricity.

Table 4-5. Scenarios regarding the evaporative surface area of reservoirs. The ranges between brackets refer to uncertainties in the evaporation estimation method.

	Reservoir areas at:		
	20% of max. capacity	56.25% of max. capacity	100% of max. capacity
Global WF_{evap} (10^9 m^3 y^{-1})	23.4 (19.2-29.7)	65.7 (54.1-83.6)	117 (96.2-149)
WF hydroelectricity generation (m^3 GJ^{-1})	5.2 (4.3-6.5)	14.6 (12.1-18.3)	25.9 (21.4-32.5)

The purposes become mutually comparable if we consider the WF per dollar of economic output (m^3 $US\$^{-1}$), or its reverse, the economic water productivity (US\$ m^{-3}). Table 4-6 shows the WF per dollar production, based on WFs per purpose averaged over the four evaporation methods. The WF per dollar of economic output is relatively low (high productivity) for residential and industrial water supply and irrigation water supply, and relatively high (low productivity) for recreation. Except for recreation, all purposes yield at least several dollars in revenue for each cubic meter of water evaporated.

Table 4-6. WF per dollar of economic output and economic water productivity per purpose.

Reservoir purpose	WF per dollar (m^3 $US\$^{-1}$)	Economic water productivity ($US\$$ m^{-3})
Hydroelectricity generation	0.44	2,26
Irrigation water supply	0.08	12,10
Flood control	0.19	5,32
Residential/ industrial water supply	0.09	11,20
Recreation	6.31	0,16
Fishing	0.21	4,81

Zooming in to the country level, we find that Brazil has the largest total WF related to reservoirs, followed by Russia and Egypt (Figure 4-4). This is a different list than that of countries with the highest installed reservoir area, which is headed by Russia, Brazil and China. This difference can be explained by the climatic conditions, which favor high evaporation in these high WF countries. For some countries, our database included only one reservoir, which usually is a very large reservoir that experiences high evaporation rates. Examples are Lake Nasser in Egypt, Lake Volta in Ghana and the Brokopondo reservoir in Suriname. Although there are 71 reservoirs included in Zimbabwe, the total WF of reservoirs there largely results from the Kariba reservoir. Results per reservoir and per country area available in Hogeboom et al. (2018b).

4.3.3. The water footprint of reservoirs in the context of water scarcity

The largest part (57%) of the WF of reservoirs is located in river basins with a low water scarcity level (Figure 4-5a). The other part is located in basins facing 1-3 months (29%), 4-6 months (7%) or 7-11 months (5%) of moderate to severe water scarcity a year. Moderate to severe water scarcity here means that more water is consumed than sustainably available – that is, environmental flow requirements are violated (Hoekstra et al., 2012). About 1% of the WF of reservoirs lies in basins with year-round moderate to severe water scarcity. We further find that in river basins with low water scarcity throughout the year, hydroelectricity generation constitutes the largest part of WF_{res}. The relative contribution of hydroelectricity generation to the total decreases with increasing water scarcity levels (Figure 4-5b-f). In river basins with more than 7 months of moderate to severe water scarcity, residential and industrial water supply are the primary reservoir purposes. This finding confirms a previous assessment by Bakken et al. (2015), who found few reservoirs used for hydroelectricity generation are located in water-scarce areas.

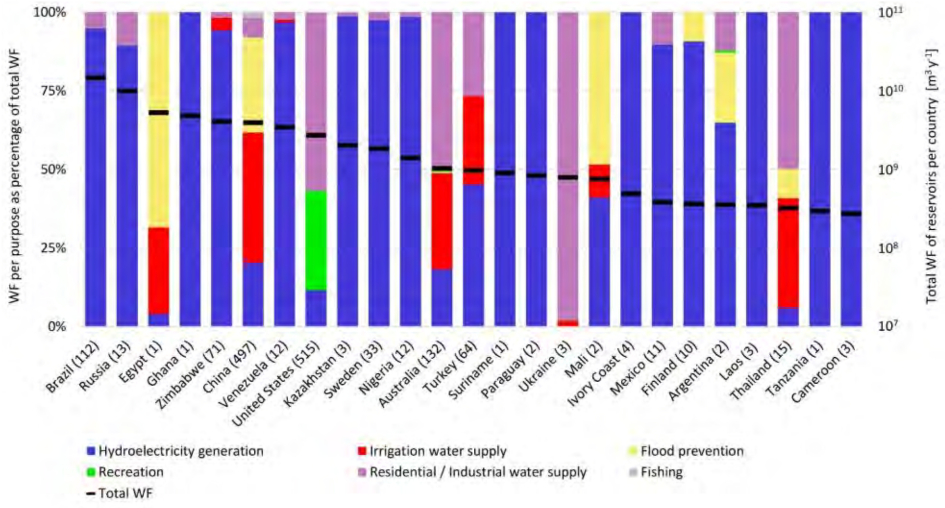

Figure 4-4. The total water footprint of reservoirs per country and the share of different reservoir purposes in the total for the 25 countries with the largest total water footprint. The number between brackets refers to the number of reservoirs in the country included in the study.

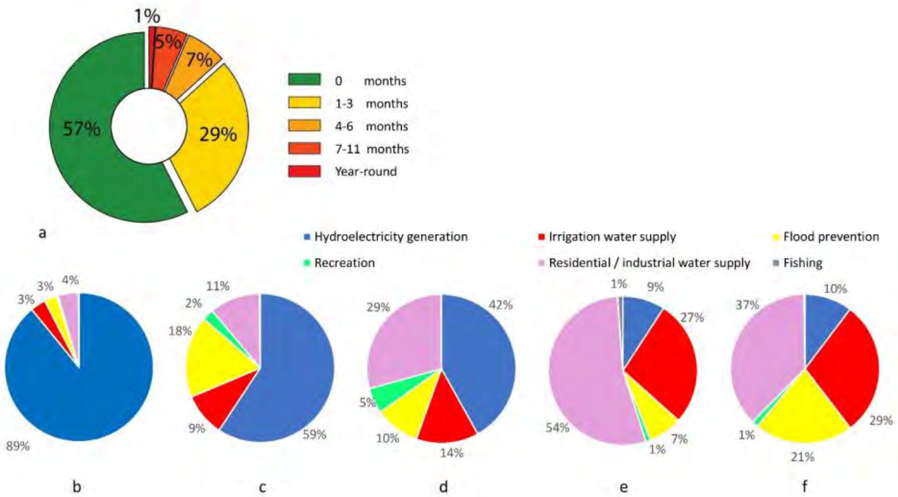

Figure 4-5. The share of the global WF of reservoirs in river basins facing moderate to severe water scarcity during 0, 1-3, 4-6, 7-11 or 12 months per year (a); and the WF share per purpose located in basins facing moderate to severe water scarcity during zero (b); 1-3 (c); 4-6 (d); 7-11 (e); or 12 (f) months per year.

4.4. Discussion

We have chosen four different evaporation methods to calculate WF_{evap} out of a host of other available approaches. We anticipated these methods would vary – as indeed they did – so we calculated an ensemble mean. Ambiguity remains on what method(s) to choose to assess open water evaporation. In addition, adding effects such as thermal heat storage, which was disregarded in this study, both complicates and improves evaporation estimations (Majidi et al., 2015; Finch, 2001). It is worth noticing that the simple Kohli and Frenken method – which is often used to calculate open water evaporation – turned out to be the most deviant method.

Beside methodological considerations, resulting evaporation fluxes depend to a large extent on the shape and surface area of the reservoir, as is shown in Table 4-5. Although we touched on the uncertainty associated with keeping areas constant, adding time series on fluctuating reservoir areas would reduce uncertainty and allow for more detailed, time-dependent WF estimations.

The difficulty to determine surface area (or any dam or reservoir parameter for that matter) became clear when we combined the ICOLD and GRanD databases. Incompleteness and differing definitions, naming or surveying methods between the two, inevitably propagated to our resulting reservoir database. Especially the lack of data on abstractions for domestic and industrial water supply took away from ICOLD's usefulness. Moreover, data availability of most variables needed to estimate economic value of reservoirs was low. Especially national values of recreation and of fishing yields and prices were often approximated, and data on volumes abstracted for industrial and residential supply should be considered with caution.

The estimated global WF of reservoirs is based on a selected set of reservoirs for which enough data was available. This set represents a combined surface area of 129,000 km^2 (~50% of full GRanD database surface area of manmade reservoirs or ~30% if all GRanD reservoirs are included). Shiklomanov & Rodda (2003) estimated an installed reservoir area of 500,000 km^2 around the beginning of the 21st century, which according to their estimate evaporated $208 \times 10^9\ m^3\ y^{-1}$. The total WF_{res} of $66 \times 10^9\ m^3\ y^{-1}$ found in this study could prudently be extrapolated to account for the excluded reservoir surfaces. A tentative estimate of global reservoir WF, then, is about $250 \times 10^9\ m^3\ y^{-1}$. This final figure corresponds to ~25% of the total human blue water consumption ($1{,}025 \times 10^9\ m^3\ y^{-1}$) as estimated by Hoekstra & Mekonnen (2012).

We found the global average WF of hydroelectricity is 14.6 $m^3\,GJ^{-1}$ (Table 4-4), and varies highly among reservoirs. Gerbens-Leenes et al. (2008) report 22 $m^3\,GJ^{-1}$, Mekonnen et al. (2015) 15.1 $m^3\,GJ^{-1}$, Scherer & Pfister (2016) 17.1 $m^3\,GJ^{-1}$ (median) or 38.9 $m^3\,GJ^{-1}$ (global average), and Bakken et al. (2017) give a range of WF of hydroelectricity (determined via the gross approach) of 1.5 - 65 $m^3\,GJ^{-1}$. Despite the differences in methods and data used, these values indicate that hydroelectricity is a water intensive form of energy compared to other energy sources (cf Mekonnen et al., 2015). If we again extrapolate our findings, by applying the average WF of hydroelectricity generation of 14.6 $m^3\,GJ^{-1}$ to the global hydroelectricity production of 3940 TWh in 2015 (World Energy Council, 2016), we prudently estimate the global WF of hydroelectricity production is $207 \times 10^9\ m^3\,y^{-1}$ – adding over 20% to the total global blue water footprint as estimated earlier by Hoekstra & Mekonnen (2012).

We confined the spatial system boundaries to the reservoir. The influence of reservoirs on evaporation, especially in cascaded systems, extends beyond the reservoir to the rivers below, because of a change in flow regime (Bakken et al., 2013). Depending on the system, this regime change can lead to decreased downstream evaporation (because of decreased flood duration and associated evaporation from flooded land), or an increase in evaporation (because of raised groundwater levels due to additional percolation and associated evaporation from groundwater). Although these processes and their importance differ at the individual reservoir level, they may cancel out at the larger scale (Shiklomanov & Rodda, 2003).

This study gives a first glimpse of how the beneficiary purposes of reservoirs share the burden of water consumption. The reservoir and climate data are all taken from global databases, with all accompanying restraints. For individual reservoirs – both those covered in this study and those to be developed in the future – we recommend to redo this analysis using local data whenever possible.

4.5. Conclusion

Building a dam and reservoir can be a valuable measure to address a host of water related issues. This study estimates the economic benefits from reservoir products and services is US$ 265 × 10^9 globally, mainly because of value added by hydroelectricity generation, residential water supply and industrial water supply. We also show that these benefits come at a significant cost in terms of water loss. The total blue water footprint of 2,235 reservoirs included in this study, related to both dam construction and evaporation losses from reservoir surfaces, is 66×10^9 m^3 y^{-1}. Water use studies, dam development plans, and water-for-energy scenarios seldom account for this reservoir water footprint. Paradoxically, a reservoir may be an apt measure to increase water availability during a certain time of the year, but only at the cost of reducing total water availability over the whole year.

Since reservoirs typically serve multiple purposes, the total WF of reservoirs must be assigned to those purposes. Rather than leaving them implicit, we explicate quantitative WFs per reservoir purpose, by attributing the total WF to purposes based on their estimated economic value – for the first time, and on a global scale. From the reservoir purposes considered, hydroelectricity generation constitutes the largest share of the total WF, followed by residential and industrial water supply. The global average WF of hydroelectricity is estimated at 14.6 m^3 GJ^{-1}, which is in line with estimates by previous studies. It demonstrates that hydroelectricity – on average – is a water intensive form of energy. On the positive side, economic water productivity (US\$ m^{-3}) is high for all purposes except recreation.

For each reservoir purpose, the WF per unit of production shows substantial variability around the global average. One factor contributing to the spread is the choice of method to estimate open water evaporation. Another is climate, because cold temperate climates give rise to low WFs and equatorial and A climates to high WFs. However, the reservoir surface size in relation to the production size of each purpose contributes most to the variability.

We investigated the water scarcity levels of the basins in which reservoirs are located and found the majority (57%) of reservoir-related WFs is located in water-abundant basins. The remainder is located in basins with one or more months of moderate to severe water scarcity. The primary reservoir purpose changes with changing water scarcity levels. While hydroelectricity generation is the primary purpose in non-scarce basins, in scarcer areas residential and industrial water supply and irrigation water supply are the purposes for which the reservoir is mostly used.

Because of growing freshwater demand, increasing water-scarcity levels worldwide, and continuing dam developments, water consumption from artificial reservoirs needs to be accounted for. All value-generating purposes of a reservoir share in this WF burden. We therefore recommend to build on this methodology, and apply it to future dam development and water-for-energy scenario studies in specific, and to water use assessments in general.

5

WATER AND LAND FOOTPRINTS AND ECONOMIC PRODUCTIVITY AS FACTORS IN LOCAL CROP CHOICE: THE CASE OF SILK IN MALAWI

5. Water and Land Footprints and Economic Productivity as Factors in Local Crop Choice: The Case of Silk in Malawi *

Abstract

In deciding what crops to grow, farmers will look, among other things, at the economically most productive use of the water and land resources that they have access to. However, optimizing water and land use at the farm level may result in total water and land footprints at the catchment level that are in conflict with sustainable resource use. This study explores how data on water and land footprints, and on economic water and land productivity can inform micro-level decision making of crop choice, in the macro-level context of sustainable resource use. For a proposed sericulture project in Malawi, we calculated water and land footprints of silk along its production chain, and economic water and land productivities. We compared these to current cropping practices, and addressed the implications of water consumption at the catchment scale. We found that farmers may prefer irrigated silk production over currently grown rain-fed staple crops, because its economic water and land productivity is higher than that for currently grown crops. However, because the water footprint of irrigated silk is higher, sericulture will increase the pressure on local water resources. Since water consumption in the catchment generally does not exceed the maximum sustainable footprint, sericulture is a viable alternative crop for farmers in the case study area, as long as silk production remains small-scale (~3% of the area at most) and does not depress local food markets.

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* This chapter has been published as:

Hogeboom, R.J. & Hoekstra, A.Y. (2017). Water and Land Footprints and Economic Productivity as Factors in Local Crop Choice: The Case of Silk in Malawi. *Water* 9(802): w9100802.

5.1. Introduction

Suppose you are a farmer in Malawi. What crops would you grow, and on what factors would you base that decision? You would probably consider the availability, quality and cost of seeds, labour, land, water, fertilizers and technology, the access to markets, available capital to invest, insurance, and what alternative options you have to feed your family if crops fail. Now you are aware that pressures on water and land resources are increasing – due to climate change, growing populations and more demanding lifestyles – and you want to find out how your operations affect overall questions of sustainability, efficient resource use, and equity. How can you make sure you maximize your farming operations' profitability, while at the same time minimizing harmful impacts on both others in your area and on the next generation? After all, they will also need the natural resources to support their livelihoods.

This stream-of-thought sketches the tension between micro-level decision making in agriculture and its macro-level effects. Much research has been done to identify factors that influence local crop choice (Dercon, 1996; Raina, 2000; Qiu et al., 2016; Sherrick et al., 2004; Wineman & Crawford, 2017; Schmautz et al., 2016; Tan et al., 2017). In the current study, we focus on water and land availability and consider indicators such as water and land footprints and economic water and land productivity (Hoekstra, 2017; Bruckner et al., 2015; Aldaya et al., 2010; Gutierrez-Martin et al., 2017). Water footprints (WF) and land footprints (LF) of crop production represent the volume of water (m^3) and area of land (m^2) that are appropriated to produce a crop (kg) (Hoekstra et al., 2011). Footprints inform the farmer how much water and land the intended crop requires in absolute terms, or, if compared to a benchmark footprint for that crop, in relative terms (Hoekstra, 2015; Chukalla et al., 2017). Economic water productivity (EWP, in € m^{-3}) and economic land productivity (ELP, in € m^{-2}) address economic considerations, by showing how much money each cubic meter of water or square meter of land generates.

Whereas micro-level questions focus on efficiency and productivity, macro-level questions are concerned with the sustainability and equity of resource use at the higher system level, such as the catchment, biome or even global level (Hoekstra & Wiedmann, 2014). Total footprints at the system level result from the pressures placed on the system by all individual water and land using activities combined. Studies concerned with macro-level questions typically try to quantify total pressure limits of the system, also termed assimilation capacity, operation space or boundaries (Steffen et al., 2015; Vörösmarty et al., 2010; Hoekstra & Wiedmann, 2014). Exceeding these leads to undesirable consequences. Defining maximum sustainable footprints is one way to quantify such macro-level limits to resource use (Chenoweth et al., 2014). If farmers are only guided by micro-level factors – such as local water and land footprints, or economic and land productivities of their intended crops – then maximum sustainable system footprints may eventually be violated at the macro-level. On the other hand, total footprint limits at the system level only become practical if they can be translated to implications at the local level.

The aim of this study is therefore to explore how data on water and land footprints and economic water and land productivity can inform micro-level decision making on crop choice, in the context of macro-level sustainability of resource use, for a case study of proposed silk production in Mzimba District in Malawi. Malawi is economically poor, but relatively rich in arable land and water resources. It has a large untapped potential for irrigation expansion (IWMI, 2010). Nevertheless, agricultural output is low and about a quarter of the population is unable to secure its minimum daily recommended food intake, despite enough food being produced at the national level (FAO, 2015b). The Malawian government therefore wants to diversify the current low-value, staple-crop-only agricultural portfolio, in order to boost overall productivity and possibly increase exports. Introducing sericulture can help achieve the desired diversification, while holding the promise of providing better livelihoods to rural families. Cultivating silk is labour intensive, requires low skill levels, and silk has had and is expected to have a steady global market for years to come (ITC, 2017). However, sericulture has implications for land and water resource use, both locally for the farmers' operations and for the wider catchment. In this study, we explore the local implications of silk production based on water and land productivity, and we place water footprints in the context of catchment-level water availability. We conclude with a discussion of whether farmers should appropriate local water and land resources to sericulture based on these factors.

5.2. Method and data

5.2.1. The production chain of raw silk

The production chain of raw silk has several steps, each of which may have a water or land footprint associated with it. The total water or land footprint of raw silk is the sum of the respective footprints in each step (Hoekstra et al., 2011). The first step of silk production is the cultivation of mulberry shrubs for their leaves and the rearing of silkworms (*Bombyx Mori*). The leaves serve as feed for the silkworms, which are raised on rearing beds in special nurseries. When the worms reach maturity, they form cocoons, which, once pupation is about to complete, are harvested. After each harvest (4-7 per year), the nurseries have to be thoroughly cleaned to prevent the spread of diseases and promote general hygiene before a new batch of worms is reared (Raina, 2000). The harvested cocoons are stifled to kill the pupae inside without disturbing the structure of the silk shell. This is usually done by means of hot air-conditioning, which is why the process is referred to as drying. After drying, the cocoons are heated in boiling water in order to soften the gummy protein sericin to a point where unravelling of the silk filament is possible. The dry raw silk is then reeled onto bobbins and is ready for further processing, dyeing or direct sale. The processes that require water and land are shown in Figure 5-1. In the case of water use, we distinguish between the green WF, representing the consumptive use of rainwater, and the blue WF, referring to the consumptive use of surface or groundwater (Hoekstra et al., 2011).

5.2.2. Study area

The choice for the case study in Malawi is borne out of an intended sericulture project by a Netherlands-based NGO. This project is to be implemented around three estates and roughly 200 surrounding smallholder farms in the Mzimba District in the Northern Region of Malawi (Figure 5-2). The study area is within the Nyika Plateau catchment, with an elevation of about 1,200 m above mean sea level and temperatures ranging between 9 °C and 30 °C. With an average annual precipitation of 644 mm and an average annual potential evapotranspiration of 1,350 mm, the climate can be classified as subtropical highland variety (Kottek et al., 2006). The wet season starts in November and ends in April, and the dry season is from May to October. The main soil types are sandy loam and silty clay loam. These climate and soil conditions are favourable for mulberry cultivation (Jian et al., 2012). The perennial Runyina River close to the study location is the preferred source of irrigation water.

Smallholder farmers currently grow crops such as tobacco, groundnuts and maize, while the estates mainly grow chillies and paprika. The project intends to replace currently grown crops with mulberry shrubs for silk production on about 20 hectares of the estates, and on half a hectare of each of the smallholder farms.

Figure 5-1. Water and land footprints along the production chain of raw silk.

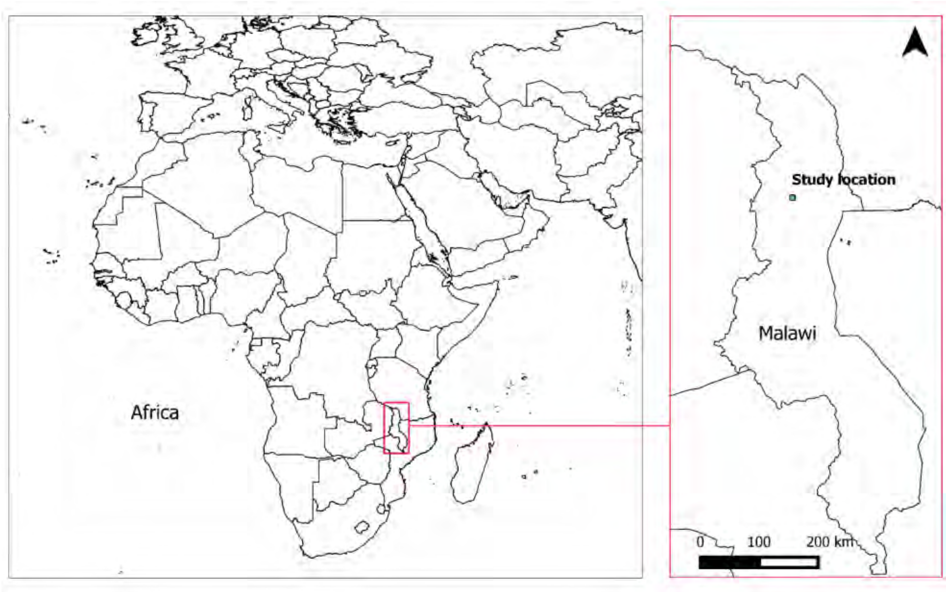

Figure 5-2. Location of the study area where switching from currently grown crops (maize, chillies, paprika, groundnuts, tobacco) to sericulture is being considered.

5.2.3. Calculation of water and land footprints and economic productivities

Water and land footprints were assessed along each step of the production chain of raw silk (Figure 5-1), following the global water footprint standard Hoekstra et al. (2011). To estimate the WF of mulberry cultivation and the currently grown crops (maize, chillies, paprika, groundnut and tobacco), we used the method as in Mekonnen & Hoekstra (2011), but replaced the CropWat model with the more advanced AquaCrop model developed by FAO (Steduto et al., 2009). AquaCrop simulates the daily soil water balance and biomass growth, in order to estimate crop water use and yield. Because mulberry is a perennial crop – and AquaCrop is developed for annuals – we set crop parameters such that AquaCrop mainly simulates canopy development and reflects local (projected) cropping practice. For mulberry shrubs, yield refers to the tons of leaves that can be harvested per year per hectare (note: not to the yield in terms of mulberries). For currently grown crops, simulated yields are scaled based on average local yields in the study area (Figure 5-2). We calculated land footprints (m^2 kg^{-1}) by taking the inverse of the yield, and we distinguished between green and blue WF based on the method described in Chukalla et al. (2015). To account for inter-annual variation in WFs, we simulate crop production for each year in the period 1986-2016. We ignored the blue WF related to energy for pumping water to the fields in case mulberry shrubs are irrigated, because the exact location, setting and types of pumps are not yet decided. We also ignored the grey WF, because of a lack of sensible data and its high dependency on local, actual practices.

We assumed that the leaves represent the full value gained from the mulberry plantation, so no value or WF is attributed to by-products such as berries. Based on estimates from the International Centre of Insect Physiology and Ecology (ICIPE, pers. comm.), we assumed that 187.5 kg of fresh mulberry leaves are needed to harvest 9.1 kg of dry cocoons, which after processing yield 1 kg of dry raw silk.

Data on soil properties are taken from De Lannoy et al. (2014) and local data. We assumed that soil fertility is good and does not hamper crop production. Crop calendars were taken from Chapagain & Hoekstra (2003) and Portmann et al. (2008). Climate data have been taken from global high-resolution datasets by Harris et al. (2014) and Dee et al. (2011). These daily fields – evaluated at the location of the estates – have been scaled such that the monthly averages match monthly fields that were observed locally, at the nearby Bolero climate station.

We evaluated five mulberry cultivation scenarios, in which we compare various irrigation strategies and techniques for growing mulberry shrubs (Table 5-1), to assess the effect of farming practice on WFs and LFs.

Table 5-1. Different scenarios of cultivating mulberry shrubs evaluated in this study.

Scenario	Irrigation strategy	Irrigation technique	Expected effect
Rain-fed	No irrigation	None	Sensitive to climate variability; a dry year leads to lower leaf yields.
Full-furrow	Full irrigation	Furrow	No water stress; optimum yields. High evaporation because large part of soil is wetted.
Full-drip	Full irrigation	Drip	No water stress; optimum yields. Lower evaporation because small part of soil is wetted.
Deficit-drip	Deficit irrigation	Drip	Some water stress, leading to lower yields. Lower evaporation because small part of soil is wetted. Smaller water footprint per ton of leaves.
Deficit-drip-organic mulching	Deficit irrigation	Drip	Some water stress, leading to lower yields. Very low evaporation because of protective organic mulching layer covering the soil. Minimum water footprint per ton of leaves.

The blue WF associated with cleaning, drying, cooking and reeling is highly dependent on local factors and practices. Due to the lack of a credible source, we assumed a water footprint of 100 L per harvest for cleaning the premises and five harvests per year, based on a one-hectare operation and a consumptive fraction of 10%. Generating electricity requires water, which needs to be accounted for (12). Singh (2011) estimates that electricity consumption of cocoon drying is 1.0 kWh per kg cocoons. We assumed a conservative blue WF of the energy mix for Malawi at 400 m^3 TJ^{-1} (or 0.00144 m^3 kWh^{-1}) based on a study by Mekonnen et al. (2015). Kathari et al. (2013) report that – using a multi-end reeling machine – cocoon cooking consumes 57 L of water per kg of raw silk and reeling 100 L per kg of raw silk. We adopted these estimates here as well, since a similar centrally operated multi-end reeling machine is anticipated to be used in the Malawi project. This machine – if wood-powered – requires 2.6 kg of wood per kg of cocoon for the cooking and reeling processes (Astudillo et al., 2014). We calculated the WF related to wood using the average (green) WF of wood in Malawi of 74 m^3 per m^3 of wet round-wood (or 137 L kg^{-1} dry firewood) as determined by Schyns et al. (2017).

However, solar power is the project's preferred source of energy to power the machine. We therefore estimated the blue WF of cooking with solar energy as well, by converting the caloric value of wood into an equivalent amount of solar energy, and multiplying solar energy demand with the blue WF of solar energy of 150 m^3 TJ^{-1} as estimated by Mekonnen et al. (2015). For the lack of a better estimate, the LF of silk processing (for the rearing facilities and equipment storage) is assumed at 100 m^2 per hectare of mulberry shrubs.

We calculated the economic water productivity (EWP, in € m^{-3}) and economic land productivity (ELP, in € m^{-2}) of silk and of the currently grown crops, by dividing the local market price (€ kg^{-1}) by the WF (m^3 kg^{-1}) or LF (m^2 kg^{-1}), respectively.

Finally, we placed the WF in the context of water availability at the catchment level. Due to the lack of local hydrological assessments for the Nyika Plateau catchment, we took data on local water scarcity levels from the high-resolution global study by Mekonnen & Hoekstra (2016) to see if sustainability levels are currently being exceeded. Also, we drew up a hypothetical case based on local precipitation figures to obtain a rough estimate of water availability levels in the catchment.

5.3. Results

5.3.1. The water and land footprint of silk production

The total WF and LF of silk production is a summation of all WFs and LFs along the production chain of silk as shown in Figure 5-1. We summarized all steps into two major components: 1) the WF and LF of silk related to cultivation of mulberry leaves, and 2) the WF and LF of silk related to the silk processing steps of cleaning, drying, cooking and reeling.

5.3.1.1. The water and land footprint of mulberry cultivation

The WF of rain-fed mulberry leaves is 423 m^3 t^{-1} and the LF 820 m^2 t^{-1} – on average over the period 1986–2016 (Table 5-2). The WF is 100% green, because only rainwater stored in the soil is consumed. Since there is no irrigation in this scenario to keep plants from suffering water stress, footprints strongly depend on the prevailing weather conditions in a given year. Temporal variability of both water and land footprints is high, as shown by their respective standard deviations of 169 m^3 t^{-1} and 537 m^2 t^{-1}.

If the mulberry fields are irrigated, the LF of leaf production goes down considerably, to 236 m^2 t^{-1} on average, and the total WF shrinks by at least 25%. The WF associated with full irrigation using the furrow technique is 314 m^3 t^{-1}, and becomes smaller with each improvement in irrigation practice. In the best-practice scenario in terms of water consumption per metric ton of leaves – i.e. deficit irrigation using drip systems while applying a layer of organic mulching – the WF is 254 m^3 t^{-1}. Temporal variability of footprints is much lower than under rain-fed conditions, because the shrubs do not suffer water stress as they do under rain-fed conditions. For example, under full drip irrigation, standard deviations are 19 m^3 t^{-1} and 10 m^2 t^{-1} for WF and LF, respectively. However, the WF does have a blue component in these scenarios.

Footprints expressed per ton of mulberry leaves are converted to footprints per kg of raw silk based on the assumed feed requirement of 187.5 kg of mulberry leaves per kg of final raw silk. Water and land footprints of silk related to mulberry leaf production are listed in Table 5-3. It shows that rain-fed silk has a green water consumption of 79,300 L kg^{-1} and irrigated silk has a total water consumption between 47,500 and 58,900 L kg^{-1}. Land footprints range from 154 m^2 kg^{-1} under rain-fed condition to 44-45 m^2 kg^{-1} under irrigation scenarios.

Table 5-2. Green and blue WF and average, minimum and maximum total WF and LF of mulberry leaf production per metric ton of leaf for five different scenarios. Average WF and LF are production weighted over the period 1986-2016.

Scenario	$WF_{avg,green}$ ($m^3\ t^{-1}$)	$WF_{avg,blue}$ ($m^3\ t^{-1}$)	$WF_{avg,total}$ ($m^3\ t^{-1}$)	WF_{min} ($m^3\ t^{-1}$)	WF_{max} ($m^3\ t^{-1}$)	LF_{avg} ($m^2\ t^{-1}$)	LF_{min} ($m^2\ t^{-1}$)	LF_{max} ($m^2\ t^{-1}$)
Rain-fed	423	0	423	340	1,336	820	532	3704
Full-furrow	117	197	314	278	356	236	217	254
Full-drip	117	180	297	265	339	236	217	254
Deficit-drip	129	142	271	239	308	243	216	278
Deficit-drip-organic mulching	122	132	254	223	288	242	212	279

Table 5-3. Green and blue WF and average, minimum and maximum total WF and LF of raw silk related to mulberry leaf production per kg of raw silk for five different scenarios. Average WF and LF are production weighted over the period 1986-2016.

Scenario	$WF_{avg,green}$ ($L\ kg^{-1}$)	$WF_{avg,blue}$ ($L\ kg^{-1}$)	$WF_{avg,total}$ ($L\ kg^{-1}$)	WF_{min} ($L\ kg^{-1}$)	WF_{max} ($L\ kg^{-1}$)	LF_{avg} ($m^2\ kg^{-1}$)	LF_{min} ($m^2\ kg^{-1}$)	LF_{max} ($m^2\ kg^{-1}$)
Rain-fed	79,300	0	79,300	63,800	250,500	154	100	694
Full-furrow	22,000	37,000	58,900	52,100	66,800	44.2	40.7	47.7
Full-drip	22,000	33,700	55,700	49,600	63,500	44.2	40.7	47.7
Deficit-drip	24,100	26,600	50,800	44,800	57,800	45.6	40.5	52.1
Deficit-drip-organic mulching	22,800	24,800	47,500	41,900	54,000	45.4	39.8	52.2

5.3.1.2. The water and land footprint of cleaning, drying, cooking and reeling

Table 5-4 shows the WF of cleaning, drying, cooking and reeling, which in each process step is fully blue. The reeling process is the major water consuming step, but this is only so if we assume that the multi-end machine runs on solar power. Alternatively, the reeling machines may run on firewood, or small-scale sericulture farmers – who cannot afford a multi-end reeling machine at all – may simply heat water in pots on firewood stoves. The use of firewood profoundly alters the water footprint. While a solar-energy powered silk processing has a total blue WF of 180 L kg^{-1}, using firewood results in a much larger green WF of firewood of over 3,200 L kg^{-1}. The choice of energy source to heat water for cooking therefore has a substantial influence on the total WF of the processing of silk.

The land footprint of the rearing facilities and equipment storage was estimated at 100 m^2 per hectare of mulberry plantation.

Table 5-4. Green, blue and total water footprint (WF) related to cleaning, drying, cooking and reeling per kg of raw silk, assuming water for cooking is heated using solar energy.

Process step	WF_{green} (L kg^{-1})	WF_{blue} (L kg^{-1})	WF_{total} (L kg^{-1})
Cleaning	0	2	2
Drying electricity	0	13	13
Cooking cocoons	0	57	57
Reeling silk	0	100	100
Multi-end machine energy when solar powered	0	8	8
Alternative: multi-end machine energy when wood powered	(3,200)	(0)	(3,200)
Total	*0*	*180*	*180*

5.3.1.3. The total water and land footprint of silk production

The total footprint of raw silk is the sum of the footprint of mulberry leaf production and the footprint of silk processing (Table 5-5). The total WF of silk decreases with each mulberry cultivation scenario, while the blue portion of 62.8% in the full-furrow irrigation scenario decreases to 52.3% in the best-practice scenario of deficit drip irrigation with organic mulching. For each scenario, a full WF split per colour and stage of the production chain is shown in Figure 5-3. We find that the largest parts of both the total LF and WF are the result of the mulberry cultivation component. The LF related to processing is around 1% of the total, while the WF related to processing is 0.2-0.4% of the total.

Table 5-5. Green, blue and total water footprint (WF) and land footprint (LF) of silk under five mulberry cultivation scenarios per kg of raw silk.

Scenario	WF_{green} (L kg^{-1})	WF_{green} (%)	WF_{blue} (L kg^{-1})	WF_{blue} (%)	WF_{total} (L kg^{-1})	LF_{total} (m^2 kg^{-1})
Rain-fed	79,300	99.7	180	0.3	79,500	155
Full-furrow	22,000	37.2	37,200	62.8	59,200	44.7
Full-drip	22,000	39.4	33,900	60.6	55,900	44.7
Deficit-drip	24,100	47.3	26,800	52.7	50,900	46.1
Deficit-drip-organic mulching	22,800	47.7	25,000	52.3	47,800	45.9

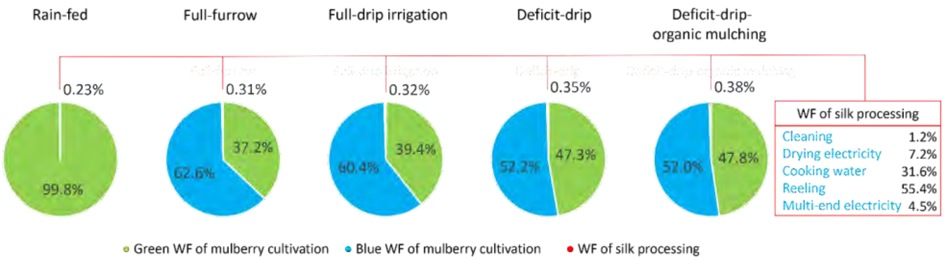

Figure 5-3. The composition of the water footprint (WF) of raw silk, by colour and by production stage, for five mulberry cultivation scenarios.

5.3.2. Economic water and land productivity

Producing one kg of silk requires far more water and land than to produce one kg of the crops currently grown by farmers (Table 5-6). The market price of silk, on the other hand, is much higher than for the other crops. Comparing economic water and land productivities of silk with those of currently grown crops confirms that silk generates more economic value per unit of natural resource used. The average ELP of silk – 0,37 € m^{-2} for the rain-fed scenario and 1,24-1,28 € m^{-2} for the drip irrigation scenarios – is considerably higher than the ELP of currently grown crops – which ranges from 0,04 € m^{-2} for maize to 0,19 € m^{-2} for chillies. The average EWP of silk for the rain-fed scenario, 0,72 € m^{-3}, is much larger than the EWP of maize, groundnuts and tobacco, slightly larger than the EWP of paprika and similar as the EWP of chillies. Under drip irrigation, the EWP of silk is estimated at 1,02 - 1,20 € m^3, which is much higher than for all currently grown rain-fed crops. The large range for the EWP of rain-fed silk (0,23 – 0,89 € m^{-3}) compared with, for example, silk production under full drip irrigation (0,90 - 1,15 € m^{-3}), demonstrates the higher variability of rain-fed versus irrigated production.

EWP and ELP vary with WF and LF, respectively, as well as with changing market prices. With a local estimate of a bottom market price for raw silk of 54 € kg^{-1}, average EWP and ELP of rain-fed silk (the least productive form of silk production) lower to 0,68 € m^{-3} and 0,35 € m^{-2}, respectively. When we would assume a low market price of raw silk of 42 € kg^{-1}, as has been reported in India (Central Silk Board, 2015), EWP and ELP of rain-fed silk would be 0,53 € m^{-3} and 0,27 € m^{-2}, respectively. Under such low silk prices, average water productivities of chillies and paprika – if unchanged themselves – become higher than for rain-fed silk; land productivity of silk remains higher than for currently grown crops regardless such low silk prices. Both average EWP and average ELP of irrigated silk remain higher than those for currently grown crops even under low silk price estimates.

Table 5-6. Economic water productivity (EWP) and land productivity (ELP) for silk under three scenarios, and for five currently grown crops. Minimum and maximum EWP are based on highest and lowest WF over the period 1986-2016, respectively. Silk yields are simulated; yields of current crops and market prices are based on local data.

Crop	WF_{total} (L kg^{-1})	$Yield_{avg}$ (kg ha^{-1})	Market price (€ kg^{-1})	EWP_{min} (€ m^{-3})	EWP_{avg} (€ m^{-3})	EWP_{max} (€ m^{-3})	ELP_{avg} (€ m^{-2})
Silk, rain-fed	79,500	65	€ 57,00	€ 0,23	€ 0,72	€ 0,89	€ 0,37
Silk, full drip irrigation	55,900	226	€ 57,00	€ 0,90	€ 1,02	€ 1,15	€ 1,28
Silk, def. drip irr., organic mulch	47,800	220	€ 57,00	€ 1,05	€ 1,20	€ 1,35	€ 1,24
Maize	2,500	1,500	€ 0,26	€ 0,01	€ 0,10	€ 0,16	€ 0,04
Chilly	3,400	750	€ 2,50	€ 0,42	€ 0,74	€ 0,84	€ 0,19
Paprika	1,900	1,350	€ 1,20	€ 0,36	€ 0,64	€ 0,73	€ 0,16
Groundnuts	3,300	1,250	€ 0,48	€ 0,03	€ 0,15	€ 0,23	€ 0,06
Tobacco	3,300	1,250	€ 1,05	€ 0,00	€ 0,32	€ 0,40	€ 0,13

5.3.3. Macro-level sustainability

Current consumption of blue water resources for agricultural and domestic purposes in the Nyika Plateau watershed is low and remains within sustainable limits for most of the year according to Mekonnen & Hoekstra (2016). Only toward the end of the dry season, in October and November, total blue WFs in the watershed are slightly higher than the volume of water that is sustainably available, potentially causing moderate water scarcity in that part of the year. This estimate is based on the assumption that 80% of runoff is to be reserved to maintain environmental flows. Due to the lack of a reliable catchment-level assessment, no exact sustainability limit could be given. However, small-scale mulberry cultivation in the order of magnitude proposed in the project is not expected to cause water scarcity in the catchment.

To sketch out what would happen if silk production in the area takes off on a larger scale, we considered the following hypothetical case. Based on local data, average rainfall over the period 1986-2016 is 644 mm per year. The Malawi Government estimates the local runoff coefficient at 20% (Malawi Government, 2010); Ghosh & Desai (2006) report a runoff coefficient of 25% for the nearby Rukuru River and 34% for the also nearby Luweya River. We conservatively assume here that 20% of annual precipitation around the study location becomes runoff, and thus becomes a blue water resource. Also, from a precautionary principle, we assume that 80% of this runoff is to remain in rivers and streams to protect riparian ecosystems (Richter et al., 2012). Given these assumptions, local total blue WFs are sustainable as long as they do not exceed about 25 mm per year on average (the macro-level sustainability limit). The blue WF of mulberry shrubs under full drip irrigation is about 750 mm per year. This implies that up to 3.3% of the local watershed area could be used for irrigated mulberry cultivation, before water consumption exceeds 20% of annual runoff potentially and environmental flow requirements are violated. Coverage of the area with irrigated mulberry shrubs beyond this share could lead to moderate water scarcity. In this scenario, we did not consider the blue WF of other activities, such as the presence of other irrigated agriculture. However, we know that the agricultural area equipped for irrigation (in the whole of Malawi) is low, at only 2.3 % of the total (IWMI, 2010). Unfortunately, we could not evaluate locally what flow is sustainably available throughout the year in the Runyina River.

5.4. Discussion

We calculated WFs and LFs of silk and currently grown crops using FAO's AquaCrop model, which yielded several uncertainties. Firstly, AquaCrop is not calibrated for mulberry shrubs nor for local Malawian circumstances. Secondly, although we accounted for variations in time by performing multi-year analyses, the sensitivities of yield and biomass build-up to specific weather conditions in a given year may not be fully captured by the model. Leaf yield also will depend on crop genetic make-up, since different mulberry varieties respond differently to different conditions. Nonetheless, simulated yields were about the same as anticipated yields of mulberry shrubs (ICIPE, pers. comm.).

Another source of uncertainty is the conversion factor of mulberry leaves to raw silk. The estimate of 187.5 kg of leaves to produce 9.1 kg of cocoons and 1 kg of raw silk (as expressed by ICIPE, pers. comm.) is slightly lower than the estimate by Astudillo et al. (2014) of 238 kg leaves per kg raw silk and slightly higher than the 8.6 kg of cocoons per kg of silk by Patil et al. (2009). Any changes in this conversion factor directly translate into changes in the footprints of silk. Literature estimates of water consumption in silk processing also show a spread. For example Kathari et al. (2013) estimate that 100 L of water is needed per kg of raw silk in the reeling process versus 1,000 L by FAO (2003) for the same process. However, since processing hardly contributes to overall footprints, the associated uncertainty is negligible.

There are no other studies to our knowledge that quantify the total WF of silk. Astudillo et al. (2014) estimated the blue WF component of silk in an Indian setting at 54.0 m^3 kg^{-1} and 26.7 m^3 kg^{-1}, for conditions following recommended guidelines and under actual farm practices, respectively. These numbers match our estimates (25.0-37.2 m^3 kg^{-1} for irrigation scenarios), but it has to be noted that climatic conditions are not necessarily comparable among the studies. Karthik & Rathinamoorthy (2017) and Central Silk Board (2015) estimate the LF of silk at 256 m^2 kg^{-1} and 103 m^2 kg^{-1}, respectively. Especially for irrigated scenarios, our estimate is significantly lower (around 45 m^2 kg^{-1}), which can probably be explained by the previously mentioned leaves-to-cocoons-to-silk conversion factors. This provides one more argument to assess thoroughly these conversion factors before embarking on sericulture.

We only considered the green and blue WF of silk production, and not the grey WF related to pollution. Sericulture has more than once been associated with pollution (Raina, 2000; FAO, 2003). Depending on farming practices, such as fertilizer and pesticides application, this component may therefore add to the total WF. Also, chemicals and disinfectants used in the silk processing stages may increase the WF if wastewater is not treated properly before disposal.

Like cotton, silk is a fibre harnessed by the apparel sector, so we thought it relevant to compare the water and land implications of silk versus cotton fibre. The global average WF of cotton of 9,100 L kg^{-1} and LF of 4.2 m^2 kg^{-1} (Hoekstra, 2013) are much lower than those for silk. Silk therefore is not the preferred source of fibre to replace cotton on a large scale. The cotton market price in Malawi estimated by Bisani (2016) is 0,46 € kg^{-1}. Therefore, the economic value of cotton is much lower than that of silk. EWP and ELP of cotton (0,05 € m^{-3} and 0,11 € m^{-2}, respectively), are lower still than their silk equivalents (see Table 5-6). Considering only water and land, this implies that farmers would prefer sericulture over cotton production if they act as rational economic agents.

The same argument goes for the currently grown crops. Land and water requirements of silk – which is a luxury item – are higher than for low-value staple food crops, but the monetary added value per unit of resource is higher still for sericulture. Silk's advantages hold as long as i) market prices for silk remain high; ii) sericulture does not depress local food markets; and iii) total (blue) water consumption does not exceed sustainability limits at the catchment level. The implication is that silk has to remain a marginally produced product, in the case of our study area at no more than ~3% of available land in the catchment area.

Clearly, water and land are not the sole factors a farmer considers in choosing what crop to grow (Chenoweth et al., 2014; Hoekstra, 2017). But footprints and economic productivities – calculated at the local level and placed in the wider environmental context of catchment-level sustainability – proved useful factors in our Malawi case study. It helps farmers to link implications of their crop choice to natural resources use and catchment-level sustainability limits (Herva et al., 2011). Especially the estate owners could thereby – however partially and by no means exhaustively – give substance to their Corporate Social Responsibility (CSR) programs.

5.5. Conclusion

This study set out to explore how data on water and land footprints and economic productivity can inform micro-level decision making on crop choice – in the context of macro-level sustainability of resource use – with a study of proposed silk production in Malawi.

The total WF and LF of silk depend on the farming practices under which mulberry shrubs are cultivated. We found the total WF and LF of silk at the study location ranges from 79,500 L kg^{-1} and 155 m^2 kg^{-1}, respectively, under rain-fed conditions, to 47,800 L kg^{-1} and 45 m^2 kg^{-1} under the best farming practices. Here, best practice entails the use of deficit drip irrigation with organic mulch application. Over 99% of both the WF and LF relates to mulberry leaf production. The rest relates to silk processing, that is cleaning the nurseries, drying and cooking of the cocoons and reeling the silk. The WF of mulberry cultivation is all green in rain-fed agriculture and a mix of green and blue under irrigated conditions. The blue WF makes up 52% to 63% of the total WF, depending on the irrigation strategy and technique. Variability in time is considerably lower in irrigated than in rain-fed agriculture. A more constant silk production is therefore expected under irrigated farming conditions.

The WF and LF of silk are higher than those of currently grown rain-fed crops (maize, groundnuts, chilly, paprika and tobacco) and cotton, but the economic water and land productivities are also higher. Average EWP of silk ranges from 0,72 € m^{-3} (rain-fed conditions) to 1,20 € m^{-3} (deficit drip irrigation with mulching). EWP of cotton is much lower at 0,05 € m^{-3}, and EWPs of currently grown crops range from 0,10 € m^{-3} (maize) to 0,74 € m^{-3} (chilly). Average ELP of silk ranges from 0,37 € m^{-2} (rain-fed conditions) to 1,24

€ m^{-2} (deficit drip irrigation with mulching) and is considerably higher than ELP of the currently grown crops (0,04 - 0,19 € m^{-2}).

The blue WF resulting from the introduction of irrigated mulberry plantations will increase the pressure on blue water resources compared with current rain-fed cropping practices. Current total water footprints in the Nyika Plateau catchment remain below the maximum sustainable footprint during most months of the year; only toward the end of the dry period is a moderate scarcity reported. Therefore, as long as irrigated mulberry cultivation takes place on a relatively small scale – but not exceeding ~3% of the catchment area – no harmful environmental effects are expected.

Sericulture holds the promise of creating agricultural diversity, income and employment for the rural Malawian setting of our study case. Based on our assessment of water and land productivity, we conclude that sericulture is a viable alternative for farmers to currently grown crops – especially if they can irrigate their fields. This conclusion holds as long as prices of silk stay high, production remains marginal, and local food markets are not repressed. We recommend, however, to more closely evaluate both catchment hydrology and mulberry leaves-to-cocoons-to-raw silk conversion factors before a decision to grow silk is made.

With the case study of proposed silk production in Malawi, we have shown how water and land footprints and economic productivity data can be useful to farmers in choosing their crops. Moreover, these indicators provide a means for the farmers to give substance to their Corporate Social Responsibility (CSR) programs. However, final decision making should include considerations of other relevant factors (about seeds, labour, technology, access to markets, capital and so on) for a fully comprehensive assessment.

6

WATER SUSTAINABILITY OF INVESTORS: DEVELOPMENT AND APPLICATION OF AN ASSESSMENT FRAMEWORK

6. Water Sustainability of Investors: Development and Application of an Assessment Framework *

Abstract

Although corporate social responsibility in general and corporate water stewardship specifically are of increasing concern to businesses, investors are lagging behind in fostering water sustainable investment practices – despite the large impact their investment decisions have on the state and shape of tomorrow's water resources. This paper is the first-ever study to assess whether and how investors include water sustainability criteria in their investment decisions, by scrutinizing their publicly released policies on the topic. We hereto i) developed an assessment framework using the water footprint concept, ii) applied it to twenty large investors in a case study for the Netherlands, and iii) ranked them accordingly. We found that, by and large, water sustainability is a blind spot to investors, resulting in disclosed policies being neither well-demarcated nor clearly formulated, especially regarding the supply chain of the activities invested in. There is a long way to go before investors can ensure efficient, sustainable and fair water use in their investment policy, but our framework helps investors direct their urgently needed improvement process, to transition toward water sustainable production systems in a circular economy.

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* This chapter has been published as:
Hogeboom, R.J., Kamphuis, I & Hoekstra, A.Y., 2018. Water Sustainability of Investors: Development and Application of an Assessment Framework. *Journal of Cleaner Production* 202: 642-648.

6.1. Introduction

Sustainable use of the Earth's finite freshwater resources is imperative for future economic development (UN-WWAP, 2017). Without adequate water supply, factories come to a halt, food production hampers, and eventually entire economies falter. However, managing this precious resource wisely is not a commonplace, as is illustrated by the World Economic Forum (WEF) consistently ranking water crises in the top-three of systemic risks posed to the global economy in terms of impact (WEF, 2017). Saliently, before 2010 water did not make it even to their top-twenty. Reaching Sustainable Development Goal (SDG) 6 – ensuring availability and sustainable management of water and sanitation for all – requires a financial injection of at least US$ 2.6 trillion until 2050, part of which inevitably will have to come from private investors (Kolker et al., 2016). Not acting on water sustainability could diminish national growth rates by as much as 6 percent of GDP by 2050 (World Bank, 2016). On the positive side, there are substantial opportunities for companies and investors alike to start and fund (water) sustainable business models (Damania et al., 2017), which can realise large sustainability gains (Sumaila et al., 2017) at even no cost to risk and return (Utz et al., 2015).

Recently, the corporate world started waking to the realisation that improved water management is fundamental for future prosperity and human wellbeing (Roca & Searcy, 2012). Responses and action initiatives include disclosure of water use and pollution (CDP, 2015), identifying water risks (Larson et al., 2012), and striving toward good water stewardship (Kelly, 2014) through corporate certification schemes (AWS, 2017), developing business platforms to share best and emerging practices (CEO Water Mandate, 2017) and suggesting context-based water targets for companies (Pacific Institute, 2017).

Regarding disclosure, CDP releases reports for investors seeking assurance that their investments are well placed to generate favourable returns and avoid value destruction because of negative impacts on water systems (CDP, 2015). In their survey of over 1,000 publicly listed companies, managing US$ 63 trillion in assets, almost two-thirds of respondents reported exposure to substantive water risk, both in their operations (2,400 risks identified) and supply chain (800 risks identified). However, the same survey showed that only 11% of companies publish a company-wide water policy that includes, among other, the setting of performance standards for both direct operations and suppliers. Beyond disclosure, water scarcity and pollution may pose physical, regulatory and reputational risks (Morrison et al., 2010). Companies are setting up ways of dealing with such risks (Larson et al., 2012), partly because of an intrinsic believe action is needed, but also – perhaps more cynically viewed – in an attempt to extend their business' control over water resources or to protect their brand name (Hepworth, 2012). Water stewardship, then, goes a step further, by evaluating water use sustainability across the entire value chain, the formulation of water consumption and pollution reduction targets, an adequate implementation plan, and proper reporting on all of the above (Hoekstra, 2014b).

While companies are thus trying to make current business models more water sustainable or water-proof for the future, to date, the role of investors is underexposed (Vörösmarty et al., 2018). Moreover, there are signs that the role of investors is modest at best, as they are lagging behind in fostering and facilitating more (water) sustainable business practices (Busch et al., 2016).

Despite these observations, a host of initiatives to help and guide investors to become more sustainable emerged over the past decade, each with its own approach, method and definitions. For example, Responsible Investment (RI) – or Sustainable Responsible Investment (SRI) – aims to integrate environmental, social and governance (ESG) factors into investment decisions (Eurosif, 2016). Such factors are often incorporated to a limited extent, due to lack of trustworthy ESG indicators and data (Busch et al., 2016), and varying collective beliefs on what RI ought to entail (Louche & Dumas, 2016). Investors may also incorporate ESG or similar criteria through Corporate Social Responsibility (CSR) programmes (see Carroll & Shabana (2010) for a review and Oh et al. (2013) for two case studies); by carrying out triple bottom line assessments (Norman & MacDonald, 2004); by adhering to the United Nations' initiated Principles of Responsible Investment (UN PRI, 2017); or by becoming certified by the Alliance for Water Stewardship (AWS, 2017). Topics encompass various sustainability domains, ranging from child labour to carbon emissions, and from deforestation to human rights. Although the long lists of both initiatives and topics suggests that sustainability is high on investors' agendas, the gist of recent literature reveals investor progress on water sustainability is skin deep at best, hindered primarily by a lack of perceived urgency about, a shared taxonomy on and meaningful indicators regarding water sustainability of their investments.

The ill-addressing of water sustainability by investors is particularly alarming in regard to future water issues, since the economy of tomorrow is shaped to a large extent by choices made by investors today. After all, investments today in new or updated farms, firms and factories will have ramifications for future water resource use and pollution. Failure by investors to transition from business-as-usual to more sustainable water practices implies water resources will continue to be further depleted, polluted and needlessly wasted, while prolonged inconsideration of sharing water resources fairly among users increases the likelihood of conflicts. Current investment practices thus sustain unsustainable water management (Lambooy, 2011), while rendering sustainable production systems in a circular economy a utopian vista.

This paper aims to contribute to the slim body of knowledge on how investors relate to water concerns, by investigating whether and how investors currently include water sustainability criteria in their investment decisions. Ideally, the facts on the ground are assessed, namely the water sustainability of actually invested-in projects or portfolios. Since such ground-truthing is an unreachable goal because of the inaccessibility of relevant data, we resorted to drawing on policy documents released to the public to make the assessment instead.

We developed and applied a framework to assess these policies of investors regarding their incorporation of water sustainability criteria. The application is done for a case study including twenty large investors - banks, pension funds and insurers – in the Netherlands. The assessment of investors' policies is concluded with a ranking of the investigated investors, to distinguish leaders and frontrunners from followers and stragglers, and to incentivise investors to improve their business practice with regards to water resources.

Although the focus is on Dutch (institutional) investors, the scientific soundness and comprehensiveness of the proposed framework render it suitable for wider application. Investors form a major actor group that is being overlooked in contemporary water management discourse. This paper provides a first and timely attempt to systematically address the role investors play in contributing to sustainable water use. By bridging the worlds of investors and water managers, this study combines perspectives for mutual learning.

6.2. Method and Data

The method consists of three main parts: i) the definition of water sustainability in this study, ii) the development of a framework to assess investors' investment policies on inclusion of water sustainability criteria, and iii) the procedure of applying this framework to twenty Dutch investors.

6.2.1. Water sustainability defined

In this study, water sustainability is defined along three dimensions of sustainability – efficient allocation, sustainable scale and equitable distribution – as first proposed by Daly (1992), and refined and tailored to water by Hoekstra & Wiedmann (2014). These three dimensions, encompassing economic, environmental and social concerns, can be operationalized using the water footprint (WF), a temporally and spatially explicit indicator of water consumption and pollution (Hoekstra et al., 2011). The green WF represents rainwater consumption of a human activity, the blue WF refers to surface water and groundwater consumption, and the grey WF provides a measure of water pollution. Using WF tools and these three dimensions, connections between water use, economic development, business practice, and social and environmental risks can be better understood (Herva et al., 2011). The dimension of efficient allocation of water can be operationalized by formulating water footprint benchmarks per product or process; sustainable scale by defining a water footprint cap per river basin; and equitable distribution by defining fair water footprint shares per community (Hoekstra, 2013). These dimensions have to be assessed in both the direct operations of any prospective investment and its supply chain, since in many cases direct water use comprises only a fraction of supply chain water consumption (Linneman et al., 2015).

6.2.2. Framework development

A framework was developed to assess how investors' investment policies incorporate aspects relevant to sustainable use of water. The framework is inspired by that of Linneman et al. (2015), who developed a framework to assess water transparency of stock-listed companies (rather than water sustainability of institutional investors). The framework consists of nine categories – labelled A to I and shown in Figure 6-1 – which collectively cover criteria relevant to water use and pollution associated with prospective investments. Each category contains three to seven equally weighted assessment criteria, which are formulated as closed questions. Answers to the questions result in a score of zero to two or three points per question, where a zero-point score indicates the investor does not consider the criterion at all, and the maximum score – two or three points, depending on the question – implies the investor considers the criterion to its best capacity. We roughly followed a progressive scoring approach, meaning that points can be scored fairly easily on the first question within a category (e.g. testing if the investor is aware of the particular criterion at all, without further qualifications), but that scoring on subsequent questions becomes increasingly difficult (e.g. questioning that goes beyond mere awareness, testing if specific metrics are employed and/or evaluated against set targets).

In order to aggregate criterion scores to the category level, the sum of the points scored on each question within the category is taken. The resulting total category score in points is subsequently expressed as a percentage: a total of 0 points corresponds to a 0% category score, while all points scored on the category questions translates to a 100% score. This is done to avoid that categories with more questions attain a higher weight. Thus, the final investor score on water sustainability is calculated as the average percentage score over all nine categories, where each category contributes an equal weight of 11.1%.

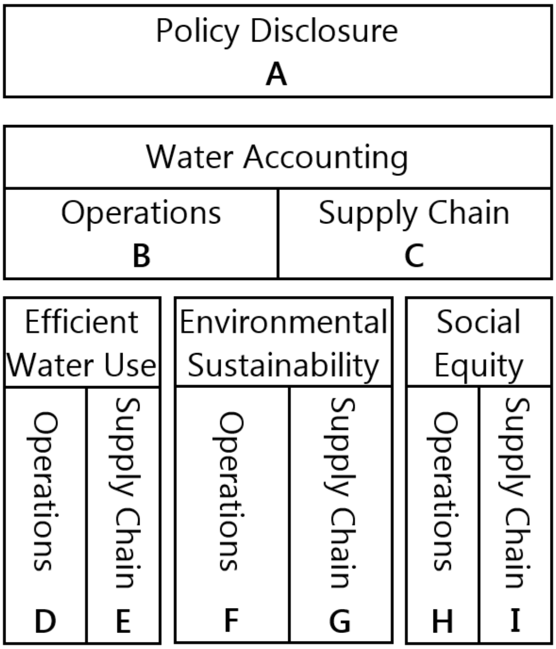

Figure 6-1. Assessment framework with nine categories.

Category A on "Policy Disclosure" assesses the transparency of an investor regarding sustainability issues in its investment policy in general and water sustainability specifically. Although reporting is not the same as actually acting in a sustainable manner, disclosure facilitates the enfolding of a debate on water, and allows the general public to hold the investor accountable. This category is prerequisite, since without disclosure scores cannot be assigned to remaining categories. Questions range from whether the investor discloses information on sustainability in general at all, to whether the investor reports on following certain sustainability guidelines, or frameworks or directives regarding water sustainability.

Categories B and C on "Water Accounting" cover the quantification (measuring and monitoring) of water consumption and pollution, both in direct operations and in the supply chain of the activity created by the proposed investment. Accounting water use and pollution is imperative in any complete assessment of water resources, because water use or pollution reduction targets can only be formulated once direct and indirect claims to freshwater are known; you cannot manage what you do not measure. Water accounts in and by themselves do not provide a comprehensive indication of sustainable water practice; rather, such accounts serve as a basis for the remaining categories, which relate to three dimensions of wise water use (Hoekstra & Wiedmann, 2014).

Categories D and E on "Efficient Water Use" concern the efficient use of water resources by the activity emerging from the prospective investment. Criteria range from simply showing awareness of the notion of water efficiency in direct operations of an investment, to adopting benchmarking procedures in the supply chain and setting reasonable water use and pollution targets against which to compare water use and pollution caused by prospective investment activities.

Categories F and G on "Environmental Sustainability" put the water use and pollution resulting from an investment (as quantified in categories B and C) in the context of locally available water resources – both at the location(s) of the direct operations and at the locations where supply chain activities will take place. Questions probe whether the investor considers potential water scarcity issues, such as violation of environmental flow requirement in the basin(s) where the activities are planned, and how it anticipates on resolving these issues through response strategies.

The last categories H and I on "Social Equity" cover an investor's awareness of and response to social equity concerns that may result from the water use and pollution that will come along with the activity targeted by the investment. Of interest are both community concerns in the place(s) of the activity itself and community concerns in the locations of the supply chain of the activity.

The questions in each category and a format for how points are assigned to each question can be found in Hogeboom et al. (2018a).

6.2.3. Framework application and ranking

The assessment framework is applied to the eighteen largest investors in the Netherlands based on their asset value in 2016. Asset values, viz. the worth of the investor, serve as a proxy for assets under management of the investor (i.e. the total market value of assets an investment company manages on behalf of its clients). Although the latter is the preferred indicator of an investor's size, no unambiguous data could be retrieved for this parameter. Asset values for 2016 are taken from the Dutch National Bank (DNB, 2016). Since hardly any published policy briefs by private investors are available to the public, selection eligibility was limited to banks, insurance companies and pension funds.

Triodos Bank and ASN Bank were added to the subset of the eighteen largest investors, on the basis of their perceived leading role in sustainable investing and the accompanying expectation that we might uncover examples of best practices by including them. Although ASN Bank is a brand in the SNS Bank holding, it has its own policies and operates independently. The resulting selection of twenty investors includes nine banks, five insurance companies, and six pension funds, as listed in Table 6-1.

To assign scores to the criteria of the framework, we solely relied on publicly disclosed information on an investor's investment policy. We analysed information published or referred to by the investor itself on its official websites, in both English and Dutch languages, and on multiple webpage domains if applicable. All webpages and (policy) documents were used which contained or pointed to information regarding investment policy, water, sustainability, and other relevant search terms such as Corporate Social Responsibility, Environment Social and Governance factors, Responsible Investing, Triple Bottom Line, and People Planet Profit. For the selected twenty investors, we scrutinized 44 unique websites and 226 relevant documents published on or linked to by these websites. Webpages and documents thus found and analysed are listed in the supplementary materials to Hogeboom et al. (2018a).

The final investor score on water sustainability directly determines the ranking of the selected investors.

Table 6-1. The twenty selected Dutch investors and their asset values in 2016 as taken from DNB (2016).

Investor Name	Investor Type	Assets (EUR billion)
ING Bank	bank	886
Rabobank	bank	687
ABN AMRO Bank	bank	416
ABP	pension fund	384
PFZW	pension fund	186
BNG Bank	bank	163
NWB Bank	bank	104
Nationale-Nederlanden	insurance company	94
Aegon Levensverzekering	insurance company	75
Shell Pensioenfonds	pension fund	68
PMT	pension fund	66
SNS Bank	bank	64[1]
bpfBOUW	pension fund	57
REAAL	insurance company	56
Achmea Verzekeringen	insurance company	52
PME	pension fund	43
ASR Levensverzekering	insurance company	42
NIBC Direct	bank	24
ASN Bank	bank	11[2]
Triodos Bank	bank	9

[1]*Including ASN Bank assets.*
[2]*Assets under management in 2016 (ASN Bank, 2016).*

6.3. Results

The ranking of the twenty selected Dutch investors on how well they incorporate water sustainability criteria in their investment policy is shown in Figure 6-2. The colours represent the various categories of the assessment framework. Table 6-2 provides a more detailed scoring overview, containing percentage scores per investor per category.

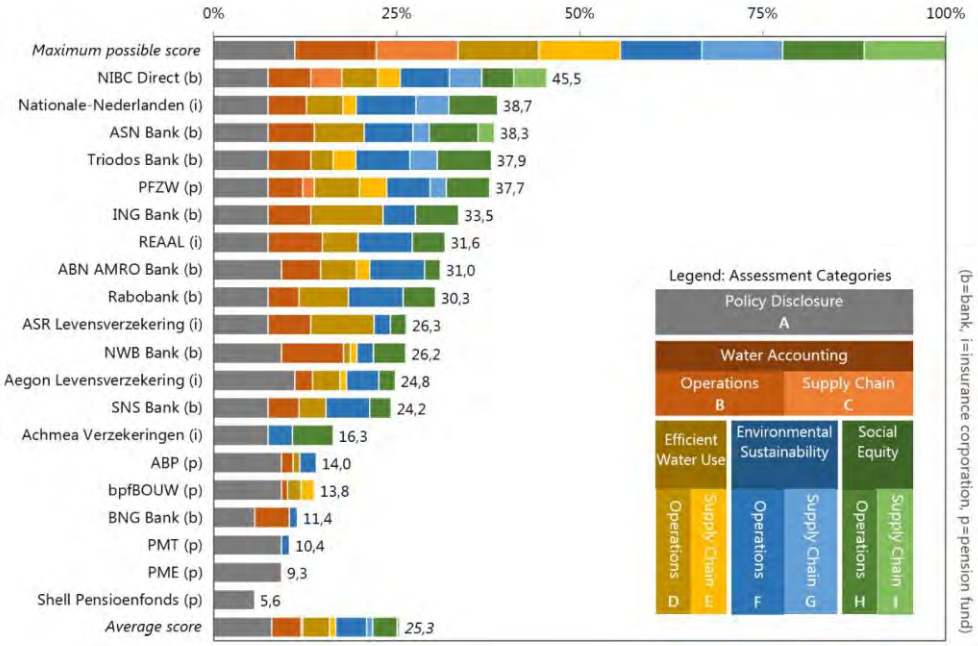

Figure 6-2. Ranking of selected large Dutch investors on incorporation of water sustainability criteria in their investment policy.

The top-three highest ranked investors comprises NIBC Direct (46%), Nationale-Nederlanden (39%) and ASN Bank (38%). The following excerpt from NIBC Direct's policy illustrates its awareness of the various dimensions related to water sustainability:

"At NIBC, we recognize that we operate in a complex world, where climate change, water scarcity, biodiversity loss and population growth create significant sustainability challenges and unprecedented pressures on natural and human systems. The increasing demand for - and scarcity of - resources may lead and has led to conflicts, political and economic instability. We are committed to take environmental criteria into consideration in our business activities, including protection and conservation of biodiversity and maintaining the benefits of ecosystem services. (...) In addition to the risks and standards mentioned in our Sustainability Policy and sector specific policies, NIBC considers the following:

- *Impacts on natural resources and ecosystem services;*
- *Pollution to air, water, and land resulting from the client's operations (land or ground water); (...)*

- *Environmental impact assessments and taking appropriate measures to manage environmental impacts, including policies, management systems, or supply chain criteria."* NIBC Direct (2017).

The scores show that typically investors score highest in category A "Policy Disclosure", with an average score across investors of 70%. Of the remaining categories, those concerning direct operations yield substantially higher scores (average scores category B: 36%; D: 33%, F: 38%; H: 30%) than the categories assessing supply chains (average scores category C: 3%; E: 8%, G: 8%; I: 3%). All individual investors score higher in operations than supply chain categories as well. While thirteen out of twenty investors score points in all four operations categories, only one (NIBC Direct) scores points in all four supply chain categories. Moreover, eleven out of twenty investors do not score any points in the supply chain categories at all. The scores indicate that generally, the role of the supply chain in water sustainability receives little to no attention in investment policy.

Regarding the type of investor, the pension funds score lower than banks and insurance companies. Five out of six pension funds are ranked in the bottom-six of twenty investors investigated. There appears to be no noteworthy difference between the scores of banks and insurance companies. Regarding the asset size of investors, no pattern emerges for the results either. Both smaller and larger investors are ranked in the top-five, as well as in the bottom-five.

Table 6-2. Overview of scores per assessment category.

Investor Name	A (%)	B (%)	C (%)	D (%)	E (%)	F (%)	G (%)	H (%)	I (%)	Total (%)
ABN AMRO Bank	83.3	47.6	0.0	44.4	16.7	66.7	0.0	20.0	0.0	31.0
ABP	83.3	14.3	0.0	8.3	0.0	20.0	0.0	0.0	0.0	14.0
Achmea Verzekeringen	66.7	0.0	0.0	0.0	0.0	30.0	0.0	50.0	0.0	16.3
Aegon Levensverzekering	100.0	21.4	0.0	33.3	8.3	40.0	0.0	20.0	0.0	24.8
ASN Bank	66.7	57.1	0.0	61.1	0.0	60.0	20.0	60.0	20.0	38.3
ASR Levensverzekering	66.7	52.4	0.0	77.8	0.0	20.0	0.0	20.0	0.0	26.3
BNG Bank	50.0	42.9	0.0	0.0	0.0	10.0	0.0	0.0	0.0	11.4
bpfBOUW	83.3	7.1	0.0	16.7	16.7	0.0	0.0	0.0	0.0	13.8
ING Bank	66.7	52.4	0.0	88.9	0.0	40.0	0.0	53.3	0.0	33.5
Nationale-Nederlanden	66.7	47.6	0.0	44.4	16.7	73.3	40.0	60.0	0.0	38.7
NIBC Direct	66.7	52.4	38.1	44.4	27.8	60.0	40.0	40.0	40.0	45.5
NWB Bank	83.3	76.2	0.0	8.3	8.3	20.0	0.0	40.0	0.0	26.2
PFZW	66.7	42.9	14.3	55.6	33.3	53.3	20.0	53.3	0.0	37.7
PME	83.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	9.3
PMT	83.3	0.0	0.0	0.0	0.0	10.0	0.0	0.0	0.0	10.4
Rabobank	66.7	38.1	0.0	61.1	0.0	66.7	0.0	40.0	0.0	30.3
REAAL	66.7	66.7	0.0	44.4	0.0	66.7	0.0	40.0	0.0	31.6
Shell Pensioenfonds	50.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.6
SNS Bank	66.7	38.1	0.0	33.3	0.0	53.3	0.0	26.7	0.0	24.2
Triodos Bank	66.7	52.4	0.0	27.8	27.8	66.7	33.3	66.7	0.0	37.9
Maximum Possible Score	100	100	100	100	100	100	100	100	100	100
Lowest Score	50.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.6
Highest Score	100.0	76.2	38.1	88.9	33.3	73.3	40.0	66.7	40.0	45.5
Average Score	*71.7*	*35.5*	*2.6*	*32.5*	*7.8*	*37.8*	*7.7*	*29.5*	*3.0*	*25.3*

6.4. Discussion

All assessed investors in some form express a willingness to contribute to a more sustainable world by adopting or supporting sustainability guidelines, frameworks or principles. All twenty investors, for example, are signatories of the United Nations' initiated Principles for Responsible Investment (UN PRI, 2017). However, in analysing policy documents to assign scores to water sustainability criteria, it was found that their good intentions did not trickle down to effectuate clear water policy. In many cases, policy is formulated in general, ambiguous, superficial or even meaningless terms (cf Scholtens (2014)). For example, eleven out of twenty investors state water considerations – in its widest sense – influence their investment decisions, but without bothering to give any further explanation. Alternatively, some (mainly low scoring) investors publish several policy documents amply stressing the importance of sustainability, but do not specify the role of water. Shell Pensioenfonds, for example, only mentions the word 'water' twice in all sustainability documents investigated, and PME thrice. Apparently, water does not have a place in their perspective on sustainability. Another example is from SNS Bank's holding company De Volksbank, which mentions the word 'water' 32 times in its 'Carbon Profit and Loss Methodology' – more than in all of their other documents combined – but only in the context of translating water consumption to CO_2-equivalents (De Volksbank, 2016). Such attention for detail regarding CO_2 is absent in their water policies, and so is a definition or explanation of how exactly they account for water use. In addition, many investors mention examples of a reduction in water consumption of their investments in absolute terms, conveying the impression of sustainable business practice. However, without any further explanation or comparison, such statements are meaningless. Only once consumption is considered per unit of output and compared to benchmark values, meaningful sustainable water practice can be claimed. ASR Levensverzekering's real estate branch ASR Vastgoed provides an example of good practice in this regard. In assessing a prospective investment,

"*ASR Vastgoed performs a BREEAM assessment, which uses recognized measures of performance set against established benchmarks, to evaluate (...) a broad range of categories and criteria. They include aspects related to (...) water use, (...), and pollution.*" (UN PRI, 2015).

The less exemplary findings, however, confirm a study by Daub (2007) of a decade ago, stating that disclosed documents may provide only superficial information, using interchangeable terminology and even leading to a value of the disclosed information that “tends to hover around zero”.

Another finding that emerged from the assessment is that water policy is often fragmented and lacking coherence. For all investors with an above-average ranking, the scores are based on at least three and up to ten different documents or webpages. No investor received its total score solely based on a single document. Even if dedicated water chapters were available, additional points could still be assigned based on other documents. Because of the fragmentation of relevant information on water sustainability, it proved cumbersome to isolate or define a coherent water policy for most investors.

Related to the fragmentation is the ambiguity surrounding who is responsible when an investor outsources the management of its assets, especially if both asset manager and investor have their own, potentially conflicting policies about dealing with water sustainability. A similar confusion arises when an investor is a subsidiary of a larger holding company, with similarly potential discrepancies in individual policies. Even in absence of such institutional difficulties, is was often unclear which documents should be leading, if fragmented documents contradicted each other. As a rule, scores were assigned based on the most favorable documents. That being said, some investors have dedicated ‘green’ or special focus funds with stricter policies than the parent company or other funds held by the investor. In such cases, we let the policy representative of the majority of the investors’ activities be guiding in assigning scores. In any case, the complexity of the investors’ organizational set-up should not inhibit sustainable water practice (Lagoarde-Segot & Paranque, 2018).

Some investors include water aspects in their policy not assessed by our framework, such as flooding, sea level rise, biodiversity and hydroelectricity. These were not the focus of our resources-based assessment, but relevant nonetheless to different sustainability contexts.

While the focus of this study is the investor’s consideration of water sustainability criteria in prospective investments, some investors report on water use of their own in-house operations. Rabobank (Rabobank, 2015) and Aegon (Aegon Asset Management, 2016), for example, both explicitly account for water use in their offices. However, the arguably much larger water use by their investments is largely left unnoticed.

The scoring of investors is based disclosed policies rather than on actual ground-truthed performance, which limits the interpretation of our resulting water sustainability.

Moreover, in addition to disclosed polies investors may comply with internal, possibly confidential procedures. Aegon, for example, hints to the existence of internal procedures in an informal Q&A interview format presented in its Responsible Investment Report 2016 (Aegon Asset Management, 2016). If included in our assessment, these internal documents, might have given rise to higher scores than assigned in this study. On the other hand, publicly available documents might keep up appearances of actual investment practice, indicating too high scores might have been assigned.

The ranking of investors based on the assessment framework is subjective to some degree. Firstly in the composition of the framework itself, but also in the weighting of categories and the scoring within each category. While the former is an inherent design consequence, for the latter two, a closer look at the distribution of points scored over the nine categories shows that investors with a high total score received this score based on points assigned in multiple categories. No investor that scores high points in operations categories outranks investors with fewer points spread over both operations and supply chain categories. We would not expect differently, since typically investors start considering supply chains only after covering water criteria in direct operations. A change in weighting may therefore alter the absolute scores, but will affect the ranking only to a limited extent.

Future research may refine, test and build upon our framework and findings, especially in capturing and describing intricacies related to which policies are guiding in practice, how these policies are applied, and the actual ground-truthing of water sustainability of activities invested in. This may for instance be done in a case study setting, where collaboration is sought with selected investors and their local investees.

Few rankings of Dutch investors are available for comparison. In the Fair Finance Guide (FFG) ranking of general sustainability of Dutch financial institutions, ASN Bank, SNS Bank and Triodos Bank score high points, followed at a distance by NIBC Direct, in the category 'Climate Change' (Brink et al., 2016a). In FFG's insurers subsection, Nationale-Nederlanden is in the bottom of the list (Brink et al., 2016b), while they lead the water sustainability ranking in our study. The main reason for the difference in rankings appears to be the main focus of FFG on CO_2 reduction measures, while we confined ourselves to water criteria.

Although the framework is applied to Dutch banks, pension funds and insurance companies, it can readily be used for other countries and types of investors as well. The framework provides a reference for the inception of new investment policy on incorporating water sustainability criteria, or for uptake in Corporate Social Responsibility practice or ESG frameworks (Peiró-Signes et al., 2013). Given the plethora of sustainability frameworks, principles and standards available, and the resulting dilution of focus (Krajnc & Glavič, 2005), the water sustainability criteria of our framework are preferably integrated with existing efforts on developing common taxonomies, such as e.g. those by the Pacific Institute (2017).

Since certain sectors, such as the Food and Beverage industry and Mining, are more water-intensive or susceptible to water risks – in either their operations or supply chain – we recommend investors to start implementing water sustainability criteria specifically in these sectors (Rueda et al., 2017). Some investors, such as ABN AMRO (ABN AMRO, 2017) and Aegon Levensverzekering (Aegon Asset Management, 2016), already formulate their Responsible Investment policy, including water aspects, sector-specific.

6.5. Conclusion

The state of water resources in the future greatly depends on the extent to which investors today include water criteria in their investment decisions. This study set out to find out how investors include water sustainability criteria in their investment decisions, thereby contributing to the body of knowledge on how the currently overlooked and under-addressed actor group of investors relate to water concerns.

The main conclusion is that despite their expressed good intentions, the low total scores of even the highest scoring investors in the Netherlands (<46%) indicate their ambitions have not trickled down (yet) to effectuate clear water policy. Investors score points on disclosure and reporting (the first, readily doable step), but that in itself does not guarantee actual water sustainable investment practice. Assessing numerous policy documents revealed that, by and large, disclosed policy on water sustainability is neither well-demarcated nor clearly formulated. This confirms earlier observations by Scholtens (2014), Lambooy (2011) and Daub (2007), and bolster skepticism about an imminent and prompt transition to water sustainable production systems in a circular economy. That being said, preliminary but promising water policies, such as that of frontrunner NIBC Direct, or a sector-specific approach as pursued by ABN AMRO and Aegon Levensverzekering, lead the path in the right direction and deserve recognition.

The practice of accounting for water use and pollution in both operations and supply chains of the activities emerging from prospective investments is imperative, but we found that especially the supply chain part of the value chain is being overlooked by most investors. In addition, low scores in Categories D – I (Table 2) show that investors are still a long way from guaranteeing that their prospective investments i) safeguard efficient water use, ii) fit a sustainable scale, and iii) ensure a fair sharing of limited water supplies.

Moreover, we discovered how a lack of mutual understanding and exchange between the investor community and the water management community inhibits the development of sound, practical and science-based water investment policy. Concurrently, increased collaboration may hold the linchpin for developing water sustainable investment practices. The assessment framework developed and applied in this study is a first attempt to straddle the gap by providing investors with systematic handles to give substance to their own improvement process to incorporate water criteria into investment decisions. In the end, the purpose of assessing and ranking investors is to incentivise them to improve.

In light of the severity of water issues faced today, we stress the urgency to take action, both specifically toward incorporating water criteria into investment decisions and generally toward more water sustainable economies.

7

CONCLUSION

7. Conclusion

This research set out to feed the scientific discourse on sustainable and efficient consumption of freshwater resources worldwide. I investigated various avenues on how the field of Water Footprint Assessment can advance from water footprint accounting to formulating quantified water footprint reduction targets. In particular, I zoomed in on two promising policy instruments, i.e. setting water footprint caps at the river basin level and formulating water footprint benchmarks for water-using activities.

7.1. Setting water footprint caps per river basin

The world's water resources are shared by humans and nature. If humanity is to prevent overshoot of limited natural endowments, to stay within ecological boundaries and to reconcile human freshwater appropriation with conservation, blue WFs will have to be reduced in about half the world's basins (Hoekstra et al., 2012). The notion of setting a quantified target that caps the WF in a river basin at an ecologically sustainable level has evident merit in both its necessity and clarity of concept.

Based on the global assessment of **Chapter 2**, I found that setting WF caps at various historical blue water availability statistics calls for seeking a compromise between having incidental underutilization of the potential of sustainable water consumption on the one hand, and accepting incidental violations of environmental flow requirements on the other. This trade-off is particularly pronounced in basins with a high seasonal and high inter-annual variability. Consequently, severely stressed basins that experience highly variable water availability will more often be faced with this trade-off under a capped regime compared to moderately stressed basins with less variability. At the same time, it is particularly this first category of both variable and scarce basins that should feel the strongest imperative to reduce their WF.

While this work on WF caps unveiled the intrinsic difficulty of dealing with variability toward any practical policy arrangement, it is the first-ever study to quantitatively explore the concept of setting WF caps per river basin, on a global scale. The study thereby enriched the debate on how freshwater consumption worldwide can be achieved in a sustainable way.

7.2. Formulating water footprint benchmarks for water-using activities

The global food system is a major driver of depletion of freshwater resources, with agriculture accounting for 92% of humanity's WF (Hoekstra & Mekonnen, 2012). In the global study presented in **Chapter 3**, I found that substantial efficiency gains are possible in agriculture by meeting WF benchmarks in crop production. I identified both consumptive (green plus blue) and specific blue WF benchmarks, differentiated per climate zone, for the world's major crops. These crops collectively constitute the lion's share of global food production and of global water consumption. If crop WFs would be reduced to consumptive and blue WF benchmark levels of the best-25th percentile of production, respectively, global annual average consumptive and blue water savings are 44% and 31% compared to the reference consumption. Moreover, I discovered 89% of this blue water saving can be achieved in water-scarce areas.

Although agriculture assumes the largest share of humanity's WF, water consumption from reservoirs had not been included in blue WF studies. In **Chapter 4**, I found reservoirs can consume large amounts of water through evaporation from their surfaces, amounting to 66×10^9 m^3 y^{-1} globally. Extrapolating to reservoirs missing in this study, man-made reservoirs may thus add another ~25% to humanity's blue WF (Hoekstra & Mekonnen, 2012). I attributed the blue WF of reservoirs to its various purposes (including irrigation and hydroelectricity generation) based on their economic value, to find substantial spatiotemporal variability in resulting WFs. For example, while I estimated that globally and on average a GJ of hydroelectricity requires 14.6 m^3, this value varies from 0.1 m^3 GJ^{-1} to 207.1 m^3 GJ^{-1} depending on the location, purpose and shape of the reservoir. While WFs towards the higher end of the estimated spectrum of each purpose might be labelled 'inefficient', the study also revealed that the majority of reservoir-related WFs (57%) – and particularly those with hydroelectricity generation as a purpose – are located in water-abundant basins. When planning reservoirs, it is crucial yet today not always explicitly included, to estimate their blue WF and include this factor in the decision to construct them or not. Once constructed, there are few ways to reduce evaporation else than covering the water with for instance solar panels or shade balls, which may have its pros and cons (Haghighi et al., 2018).

In **Chapter 5**, I moved from a global to a local perspective and zoomed in on a case study on silk production in Malawi. I calculated WFs associated with various farm management systems (cf Chukalla et al. (2015)). Here, the benchmark is considered as the WF of the most water-efficient farm management system – in the Malawi case deficit drip irrigation with organic mulch application. However, the resulting blue WF of silk (in m^3 ha^{-1}) was found to be considerably higher than that of currently grown (rainfed) crops. Producing silk would therefore increase pressure on local blue water resources if it were to be produced an a large scale. As long as silk production remains marginal, the added blue water consumption in the case study catchment will not exceed the maximum sustainable footprint. It does show, however, that water-efficient production at the micro-level may discord with maintaining a sustainable scale of water consumption at the macro-level.

WF benchmarks – be it for crops, reservoir purposes or other water-using activities – provide a quantified reference target against which to measure water productivity. Formulating feasible benchmark levels and striving to meet these benchmarks has the potential to save substantial amounts of water and achieve efficient consumption of freshwater both locally and globally. If savings made through benchmarking are kept in or returned to the water system, policies that encourage adoption of WF benchmarks may not only boost efficient water consumption, but also help alleviate water scarcity, as Chapter 3 showed. If, on the other hand, savings are used to increase production in already stressed regions – efficient as the supplemental production may be – these regions remain stressed and sustainable consumption at larger spatial scales may be compromised (Grafton et al., 2018; Hoekstra, 2013). Trade-offs between meeting efficiency and sustainability targets should therefore explicitly be addressed in developing policies aimed at promoting the transition towards both efficient and sustainable consumption of freshwater.

7.3. Sustainable and efficient water consumption by investors

Establishing and maintaining sustainable and efficient water consumption is a shared responsibility, where a role is to be played by each of the actors involved (UN-WWAP, 2019; Hoekstra, 2013). In Chapter 6, I developed a framework applicable to investors and assessed to what extent investors incorporate water sustainability targets in their investment decisions. The study revealed that concerns over widespread water scarcity and inefficient water use are largely invisible to investors. While they expressed due concern, I found investors' good intentions did not trickle down to effectuate clear water criteria to be included in investment decisions that can meaningfully abate unsustainable and inefficient water use. These findings sketch a disturbing prospect about a powerful actor group. After all, investment decisions made in investors' board rooms today profoundly influence the state and shape of tomorrow's water resources. To ignite their improvement process, the assessment framework developed and applied in this study provides investors with systematic handles to give substance to their amelioration efforts.

7.4. Future research

7.4.1. WF caps

Regarding setting WF caps per river basin, several concerns have not or not thoroughly enough been addressed in the study in Chapter 2, that warrant additional scrutiny in future research.

First, both the Global Hydrology models and the EFR-methods I used to quantify WF caps – despite their state-of-the-art connotation – yielded considerable spread in outputs (for both natural runoff and environmental flow estimates). This model output uncertainty propagates into quantification of the WF caps. While resulting WF cap values sufficed for this explorative research, I recommend that any policy attempt to set a cap be based on local or validated data on runoff and environmental flow requirements. In addition, further research may buttress practical uptake by exploring whether a dynamic WF cap regime can be developed, whereby monthly caps partially depend on runoff forecasts.

Notable omissions and limitations that deserve additional consideration include: separate dynamics and interlinkages between groundwater and surface water potentially undermining setting a 'lumped' blue WF cap; sub-basin spatial distribution of water availability and the appropriate spatial scale of setting a WF cap; effects of man-made alterations to the water system (e.g reservoir storage and operation rules) that affect basin hydrographs; additional implications (e.g. economic, social or environmental) of setting a cap; institutional arrangements for practical uptake, particularly relating to transboundary basins and issues regarding equitable distribution of water.

Last, while the study found that on a global level humanity's annual WF probably does not exceed the 'planetary' cap (yet), scarcity does occur at finer spatiotemporal scales, in particular months in particular basins. To reveal when and where consumption overshoots sustainable supply, actual monthly WFs in river basins should be compared to WF cap for these basin. Such an analysis would additionally reveal the size of the challenge humanity is up for, in its quest to reduce its water consumption to sustainable scales.

7.4.2. WF benchmarks

Regarding formulating WF benchmarks for water-using activities, the global datasets used in the underlying *Aqua21* model framework vary in quality and resolution – be it climate, soil characteristics, crop maps, irrigated areas, agricultural management or yield statistics. Numerous cogent assumptions have been made in this extensive modelling exercise, most notably regarding the temporal dimension. Caution is therefore warranted in interpreting results at finer spatiotemporal scales.

Chapter 3 revealed that certain WFs are realised by the 'best' producers, but it does not tell *how* the proposed WF benchmarks can be achieved. The alternative route of identifying WF benchmarks (i.e. based on best management practice, as was explored in the case study for silk in Chapter 5), provides a more actionable pathway to materialize efficiency gains through benchmarking. Future studies could therefore attempt to recalculate WF benchmarks of crops based on this latter approach. Such an effort would require filling gaps in global spatiotemporal data on agricultural management.

Moreover, I did not develop grey WF benchmarks, which would complement consumptive water use benchmarks with a water quality target.

The reservoir study in Chapter 4 showed that part of the variability in WFs across reservoirs is caused by the climate of the reservoir location. Yet the reservoir surface size in relation to the production (viz. output) of each purpose contributes most. Future research may propose criteria based upon which (purpose-specific) blue WF benchmarks can be formulated for reservoirs. Particular focus should be put on how such WF benchmarks would be reasonable or plausible in relation to other (e.g. guarantee of delivery, ecological) factors that planners rely on in developing a reservoir.

7.4.3. Sustainable investments

The investor study assessed 20 Dutch institutional investors, based on their publicly disclosed policy documents. Future research could expand the pool of investors, and investigate whether results found in Chapter 6 can be generalized or extrapolated to investors in foreign or private markets. Alternatively, a case study analyzing the full portfolio of one or two selected and collaborative investors may allow for a scoring of framework criteria based on actual ground-truthed performance.

7.5. Final remarks

Research on setting WF caps at the river basin level and formulating WF benchmarks for water-using activities is in its infancy. Outside academia, to date one practical policy arrangement sets a WF cap for a river basin, i.e. the Murray-Darling basin in Australia (Grafton et al., 2014). Given both the rising pressure on water resources globally and the increased understanding of both instruments, however, I anticipate more practical examples applying WF caps and benchmarks will follow soon.

It is tempting to expect setting water targets is a concern exclusively aimed at scarce regions or basins. While it is indispensable for such areas to curb their WFs, a strategic pathway to conserve limited blue water resources in stressed basins is – perhaps somewhat counterintuitively – to increase water productivity in basins that still have the potential to do so. Setting water targets could therefore equally well enrich policy arrangements in water-abundant regions.

My research alluded that, in setting water targets, potential discord between efficient and sustainable water consumption could arise. Additional trade-offs between (targets emerging from) food, energy, foreign trade and other policy domains were not in the scope of this work. However, achieving such (non-water) targets may come at the expense of meeting a WF cap or benchmark. Examples include equitable sharing of freshwater between various user groups, generating sufficient income for (subsistence) farmers, lowering carbon footprints of energy and food production, and achieving self-sufficiency in national food security. Given these interdependencies, policies employing targets that affect other domains should ideally be considered collectively rather than in isolation.

The complexity surrounding the development of well-intending water policies underscores the necessity to include a wide array of actors – as without such concerted efforts the multiheaded global water crises will overcome us. I am convinced, however, that if we recognize the shared responsibility for what it is and take it on, humanity can transition to sustainable and efficient use of freshwater resources worldwide.

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Supplementary Materials

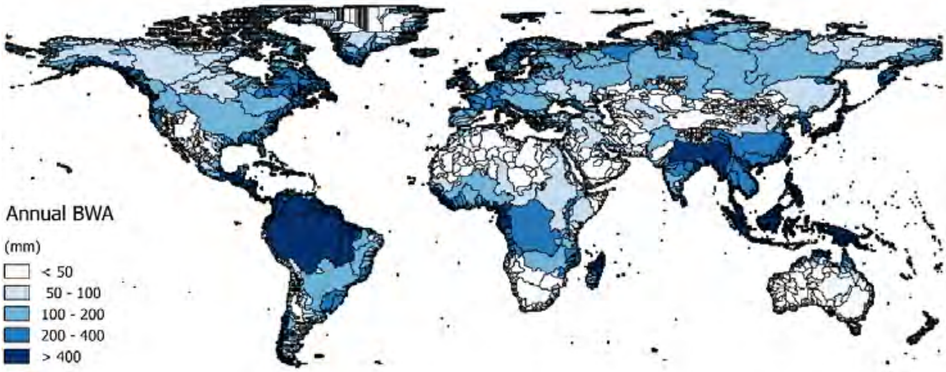

Figure S 1. Annual BWA in mm for all basins considered.

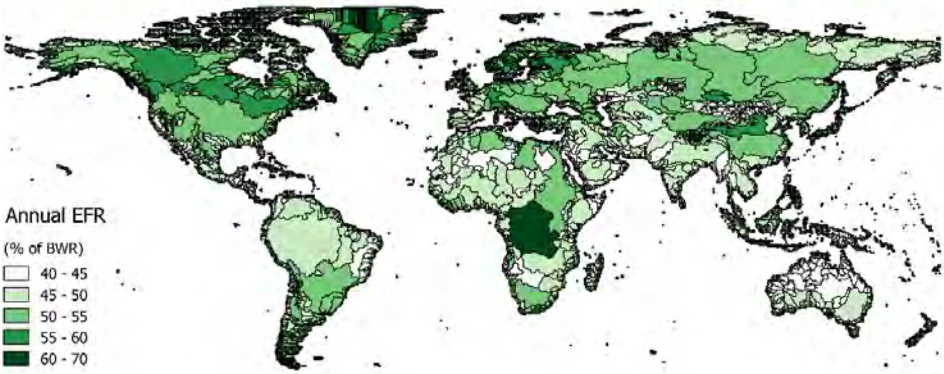

Figure S 2. EFR as percentage of BWR on annual basis, based on ensemble means of both EFR-methods and BWR models.

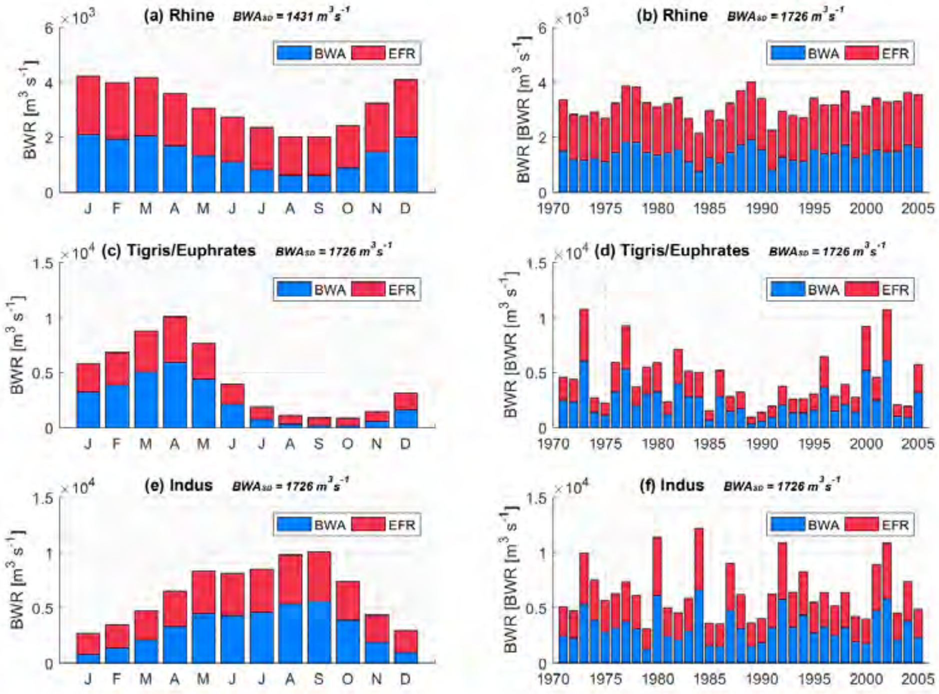

Figure S 3. Monthly blue water runoff (BWR) partitioned into environmental flow requirements (EFR) and blue water availability (BWA), for three selected basins, with mean and standard deviation (SD) values of BWA within an average year (a,c,e); and between years (b,d,f) in the period 1970-2005.

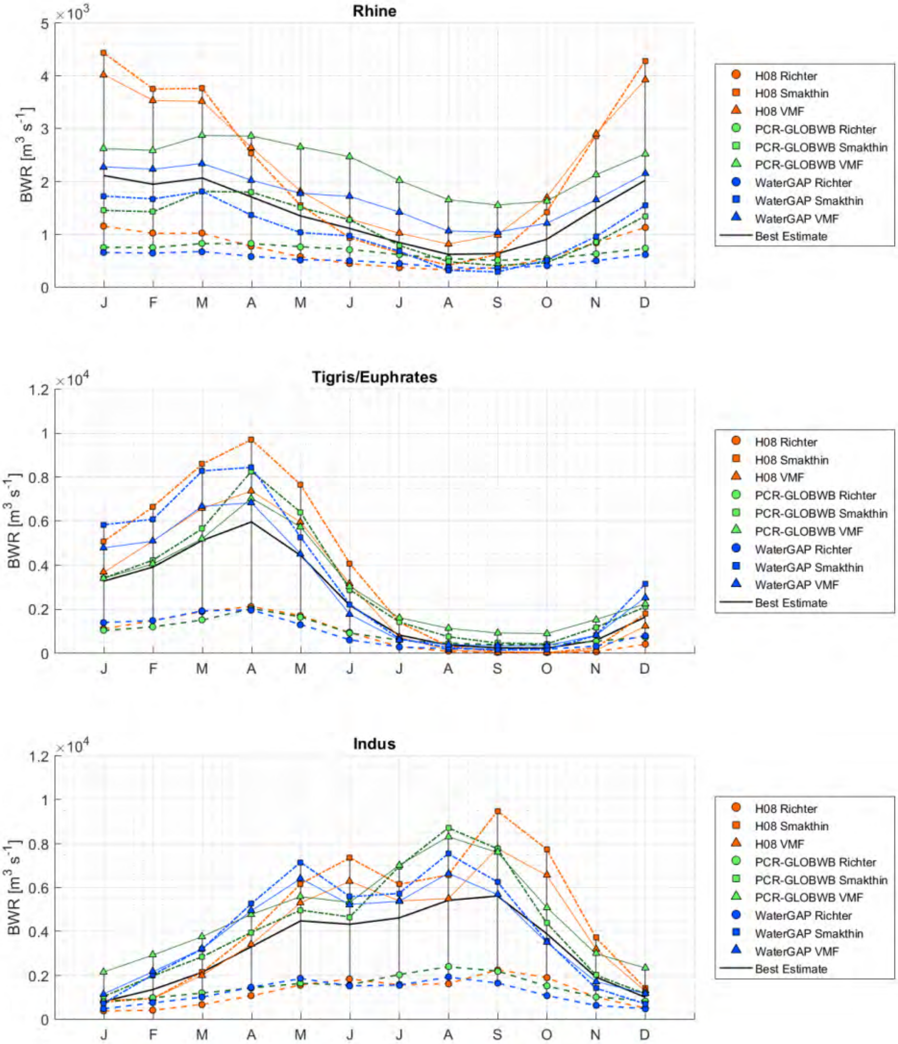

Figure S 4. Spread in model output for both GHMs and EFR methods for three selected basins. The black line shows the BWA estimate used in this study based on ensemble mean BWR minus ensemble mean EFR.

Table S 1. WF benchmarks ($m^3\ t^{-1}$) and yield (Y, $t\ ha^{-1}$) for all crops at various percentiles of global production, differentiated per climate zone (HA: Hyper-arid; A: Arid; SA: Semi-arid; DH: Dry-subhumid; H: Humid) and farming system (I: Irrigated; R: Rainfed), over the period 1996-2015.

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
Barley	HA	I	213	25.5	357	18.9	1195	6.1	4122	1.6	*1936*	2.3
	A	I	182	15.1	257	12.6	531	8.4	3215	2.8	*1288*	3.1
	SA	R	431	4.2	748	3.8	1149	3.0	3328	1.4	*1722*	1.5
	SA	I	323	9.4	465	8.6	854	7.1	2266	3.8	*1255*	3.6
	DH	R	301	7.3	584	5.8	942	4.0	2356	2.2	*1257*	2.3
	DH	I	402	10.1	593	8.0	1089	5.4	2583	3.1	*1713*	2.9
	H	R	420	8.1	525	6.7	660	5.8	1551	3.7	*894*	3.6
	H	I	287	10.8	397	9.7	604	9.6	2084	4.9	*1112*	4.2
Maize	HA	I	801	7.9	900	8.1	1044	8.0	1443	8.1	*1121*	7.7
	A	I	562	11.5	611	10.7	688	9.5	1809	7.4	*973*	6.7
	SA	R	533	8.8	712	6.8	1513	3.6	3515	2.0	*1940*	2.0
	SA	I	500	12.9	569	11.2	691	10.9	1229	10.5	*821*	8.5
	DH	R	479	10.3	588	9.1	808	6.8	2374	2.9	*1242*	3.2
	DH	I	461	12.9	494	11.9	570	11.6	893	10.5	*670*	9.0
	H	R	426	11.8	495	11.6	587	10.7	1843	5.0	*962*	5.1
	H	I	379	14.4	438	11.9	502	11.7	1160	9.4	*651*	8.0
Rye	HA	I	430	25.3	447	23.9	580	19.6	953	12.2	*671*	14.9
	A	I	356	21.4	431	20.0	694	12.9	6187	2.1	*2168*	2.4
	SA	R	765	4.9	1186	3.4	1849	2.4	6417	1.0	*3191*	1.0
	SA	I	944	8.1	1437	5.6	2003	4.0	4107	2.6	*2340*	2.7
	DH	R	506	6.4	819	4.4	1836	2.8	5415	1.3	*2790*	1.4
	DH	I	459	14.1	653	9.8	1293	5.8	2945	3.9	*1679*	3.6
	H	R	553	7.7	765	5.7	1324	3.7	2672	2.4	*1589*	2.6
	H	I	273	17.8	523	10.3	810	6.1	2012	4.7	*1084*	5.2
Oats	HA	I	927	9.6	2620	2.7	3058	1.7	4730	1.2	*3220*	1.6
	A	I	641	9.8	830	10.6	1049	8.6	2844	5.8	*1443*	4.5
	SA	R	487	4.7	819	3.4	1208	2.2	2797	1.2	*1558*	1.3
	SA	I	663	7.0	876	6.1	1148	4.7	2370	3.1	*1403*	3.4
	DH	R	383	5.3	538	4.2	1022	2.6	1824	1.8	*1129*	1.8
	DH	I	698	6.9	885	4.5	1156	3.6	2332	2.4	*1403*	2.7
	H	R	484	5.2	837	4.0	1040	3.3	1660	2.3	*1138*	2.4

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
	H	I	658	8.6	832	6.6	1146	5.1	1997	4.1	*1303*	4.1
Millet	HA	I	2046	4.2	2344	2.5	2911	1.8	3857	1.6	*2962*	1.8
	A	I	1409	5.4	2966	3.6	4761	1.7	10337	0.9	*5519*	1.2
	SA	R	2877	2.0	3612	1.5	4672	1.2	9988	0.7	*5972*	0.7
	SA	I	2670	3.8	3377	1.9	4078	1.8	6115	1.4	*4280*	1.5
	DH	R	1383	4.2	2554	1.9	3641	1.4	6325	1.1	*3998*	1.2
	DH	I	1167	5.5	2920	3.0	3844	2.9	5431	2.4	*3651*	2.5
	H	R	1905	2.8	2459	1.8	3042	1.5	5445	1.2	*3651*	1.2
	H	I	793	5.3	984	4.6	2018	3.1	4689	2.1	*2517*	2.4
Sorghum	HA	I	491	8.4	760	6.4	1183	5.2	2550	4.3	*1491*	4.1
	A	I	691	8.5	3079	4.8	4144	2.5	7073	1.8	*4229*	2.0
	SA	R	1260	5.2	2479	2.6	4263	1.4	8881	0.9	*5309*	0.9
	SA	I	922	8.6	1133	6.9	1466	7.3	5245	3.0	*2343*	3.5
	DH	R	894	7.4	1109	6.2	1607	4.8	5196	1.5	*2827*	1.8
	DH	I	573	12.8	844	8.5	1015	6.9	4387	4.3	*1618*	4.8
	H	R	828	7.8	1034	7.0	1641	4.6	3981	1.7	*2222*	2.3
	H	I	380	12.0	467	10.6	877	8.2	1485	5.6	*1054*	5.2
Fonio	A	I	6772	1.0	7801	1.0	8497	1.0	11567	0.8	*9005*	0.8
	SA	R	5838	1.0	7126	0.8	8496	0.7	12568	0.5	*9307*	0.5
	SA	I	4357	1.3	5969	1.0	6572	0.9	9172	0.8	*6794*	0.8
	DH	R	4603	1.2	5345	0.9	5770	0.9	7530	0.8	*5976*	0.8
	DH	I	5380	1.1	5698	0.9	6068	0.8	7184	0.8	*6135*	0.8
	H	R	3076	1.4	3292	1.3	3625	1.2	5360	1.0	*3957*	1.1
	H	I	3337	1.3	3473	1.3	4059	1.1	5806	1.0	*4291*	1.0
Triticale	A	I	524	18.6	631	17.1	678	14.4	1095	12.5	*763*	12.6
	SA	R	550	5.8	685	5.3	993	2.6	2704	1.2	*1546*	1.1
	SA	I	263	28.8	326	26.2	603	15.6	1072	10.4	*756*	4.5
	DH	R	115	8.0	183	6.1	331	4.9	1413	1.9	*731*	1.9
	DH	I	97	7.2	187	6.7	258	10.4	741	6.2	*387*	4.5
	H	R	11	8.7	34	5.1	99	4.6	330	4.2	*162*	4.0
	H	I	30	10.4	79	7.2	176	11.1	488	6.3	*261*	5.7
Potatoes	HA	I	238	29.5	268	29.6	334	28.6	471	28.7	*344*	28.2
	A	I	180	54.7	211	44.4	286	32.7	439	25.8	*306*	27.8
	SA	R	135	15.4	217	15.5	315	11.7	793	5.9	*399*	7.1
	SA	I	160	42.2	181	42.1	207	42.0	336	31.0	*233*	31.8

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
	DH	R	175	23.1	230	22.7	330	13.7	660	8.8	*387*	9.5
	DH	I	154	43.3	177	41.3	206	39.4	362	25.6	*246*	26.9
	H	R	113	47.6	179	30.7	251	20.2	389	15.9	*263*	16.9
	H	I	113	53.5	125	48.0	153	44.5	331	27.9	*197*	31.1
Sweet potatoes	HA	I	190	31.8	204	32.7	226	31.5	315	27.1	*246*	27.2
	A	I	185	38.9	202	35.0	227	34.5	395	24.9	*299*	23.7
	SA	R	91	56.3	108	45.5	156	29.9	1450	5.6	*452*	6.9
	SA	I	88	78.6	103	63.9	149	46.5	274	27.9	*187*	31.8
	DH	R	74	72.0	87	63.3	105	52.7	533	18.9	*241*	14.6
	DH	I	80	80.4	88	73.0	101	65.1	173	49.7	*128*	45.4
	H	R	70	36.6	80	41.4	98	34.8	792	10.8	*261*	12.4
	H	I	72	69.1	78	70.9	87	63.3	202	37.5	*137*	33.6
Cassava	HA	I	439	19.6	457	18.1	504	16.3	653	14.6	*521*	15.4
	A	I	444	22.8	494	21.3	540	19.0	683	16.1	*560*	17.5
	SA	R	351	12.2	508	9.5	702	9.4	1606	5.6	*883*	6.2
	SA	I	444	23.6	489	20.8	537	19.1	688	16.1	*556*	17.7
	DH	R	309	24.1	447	17.2	654	12.5	1417	6.8	*796*	8.0
	DH	I	526	24.6	584	25.2	625	26.8	691	25.1	*625*	25.2
	H	R	330	29.9	424	20.6	567	16.2	1064	10.6	*656*	11.8
	H	I	474	27.9	499	25.1	563	21.9	712	17.2	*580*	19.7
Yams	A	I	1138	19.6	1295	6.2	1386	6.2	1658	6.1	*1326*	6.4
	SA	R	71	8.1	125	5.3	260	3.0	1151	3.2	*500*	3.3
	SA	I	50	131.0	66	114.2	320	32.4	762	11.2	*361*	15.9
	DH	R	106	8.0	150	7.4	192	9.0	358	6.4	*224*	6.7
	DH	I	94	94.4	167	42.3	300	25.2	498	13.9	*301*	18.7
	H	R	199	15.4	228	14.0	273	11.9	413	9.2	*294*	10.0
	H	I	181	40.8	213	18.1	241	16.3	395	13.9	*264*	15.2
Sugar beet	HA	I	108	133.2	146	88.7	191	61.3	301	55.4	*205*	63.1
	A	I	110	98.0	143	56.1	176	51.6	319	44.1	*202*	47.6
	SA	R	41	102.6	60	66.2	103	43.6	267	17.9	*145*	19.7
	SA	I	85	100.6	99	91.0	128	71.9	186	56.0	*134*	62.1
	DH	R	58	90.4	78	64.8	123	34.9	265	19.4	*151*	22.4
	DH	I	88	93.7	108	74.8	132	62.9	184	52.9	*135*	56.7
	H	R	50	98.0	58	80.8	72	67.3	146	44.9	*88*	46.8

Crop			WF$_{p10}$	Y$_{p10}$	WF$_{p25}$	Y$_{p25}$	WF$_{p50}$	Y$_{p50}$	WF$_{p90}$	Y$_{p90}$	WF$_{avg}$	Y$_{avg}$
	H	I	63	100.9	71	81.2	83	71.7	121	60.9	*90*	64.5
Beans	HA	I	1677	3.3	2410	3.2	2998	3.2	4339	2.8	*3015*	2.9
	A	I	1643	3.6	2100	3.4	2817	3.3	11037	1.3	*4690*	1.5
	SA	R	1284	0.5	2098	1.0	4674	0.6	10658	0.5	*5618*	0.5
	SA	I	1828	3.5	2023	3.2	2679	2.8	10716	0.8	*4730*	1.1
	DH	R	1272	1.0	1773	1.6	4206	0.9	10308	0.5	*5282*	0.6
	DH	I	1786	3.6	6688	1.0	8805	0.5	11588	0.5	*8111*	0.6
	H	R	1407	2.0	1806	1.7	2531	1.3	6410	0.8	*3374*	0.9
	H	I	1527	3.1	2132	2.2	3981	1.2	8810	0.7	*4642*	0.9
Peas	HA	I	1276	2.9	1392	2.7	1530	2.6	2070	2.5	*1616*	2.5
	A	I	1118	7.2	1342	6.2	1798	3.6	8027	1.1	*3613*	1.6
	SA	R	835	2.4	1204	2.5	1855	1.7	4093	1.0	*2249*	1.1
	SA	I	1059	7.3	1216	6.9	1665	5.3	5832	1.6	*2784*	2.0
	DH	R	816	5.0	1037	4.0	1583	2.7	3365	1.4	*1947*	1.6
	DH	I	1342	6.8	2238	3.5	2890	1.7	4060	1.4	*2962*	1.5
	H	R	638	5.5	836	4.4	1324	3.2	2546	1.9	*1542*	2.0
	H	I	498	10.0	578	9.1	683	7.8	2117	4.8	*1008*	4.4
Cow peas	SA	R	5411	0.8	7046	0.7	9027	0.5	20853	0.3	*11291*	0.3
	SA	I	2592	2.3	2753	2.2	4413	1.6	7549	0.9	*4801*	1.2
	DH	R	2351	1.0	3440	1.1	4963	0.7	7165	0.6	*4813*	0.7
	DH	I	2518	1.7	4643	1.2	5016	0.7	6938	0.7	*4930*	0.7
	H	R	2020	1.2	2663	1.2	4255	0.7	5814	0.6	*4001*	0.7
	H	I	2515	1.6	3841	1.6	4527	0.8	6281	0.7	*4507*	0.8
Bambara beans	SA	R	702	1.1	807	1.0	1053	0.9	2066	0.9	*1238*	0.9
	DH	R	735	1.6	799	1.4	881	1.3	1228	1.1	*968*	1.1
	H	R	997	1.6	1442	1.3	1871	0.9	6647	0.6	*3032*	0.7
Lupins	HA	I	2161	2.0	2457	1.9	2618	1.8	3635	1.6	*2779*	1.6
	A	I	2260	0.5	2656	1.8	3053	1.7	12507	1.0	*5184*	0.9
	SA	R	390	6.2	663	3.3	906	2.4	2545	1.3	*1360*	1.2
	SA	I	1928	3.6	2413	2.7	2571	2.9	5219	2.1	*3003*	2.3
	DH	R	367	2.4	520	3.3	816	2.7	2326	1.3	*1229*	1.3
	H	R	134	2.4	247	2.3	547	2.5	1457	1.9	*1098*	1.5
Almonds	HA	I	5387	4.3	8125	2.2	9303	1.9	10900	1.8	*9071*	1.9
	A	I	1435	9.5	1571	7.9	1953	6.9	6055	4.1	*2760*	4.4
	SA	R	1473	6.6	1898	5.6	3431	2.7	26416	0.4	*10730*	0.4

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
	SA	I	1503	9.0	1776	6.8	2651	4.9	10511	1.6	*4727*	2.1
	DH	R	2255	3.4	4504	1.3	9981	0.7	26617	0.3	*12687*	0.4
	DH	I	921	13.2	1344	8.7	2392	5.4	10647	1.8	*4215*	2.1
	H	R	1919	5.1	2717	2.7	4025	1.5	12273	0.7	*5839*	0.8
	H	I	1403	6.4	1766	6.2	2581	3.4	5682	2.3	*3456*	2.2
Soybeans	HA	I	2066	4.2	2606	4.0	3323	3.5	7371	2.2	*4171*	2.7
	A	I	2149	3.6	2356	3.1	2597	2.9	3308	2.8	*2762*	2.8
	SA	R	2134	2.2	3221	1.5	4668	1.2	7022	0.9	*4860*	0.9
	SA	I	1880	4.6	2157	3.5	2404	3.3	3127	3.0	*2532*	3.0
	DH	R	1579	3.9	1814	3.4	2485	2.5	5377	1.3	*3145*	1.5
	DH	I	1682	5.1	1885	4.2	2107	3.7	2656	3.3	*2193*	3.4
	H	R	1539	4.2	1728	3.8	2063	3.4	3019	2.7	*2292*	2.6
	H	I	1545	4.8	1721	3.7	1925	3.5	2532	3.3	*2025*	3.3
Ground nuts	HA	I	2349	3.9	2652	3.5	2977	3.4	4108	3.4	*3185*	3.3
	A	I	1681	4.5	1860	4.0	2170	3.7	8206	2.2	*3702*	2.4
	SA	R	2516	2.4	3510	1.7	4738	1.4	9819	0.8	*5779*	0.8
	SA	I	1672	4.3	2451	3.3	3458	3.0	5268	2.3	*3560*	2.5
	DH	R	1467	4.2	1860	3.7	3161	2.2	6230	1.2	*3705*	1.4
	DH	I	1431	4.9	1606	4.2	2888	3.1	4566	2.1	*2966*	2.5
	H	R	1180	4.8	1327	4.3	1921	3.6	4765	1.8	*2556*	1.9
	H	I	1147	4.8	1272	4.6	1432	4.2	2308	3.9	*1622*	3.8
Coconuts	SA	R	1540	5.3	1777	4.3	2115	3.7	4928	2.2	*3022*	2.1
	SA	I	1577	9.7	1708	9.2	1860	9.0	2208	8.5	*1941*	8.3
	DH	R	1142	8.0	1508	5.6	1861	4.7	3402	3.3	*2367*	3.0
	DH	I	1253	12.5	1519	10.1	1725	9.1	2051	8.6	*1715*	8.9
	H	R	1169	11.0	1443	6.9	1710	6.3	2681	4.8	*1876*	5.2
	H	I	868	16.4	1168	11.1	1426	9.6	2001	8.1	*1528*	8.4
Oil palm fruit	SA	R	280	18.3	346	15.8	420	12.7	783	9.6	*485*	10.2
	DH	R	508	15.8	602	12.5	698	11.1	1060	8.1	*825*	8.3
	H	R	432	23.1	477	21.8	523	20.3	716	18.6	*673*	13.6
	H	I	2634	3.5	2797	3.4	2986	3.3	3476	3.3	*3031*	3.3
Olives	HA	I	915	10.8	1073	9.7	1309	9.2	2402	6.6	*1627*	6.8
	A	I	1046	11.1	1416	7.8	1915	6.0	3867	3.9	*2252*	4.7
	SA	R	1505	3.3	1870	2.1	2522	1.5	7968	0.8	*4060*	0.8

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
	SA	I	889	9.5	1018	8.3	1319	6.6	2130	4.9	*1480*	5.3
	DH	R	1075	3.9	1328	2.9	1771	2.2	4482	1.3	*2450*	1.4
	DH	I	836	8.7	987	7.0	1202	6.0	1943	4.7	*1344*	5.0
	H	R	1037	4.1	1289	3.1	1640	2.5	2957	1.8	*1877*	1.9
	H	I	852	7.8	1024	6.3	1243	5.3	2147	4.0	*1408*	4.2
Sunflower seed	HA	I	2781	3.3	3066	2.7	4059	2.6	8917	2.1	*4966*	2.2
	A	I	1272	5.9	1621	4.4	4234	2.6	8412	1.4	*4595*	1.9
	SA	R	1466	3.1	2377	1.8	3468	1.3	8885	0.6	*4801*	0.7
	SA	I	1341	5.8	1827	4.5	2445	4.0	6423	2.0	*3114*	2.2
	DH	R	1451	3.1	1893	2.3	2507	2.0	4459	1.3	*2880*	1.4
	DH	I	1056	6.8	1258	5.5	1728	4.3	3589	2.9	*2113*	3.0
	H	R	1296	3.7	1580	2.8	2016	2.4	3899	1.7	*2395*	1.7
	H	I	761	7.4	937	6.0	1135	5.2	2125	3.9	*1399*	4.0
Rapeseed	HA	I	3190	2.3	5281	2.0	6187	2.6	7963	2.2	*6293*	2.1
	A	I	2126	1.7	2401	1.7	2761	1.6	3781	1.5	*2905*	1.5
	SA	R	1030	1.5	1466	1.6	2128	1.1	4170	0.7	*2455*	0.8
	SA	I	2357	1.7	2580	1.8	2860	1.8	3522	1.7	*2931*	1.6
	DH	R	1044	4.4	1320	3.3	1675	2.7	3532	1.5	*2049*	1.6
	DH	I	2183	1.9	2359	1.8	2585	1.8	3278	1.7	*2706*	1.7
	H	R	466	7.0	867	4.1	1187	3.5	2474	2.2	*1449*	2.1
	H	I	990	7.5	1306	5.5	2469	2.1	3568	1.8	*2429*	1.9
Seed cotton	HA	I	2162	3.4	2419	3.0	3134	2.8	5142	2.4	*3515*	2.6
	A	I	2216	4.6	2867	2.9	3447	2.6	5827	2.0	*3833*	2.2
	SA	R	1760	2.7	2913	1.6	4532	1.2	9843	0.7	*5375*	0.9
	SA	I	1444	5.4	2440	3.8	3256	2.7	5459	2.1	*3505*	2.3
	DH	R	929	6.6	1239	5.6	2335	2.8	6390	1.3	*3163*	1.6
	DH	I	917	9.8	1210	5.7	1730	4.5	3782	2.8	*2228*	3.2
	H	R	987	7.4	1204	6.1	2121	4.2	5004	2.2	*2758*	2.2
	H	I	806	9.9	1006	7.0	1225	6.0	3120	3.8	*1689*	4.3
Cabbage and other brassicas	HA	I	216	33.2	232	33.5	255	31.5	407	30.0	*286*	28.0
	A	I	147	54.0	181	50.6	225	39.0	375	30.6	*261*	33.1
	SA	R	104	31.9	123	25.3	143	24.8	270	17.3	*176*	16.6
	SA	I	109	73.1	125	56.2	150	51.8	290	33.9	*177*	40.7

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
	DH	R	105	39.2	121	36.9	147	29.9	263	20.2	*171*	21.2
	DH	I	110	69.1	127	57.1	156	38.9	184	31.4	*150*	37.8
	H	R	71	58.9	83	45.1	107	38.7	243	25.7	*140*	27.2
	H	I	80	60.5	98	47.2	112	50.0	186	42.3	*127*	42.4
Lettuce and chicory	HA	I	80	42.5	88	42.0	93	42.2	103	41.4	*99*	39.1
	A	I	49	36.3	76	36.1	92	39.3	113	37.3	*97*	34.5
	SA	R	35	36.2	42	32.5	60	25.6	202	18.5	*103*	19.1
	SA	I	36	39.3	39	35.4	48	36.0	184	32.8	*77*	32.1
	DH	R	46	30.8	95	21.6	169	23.8	294	18.2	*168*	17.1
	DH	I	42	36.3	47	36.0	68	33.1	189	29.6	*111*	30.5
	H	R	45	28.5	59	28.0	76	26.0	193	23.1	*109*	18.6
	H	I	42	33.0	57	30.8	108	29.1	163	33.3	*108*	29.8
Spinach	HA	I	108	39.2	120	29.2	132	25.6	229	21.2	*152*	21.3
	A	I	90	27.9	104	28.1	122	28.0	185	22.8	*131*	23.0
	SA	R	90	22.7	101	31.1	128	24.9	218	14.8	*149*	14.8
	SA	I	85	21.5	95	34.2	106	35.4	178	26.8	*119*	26.9
	DH	R	88	30.0	99	33.9	121	28.3	208	16.2	*142*	17.1
	DH	I	86	28.7	93	41.8	105	39.5	171	29.9	*117*	31.5
	H	R	43	48.4	53	36.3	74	26.9	159	19.5	*102*	19.2
	H	I	55	47.4	71	35.2	85	35.4	143	26.8	*99*	28.5
Tomatoes	HA	I	129	47.4	145	40.1	172	38.6	237	37.6	*180*	38.0
	A	I	47	98.0	80	78.6	117	47.7	218	39.2	*128*	46.6
	SA	R	36	44.9	48	52.8	73	40.2	226	18.3	*110*	22.6
	SA	I	41	109.8	50	91.8	94	75.2	213	39.4	*116*	50.5
	DH	R	44	70.7	65	51.2	104	44.1	242	20.6	*135*	24.7
	DH	I	45	102.3	64	77.1	107	67.6	201	35.7	*122*	46.2
	H	R	49	81.6	60	56.5	78	53.2	250	30.4	*121*	28.8
	H	I	47	115.0	58	83.8	72	71.0	126	54.4	*85*	57.8
Pumpkin, squash and gourds	HA	I	106	44.0	190	23.6	225	19.0	376	18.8	*246*	19.8
	A	I	162	26.3	192	20.6	215	19.7	339	17.0	*237*	17.1
	SA	R	65	9.7	128	14.3	234	13.1	403	9.3	*278*	8.3
	SA	I	87	46.7	136	33.4	205	23.7	340	15.0	*213*	18.8

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
	DH	R	106	21.3	150	20.4	198	20.5	367	12.1	*240*	12.3
	DH	I	89	42.2	148	31.4	205	25.1	322	14.2	*217*	18.3
	H	R	117	26.5	142	22.2	186	20.2	407	11.4	*255*	11.2
	H	I	85	45.6	137	25.5	163	25.6	296	20.3	*189*	20.5
Cucum-bers and gherkins	HA	I	97	92.1	134	72.6	197	33.7	283	29.9	*220*	24.5
	A	I	89	85.9	147	41.0	193	36.1	299	31.1	*205*	33.7
	SA	R	35	16.7	51	58.1	81	53.7	238	25.4	*143*	20.0
	SA	I	43	133.9	58	102.3	105	68.3	331	28.6	*163*	35.3
	DH	R	43	78.5	55	86.2	84	66.7	204	32.0	*118*	33.2
	DH	I	44	133.9	53	110.3	71	89.8	141	59.6	*93*	58.1
	H	R	39	112.0	48	99.2	78	64.8	284	22.8	*152*	20.6
	H	I	41	155.8	47	116.4	63	95.8	157	53.2	*98*	44.1
Green chillies and peppers	HA	I	136	19.8	167	18.1	200	16.7	307	16.2	*214*	16.7
	A	I	70	33.4	127	20.4	176	19.2	508	14.8	*232*	15.6
	SA	R	15	67.1	36	33.2	101	21.4	305	15.1	*136*	15.0
	SA	I	21	75.6	45	59.3	101	42.1	171	35.3	*108*	33.7
	DH	R	61	31.1	108	28.6	136	34.7	305	19.3	*166*	18.5
	DH	I	42	52.4	92	57.0	129	44.5	203	37.2	*143*	34.4
	H	R	87	28.7	105	28.0	133	26.6	672	10.5	*254*	12.4
	H	I	70	77.2	106	42.2	125	41.6	249	30.3	*184*	24.6
Onions	HA	I	125	80.4	171	60.4	231	36.4	382	33.3	*335*	29.9
	A	I	90	70.2	134	62.7	194	47.1	557	21.6	*329*	25.7
	SA	R	80	40.9	130	28.8	223	19.5	567	11.1	*289*	11.3
	SA	I	124	60.0	165	47.4	234	41.5	374	33.9	*242*	36.6
	DH	R	121	37.2	181	24.1	287	17.0	569	11.9	*319*	12.5
	DH	I	142	47.6	200	41.2	260	40.1	356	31.2	*255*	33.8
	H	R	100	54.6	138	33.3	179	27.1	566	15.7	*283*	16.5
	H	I	108	60.5	155	38.3	225	32.6	337	29.5	*241*	29.6
Garlic	HA	I	251	27.4	293	24.7	332	23.5	417	22.9	*338*	23.4
	A	I	177	29.0	204	27.3	289	26.9	633	16.2	*385*	18.6
	SA	R	181	25.6	209	28.4	292	18.6	772	7.5	*401*	8.7
	SA	I	183	36.2	204	33.8	269	26.6	880	13.5	*405*	16.7

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
Green beans	DH	R	169	29.0	197	29.5	237	27.9	619	11.5	*331*	12.4
	DH	I	185	36.5	200	36.8	226	34.8	829	16.7	*355*	20.0
	H	R	122	41.3	149	33.0	201	26.4	612	12.8	*292*	13.7
	H	I	152	41.4	177	35.5	204	33.2	392	19.8	*257*	22.6
	HA	I	212	12.2	262	11.3	299	11.0	456	9.7	*323*	10.0
	A	I	168	18.4	192	12.5	249	12.2	556	11.0	*328*	10.7
	SA	R	79	7.5	104	26.2	143	25.2	377	9.3	*197*	9.2
	SA	I	89	42.1	101	43.7	122	36.1	305	21.3	*184*	19.0
Green peas	DH	R	83	33.4	100	42.4	124	38.4	288	18.9	*171*	17.6
	DH	I	92	50.0	101	49.9	119	43.7	257	23.1	*171*	21.7
	H	R	60	22.0	76	23.1	110	26.1	507	9.5	*235*	9.1
	H	I	81	27.1	92	42.1	111	38.1	442	20.6	*181*	17.9
	HA	I	313	12.9	346	12.8	398	12.3	707	8.8	*476*	8.8
	A	I	296	13.2	337	12.7	444	12.2	781	10.5	*539*	10.2
	SA	R	185	5.3	247	6.7	351	8.8	699	8.6	*410*	6.2
	SA	I	236	18.4	294	15.8	357	16.5	656	13.0	*429*	12.3
Carrots and turnips	DH	R	219	8.6	306	11.7	353	16.5	696	10.3	*426*	8.7
	DH	I	199	17.3	269	16.1	332	18.4	429	17.6	*348*	15.1
	H	R	158	11.6	216	8.8	296	11.3	677	8.5	*402*	6.4
	H	I	235	18.9	278	19.6	312	18.8	407	16.8	*350*	12.7
	HA	I	185	49.8	200	58.0	219	58.6	378	39.0	*264*	40.6
	A	I	138	30.0	158	39.3	172	51.3	228	49.9	*180*	40.9
	SA	R	70	23.1	85	43.7	127	27.2	256	15.7	*154*	16.8
	SA	I	58	132.6	80	79.2	107	56.9	189	40.1	*118*	45.6
Okra	DH	R	80	39.3	98	40.3	127	27.6	229	18.1	*150*	18.6
	DH	I	70	81.2	87	68.1	108	57.2	195	37.0	*127*	41.0
	H	R	38	70.0	54	57.4	76	45.0	174	22.9	*97*	26.4
	H	I	49	72.1	60	70.3	76	61.7	151	37.5	*93*	41.1
	HA	I	126	16.0	150	15.4	233	14.3	540	12.0	*284*	11.9
	A	I	116	16.1	126	16.1	151	15.3	832	12.4	*304*	11.8
	SA	R	125	7.5	149	10.4	563	6.0	1391	2.1	*651*	2.7
	SA	I	145	13.6	161	13.8	184	15.2	251	14.0	*258*	12.2
	DH	R	113	13.2	125	12.0	143	11.2	966	4.3	*340*	3.9

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
	DH	I	121	18.2	133	17.3	142	18.0	175	16.1	*149*	16.1
	H	R	113	16.1	129	13.6	152	11.4	738	6.5	*299*	6.2
	H	I	113	19.8	125	18.6	138	18.4	165	20.0	*149*	18.4
Bananas	HA	I	359	44.8	409	43.5	485	43.3	599	44.8	*503*	41.9
	A	I	324	65.8	366	44.6	866	37.1	1486	18.1	*828*	24.2
	SA	R	236	36.1	272	32.1	356	23.0	1138	11.5	*606*	10.0
	SA	I	233	85.7	256	75.1	277	74.5	360	60.5	*307*	59.1
	DH	R	231	41.3	258	38.1	321	29.8	1016	14.0	*534*	13.9
	DH	I	218	86.1	238	75.7	262	73.7	367	59.5	*283*	60.2
	H	R	186	59.6	228	50.2	380	34.1	948	16.7	*521*	18.5
	H	I	211	71.3	239	65.7	279	59.0	712	32.4	*377*	36.6
Plantains	HA	I	468	30.3	490	31.0	533	28.0	680	26.6	*560*	27.2
	A	I	509	32.7	531	30.8	575	28.4	817	25.3	*693*	21.9
	SA	R	1158	9.4	2019	4.9	2383	3.8	3157	2.8	*2589*	2.7
	SA	I	513	31.7	679	27.6	998	18.2	2609	9.0	*1537*	9.0
	DH	R	917	13.3	1815	5.9	2105	4.8	2641	4.0	*2110*	3.9
	DH	I	471	32.1	523	28.0	908	18.8	2011	9.6	*1411*	9.0
	H	R	750	17.9	1015	10.7	1436	8.1	2303	5.8	*1498*	6.6
	H	I	454	35.0	773	23.0	971	11.0	1541	9.5	*1133*	9.7
Oranges	HA	I	356	26.9	381	24.6	429	24.4	565	22.6	*446*	23.4
	A	I	228	49.8	270	36.7	325	32.5	737	20.9	*405*	25.0
	SA	R	194	30.2	283	22.0	479	12.5	985	7.4	*592*	7.8
	SA	I	136	69.3	182	49.6	229	38.3	529	25.0	*280*	28.7
	DH	R	187	34.0	260	25.1	359	17.6	788	9.7	*473*	9.8
	DH	I	118	78.3	171	44.3	224	34.5	509	22.6	*268*	26.3
	H	R	191	41.5	263	26.1	299	23.0	524	18.1	*347*	17.8
	H	I	154	46.4	167	46.5	196	42.9	456	26.0	*262*	27.7
Apples	HA	I	482	44.4	503	24.9	575	24.2	736	22.3	*588*	23.6
	A	I	228	62.7	262	41.9	331	34.5	622	27.3	*405*	28.9
	SA	R	268	27.2	334	21.2	479	13.0	1186	6.2	*714*	6.0
	SA	I	161	71.0	215	55.4	301	37.5	849	20.0	*427*	23.8
	DH	R	270	30.1	330	24.6	444	14.9	1022	7.5	*584*	8.4
	DH	I	124	91.1	164	68.6	266	43.2	664	23.7	*330*	28.2
	H	R	147	46.8	217	31.4	315	23.8	769	12.5	*406*	13.5
	H	I	104	80.5	135	65.0	214	45.8	459	27.1	*260*	30.0

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
Pears	HA	I	435	34.0	713	22.7	936	17.4	1419	13.3	*950*	15.9
	A	I	155	110.9	188	63.0	250	52.7	637	23.9	*334*	31.3
	SA	R	246	45.8	379	21.5	530	13.8	1144	6.8	*686*	7.1
	SA	I	164	73.7	194	54.0	243	44.8	695	24.4	*361*	26.8
	DH	R	348	25.5	418	20.9	549	13.0	1057	6.9	*665*	7.4
	DH	I	174	62.8	224	48.9	281	37.8	487	25.5	*320*	28.3
	H	R	199	40.8	260	30.6	350	23.9	662	14.0	*410*	14.6
	H	I	154	53.4	185	45.9	249	35.9	434	25.2	*282*	27.8
Peaches and nectarine	HA	I	931	17.6	1064	12.6	1343	11.6	1799	10.5	*1347*	11.3
	A	I	321	56.3	442	33.2	492	27.0	1101	18.5	*636*	20.0
	SA	R	285	33.4	470	16.5	648	12.2	1578	5.2	*1040*	4.4
	SA	I	183	64.4	213	54.5	263	45.2	603	26.3	*369*	28.8
	DH	R	345	23.1	497	14.5	666	11.5	1459	5.7	*864*	5.8
	DH	I	155	69.5	188	54.1	225	44.9	547	29.8	*331*	29.8
	H	R	295	25.3	341	24.0	432	19.8	803	12.8	*522*	12.4
	H	I	171	50.9	201	42.2	281	33.1	523	21.9	*336*	24.7
Rasp-berries	SA	R	456	18.0	586	13.2	830	8.8	1886	3.5	*1136*	3.8
	SA	I	778	15.5	1058	9.0	1163	8.2	1443	6.8	*1228*	7.0
	DH	R	567	10.6	696	8.7	888	6.2	1489	4.3	*974*	4.8
	DH	I	1030	7.1	1257	6.1	1528	5.2	2702	3.8	*1824*	3.8
	H	R	522	11.3	724	8.6	912	6.1	1453	4.3	*982*	4.9
	H	I	583	8.9	812	7.2	978	6.2	1275	5.3	*974*	5.7
Grapes	HA	I	409	31.1	432	31.7	473	30.2	612	28.2	*580*	24.6
	A	I	382	26.4	426	24.3	564	21.7	1231	13.3	*697*	14.8
	SA	R	253	24.7	345	17.8	591	11.0	1288	5.2	*729*	6.1
	SA	I	416	23.9	505	20.4	623	17.5	1180	12.2	*733*	12.6
	DH	R	241	35.1	410	19.0	580	10.6	1177	6.9	*670*	8.0
	DH	I	346	25.6	398	22.3	489	19.1	834	13.9	*565*	14.8
	H	R	249	35.9	414	16.2	572	12.1	995	8.0	*630*	8.9
	H	I	333	24.1	382	20.7	462	18.4	827	13.0	*552*	13.6
Water-melons	HA	I	136	63.7	202	36.3	258	32.9	438	25.1	*291*	27.7
	A	I	124	73.9	179	40.2	219	38.0	350	28.7	*244*	30.4
	SA	R	44	23.2	60	43.1	80	47.9	231	20.2	*122*	20.3

Crop			WF_{p10}	Y_{p10}	WF_{p25}	Y_{p25}	WF_{p50}	Y_{p50}	WF_{p90}	Y_{p90}	WF_{avg}	Y_{avg}
	SA	I	55	103.2	70	80.0	120	57.1	344	28.8	*169*	34.0
	DH	R	52	60.5	63	60.9	76	63.8	186	32.9	*110*	30.8
	DH	I	53	109.0	61	86.7	73	79.3	148	57.2	*95*	54.3
	H	R	48	86.0	57	76.9	77	57.6	225	22.8	*123*	25.7
	H	I	53	95.8	58	88.0	64	81.9	157	45.6	*92*	45.0
Mangoes, mango-steens, guavas	HA	I	575	27.5	799	21.9	1423	12.7	2147	10.0	*1354*	12.4
	A	I	760	24.8	941	19.3	1034	16.1	1406	14.6	*1120*	14.6
	SA	R	965	10.0	1163	6.7	1386	5.4	2054	3.9	*1488*	4.3
	SA	I	863	21.9	976	16.5	1185	15.3	1392	13.0	*1149*	14.3
	DH	R	861	11.1	1009	8.3	1189	6.9	1571	5.2	*1219*	5.5
	DH	I	877	17.3	1001	16.3	1151	14.3	1323	12.4	*1128*	13.6
	H	R	564	22.0	730	14.0	962	10.5	1668	7.1	*1118*	7.4
	H	I	710	26.2	860	15.8	1012	14.0	1250	12.5	*995*	13.5
Pine-apples	HA	I	68	61.2	71	59.1	88	46.3	133	31.2	*96*	37.6
	A	I	75	58.6	81	61.6	109	44.1	186	28.2	*138*	28.3
	SA	R	83	50.6	102	41.7	169	29.5	788	8.3	*334*	12.1
	SA	I	119	43.7	163	26.4	234	18.4	421	13.4	*281*	13.6
	DH	R	93	45.7	103	39.8	118	37.6	401	24.7	*179*	21.3
	DH	I	92	50.8	133	32.6	175	27.0	405	13.3	*275*	13.5
	H	R	60	86.4	84	54.3	115	38.8	252	24.7	*152*	23.4
	H	I	65	81.0	79	65.2	121	44.2	172	28.3	*125*	34.1
Dates	HA	I	2285	10.3	3129	7.0	3448	6.4	4488	5.2	*3490*	5.8
	A	I	1459	10.6	1702	9.0	2114	7.7	4397	5.3	*2591*	6.1
	SA	R	129	69.2	488	27.1	878	11.9	1423	8.5	*1000*	6.5
	SA	I	1618	6.9	1907	5.8	2358	4.9	3474	3.8	*2556*	4.2
	DH	R	462	20.3	590	14.2	735	11.1	1258	8.1	*937*	6.3
	DH	I	2216	5.2	2528	4.5	2888	3.7	3563	3.2	*2962*	3.4
	H	R	462	17.4	531	15.3	664	13.5	1001	9.8	*768*	9.7

Table S 2. Irrigation water fractions in consumptive WFs of production locations that meet WF_{p25} in irrigated crop production (m^3 t^{-1}), per climate zone, at the 5^{th}, median and 95^{th} percentile.

Crop	Climate zone	WF_{p25}	$Frac_{q5}$	$Frac_{median}$	$Frac_{q95}$
Wheat	Hyper-arid	979	0,87	0,94	0,99
	Arid	869	0,46	0,68	0,86
	Semi-arid	1354	0,30	0,57	0,76
	Dry-subhumid	1061	0,22	0,44	0,61
	Humid	493	0,02	0,10	0,26
Rice	Hyper-arid	771	0,59	0,92	0,99
	Arid	662	0,43	0,83	0,95
	Semi-arid	637	0,14	0,55	0,72
	Dry-subhumid	556	0,13	0,46	0,61
	Humid	517	0,06	0,22	0,46
Barley	Hyper-arid	357	0,78	0,93	0,99
	Arid	257	0,34	0,61	0,91
	Semi-arid	465	0,11	0,34	0,72
	Dry-subhumid	593	0,08	0,24	0,60
	Humid	397	0,03	0,12	0,34
Maize	Hyper-arid	900	0,68	0,93	0,98
	Arid	611	0,69	0,81	0,92
	Semi-arid	569	0,37	0,57	0,73
	Dry-subhumid	494	0,25	0,40	0,58
	Humid	438	0,04	0,18	0,45
Rye	Hyper-arid	447	0,93	0,96	0,97
	Arid	431	0,73	0,82	0,91
	Semi-arid	1437	0,08	0,27	0,73
	Dry-subhumid	653	0,07	0,30	0,49
	Humid	523	0,04	0,14	0,31
Oats	Hyper-arid	2620	0,84	0,91	0,95
	Arid	830	0,46	0,71	0,91
	Semi-arid	876	0,24	0,37	0,69
	Dry-subhumid	885	0,14	0,30	0,49
	Humid	832	0,04	0,16	0,39
Millet	Hyper-arid	2344	0,87	0,93	0,96
	Arid	2966	0,54	0,76	0,92

Crop	Climate zone	WF_{p25}	$Frac_{q5}$	$Frac_{median}$	$Frac_{q95}$
	Semi-arid	3377	0,19	0,45	0,68
	Dry-subhumid	2920	0,20	0,36	0,49
	Humid	984	0,03	0,12	0,32
Sorghum	Hyper-arid	760	0,85	0,95	0,99
	Arid	3079	0,63	0,81	0,93
	Semi-arid	1133	0,31	0,58	0,79
	Dry-subhumid	844	0,13	0,37	0,51
	Humid	467	0,04	0,15	0,37
Fonio	Arid	7801	0,54	0,63	0,68
	Semi-arid	5969	0,09	0,24	0,39
	Dry-subhumid	5698	0,02	0,08	0,13
	Humid	3473	0,01	0,02	0,04
Triticale	Arid	631	0,76	0,82	0,89
	Semi-arid	326	0,49	0,69	0,83
	Dry-subhumid	187	0,05	0,12	0,23
	Humid	79	0,02	0,06	0,11
Potatoes	Hyper-arid	268	0,89	0,95	0,99
	Arid	211	0,56	0,81	0,91
	Semi-arid	181	0,23	0,60	0,79
	Dry-subhumid	177	0,15	0,40	0,66
	Humid	125	0,08	0,30	0,49
Sweet potatoes	Hyper-arid	204	0,84	0,97	1,00
	Arid	202	0,72	0,81	0,89
	Semi-arid	103	0,36	0,47	0,58
	Dry-subhumid	88	0,28	0,38	0,49
	Humid	78	0,05	0,12	0,28
Cassava	Hyper-arid	457	0,86	0,91	0,95
	Arid	494	0,73	0,85	0,87
	Semi-arid	489	0,69	0,83	0,86
	Dry-subhumid	584	0,44	0,56	0,65
	Humid	499	0,03	0,32	0,49
Yams	Arid	1295	0,62	0,75	0,86
	Semi-arid	66	0,43	0,59	0,68
	Dry-subhumid	167	0,43	0,48	0,57
	Humid	213	0,02	0,23	0,53

Crop	Climate zone	WF_{p25}	$Frac_{q5}$	$Frac_{median}$	$Frac_{q95}$
Sugar cane	Hyper-arid	88	0,84	0,96	0,99
	Arid	146	0,69	0,84	0,94
	Semi-arid	113	0,38	0,53	0,72
	Dry-subhumid	105	0,30	0,44	0,57
	Humid	102	0,03	0,19	0,43
Sugar beet	Hyper-arid	146	0,92	0,97	0,99
	Arid	143	0,72	0,83	0,94
	Semi-arid	99	0,52	0,67	0,82
	Dry-subhumid	108	0,38	0,55	0,71
	Humid	71	0,07	0,31	0,48
Beans	Hyper-arid	2410	0,80	0,93	0,98
	Arid	2100	0,61	0,80	0,91
	Semi-arid	2023	0,45	0,62	0,78
	Dry-subhumid	6688	0,07	0,46	0,69
	Humid	2132	0,03	0,13	0,57
Peas	Hyper-arid	1392	0,65	0,86	0,97
	Arid	1342	0,72	0,82	0,87
	Semi-arid	1216	0,53	0,70	0,81
	Dry-subhumid	2238	0,40	0,50	0,61
	Humid	578	0,22	0,36	0,50
Cow peas	Semi-arid	2753	0,12	0,34	0,46
	Dry-subhumid	4643	0,01	0,06	0,57
	Humid	3841	0,01	0,07	0,46
Lupins	Hyper-arid	2457	0,89	0,93	0,94
	Arid	2656	0,84	0,87	0,90
	Semi-arid	2413	0,61	0,73	0,81
Almonds	Hyper-arid	8125	0,90	0,96	1,00
	Arid	1571	0,78	0,88	0,95
	Semi-arid	1776	0,55	0,70	0,82
	Dry-subhumid	1344	0,42	0,56	0,69
	Humid	1766	0,18	0,40	0,59
Soybeans	Hyper-arid	2606	0,93	0,97	1,00
	Arid	2356	0,73	0,83	0,92
	Semi-arid	2157	0,39	0,58	0,77
	Dry-subhumid	1885	0,22	0,41	0,60

Crop	Climate zone	WF_{p25}	$Frac_{q5}$	$Frac_{median}$	$Frac_{q95}$
	Humid	1721	0,02	0,12	0,42
Groundnuts	Hyper-arid	2652	0,93	0,96	1,00
	Arid	1860	0,73	0,84	0,94
	Semi-arid	2451	0,24	0,56	0,77
	Dry-subhumid	1606	0,26	0,40	0,62
	Humid	1272	0,02	0,08	0,26
Coconuts	Semi-arid	1708	0,51	0,60	0,70
	Dry-subhumid	1519	0,44	0,54	0,63
	Humid	1168	0,11	0,36	0,55
Oil palm fruit	Humid	2797	0,03	0,15	0,30
Olives	Hyper-arid	1073	0,92	0,96	0,99
	Arid	1416	0,73	0,84	0,93
	Semi-arid	1018	0,43	0,59	0,73
	Dry-subhumid	987	0,36	0,49	0,63
	Humid	1024	0,22	0,40	0,55
Sunflower seed	Hyper-arid	3066	0,92	0,96	1,00
	Arid	1621	0,71	0,81	0,91
	Semi-arid	1827	0,40	0,61	0,75
	Dry-subhumid	1258	0,29	0,46	0,60
	Humid	937	0,02	0,11	0,37
Rapeseed	Hyper-arid	5281	0,77	0,92	0,98
	Arid	2401	0,57	0,77	0,93
	Semi-arid	2580	0,30	0,48	0,70
	Dry-subhumid	2359	0,23	0,41	0,56
	Humid	1306	0,31	0,45	0,56
Seed cotton	Hyper-arid	2419	0,90	0,95	0,99
	Arid	2867	0,67	0,84	0,94
	Semi-arid	2440	0,33	0,55	0,74
	Dry-subhumid	1210	0,23	0,36	0,54
	Humid	1006	0,03	0,11	0,32
Cabbages and other brassicas	Hyper-arid	232	0,93	0,98	1,00
	Arid	181	0,74	0,86	0,94
	Semi-arid	125	0,44	0,64	0,79
	Dry-subhumid	127	0,33	0,50	0,75
	Humid	98	0,04	0,21	0,45

Crop	Climate zone	WF_{p25}	$Frac_{q5}$	$Frac_{median}$	$Frac_{q95}$
Lettuce and chicory	Hyper-arid	88	0,74	0,89	0,95
	Arid	76	0,23	0,52	0,78
	Semi-arid	39	0,05	0,18	0,39
	Dry-subhumid	47	0,03	0,10	0,35
	Humid	57	0,02	0,07	0,24
Spinach	Hyper-arid	120	0,73	0,89	0,98
	Arid	104	0,27	0,63	0,80
	Semi-arid	95	0,10	0,37	0,65
	Dry-subhumid	93	0,05	0,45	0,61
	Humid	71	0,04	0,19	0,41
Tomatoes	Hyper-arid	145	0,85	0,93	1,00
	Arid	80	0,49	0,70	0,89
	Semi-arid	50	0,25	0,39	0,61
	Dry-subhumid	64	0,16	0,28	0,63
	Humid	58	0,05	0,22	0,54
Pumpkins, squash and gourds	Hyper-arid	190	0,81	0,90	0,98
	Arid	192	0,59	0,72	0,87
	Semi-arid	136	0,38	0,52	0,65
	Dry-subhumid	148	0,24	0,41	0,55
	Humid	137	0,03	0,15	0,40
Cucumbers and gherkins	Hyper-arid	134	0,91	0,95	0,98
	Arid	147	0,69	0,81	0,93
	Semi-arid	58	0,39	0,59	0,70
	Dry-subhumid	53	0,34	0,51	0,75
	Humid	47	0,05	0,15	0,42
Green Chillies and peppers	Hyper-arid	167	0,83	0,93	1,00
	Arid	127	0,38	0,61	0,86
	Semi-arid	45	0,09	0,23	0,62
	Dry-subhumid	92	0,06	0,20	0,63
	Humid	106	0,04	0,17	0,49
Onions	Hyper-arid	171	0,86	0,96	1,00
	Arid	134	0,57	0,74	0,89
	Semi-arid	165	0,28	0,51	0,76
	Dry-subhumid	200	0,18	0,34	0,70
	Humid	155	0,06	0,23	0,47

Crop	Climate zone	WF_{p25}	$Frac_{q5}$	$Frac_{median}$	$Frac_{q95}$
Garlic	Hyper-arid	293	0,93	0,98	1,00
	Arid	204	0,57	0,75	0,90
	Semi-arid	204	0,28	0,48	0,69
	Dry-subhumid	200	0,24	0,54	0,66
	Humid	177	0,09	0,22	0,44
Green beans	Hyper-arid	262	0,83	0,93	0,99
	Arid	192	0,52	0,72	0,84
	Semi-arid	101	0,12	0,47	0,61
	Dry-subhumid	101	0,16	0,35	0,50
	Humid	92	0,04	0,10	0,30
Green peas	Hyper-arid	346	0,86	0,92	0,96
	Arid	337	0,66	0,81	0,90
	Semi-arid	294	0,31	0,48	0,61
	Dry-subhumid	269	0,25	0,42	0,55
	Humid	278	0,13	0,34	0,58
Carrots and turnips	Hyper-arid	200	0,73	0,96	0,99
	Arid	158	0,46	0,65	0,88
	Semi-arid	80	0,23	0,55	0,77
	Dry-subhumid	87	0,20	0,47	0,68
	Humid	60	0,04	0,20	0,42
Okra	Hyper-arid	150	0,76	0,88	0,98
	Arid	126	0,53	0,69	0,80
	Semi-arid	161	0,21	0,34	0,48
	Dry-subhumid	133	0,16	0,25	0,38
	Humid	125	0,12	0,21	0,34
Bananas	Hyper-arid	409	0,95	0,98	1,00
	Arid	366	0,81	0,91	0,96
	Semi-arid	256	0,57	0,68	0,79
	Dry-subhumid	238	0,50	0,61	0,72
	Humid	239	0,08	0,45	0,63
Plantains	Hyper-arid	490	0,80	0,96	1,00
	Arid	531	0,70	0,86	0,97
	Semi-arid	679	0,48	0,66	0,85
	Dry-subhumid	523	0,36	0,53	0,65
	Humid	773	0,06	0,28	0,52

Crop	Climate zone	WF_{p25}	$Frac_{q5}$	$Frac_{median}$	$Frac_{q95}$
Oranges	Hyper-arid	381	0,89	0,95	0,99
	Arid	270	0,71	0,83	0,92
	Semi-arid	182	0,41	0,58	0,75
	Dry-subhumid	171	0,31	0,45	0,58
	Humid	167	0,08	0,30	0,48
Apples	Hyper-arid	503	0,93	0,96	0,99
	Arid	262	0,76	0,86	0,94
	Semi-arid	215	0,53	0,68	0,82
	Dry-subhumid	164	0,42	0,56	0,70
	Humid	135	0,13	0,37	0,59
Pears	Hyper-arid	713	0,93	0,95	1,00
	Arid	188	0,75	0,82	0,89
	Semi-arid	194	0,54	0,67	0,80
	Dry-subhumid	224	0,43	0,57	0,71
	Humid	185	0,07	0,37	0,61
Peaches and nectarines	Hyper-arid	1064	0,92	0,97	1,00
	Arid	442	0,79	0,87	0,94
	Semi-arid	213	0,54	0,67	0,81
	Dry-subhumid	188	0,44	0,57	0,70
	Humid	201	0,11	0,40	0,59
Raspberries	Semi-arid	1058	0,53	0,65	0,72
	Dry-subhumid	1257	0,44	0,53	0,59
	Humid	812	0,07	0,20	0,40
Grapes	Hyper-arid	432	0,92	0,97	0,99
	Arid	426	0,73	0,83	0,93
	Semi-arid	505	0,45	0,64	0,78
	Dry-subhumid	398	0,35	0,49	0,66
	Humid	382	0,12	0,35	0,54
Watermelons	Hyper-arid	202	0,92	0,97	1,00
	Arid	179	0,69	0,84	0,94
	Semi-arid	70	0,43	0,58	0,71
	Dry-subhumid	61	0,40	0,53	0,71
	Humid	58	0,05	0,17	0,45
Mangoes, mangosteens, guavas	Hyper-arid	799	0,88	0,96	0,99
	Arid	941	0,66	0,89	0,97

Crop	Climate zone	WF_{p25}	$Frac_{q5}$	$Frac_{median}$	$Frac_{q95}$
	Semi-arid	976	0,53	0,67	0,82
	Dry-subhumid	1001	0,45	0,58	0,69
	Humid	860	0,12	0,32	0,57
Pineapples	Hyper-arid	71	0,75	0,87	0,99
	Arid	81	0,34	0,56	0,79
	Semi-arid	163	0,08	0,28	0,45
	Dry-subhumid	133	0,04	0,15	0,25
	Humid	79	0,01	0,09	0,26
Dates	Hyper-arid	3129	0,92	0,97	1,00
	Arid	1702	0,78	0,87	0,94
	Semi-arid	1907	0,58	0,71	0,83
	Dry-subhumid	2528	0,45	0,60	0,69

List of Publications

Peer-Reviewed Journal Articles

van Oel, P.R., Mulatu, D.W., Odongo, V.O., Meins, F.M., **Hogeboom, R.J.**, Becht, R., Stein, A., Onyando, J.O. & van der Veen, A. (2013). The Effects of Groundwater and Surface Water Use on Total Water Availability and Implications for Water Management: The Case of Lake Naivasha, Kenya, *Water Resources Management*, 27(9): 3477-3492.

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[**Hogeboom, R.J.**, De Bruin, D., Schyns, J.F., Krol, M.S. & Hoekstra, A.Y. (*under review*). Capping Human Water Footprints in the World's River Basins.]

[**Hogeboom, R.J.**, Schyns, J.F., Krol, M.S. & Hoekstra, A.Y. (*submitted*). Global Water Saving Potential and Water Scarcity Alleviation by Reducing Water Footprints of Crops to Benchmark Levels.]

[Vanham, D., Schyns, J.F., **Hogeboom, R.J.**, Magagna, D., Medarac, H. (*under review*). The water footprint of the European Union energy sector.]

[Karandish, F., Hoekstra, A.Y. & Hogeboom, R.J. (*under review*). Effective pathways to reduce blue water consumption and pollution in agriculture: A water footprint assessment for cereals in Iran.]

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About the Author

Rick Hogeboom has a passion for water. His fascination with this vital resource led him to develop a strong background in international water management and a thorough understanding of global water consumption, scarcity and security. In his various professional roles, Rick tries to make his vision reality to positively impact the world by promoting wise water stewardship whenever and wherever he can. Science-based, practical and inclusive. Always with a big smile.

As Executive Director at the Water Footprint Network, Rick works with his team to promote research, education and knowledge development related to the water footprint concept, develop practical tools and solutions to assess and reduce water footprints, raise awareness, help shape policy, and connect businesses, governments and NGO's in their quest to a more efficient, sustainable and fair use of water.

As Programme Manager at the Wetskills Foundation, Rick leads and (co-)organize inter-cultural learning and networking events around the world. During two-week Wetskills events, students and young professionals work in mixed, inter-disciplinary teams to provide out-of-the-box solutions to real-life study cases provided by the water sector.

Rick holds a BSc degree and an MSc degree (*with distinction*) in Civil Engineering & Management from the University of Twente, with a specialization in water management.

Rick will continue his research at the University of Twente as Assistant Professor.