Safeguarding water for food and ecosystems.

Amandine Pastor

Safeguarding water availability for food and ecosystems under global change

Modelling and assessment of the role of environmental flows

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Safeguarding water availability for food and ecosystems under global change

Modelling and assessment of the role of environmental flows

Amandine V. Pastor

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Chapter 1

General introduction

1.1. Global water resources estimates during the 20th century

Water is the "elixir of life" and is not an infinite resource (Lal 2015). One of the main challenges of the 21st century is to manage natural resources so that human needs can be satisfied without harming the environment especially freshwater resources (West, Gerber et al. 2014). In terms of the global water cycle this means to manage the system in such a way that enough water is available for both food production and for environmental needs. First estimates indicate that the potentially available runoff for human use lies between 12,500 and 15,000 km^3 year^{-1} (Biemans, Hutjes et al. 2009, Rockström and Karlberg 2010, Haddeland, Clark et al. 2011). Traditionally, water scarcity assessments assume that regions experience severe water scarcity when withdrawals exceed 40 to 60% of available resources (Oki and Kanae 2006). At global scale this corresponds to 5,000 to 6,000 km^3 year^{-1}. As there are large uncertainties related to human manipulations to the global water cycle, the proposed boundary for 'safe' freshwater use were set to 4000 km^3 year^{-1} (Rockström and Karlberg 2010) which have been revised to a lower amount close to 2,100 km^3 year^{-1} (Gerten, Hoff et al. 2013). Currently, human blue water withdrawals are close to 4,000 km^3 year^{-1} (Oki and Kanae 2006, FAO 2016). Agriculture is by far the largest user of water resources and accounts for about 70% of total water withdrawals and about 92% of water consumption (Hoekstra and Mekonnen 2012, Haddeland, Heinke et al. 2014).

1.2. Population increase and rising water demand

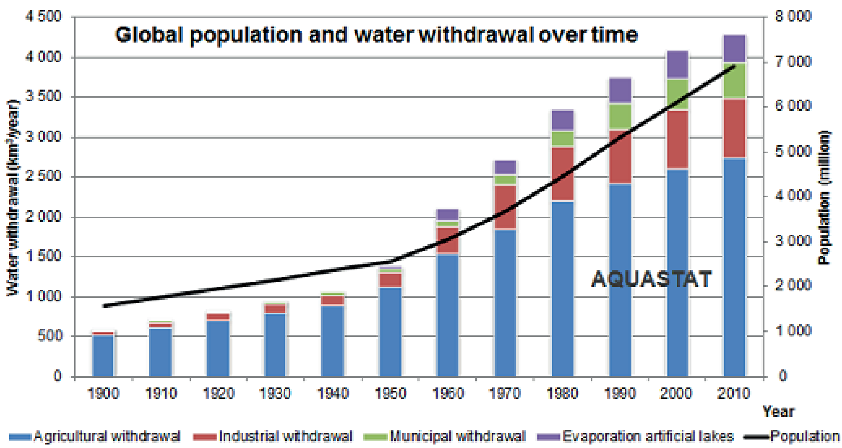


Figure 1.1. Global population and water withdrawal over time by sector (FAO 2016)

The last decades, water use has increased proportionally to population rise (Figure 1.1) (FAO 2016). Increasing population size and improving diets are likely to increase food demand in the coming decades (Stehfest, Bouwman et al. 2009, de Fraiture and Wichelns 2010) and these higher food demands are likely to increase agricultural water use (Haddeland, Heinke et al. 2014, Schewe, Heinke et al. 2014). There are however large uncertainties about how agricultural water demand and availability will change in the future. Two of the largest sources of uncertainty in future water use are probably land use change and changes in water availability caused by a changing climate. The expansion of irrigated areas is likely to increase linearly with water use and increase co;petition between water sectors (Heistermann 2006, Fischer, Tubiello et al. 2007, Biemans, Haddeland et al. 2011, Elliott, Deryng et al. 2014). Nowadays, global freshwater use exceeds long-term accessible supplies by 5 to 25 % (Viala 2008, Grantham, Viers et al. 2014) and 30% of water use is coming from groundwater of which a part of groundwater resources is already in depletion (Wada, van Beek et al. 2010, Famiglietti 2014). By 2025, population living with less than 1000 km^3 per capita will increase from 19.4% to 33.1% mainly located in North African and Mediterranean areas and in Asia (Revenga and Mock 2000). Additionally, 82% of the world's population served by upstream areas are exposed to high levels of threat especially in South-East Asia (Figure 1.2) (Green, Vörösmarty et al. 2015).

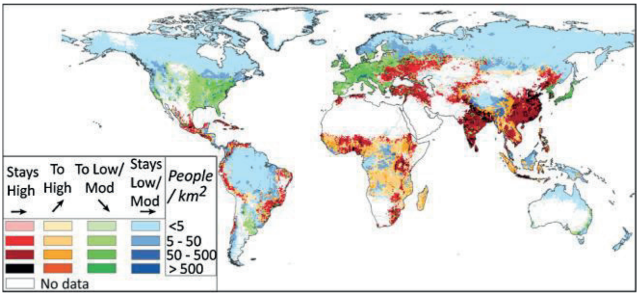


Figure 1.2. Hydrological Water Stress index on freshwater provisions supporting downstream users (from (Green, Vörösmarty et al. 2015))

1.3. Intensification of global water cycle under global change

Climate change is likely to affect water availability through changes in runoff patterns; and might impact plant water demand through changes in evapotranspiration and elevated CO_2 concentrations (Palmer, Reidy Liermann et al. 2008, Field, Barros et al. 2014, Davis, O'Grady et al. 2015). Until now, few studies were combining impact of land use change and climate change on future water use (Elliott, Deryng et al. 2014, Bonsch, Popp et al. 2015). Most studies looking at the impact of climate change on future water availability and use, were conducted separately from land use change analyses (Barnett, Adam et al. 2005). Vice versa, studies on future land use change usually ignored water availability (Foley, DeFries et al. 2005). Haddeland et al. 2014 predicted that land use change would be as important as the impact of climate change on water use except in Asia and West of US where irrigation demand would outpace the impact of climate change. Tilman, Balzer et al. (2011) showed that to double future food demand, 0.2 to 1 billion land would need to be cleared for agriculture use.

To define accurate estimates of future irrigated areas, it is essential to define how much water is available and which share of water is needed by the environment. Moreover future freshwater ecosystems requirements and food security need to be sustained under global change.

1.4. Trends in macro hydro-ecology

1.4.1. *River fragmentation*

Over half of the rivers are being fragmented and assuming the construction of all projected dams this number could nearly double, especially in the Amazon (Nilsson, Reidy et al. 2005, Grill, Lehner et al. 2015). Harrison, Green et al. (2016) report that 10% of water provisions in protected areas and nearly a quarter of water provisions in non-protected areas are exposed to high level of threat. Some of the biggest rivers of the world do not reach the sea anymore such as the Indus, Nile and Colorado which are heavily modified rivers (Gleick 2003).

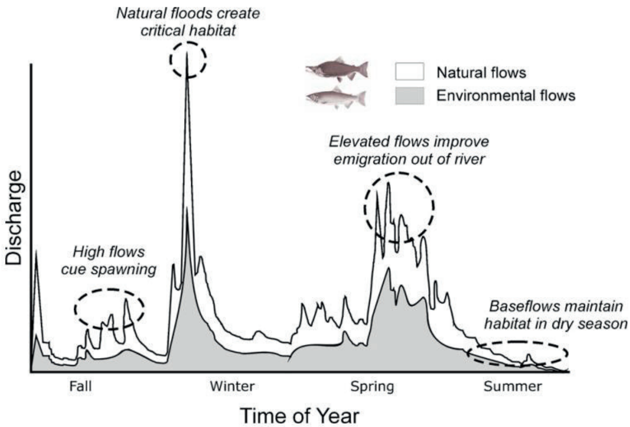


Figure 1.1. Temporal representation of EFRs mimicking natural discharge. EFRs functions are described for the Chinook Salmon (Naiman, Latterell et al. 2008).

Streamflow defines the physical pattern upon which freshwater ecosystems depends on (Poff, Allan et al. 2003, Mims and Olden 2013). Natural variability of flows such as floods and low flows are necessary for the functioning of freshwater ecosystems such as fish migration and spawning (Figure 1.3) (Dudgeon, Arthington et al. 2006). River regulation and fragmentation are already heavily affecting freshwater ecosystems with a loss of 81% of species in the last 40 years (Figure 1.4) (WWF/ZSL 2016). For example, in some heavily exploited rivers such as the Yellow river, fishing has even been prohibited due to the species extinctions of the yellow croaker and the hairtail (Liu and Diamond 2005).

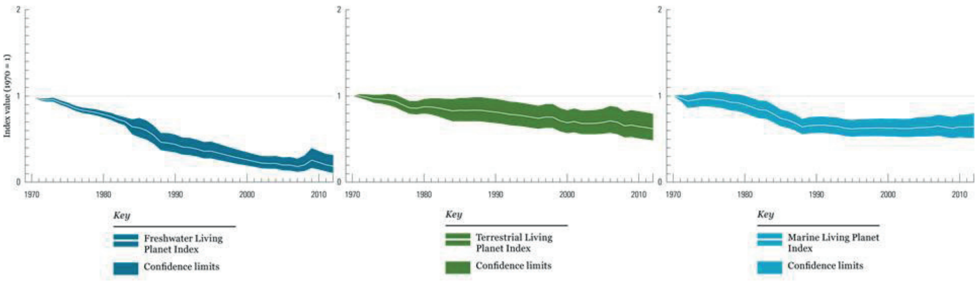


Figure 1.2. The living planet index showing the decline in freshwater (81%), terrestrial (38%) and marine (36%) species between 1970 and 2012 (WWF/ZSL 2016).

1.4.2. *Hydro-ecology in regional context*

Hydro-ecological relationships for freshwater ecosystems have been widely studied (Bunn and Arthington 2002, Bond, Lake et al. 2008, Poff and Zimmerman 2010, Baumgartner, Conallin et al. 2014, Sagouis, Jabot et al. 2016). Flow alteration can impact the densities, size, age, structure of populations, community composition and the diversity of freshwater ecosystems (Lake 2003). It was shown that food chain length (FCL) increases with drainage area and decreases with hydro-variability and intermittency (Sabo, Finlay et al. 2010). The impact of seasonal and predictable droughts on freshwater ecosystems are less likely to be harmful than supra-seasonal droughts and the recovery of species mainly depends on refugia (Lake 2003). On one side, intermittent rivers have shown the tendency to have a higher hydrological resilience than stable rivers due to frequent extreme events (Botter, Basso et al. 2013). However, resilience of freshwater ecosystems tends to decrease in conditions of repeated droughts with long duration, is is likely to be exacerbated by intense irrigation extraction (Bond, Lake et al. 2008). For example, in intermittent streams of California, it was shown that prolonged droughts caused the reduction of native species and the increase of invasive species (Bêche, Connors et al. 2009). On the other side, Nel, Roux et al. (2007) show that perennial rivers tend to have be more suitable for dam construction than intermittent rivers due to their stable flow. Perennial rivers also show a high proportion of potentially impacted ecosystems than intermittent rivers due to their high biodiversity and specie richness (Abell, Thieme et al. 2008). Finally, restoration of perennial rivers tends to be more difficult than restoration of intermittent rivers because of the presence of big infrastructure constructions (e.g. dams and reservoirs) such as in the Amazon and in the Yellow rivers.

1.4.3. *Hydro-ecology in global context*

At global scale, knowledge is lacking on "how climate, stream flow and stream temperature affect species and community composition of freshwater ecosystems ?". Progress has been made in collecting global fish database (Oberdorff, Tedesco et al. 2011, Brosse, Beauchard et al. 2013, Tisseuil, Cornu et al. 2013, van Vliet, Franssen et al. 2013). However only few studies show eco-hydrological relationships at global scale (Xenopoulos, Lodge et al. 2005). Davis, O'Grady et al. (2015) show that hydrological intensification and impact of land use are likely to increase their impact on freshwater ecosystems. Land-use change endorses longer dry season flows via irrigation use altering water quality composition and decreasing dilution flow capacity of detritus, algae and plants (Cooper, Lake et al. 2013). Furthermore, climate change is likely to impair freshwater ecosystems via intensification of the hydrological cycle including droughts, and species might migrate according to stream temperature gradient changes (Comte, Buisson et al. 2013).

1.4.4. *EF methods*

Table 1.1. Categorization of existing EF methods

Types of EFR methods	Principles	Examples
Hydrological (ad-static, rule of thumbs)	Based on hydrological parameters such as Q90 or percentage of runoff	IHA (Mathews and Richter 2007)
Hydraulic	Uses hydraulic parameter such as cross section area of a river	R2 cross method (Armstrong, Todd et al. 1999)
Habitat-simulation	Uses a key specie to define eco-hydrological conditions	Phabsim (Bourgeois 1994)
Holistic	Uses a combination of upper 3 methods and flow alteration deviation to allocate minimum standards. Expert knowledge via stakeholders' participation is included.	ELOHA (Poff, Richter et al. 2009)

In 2007, Environmental Flow Requirements (EFRs) were defined by international scientists as the required flow to sustain freshwater ecosystems and its human livelihoods (Brisbane declaration, 2007). Currently, more than 200 methods are able to calculate EFRs and these last can be classified in four categories (Table 1.1) (Tharme 2003). However most of the methods including habitat simulation and holistic methods are time-consuming and require extended ecological and hydrological data collection. Therefore, simple ad static environmental flow rules (hydrological methods) are currently often used (Smakhtin, Revenga et al. 2004, Arthington, Bunn et al. 2006, Palau 2006). However ad-static methods or rule of thumbs do not respect flow variability and ecological integrity. Temporal representation of EFRs should mimic temporal variability of natural discharge so that freshwater ecosystems can be sustained. For example, the main ecosystem functioning served by EFRs are described for the Chinook Salmon in Naiman, Latterell et al. (2008). However, it is difficult to calculate EFRs at global scale due to the lack of solid global hydro-ecological relationships. Until recently, EFRs were rarely included in global assessments or were defined with a ratio of annual water availability (Smakhtin, Revenga et al. 2004). To improve the definition of the planetary boundaries on water use this thesis aims at defining the water needed by the environment at explicit spatial and temporal scales. For that, the use of integrated models to define water resources, ecosystem requirements and future land use are necessary to anticipate a sustainable future for humans and ecosystems.

1.5. Hydrological and integrated assessment modelling

1.5.1. *Modelling approach and global hydrological models*

Making a proper assessment of future global water resources and agricultural water demand requires an extensive understanding of the global hydrological cycle and its interactions with land use, climate and humans. To fully understand the processes including future water demand and its impacts on different sectors, it is necessary to use a modelling framework which is able to integrate all driving forces leading to increased water stress.

Global hydrological models (GHMs) are useful tools to simulate water fluxes driven by diverse climate inputs such as precipitation, radiation and or temperature and soil dynamics. GHMs are usually deterministic semi-distributed models using spatial data of soil and land use. GHMs use a river routine scheme allowing the routing of runoff downstream of each catchment. The catchment is divided in sub-units (grid cells) which are laterally linked during the river routine operations (Gosling, Taylor et al. 2011). The last decades, a couple of hydrological models have been developed such as H08 (Hanasaki, Kanae et al. 2008), Watergap (Alcamo, Döll et al. 1997, Alcamo, Flörke et al. 2007), WBM (Wisser, Fekete et al. 2010), PCR (Wada, van Beek et al. 2010), and VIC (Liang, Lettenmaier et al. 1994).

1.5.2. *LPJmL*

I used the newly developed Lunt-Potsdam-Jena managed land model (LPJmL) with includes a global hydrological model and a dynamic vegetation model (Sitch, Smith et al. 2003, Gerten, Schaphoff et al. 2004, Bondeau, Smith et al. 2007). The model was extended with a river routine and a reservoir module to simulate required flow for irrigation purposes (Figure 1.5) (Rost, Gerten et al. 2008, Biemans, Haddeland et al. 2011). Discharge validation was also computed around a set of precipitation data (Biemans, Hutjes et al. 2009). In our research, LPJmL was used because of its unique coupling of vegetation and hydrology models. This model allows the simulation of river discharge under natural (pristine river flow without any modification of land use) and river flow in actual conditions (simulated flow after anthropogenic water extractions). LPJmL is also used for carbon impact studies thanks to simulations of natural vegetation (Plant functional types (PFTs)) and crop yields (Crop Functional Types (CFTs)). Climate change impact studies can therefore also be performed with this model.

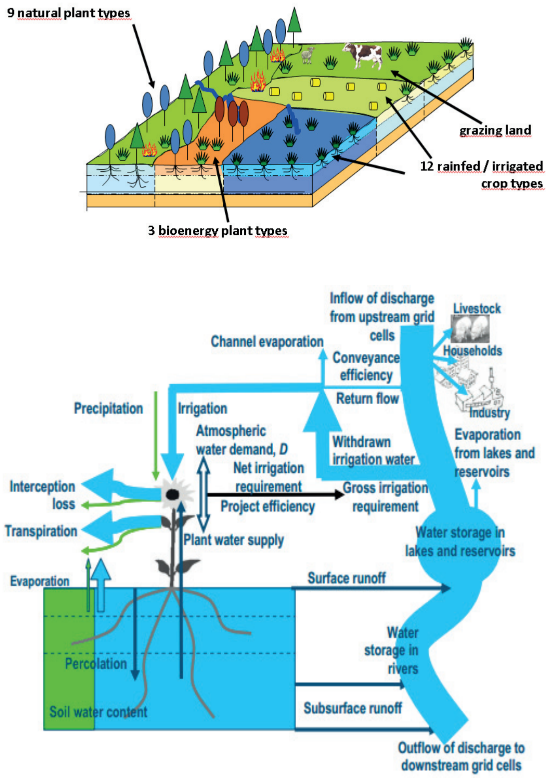


Figure 1.3. Representation of LPJmL with the dynamic module (upper plot - adapted from (Hoff 2011) and river routine scheme (down plot) (Rost, Gerten et al. 2008)

1.5.3. *Integrated assessments: IMAGE / GLOBIOM*

While global vegetation and hydrological models can simulate biophysical processes such as vegetation development and river runoff, they do not account directly for the impact of socio-economic and climate change drivers on land use allocation. Therefore, the last decades, global economic models have been used to assess plausible futures for agriculture market and food security. The aim of these models is to assess alternative socio-economic, climate change and bioenergy scenarios on land-use dynamics (Verburg, Schot et al. 2004, Lampe, Willenbockel et al. 2014). Economic models are classified into computable general equilibrium (CGE) models and partial equilibrium (PE) models (Robinson, Meijl et al. 2014). CGE require general production/cost functions for all sectors, whereas PE models emphasize on a more comprehensive description of technology in agriculture. Agriculture inputs and their impact on land use and crop yields are included in PE models. Agricultural commodities are exogenously included into PE models such as detailed cropland allocation. PE models usually require exponential demand curves with constant price and income elasticity to simulate land use allocation.

1.5.4. *GLOBIOM Global optimization model*

In this study, the Global Biosphere Management Model (GLOBIOM) was used, this model is a global recursive dynamic PE model that combines the agricultural, bioenergy and forestry sectors (Havlík, Schneider et al. 2011). It is a tool to provide analyses to policymakers on global issues such as land use repartition between different production sectors. GLOBIOM covers 30 economical world regions. Inside a region the model include specific data on agricultural and forestry production as well as bioenergy production. GLOBIOM includes 20 crops, livestock activities and forestry commodities. Crop yields are spatially determined by the Environmental Policy Integrated Climate Model (EPIC) (Liu, Williams et al. 2007). The model accounts for competition via the use of price and productivity changes. The model can address various land-use topics such as bioenergy policy impacts, climate change adaptation etc. In GLOBIOM, management options are defined by sector such as input requirements, production cost and efficiency. The best system is chosen based on the most-cost efficient system which is constrained by land availability and price of resources (Valin, Havlík et al. 2013, Havlík, Valin et al. 2014, Nelson, Valin et al. 2014).

1.5.5. *Integrated assessments*

The proposed work will focus on studying how future agricultural and environmental water demand will impact future land use under global change. A modelling exercise coupling a global hydrological model (LPJmL) with an optimisation model (GLOBIOM) will be done to allow for a fully integrated simulation of the impact of climate and water restrictions on agricultural water use and production. This modelling system will be used to evaluate a combination of socio economic and climate change scenarios. However, to simulate accurate future irrigated area, GLOBIOM needs to be extended with detailed spatial inputs on water availability for humans and ecosystems.

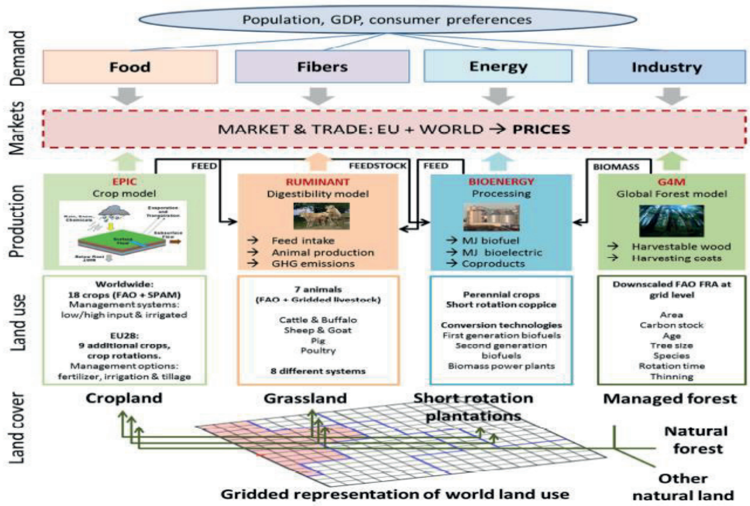


Figure 1.4. GLOBIOM (Valin, Havlík et al. 2013)

In this thesis, water constrains (water availability and EFRs) were simulated with the global hydrological model LPJmL to be included in the Integrated Assessment Model GLOBIOM (Figure 1.6). With this coupled system it is possible to estimate the impact of socio-economic development and land use changes on hydrology on the one hand, and evaluating the impact of water shortage on crop yields and agricultural development on the other hand. Although integrated assessment models like GLOBIOM, often have a simpler representation of the physical system than Earth System Models, their integrative character makes them more suitable to study complex issues with many drivers like in this study. As part of the proposed research the quantitative tools to evaluate the water availability and agriculture water use will be improved.

1.6. Research objectives and questions

The overall objectives of the thesis are to find solutions to meet future food demand while sustaining water demand for freshwater ecosystems. For that, the research question are divided into four parts:

1.6.1. *Research questions*

Q1. How to define EFRs with an explicit spatial and temporal representation at global scale? (Ch. 2)

Q2. Where and when EFRs are not met due to natural climate variability and anthropogenic water use? (Ch 3)

Q3. What is the impact of implementing EFRs on food security? (Ch 4)

Q4. How to satisfy both food security and water demand for ecosystems under global change? (Ch 5)

After reviewing existing EFRs methods, a global method with improved spatial and temporal resolution will be improved (Ch. 2). The new method has the aim to be scientifically solid and easy to be implemented in GHMs and global assessments (Q1). With the newly developed method, hot-spots of water deficiency for ecosystems due to natural climate variability and anthropogenic reasons will be assessed (Q2, Ch. 3)). Then, EFRs will be included in the river routing of LPJmL to simulate its impact on food production (Q3, Ch. 4) and finally, a coupled version of LPJmL with GLOBIOM will be used to optimize future land use and trade under global change (Q5, Ch. 5).

1.6.2. *Objectives*

To answer the defined research questions, this thesis will focus on the following objectives:

1. Reviewing existing EF methods and designing an improved spatially and temporally EF method applicable globally (Ch. 2).
2. Making an improved estimation of the planetary boundaries for freshwater use, with quantification of environmental flow deficit (or the water transgression due to natural climate variability and anthropogenic water uses) at global and regional scales (Ch. 3 and 4).
3. Analyse how implementing EFRs (water demand for the environment) would affect regional and global water use and its impact on food security (Ch. 4 and 5).
4. Study the combined impacts of socio-economics drives and climate change on global future water use and land use and how international bilateral trade could compensate for eventual food production loss (Ch. 4).
5. Evaluate options to promote sustainable water withdrawals (Ch. 4,5 and 6)

1.6.3. ***Thesis outline and methodology***

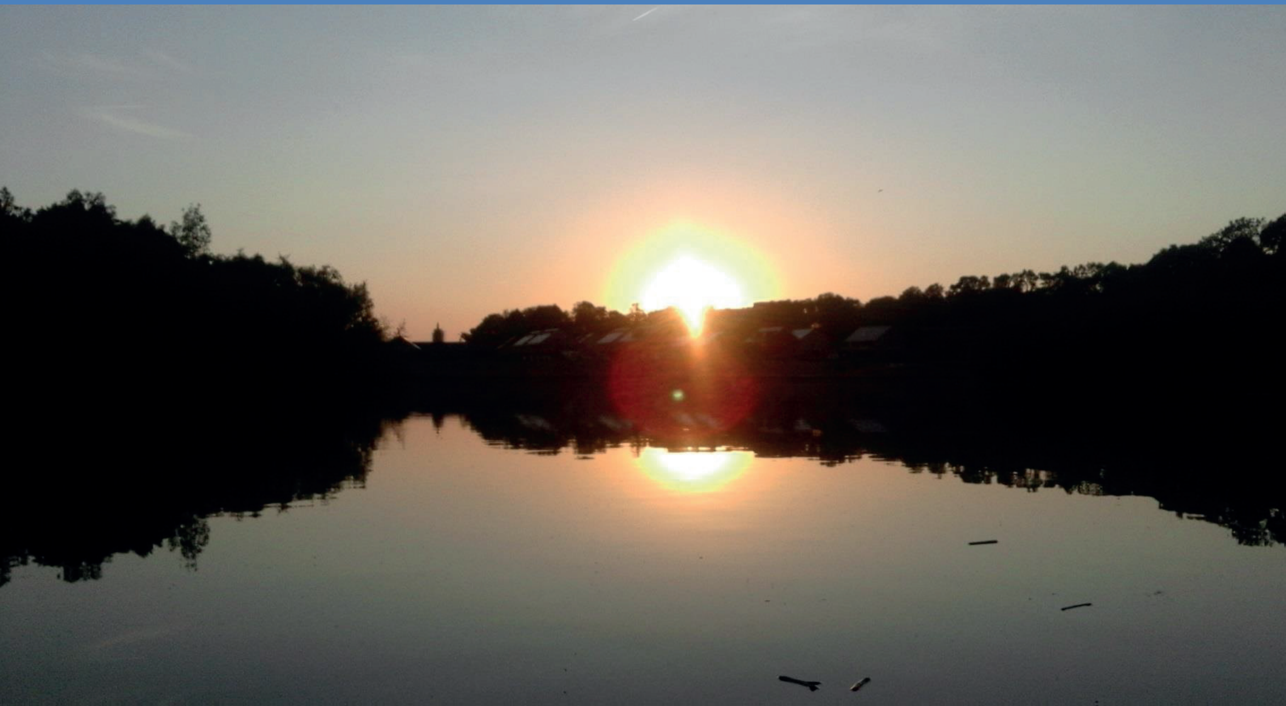
The four research questions are addressed in four scientific publications (Ch 2 to Ch 5). This thesis is composed of six chapters including the introduction (Ch 1) and the synthesis (Ch 6). Each chapter answers one or two research questions and objectives and gives inputs to the following chapters (Table 1.2). In Chapter 2, I propose a new EF model to estimate EFRs globally: the Variable Monthly Method. With this method, spatial and temporal representation of EFRs are improved by integrating natural flow variability components. The VMF method is compared with four other hydrological methods and validated with 11 local study cases using eco-hydrological data. The five hydrological methods were applied and tested with the GHM: LPJmL. One of the requirements of the new method was to be easily applicable in GHMs and in global food and water assessments. In Chapter 3, the impact of natural climate variability and anthropogenic water use on satisfying EFRs were assessed. I present the concept of "environmental flow deficit" as the water that is not met to satisfy freshwater ecosystems. EF deficit is discussed in terms of frequency, timing and duration in different river types. Hot-spots of high EF deficit are presented at global scale. In chapter 4, 3 EF methods used in Chapter 1 (VMF, Tessman and Smakhtin) are implemented in the river routine of LPJmL. The aim is to evaluate the impact of EFR implementation on food production. The impact of EFR implementation on water use and food security is tested with a sensitivity analysis. In Chapter 5, new water constraints (on water availability and EFRs) were included in the GLOBIOM model (integrated assessment model). With GLOBIOM the potential impacts of climate change (RCP8.5), socio-economic changes (SSP2) and EFR implementation (VMF method) were assessed on future land-use. Four scenarios are tested: INVEST (no limit on water use), EXPLOIT (full exploitation of water locally), ENVIRONMENT (water availability restricted by EFRs) and ENVIRONMENT+ (water availability highly restricted by EFRs). A new repartition of rainfed and irrigation areas by 2050 at global scale is proposed. To maintain food security and EFRs, trade-offs between land use, water use and bilateral trade are presented. Finally, in chapter 6, the main findings of the thesis are presented, the uncertainties of the study such as choice of methods. The impact of the research in a broad scientific context and in the agenda of SDGs is discussed. Finally, an outlook of possible future research in the global eco-hydrology world is proposed in terms of managing EFRs and water use for irrigation. Finally, I conclude with the main key finding of this study to alleviate food security and overuse of water under global change.

Table 1.2. Schematic representation of methodological framework

Research question	Chapter	Input data	Model	Simulation
Q1: How can we represent EFRs at global scale?	Ch 2: Accounting for EFRs at global scale	Precipitation Radiation Timeline: 1960-2000 Temporal scale: daily time-step Spatial scale: 0.5°	LPJmL model	Natural flow (pristine flow) Calculation of Environmental Flow Requirements (EFRs) with 5 methods and 11 case studies
Q2: Where and when EFRs are not met globally?	Ch. 3: Environmental flow deficit at global scale			Natural flow (pristine flow) Actual flow (including total withdrawals) Environmental Flow (EF) deficit (timing, duration, frequency and origin)
Q3: What is the impact of implementing EFRs on irrigated food production?	Ch. 4: Reconciling irrigated food production with environmental flows in face of SDG agenda	Precipitation Radiation Timeline: 1980-2010 Temporal scale: daily time-step Spatial scale: 0.5°	LPJmL model + EF module	Natural flow (pristine flow) Actual flow (including total withdrawals) ± EFRs Water withdrawal per river basin ± EFRs Food production ± EFRs
Q4: How to satisfy both food security and water demand for ecosystems under global change?	Ch. 5: Balancing future agriculture demand and water restrictions with global trade	Socio-economic inputs from SSP2 Climate change data for RCP8.5 - 2 GCMs Timeline: 2000-2050 Temporal scale: 10 year time-step Spatial scale: 2°	GLOBIOM model + water constrains	Future water use for different sectors Future land use repartition (rainfed, irrigated) Future food production

Chapter 2

Accounting for environmental flow requirements in global water assessments



As the requirement for water for food production and other human needs is growing, quantification of Environmental Flow Requirements (EFRs) is necessary to assess the amount of water needed to sustain freshwater ecosystems. In this study, five environmental flow (EF) methods for calculating EFRs were compared with 11 case studies of locally assessed EFRs. We used three existing methods (Smakhtin, Tennant, and Tessmann) and two newly developed methods (the Variable Monthly Flow method (VMF) and the Q_{90}_Q_{50} method). All methods were compared globally and validated at local scales while mimicking the natural flow regime. The VMF and the Tessmann methods use algorithms to classify the flow regime into high, intermediate, and low-flow months and they take into account intra-annual variability by allocating EFRs with a percentage of mean monthly flow (MMF). The Q_{90}_Q_{50} method allocates annual flow quantiles (Q_{50} and Q_{90}) depending on the flow season. The results showed that, on average, 37% of annual discharge was required to sustain environmental flow requirement. More water is needed for environmental flows during low-flow periods (46–71% of average low-flows) compared to high-flow periods (17–45% of average high-flows). Environmental flow requirements estimates from the Tennant, Q_{90}_Q_{50}, and Smakhtin methods were higher than the locally calculated EFRs for river systems with relatively stable flows and were lower than the locally calculated EFRs for rivers with variable flows. The VMF and Tessmann methods showed the highest correlation with the locally calculated EFRs (R^2=0.91). The main difference between the Tessmann and VMF methods is that the Tessmann method allocates all water to EFRs in low-flow periods while the VMF method allocates 60% of the flow in low-flow periods. Thus, other water sectors such as irrigation can withdraw up to 40% of the flow during the low-flow season and freshwater ecosystems can still be kept in reasonable ecological condition. The global applicability of the five methods was tested using the global vegetation and hydrological model LPJmL. The calculated global annual EFRs for fair ecological conditions represent between 25 and 46% of mean annual flow (MAF). Variable flow regimes such as the Nile have lower EFRs (ranging from 12 to 48% of MAF) than stable tropical regimes such as the Amazon (which has EFRs ranging from 30 to 67% of MAF).

Based on:

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2.1. Introduction

One of the main challenges of the 21st century is to manage water and other natural resources so that human needs can be satisfied without harming the environment. By 2050 agricultural production is projected to increase by 70% compared to 2000 so that enough food can be provided for 9 billion people (Alexandratos and Bruinsma 2012). This future increase in food production will result in an increase in water demand (Biemans, Haddeland et al. 2011). As a result, about 60% of the world's population could face surface water shortages from lakes, rivers, and reservoirs ((Rockström, Falkenmark et al. 2009).

Today, 65% of global rivers are considered as being under moderate-to-high threat in terms of human water security and biodiversity (Vorosmarty, McIntyre et al. 2010). Since the beginning of the 20th century, more than 800,000 dams have been built to facilitate increased withdrawals, and currently 75% of the main rivers are fragmented (Richter, Mathews et al. 2003, Biemans, Haddeland et al. 2011). Some large river basins, like the Yellow River Basin, have seen their flow reduced by almost 75% over 30 years due to increasing water withdrawals (Changming and Shifeng 2002). Moreover, in many rivers, flows are not enough to sustain the deltas. This is the case in, for example, the Colorado and the Nile (Gleick 2003). In other river basins such as the Amazon or Mekong, flow deviation and dam construction are planned with consequent losses in fish biomass and to the detriment of biodiversity (Ziv, Baran et al. 2012).

River flow is the main driver involved in maintaining a river's good ecological status (Poff, Richter et al. 2009). Human activities have impaired freshwater ecosystems through excess water withdrawal, river pollution, land use change (including deforestation), and overfishing (Dudgeon 2000). Stressors associated with reduction in flow and water quality are the most obvious causes of biodiversity hazard as they directly degrade aquatic ecosystems (Pettit, Froend et al. 2001, Doupé and Pettit 2002, O'Keeffe 2009, Vorosmarty, McIntyre et al. 2010). Between 1970 and 2000 freshwater ecosystem species declined by 36% (Loh, Collen et al. 2010). With increasing future demand for water for agriculture, industry, and human consumption, freshwater ecosystems will be under great pressure in the coming decades. Climate change is also expected to affect river discharge and river ecosystems, with decreased low-flows and rising river temperatures being predicted (Vliet, Ludwig et al. 2013).

Over the last ten years, global hydrological models (GHMs) have been used to evaluate global water assessments (GWAs) (Arnell 2004, Alcamo, Flörke et al. 2007, Hanasaki, Kanae et al. 2008, Rockström, Falkenmark et al. 2009, Hoff, Falkenmark et al. 2010, van Beek, Wada et al. 2011). Global water assessments have highlighted regions with current and future water scarcity. However, most of these studies have neglected the water required by the environment, also known as environmental flow requirements (EFRs), with only a few studies attempting to include some aspects of environmental flows (Smakhtin, Revenga et al. 2004, Hanasaki, Kanae et al. 2008, Hoekstra and Mekonnen 2011, Gleeson, Wada et al. 2012).

According to the Brisbane Declaration (2007), "environmental flows describe the quantity, quality and timing of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems." Environmental flows can also be defined as the flows to be maintained in rivers through management of the magnitude, frequency, duration, timing, and rate of change of flow events (O'Keeffe 2009). Environmental flow (EF) methods should take into account the natural variability of river flow by allocating different flow components in order to maintain and/or restore freshwater ecosystems (Acreman, Dunbar et al. 2008) and riparian vegetation (Pettit, Froend et al. 2001, Bunn and Arthington 2002, Kingsford and Auld 2005, O'Keeffe and Quesne 2009, Bejarano, Nilsson et al. 2011). For example, sustaining a minimum flow is usually important to guarantee the survival of aquatic species, while flood flows are usually crucial for sediment recruitments and for the maintenance of wetlands and floodplains (Bunn and Arthington 2002, Hugues and Rood 2003, Acreman, Dunbar et al. 2008, Bigas 2012). Disrupting a stable flow regime can also impair aquatic ecosystems and favor proliferation of invasive species and more generalist fish species (Marchetti and Moyle 2001, O'Keeffe 2009, Poff, Richter et al. 2009).

There have been major efforts to define EFRs based on eco-hydrological relationships in individual rivers (Richter, Warner et al. 2006) but there has been limited upscaling of individual methods to global or regional scales. In general, eco-hydrological relationships are far from being linear at local scales. Therefore, defining eco-hydrological relationships at global scale is even more challenging. In a recent study, a world database on fish biodiversity has been developed (Oberdorff et al., 2011) and in other studies, some efforts are shown in relating global eco-hydrological responses to flow alteration (Xenopoulos, Lodge et al. 2005, Iwasaki, Ryo et al. 2012, Yoshikawa, Yanagawa et al. 2013). However, it is still difficult to correlate freshwater biodiversity with flow metrics at both local and global scale (Poff and Zimmerman, 2010).

In current global water assessments, EFRs are almost always neglected or included in a very simplified way. Because EFRs are being ignored, the quantity of water available for human consumption globally is probably being overestimated (Gerten, Hoff et al. 2013). To be able to assess where there will be enough water available to allow a sustainable increase in agricultural production, there must be full acknowledgment that nature itself is a water user and limits must be set to water withdrawals in time and space. In the absence of global eco-hydrological assessments, we assume that locally calculated EFRs are the best estimates of the ecological needs of a river and that they can be used for validation of global EF methods.

The aim of this study is to compare different EF methods and their applicability in GHMs to set limits to water withdrawals. In this paper, we first present an overview of existing EF methods. Secondly, we present the selection and development of five hydrological EF methods that were compared with locally calculated EFRs in 11 case studies. In a final step we present a comparison of the five hydrological EF methods applied to a global hydrological and vegetation model LPJmL (Gerten, Schaphoff et al. 2004, Bondeau, Smith et al. 2007).

2.2. Review of environmental flow methods

2.2.1. *Locally defined methods*

There are currently more than 200 environmental flow methods (Tharme 2003). EF methods are classified into four types: hydrological methods; hydraulic rating methods; habitat simulation methods; and holistic methods (Table 2.1). These EF methods were mainly developed at river or basin scale, either in the context of flow restoration projects (Richter, Warner et al. 2006) or for assessing the ecological status of rivers at a regional, national, or continental level, as per, for instance, the Water Framework Directive 2000/60/EC (Council 2000).

Table 2.1. Description of regional environmental flow methods such as the DRIFT (Downstream Response to Imposed Flow Transformations), DRM (Desktop Reserve Model), and New England AquaticBase-Flow (ABF) methods. .

Type of EF method	Data input	Example	Sources
Hydrological	Long-term datasets of unregulated or naturalized daily flows (> 20 years)	Tennant, Tessman, IHA, RVA, DRM, ABF	(Tennant 1976, Tessmann 1980, Richter, Baumgartner et al. 1997, Armstrong, Todd et al. 1999, Smakhtin, Shilpakar et al. 2006, Richter 2010, Babel, Dinh et al. 2012)
Hydraulic	Flow velocity, river crossing area	R2Cross method	(Armstrong, Todd et al. 1999)
Habitat-simulation	Flow velocity, river cross section, dataset of a fish specie	PHABSIM, IFIM	(Bovee 1986, Bovee, Lamb et al. 1998, Milhous 1999, Capra, Sabaton et al. 2003)
Holistic	Combination of hydrological, hydraulics, ecological, and social sciences (expert knowledge)	Building block method (BBM), ELOHA, DRIFT	(King and Louw 1998, Hughes 2001, Bunn and Arthington 2002, Arthington, Bunn et al. 2006, Poff, Richter et al. 2009)

2.2.2. *Hydrological methods*

Hydrological methods are usually based on annual minimum flow thresholds such as *7Q10*, the lowest flow that occurs for seven consecutive days once in ten years (Telis and District 1992) or Q_{90}, where the flow exceeded 90% of the period of record (NGPRP 1974). The first step in determining the desired level of ecological condition of a river is often via, for instance, the Tennant method (Tennant

1976) which defines seven classes ranging from severe degradation (F) to outstanding ecological conditions (A). According to the Tennant classification, a different percentage of the annual flow is allocated during the high-flow and low-flow seasons. The Tessmann method (1980) considers intra-annual variability by allocating percentages of monthly flow to calculate EFRs depending on the different flow seasons (high-, intermediate-, or low-flow months). Richter, Baumgartner et al. (1997) divided the Indicators of Hydrological Alteration (IHA) into five groups: magnitude, timing, duration, frequency, and rate of change; they determined some environmental flow components (EFCs), such as the maintenance flow, during dry and normal years (Mathews and Richter 2007). Alternatively, EFRs can be calculated using a method called the Range of Variability Approach (RVA) which in non-parametric analyses calculates EFRs as a range between the 25^{th} and 75^{th} monthly flow percentile (Armstrong, Todd et al. 1999, Babel, Dinh et al. 2012) or in parametric analyses as a range of mean monthly flow (±standard deviation) (Smakhtin, Shilpakar et al. 2006, Richter, Davis et al. 2012). The advantage of hydrological methods is that they are simple and fast EF methods for use in preliminary assessments or when ecological datasets are not available. They can easily be implemented at local and global scale depending on their level of complexity and the availability of hydrological data.

2.2.3. *Hydraulic methods*

Hydraulic methods are used at a local scale when river cross-section measurements are available. They can ultimately complement habitat simulation models for calculating the area necessary for fish habitat survival (Espegren 1998, Gippel and Stewardson 1998). The inconvenience of this method is that it requires river hydraulic measurements and is specific to each river section.

2.2.4. *Habitat simulation methods*

Habitat simulation models make use of ecohydrological relationships. They are based on correlations between hydraulic parameters such as flow velocity and certain species of freshwater ecosystems. For example, the Instream Flow Incremental Methodology (IFIM) requires datasets of river discharge, river temperature, and fish species richness (Bovee 1986, Bovee, Lamb et al. 1998). The Physical Habitat Simulation Model or PHABSIM (Milhous 1999) is based on the theory that the quality and quantity of physical habitat are related to the environmental needs of aquatic ecosystems of each life stage (Jowett 1989, Palau and Alcázar 2010). The advantage of habitat simulation models is that they take into consideration riverine ecosystems; however, data collection can be costly and time-consuming. Habitat simulation models also need to be recalibrated when they are applied to a different region and are usually species-specific (McManamay, Orth et al. 2013).

2.2.5. *Holistic methods*

Holistic methods are a combination of hydrological, hydraulic, habitat simulation methods, and expert knowledge (Poff, Richter et al. 2009, Shafroth, Wilcox et al. 2009). For example, the Building Block Model is a well-documented method for estimating EFRs at local or basin scale (King and Louw 1998, Hugues and Rood 2003, Tharme 2003, King and Brown 2010). The building block method supports the principle that maintaining certain components of the natural flow is of fundamental importance. The flow blocks encompass low flows and high flows, both of which are defined for normal and dry years. The Desktop Reserve Model (Hughes, 2001) provides estimates of these building blocks for each month of the year. River streams are classified (from A to D) according to their level of flow alteration, and the decision regarding ecological flows will depend on those classes (Kashaigili, McCartney et al. 2007). The Downstream Response to Imposed Flow Transformations is a model that uses 10 ecologically relevant flow categories such as wet and dry seasonal low flows, periodicity of floods, and flow variability via flow duration curves (Arthington, Rall et al. 2003). Finally, the Ecological Limits of Hydrologic Alteration (ELOHA) approach includes both a scientific and a social approach. The method uses a hydrological classification of natural flow regime types and calculates the rate of flow alteration between natural and actual conditions. The second part of the method uses ecohydrological relations to determine EFRs, and expert knowledge is included in the final part of the assessment. Holistic methods require time to collect large amounts of data and are difficult to upscale due to the different freshwater ecosystems, flow regime types, water management techniques, and different socio-economic contexts. The strength of holistic methods is that they promote interdisciplinarity where hydrological, geo-morphological, biological, and sociological methods are used to find the best compromise between water demand for freshwater ecosystems and water requirements for anthropogenic purposes (Poff, Richter et al. 2009).

2.2.6. *Global environmental flow methods*

Global EF methods are defined using hydrological methods (section 2.1.1) because of the lack of global ecohydrological data (Richter, Warner et al. 2006, Poff and Zimmerman 2010). Smakhtin, Revenga et al. (2004) developed the first EF method for application within global hydrological models. Smakhtin, Revenga et al. (2004) defined four potential ecological river statuses: pristine, good, fair, and degraded, following the recommendations of the Department of Water Affairs and Forestry (DWAF 1997). In the study of Smakhtin, Revenga et al. (2004), a low-flow component is defined for each ecological river status such as Q_{50} for good ecological status, Q_{75} for moderate ecological status, Q_{90} for fair conditions, and NA for degraded river status. Smakhtin, Revenga et al. (2004) developed a method assuming a fair ecological status of global rivers, and Q_{90} was defined as the base flow requirement. To determine high-flow requirements, the global river discharge was classified according to a river's base flow index, which determines the river flow regime. Hanasaki, Kanae et al. (2008) developed an EF method considering intra-annual variability based on global monthly river flows. They defined four different river regimes: dry, wet, stable, and variable. For each class, they determined EFRs as a percentage of mean monthly flow (MMF) depending on the flow regime type (from 10 to 40% of MMF). EFRs are also determined with a fair ecological status based on the Tennant method (Hanasaki, personal communication). Hoekstra and Mekonnen (2012) evaluated monthly EFRs by applying the presumptive environmental flow standard defined by Richter, Davis et al. (2012). Although Hoekstra, Mekonnen et al. (2012) limited water consumption to 20% of total discharge, this did not imply that 80% of the total discharge was unavailable; they show, however, the period of the year in which net water availability fails to meet water demand. In another recent global water assessment, EFRs were defined as the monthly flow quantile Q_{90} in the PCR-GLOBWB model (Gleeson, Wada et al. 2012). In this study, locally calculated EFRs were assumed to be the best estimates of EFRs for validating global hydrological methods. We therefore selected five hydrological EF methods and compared them with 11 locally calculated EFRs cases so as to have a simple and reliable global EF method that takes into account intra-annual variability.

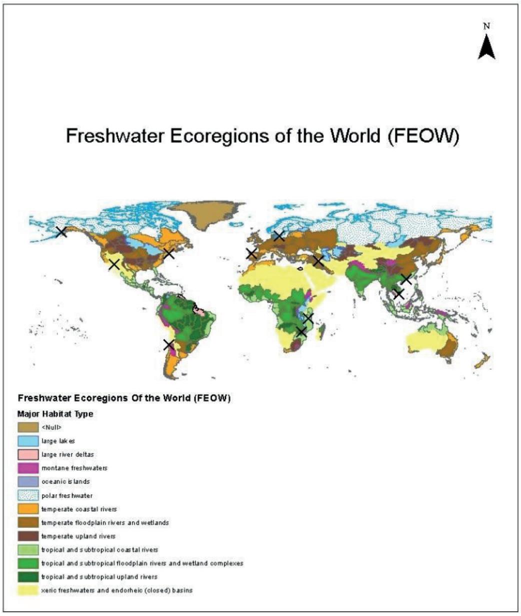

Figure 2.1. Location of 11 case studies where environmental flow requirements (EFRs) were locally defined.

2.3. Methods

2.3.1. *Selection of case studies*

Eleven case studies were selected according to their types of locally defined EF methods, river flow regimes, geo-localizations, and Major Habitat Types (MHTs) (Table 2.1, Figure 2.1). Major Habitat Types such as temperate coastal rivers and large river deltas are described in the Freshwater Ecoregions Of the World (FEOW), which classify global rivers into 426 freshwater ecoregions (Abell, Thieme et al. 2008). We chose this classification because it is more robust than a simple global river classification, which is usually based on climate zones and/or river discharge (Haines, Finlayson et al. 1988, McMahon, Peel et al. 2007). MHT classification is based on riverine species biodiversity, endemism, and river fragmentation. The description of the geo-localization of the case studies is described in Table 2.2 and Figure 2.1. In our selection of 11 case studies, five sub-groups of MHTs (xeric, temperate, tropical, and polar) were represented by at least two case studies. Five out of six continents were represented by at least one or two case studies. The type of flow regimes of the different case studies varied between stable and variable flow regime. Finally, the choice of case study was restricted to methods focusing on riverine ecosystems, such as habitat simulation, and/or hydrological methods, based on daily flow datasets.

Table 2.2. Description of geographic coordinates of the case studies and their hydrological datasets.

Case studies	Latitude	Longitude	Daily flow data used in case studies	Daily flow data used in this study
Bill William River, USA (Shafroth et al. 2009)	34.23	-113.60	Pre-dam data (1940-1965)	GRDC 4152120
Ipswich River, USA (Armstrong et al. 1999)	42.57	-71.03	Ipswich flow data (1961-1995)	20 years LPJmL simulation without landuse and irrigation (PNV run)
Silvan River, Spain (Palau and Alcázar, 2010)	42.37	-6.63	Natural flow data (1980-1998): no flow regulation	Dataset from the authors
Osborne River, Zimbabwe (Symphorian et al., 2003)	-18.75	32.25	Naturalized flow data (1961-1973)	Dataset from the authors
Voijm Dam, Sweden (Renofalt et al., 2010)	62.80	17.93	Pre-dam data (1909-1940)	Dataset from the authors
Newhalen River, Alaska (Estes 1998)	59.25	-154.75	Pre-dam data (1951-1986)	USGS 153000000
Hong Kong, China (Niu and Dudgeon, 2011)	22.27	113.95	Natural flow data (2007-2008)	20 years LPJmL simulation without landuse and irrigation (PNV run)
La Gna River, Vietnam (Babel et al., 2012)	10.82	107.15	Pre-dam data (1977-1999)	Dataset from the authors
Great Ruaha River, Tanzania (Kashaigili et al., 2007)	-7.93	37.87	Pre-dam data (1958-1973)	20 years LPJmL simulation without landuse and irrigation (PNV run)
Huasco River, Chile (UICN, 2012)	-28.43	-71.20	Historical data (1975-1988)	Dataset from the authors
Sharh Chai River, Iran (Yasi et al., 2012)	37.70	45.32	Pre-dam data (1949-2004)	Dataset from the authors

2.3.2. ***Hydrological datasets***

Hydrological datasets of individual case studies were obtained from the Global Runoff Data Centre (available at http://grdc.bafg.de) or from the authors of the case studies (Table 2.2). Mean monthly flows were calculated with historical datasets of 8–30 years to represent the "natural" or "pristine" ecological conditions of the river. In other cases, such as in the Ipswich River case study and the Hong Kong case study, a 20-year average of simulated natural monthly flow was used (section 3.6).

2.3.3. ***Hydrological indexes***

The analyses were all computed over a 40-year time period (from 1961 to 2000) to take inter-annual variability into account. The flow regimes of the selected case studies were analyzed using several hydrological indicators and river classification. To compare the case studies, we calculated some hydrological flow indexes such as the base flow index (BFI) and a hydrological variability index (HVI) as follows in Eq. (1) and Eq. (2):

$$BFI = \frac{Q90}{MAF} \qquad (1)$$

$$HVI = \frac{Q25-Q75}{\mathrm{Q50}} \qquad (2)$$

where: Q_{90} – the annual flow which is equaled or exceeded for 90% of the period of record, *MAF* – the mean annual flow, Q_{25} – the annual flow which is equaled or exceeded for 25% of the period of record, Q_{75} – the flow which is equaled or exceeded for 75% of the period of record, and Q_{50} – the flow which is equaled or exceeded for 50% of the period of record. All our calculations are in $m^3\ s^{-1}$. Finally, we classified our case studies with their respective number of high-flow (HF), intermediate-flow (IF) and low-flow (LF) months. HF is defined as MMF ≥ 80% of MAF, IF is defined as MMF ≥ 40% of MAF, and MMF < 80% of MAF, and LF is defined as MMF < 40% of MAF (Table 2.3).

Table 2.2. Inter-comparison of hydrological indicators of the case studies.

Case studies	Major Habitat Type (Abell et al., 2009)	Environmental flow method type[1]	MAF[2] (LF[3]-HF[4])	BFI[5]	HVI[6]	Nb. high-flow months	Nb. intermediate months	Nb. low-flow months
Bill William River, USA (Shafroth et al. 2009)	Xeric freshwater	4. HEC-EFM	2.7 (0.8-5.3)	5.3	2	6	0	6
Sharh Chai River, Iran (Yasi et al, 2012)	Xeric freshwater	1. GEFC (class C)	5.3 (1.6-12.7)	21.1	3.3	4	1	7
Ipswich River, US (Armstrong et al. 1999)	Temperate coastal river	2. R2Cross method	265 (120-556)	22.6	1.3	5	2	5
Silvan River, Spain (Palau and Alcázar, 2010)	Temperate coastal river	3. RHYHABSIM (class B)	0.7 (0.3-0.9)	21.5	0.9	7	2	3
Osborne Dam, Zimbabwe (Symphorian, Madamombe et al. 2003)	Temperate coastal river	1. Hugues method (class B)	39.7 (25.2-55.8)	43.6	0.6	5	5	2
Huasco River, Chile (Pouilly and Aguilera 2012)	Temperate coastal river	3. PHABSIM	6.2 (5.3-8.9)	80.6	0.2	12	0	0
Voijm Dam, Sweden (Renofalt et al., 2010)	Polar freshwater	4. Expert knowledge	39 (16.3-71)	51.3	0.7	6	2	4
Newhalen River, Alaska (Estes, 1998)	Polar freshwater	1. Tennant (fair/degrading class)	284 (98.1-544.3)	21.5	2.2	5	2	5
Hong Kong, China (Niu and Dudgeon, 2011)	Tropical floodplain	3. Macroinvertebrates sampling (degrading and outstanding classes)	1119 (317-1921)	12	1.6	6	2	4
La Gna River, Vietnam (Babel et al., 2012)	Tropical and subtropical coastal river	1. RVA approach (*Q25-Q75*)	133.5 (49.4-251.3)	15.4	1.7	5	1	6
Great Ruaha River, Tanzania (Kashaigili, McCartney et al. 2007)	Tropical and subtropical coastal river	1. Desktop Reserve Model (class C/D)	245 (45-524.4)	6.4	4.3	5	1	6

1. Environmental flow method type: 1. hydrological, 2. hydraulic 3. habitat simulation, 4. holistic,
2. MAF: Mean Annual Flow [$m^3 s^{-1}$]
3. LF: Low-flow average calculated as the average flow when MMF>MAF [$m^3 s^{-1}$]
4. HF: High-flow average calculated as the average flow when MMF≤MAF [$m^3 s^{-1}$]
5. Base flow index: Q_{90}/MAF (see Eq.1)
6. Hydrological variability index: $(Q_{25}-Q_{75})/Q_{50}$ (see Eq. 2)

2.3.4. *Description of the case studies*

The hydrological description of the 11 case studies is shown in Table 2.3. The first case is the Bill Williams River, located in Arizona, USA, which is classified as the xeric freshwater habitat type and characterized by a long low-flow season (more than 6 months) with a low Base Flow Index (BFI=5.3%). The second case is the Sharh Chai River, which also belongs to the xeric freshwater habitat type. It is characterized by a long period of low-flow (about 6 months) and by a high BFI (21%). Four temperate coastal rivers were then selected: the Ipswich River in the USA, the Silvan River in northwest Spain, the upstream flow of the Osborne River in Zimbabwe, and the Huasco River in Chile (Table 2.3; Figure 2.1). These all have relatively stable flow regimes with a strong base flow index (BFI≥20%) and a hydrological variability index (HVI<1). Two case studies were selected in the polar freshwater habitat types: the Voijm River in Sweden and the Newhalen River in Alaska, both rivers being characterized by a strong BFI of 51% and 22%, respectively. Finally, three case studies are located in tropical floodplains and coastal habitat types: a stream near Hong Kong in China, the Gna River in Vietnam, and the Great Ruaha River in Zimbabwe. These are all characterized by a monsoon season of 3-4 months with a low BFI (between 5 and 15), with the Great Ruaha River being characterized by the strongest variability index (4.3). As mentioned in section 2, case studies were selected according to whether EFRs were calculated with EF methods using ecological datasets and/or daily flow datasets. Three case studies used EF methods with eco-hydrological relationships such as PHABSIM, RHYHABSIM, and an empirical relationship between macroinvertebrate survival and river flow. One case study (Swedish case) used a holistic approach by including expert knowledge. One case study used a hydraulic method based on the river cross section in order to assess suitable habitat area for fish habitat (R2 cross method). In five case studies, hydrological methods were used to determine EFRs at local scale. Those methods were developed and validated with statistical analyses of daily flow datasets (e.g., GEFC, Hugues method, Tennant, Desktop reserve model).

2.3.5. *Selection of global environmental flow methods*

In the absence of global eco-hydrological relationships, we assumed that locally calculated EFRs were the best estimates for determining EFRs and were thus used for validation of global hydrological EF methods. In this study, we selected three existing hydrological EF methods and developed two new hydrological EF methods that were first compared with the locally calculated EFRs and then implemented in a GHM. The aim was to select and design methods that could be easily implementable in global hydrological models. We excluded EF methods that use daily flows as inputs (e.g., the Hanasaki method) because GHMs are mainly validated on a monthly or annual time scale (Döll, Kaspar et al. 2003, Portmann, Siebert et al. 2010, Werth and Güntner 2010, Biemans, Haddeland et al. 2011, Pokhrel, Hanasaki et al. 2011). The three selected existing EF methods were the Tennant, Smakhtin, and Tessmann methods. The algorithms of the Smakhtin and Tennant methods were adjusted from annual to monthly time-step in order to compare EFRs with monthly irrigation requirements in future water assessments. We therefore divided the river hydrograph into low-/high-flow months and defined EFRs algorithm for each flow season (high-flow or low-flow months). For example, in the Smakhtin method, low-flow requirements (LFRs) were allocated during low-flow months and high-flow requirements (HFRs) were allocated during high-flow months. By including intra-annual variability in our EF methods, we were improve the representation of EFRs compared with EF methods that give an annual flow threshold.

2.3.6. *Design of new EF methods*

Two of the five EF methods were newly developed for the purpose of this study (Table 2.4). One method is based on annual flow quantiles (the $Q_{90}\_Q_{50}$ method) and the other method is based on average monthly flows (the VMF method). We chose to develop a purely non-parametric method ($Q_{90}\_Q_{50}$), which uses flow quantiles to allocate minimum instream flow during the high-flow and low-flow seasons. EFRs are calculated using the allocation of the annual flow quantile (Q_{90}) during the low-flow season; the innovation in this method is that the minimum flow threshold was adapted during the high-flow season by allocating the annual flow quantile (Q_{50}) instead of (Q_{90}), based on the study of Allain and El-Jabi (2002). Flow quantiles were determined based on long-term average monthly flows between 1961 and 2000. We also developed a parametric method: the Variable Monthly Flow (VMF) method. This method follows the natural variability of river discharge by defining EFRs on a monthly basis as in the Tessmann and Hoekstra methods, except that the VMF method adjusts EFRs according to flow season. The VMF method was developed to increase the protection of freshwater ecosystems during the low-flow season with a reserve of 60% of the MMF and a minimum flow of 30% of MMF during the high-flow season. The VMF method allows other water users to withdraw water up to 40% of the MMF during the low-flow season. In all the EF methods except the VMF method and the Tessmann methods, the low-flow season was determined when the MMF was below mean annual flow (MAF) and the high-flow season when MMF was above MAF. In two of the five methods, intermediate flows were determined for a smooth transition to be made between high-flow and low-flow months (Table 2.4).

Table 2.4. Description of tested hydrological environmental flow methods with MAF (the Mean Annual Flow), MMF (the Mean Monthly Flow), Q90 (where the flow exceeded 90% of the period of record), and Q50 (where the flow exceeded 50% of the period of record). HFRs, IFR,s and LFRs are used for high, intermediate, and low-flow requirements, respectively.

Hydrological season	Smakhtin (2004)	Tennant (1976)	Q_{90}_Q_{50} (this study)	Tessman (1980)[b]	Variable Monthly Flow (this study)[b]
Determination of low-flow months	MMF<= MAF	MMF<= MAF	MMF<= MAF	MMF<= 0.4*MAF	MMF<= 0.4*MAF
Low-flow requirements (LFRs)	Q90	0.2* MAF	Q90	MMF	0.6*MMF
Determination of high-flow months	MMF>MAF	MMF>MAF	MMF>MAF	MMF>0.4*MAF & 0.4*MMF>0.4*MAF	MMF> 0.8*MAF
High-flow requirements (HFRs)	0 to 0.2*MAF[a]	0.4*MAF	Q50	0.4*MMF	0.3*MMF
Determination of intermediate-flow months	-	-	-	MMF>0.4*MAF & 0.4*MMF<= 0.4*MAF	MMF> 0.4*MAF & MMF<= 0.8*MAF
Intermediate-flow requirements (IFRs)	-	-	-	0.4*MAF	0.45*MMF

a. If Q90>30%MAF, HFRs=0,
If Q90<30% and Q90>20%, HFRs=7%MAF,
If Q90<20% and Q90>10%, HFRs=15%MAF,
If Q90<10%, HFRs=20%MAF.

b. Only the Tessmann and the Variable Monthly Flow methods require intermediate-flow determination, as their methods are based on monthly flows. The other methods (Smakhtin, Tennant, and Q_{90}_Q_{50}) only allocate EFRs in high- and low-flow seasons.

2.3.7. *Ecological conditions*

At global scale, there is no dataset indicating the level of ecological condition of rivers; nor is there a dataset with the desired ecological status of rivers worldwide. The decision on the ecological status of any river is part of an international consensus between water managers, governments, and environmental scientists. The five hydrological methods were defined with various ecological condition levels. For instance, the Smakhtin method was defined with fair ecological status, while other methods such as the Tessmann method did not define the desired ecological status but allocated at least 40% of MMF to the river. VMF was defined to reach fair ecological status with a minimum monthly flow allocation of at least 30% MMF and a higher restriction during low-flow months. We excluded methods that used good ecological conditions, such as Hoekstra, Mekonnen et al. (2012) because our aim was to validate an EF method based on locally calculated EFRs with fair-to-good ecological conditions. Finally, our focus was to improve the temporal algorithms of EF methods to restrict other water users at monthly time-steps.

2.3.8. *Validation of EF methods*

The performance of the five hydrological methods was tested against the locally calculated EFRs using the efficiency coefficient *R2* from Nash and Sutcliffe (1970). In extremely dry conditions (MMF<$1 m^3 s^{-1}$), there was no environmental flow allocation.

2.3.9. *Description of the global hydrological model LPJmL and simulations*

The global application and comparison of different EF methods require the simulation of "pristine" river discharge. For that, the Lund-Potsdam-Jena managed land (LPJmL) model was used to simulate river flow globally at a spatial resolution of 0.5' by 0.5' on a daily time step. The CRU TS 2.1 global climate data (1901–2002) was used to drive the model. LPJmL was initially a dynamic global vegetation model simulating water and carbon balances for natural vegetation (Sitch, Smith et al. 2003, Gerten, Schaphoff et al. 2004). LPJmL is different from other GHMs such as VIC (Liang, Lettenmaier et al. 1994) and HO8 (Hanasaki, Kanae et al. 2008) in that it has been extended with a crop model (Bondeau, Smith et al. 2007, Fader, Rost et al. 2010), with a river routine that simulates water withdrawal from rivers and lakes (Rost, Gerten et al. 2008), and more recently with the integration of a dam and reservoir module (Biemans et al., 2011).

Simulations were computed from 1901 to 2001 with a spin-up phase of 1,000 years for carbon and water balance. A simulation was run for naturalized river flow by using exclusively potential natural vegetation (PNV). EFR calculations were always computed with natural flows obtained from historical datasets or from simulated naturalized flow datasets. All the analyses were done on a monthly time step. In order to compare EF methods globally, the ratio of monthly EFRs to natural monthly flow was used to show the intra-annual variability of EFRs in space and time. Calculations are shown on an annual basis and for two months, January and April, averaged from 1961 to 2000. We also compared the annual ratio of EFRs for the natural flow of different river basins by giving a range of annual EFRs for the five hydrological methods.

2.4. Results

2.4.1. *Comparison of global environmental flow requirements per case study*

The overall annual average of EFRs across the 11 case studies and five methods represent 37% of MAF (Figure 2.2; Table 2.5). The range of EFRs defined locally in the case studies is from 18 to 63% of MAF, while the range of EFRs among the global EF methods is from 9 to 83% of MAF. On average, low-flow requirements represent 46–71% of mean low flows, while high-flow requirements represent 17– 45% of high-flows (Table 2.5). Low-flow requirements are usually higher than high-flow requirements relative to mean annual flow when the low-flow season is longer than four months. The correlation between the EFRs calculated with the five selected methods and the locally calculated EFRs are shown in Figure 2.3. Among the EF methods used, all the simulated EFRs were highly correlated with the locally calculated EFRs. The Tessmann and VMF methods recorded the highest correlation coefficient (R^2=0.91), while the Smakhtin, Q_{90}_Q_{50} , and Tennant methods showed a correlation (R^2) of 0.86–0.88.

Table 2.5. Comparison of annual average of environmental flow requirements (EFRs) per method and per case study (EFR: Environmental flow requirements, LFR: Low-flow requirements, HFR: High-flow Requirements). EFR is expressed as a percentage of mean annual discharge of river in "natural" conditions; LFR is expressed as a percentage of mean annual low-flow; HFR is expressed as a percentage of mean annual high-flow.

Case studies	MHT class (Abell et al., 2009)	EFR case study (LFR and HFR)	Variable Monthly Flow (LFR and HFR)	Smakhtin (LFR and HFR)	Tennant (LFR-HFR)	Tessmann (LFR-HFR)	Q_{90}_Q_{50} (LFR-HFR)	Average all EFR results (average LFR- average HFR)
Bill William River, USA (Shafroth et al. 2009)	Xeric freshwater	63 (133 -48)	33 (46-30)	12 (18-11)	27 (67-18)	46 (72-40)	6 (18-3)	46 (48-26)
Sharh Chai River, Iran (Yasi, Karimi et al. 2012)	Xeric f reshwater	51 (42-53)	35 (56-30)	19 (70-15)	27 (66-17)	50 (90-40)	19 (70-13)	33 (66-28)
Ipswich River, USA (Armstrong et al., 1999)	Temperate coastal river	25 (56-12)	35 (47-30)	25 (50-14)	27 (44-19)	49 (60-30)	37 (44-19)	33 (46-17)
Silvan River, Spain (Palau and Alcázar, 2010)	Temperate coastal river	34 (58-28)	34 (50-30)	26 (54-20)	33 (56-28)	46 (73-40)	77 (89-74)	43 (63-37)
Osborne Dam, Zimbabwe (Symphorian, Madamombe et al. 2003)	Temperate coastal river	46 (84-13)	32 (44-27)	44 (73-26)	27 (34-24)	46 (66-35)	59 (73-53)	44 (62-29)
Huasco River, Chile (Pouilly and Aguilera 2012)	Temperate coastal river	34 (30-42)	30 (30-30)	81 (94-56)	25 (23-28)	44 (47-44)	83 (94-64)	54 (53-45)
Voijm Dam, Sweden (Renofalt et al., 2010)	Polar freshwater	20 (18-21)	34 (45-30)	51 (123-28)	28 (48-22)	48 (72-40)	69 (123-52)	43 (71-32)
Newhalen River, Alaska (Estes, 1998)	Polar freshwater	18 (27-14)	35 (53-30)	20 (62-15)	32 (58-21)	30 (88-40)	50 (63-29)	30 (59-25)
Hong Kong, China (Niu and Dudgeon, 2011)	Tropical f loodplain	48 (77-44)	53 (50-30)	19 (42-16)	30 (71-23)	40 (82-40)	53 (42-54)	38 (67-32)
La Gna River, Vietnam (Babel, Dinh et al. 2012)	Tropical and subtropical coastal river	53 (50-54)	35 (52-30)	28 (31-9)	28 (54-21)	48 (75-40)	38 (42-38)	39 (51-32)
Great Ruaha River, Tanzania (Kashaigili, McCartney et al. 2007)	Tropical and subtropical coastal river	22 (19-22)	33 (54-30)	15 (35-12)	28 (109-19)	46 (92-40)	19 (58-17)	25 (61-19)
Average per method		37 (43-28)	40 (48-30)	31 (59-20)	32 (57-22)	40 (74-39)	43 (65-38)	37 (56-34)

The results show that while there is no unique method fitting a unique habitat type, two of the five methods (VMF and Tessmann) performed better than the three other methods (Smakhtin, Q_{90}_$Q_{50,}$ and Tennant). On average, the Tennant method allocated about 10% less water than the locally calculated EFRs. The Tessmann method was in general higher than the locally calculated EFRs (+24%), especially in polar freshwater ecosystems. The Smakhtin and Q_{90}_Q_{50} methods allocated less water than recommended in xeric freshwater and tropical freshwater ecosystems (variable flow regimes) and allocated more water than recommended in polar freshwater ecosystems (stable flow regime). Finally, the VMF method was the closest to the locally calculated EFRs (about 10% above average). The five methods gave lower EFRs estimates than the locally calculated EFRs in xeric freshwater ecosystems and higher estimates of EFRs than locally calculated EFRs in polar freshwater ecosystems. The methods that were closer to the locally calculated EFRs for xeric freshwater ecosystems were Tessmann and the VMF methods, and for polar freshwater ecosystems, the Tennant method. For temperate coastal rivers, the method closest to the locally calculated flow was the VMF method (Figure 2.2; Table 2.5).

EFRs of variable rivers accounted for more than 60% of the total annual flow during the high-flow season. For example, in the case of the Bill William river and the Iranian case studies, about 80% of the river flow occurs during the high-flow season which lasts three to five months. In the Tanzanian case, the high-flow season lasts five months during which 90% of the total flow occurs and about 80% of EFRs are allocated. The Tessmann, VMF, and Q_{90}_Q_{50} methods were in line with the locally calculated EFRs of variable rivers, but only the VMF and Tessmann methods could capture the intra-annual variability and allocated peak flows during the high-flow season (Figure 2.2; Table 2.5).

In perennial rivers, such as the Chilean case study, about 40% of the total flow occurs during the three wettest months of the year with the allocation of more than 50% of EFRs. The Tessmann, Tennant, and VMF methods were in line with the locally calculated EFRs, while the Smakhtin and the Q_{90}_Q_{50} methods allocated more water than recommended. In the Odzi River in Zimbabwe, only Tessmann and Q_{90}_Q_{50} could allocate an amount of water close to the locally calculated EFRs. In the Voijm River in Sweden, all the EF methods used were in line with the locally calculated EFRs with the exception of the timing of the peak flow, which was calculated as being two months later with the locally calculated EFRs.

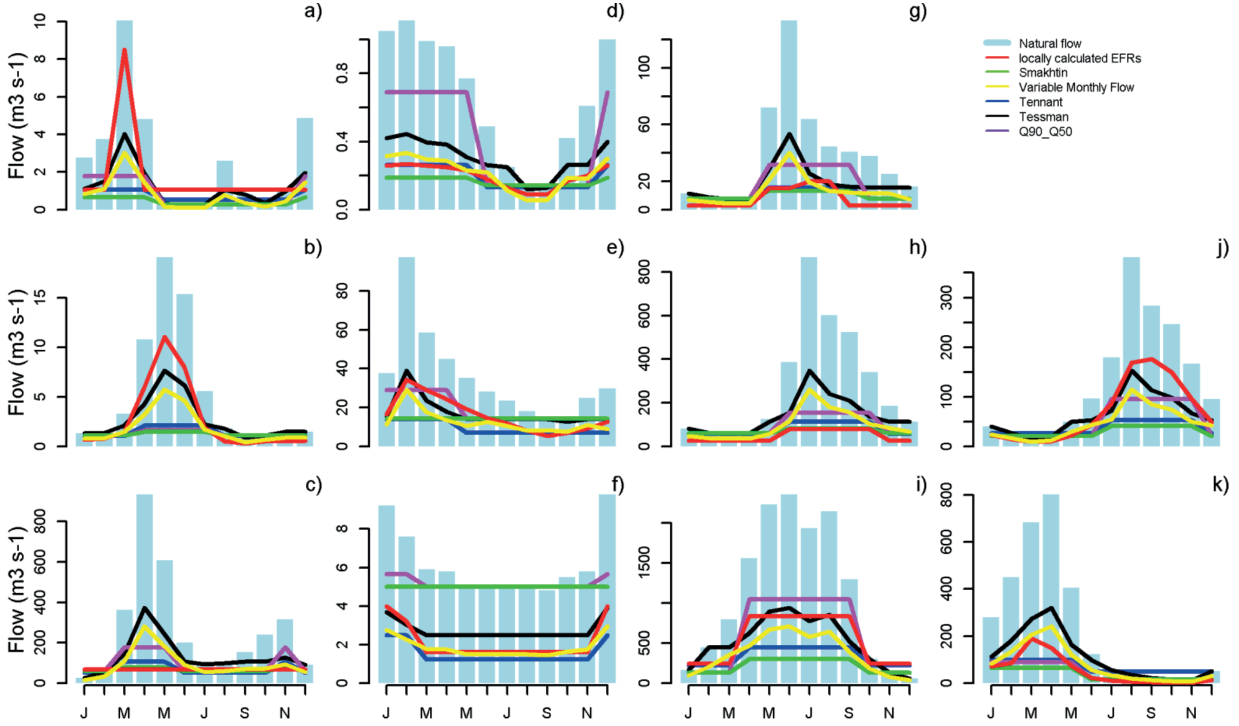

Figure 2.2. Comparison of EF methods with locally calculated EFRs in different case studies a) BWR, USA, b) Urmia Dam, Iran c) Ipswich River, US, d) Silvan River, Spain, e) Osborne Dam, Zimbabwe f) Huasco River, Chile g) Voijm Dam, Sweden, h) Newhalen River, Alaska, i) Hong Kong stream, China, j) Gna River, Vietnam, k) Great Ruaha River, Tanzania. Observed or simulated natural flows from case studies are presented in light blue, except for natural flows c) and e) which were simulated with LPJmL.

2.4.2. ***Comparison of environmental flow methods globally***

Among the methods, EFRs ranged from 25– 46% of MAF, with an increasing percentage of EFRs from the Smakhtin method to the Q_{90}_Q_{50} method. On a monthly basis, the VMF, Tennant, and Tessmann methods produced similar spatial distribution of EFRs. Similarly, the Smakhtin and Q_{90}_Q_{50} methods showed analogous spatial allocation of EFRs such as a high water allocation in perennial rivers, and a low to no-flow allocation in variable rivers. The Smakhtin method allocated 100% of MMF in the regions of the Arctic North Pole, between 40% and 60% of MMF in the tropics, and between 0 and 40% of the MMF in the rest of the world. The VMF, Tennant, and Tessmann methods allocated from at least 20 to 40% of MMF in arid regions and more than 50% of the MMF during the low-flow season. The Tennant method calculated high EFRs in the tropics (EFRs≥100% of MMF). However, the Tennant method calculated lower EFRs than the rest of the methods in temperate zones, especially during the high-flow period. In the temperate zones, the Tennant method allocated about 20% of MMF, while the VMF and Tessmann methods allocated at least 40% of MMF. A comparison of Figure 2.4 with Figures 2.5 and 2.6, shows that EFRs are more homogenous on an annual time-step compared to a monthly time-step because monthly EFRs are averaged-out. For example, the Tessmann method allocated an equal percentage of MAF worldwide and did not show strong differences between regions (Figure 2.4), whereas, on a monthly basis, the Tessmann method showed clear spatial differences in flow allocation (Figure 2.5 and 2.6).

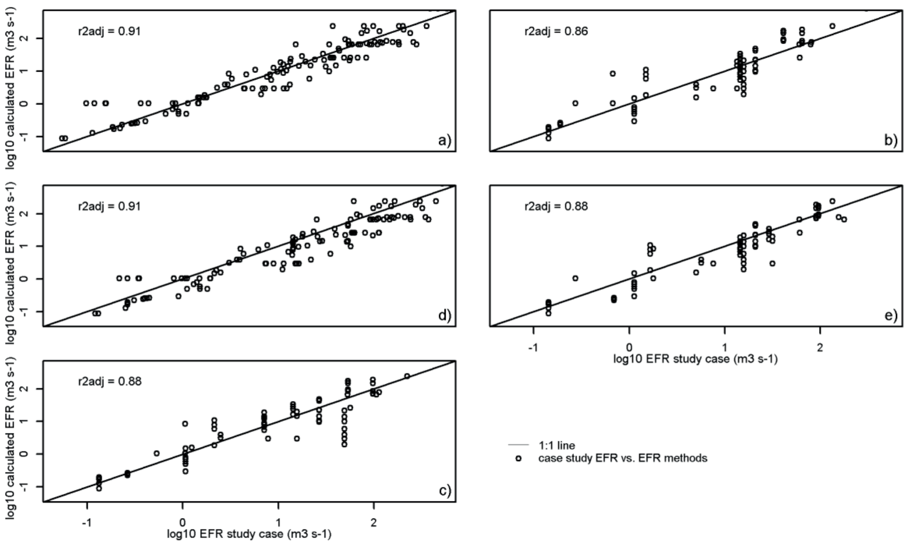

Figure 2.3. Relation between the monthly calculated EFRs and the locally calculated monthly EFRs of 11 case studies with the (a) Variable Monthly Flow, (b) Smakhtin, (c) Tessmann, (d) Q_{90}_Q_{50}, (e) Tennant methods. In each sub-figure, each dot represents EFRs for one month and for one case study.

Using a combination of the five EF methods can give a range of uncertainties of EFRs in the absence of any locally calculated EFRs. For example, we present a range of EFRs calculated with the five hydrological EF methods at the outlet of 14 of the biggest river basins. The results show that perennial rivers such as the Congo, Amazon, Rhine, and Mississippi required 30–80% of MAF (Figure 2.7). More variable river basins such as the Ganges or the Nile required 10–50% of MAF depending on the five EF methods. On average, Q_{90}_Q_{50} resulted in the highest EFRs (48% of MAF) and the Smakhtin method resulted in the lowest EFRs (26% of MAF). The VMF method allocated on average 33% of MAF, which is higher than the Tennant method (30% of MAF) and lower than the Tessmann method (43% of MAF).

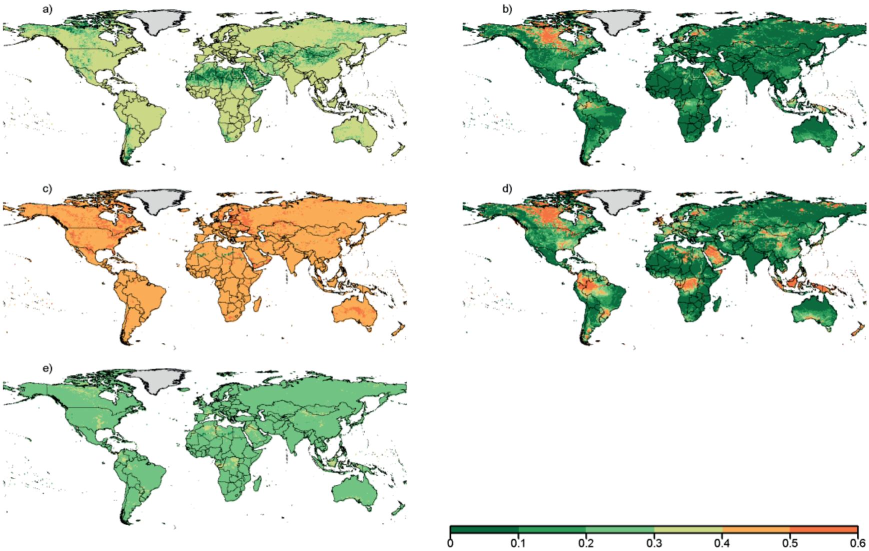

Figure 2.4. Ratios of annual environmental flow by annual natural flow within a) Variable Monthly Flow, b) Smakhtin, c)Tessmann, d) Q_{90}_Q_{50}, e) Tennant environmental flow methods.

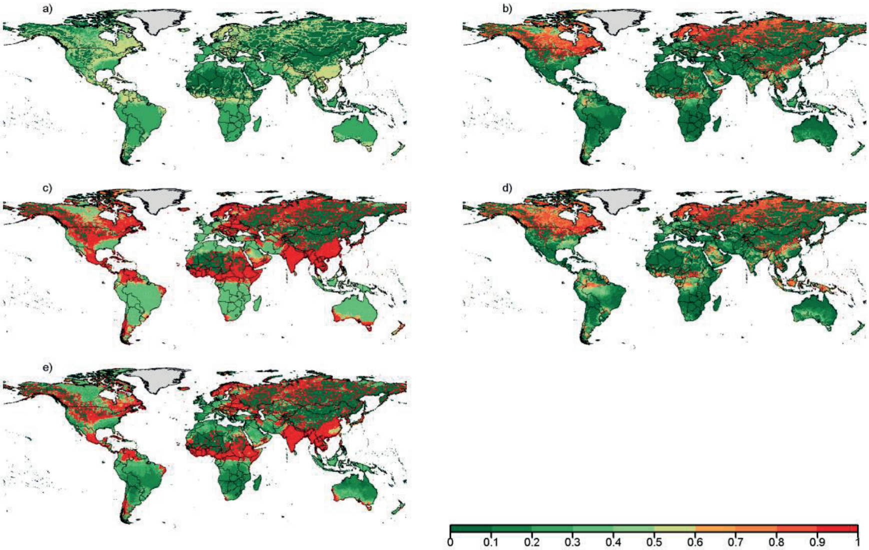

Figure 2.5. Ratios of monthly environmental flow by monthly actual flow (January) within a) Variable Monthly Flow , b) Smakhtin, c)Tessmann, d) Q_{90}_Q_{50}, e) Tennant environmental flow methods.

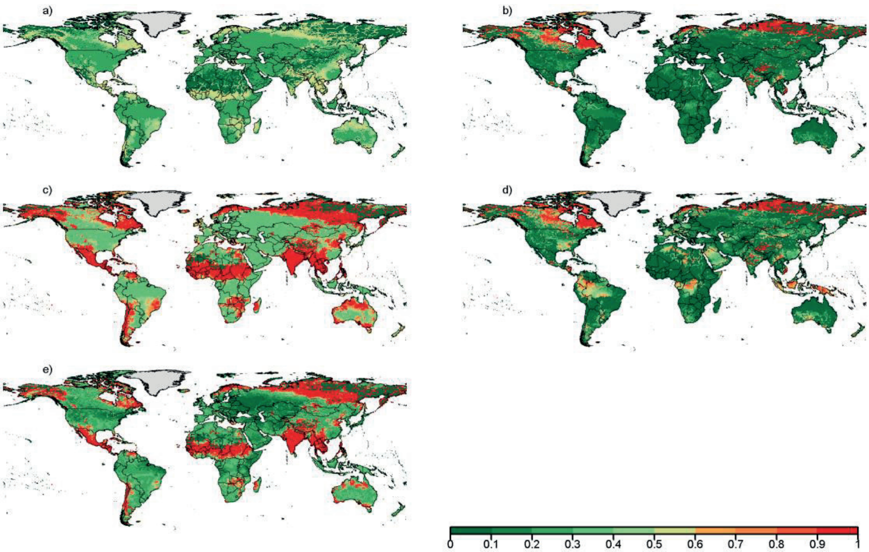

Figure 2.6. Ratios of monthly environmental flow by monthly actual flow (April) within a) Variable Monthly Flow, b) Smakhtin, c)Tessmann, d) Q_{90}_Q_{50}, e) Tennant environmental flow methods.

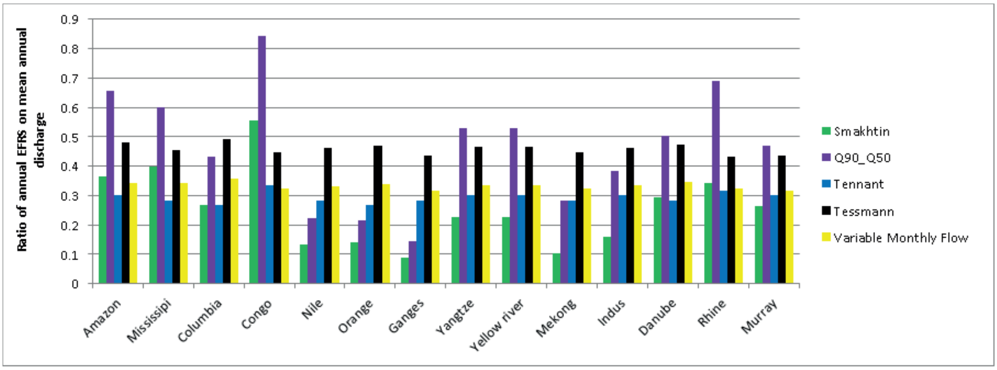

Figure 2.7. Comparison of five environmental flow methods at the outlet of 14 river basins.

2.5. Discussion

2.5.1. *Improving global environmental flow assessments*

This study compared a selection of hydrological EF methods with locally calculated EFRs while accounting for intra-annual variability. Five hydrological methods were tested using a set of local case studies to identify methods that could be used in future global water assessments. The inclusion of intra-annual variability in the algorithm of EF methods presents a significant improvement over previous global water assessments based on an annual scale (Smakhtin, Revenga et al. 2004, Vorosmarty, McIntyre et al. 2010). The VMF method was developed with the specific aim of being flexible, reliable, and globally applicable. The VMF and Tessmann showed a good correlation with the locally calculated EFRs in different case studies from a wide range of climates, flow regimes, and freshwater ecosystems (R^2 =0.91). Both methods classify flow regime into high, intermediate, and low-flow seasons and allocate monthly EFRs with different percentages of the MMF or MAF. Those two methods show some temporal and spatial improvements in the calculation of EFRs, especially for the variable flow regimes, compared with methods using annual flow thresholds such as low-flow indices (Q_{90} or 7Q10) or percentages of MAF (Palau 2006). The advantage of the VMF and the Tessmann methods is that they mimic the natural flow as suggested by Poff, Richter et al. (2009). In the case of the VMF method, the allocation of 30–60% of mean monthly flow as a degradation limit was selected because the purpose of this study was to allocate water for freshwater ecosystems in fair ecological conditions similar to Smakhtin, Revenga et al. (2004), and an allocation of 30% of MAF to calculated EFRs was widely recognized (Hanasaki, Kanae et al. 2008).

2.5.2. *Differentiation between Tessmann and VMF methods*

The main difference between the VMF and Tessmann methods is that they define high-flow, intermediate-flow, and low-flow seasons with different algorithms (Table 2.4). They allocate 60% and

100%, respectively, of MMF during the low-flow season. The relative amount of EFRs during the low-flow period is high because we considered the habitat area for freshwater ecosystems to be smaller during the low-flow season compared to the high-flow season, and we also wished to prevent the eventual impact of seasonal droughts on freshwater ecosystems (Bond, Lake et al. 2008). Saving water for the environment is thus more important during the low-flow season in order to reduce the pressure on fish survival. This assumption is confirmed in the study of Palau and Alcázar (2010) where our calculated LFRs were close to the requirements of fish habitat survival. On the other hand, water users such as industry and the irrigation sector can still withdraw up to 40% of MMF during the low-flow season (which is usually the season with the highest water demand from the irrigation sector). However, with the Tessmann method, water withdrawals are not possible during the low-flow season. During the high-flow season, allocation of HFRs does not differ significantly between the VMF and Tessmann methods because the VMF method allocates 30% of MMF and the Tessmann method allocates 40% of MMF. The determined threshold levels of the VMF method can easily be adjusted depending on the objectives of the water policy (e.g., a stricter policy on riverine ecosystems may require higher EFRs thresholds), on the ecological status of a river basin (e.g., a very altered river may never achieve the actual thresholds of VMF), and on the specific demands of other water users.

2.5.3. *Limitations of environmental flow methods based on annual thresholds*

We found that EFRs calculated with methods based on annual thresholds (Tennant, Smakhtin, and Q_{90}_Q_{50}) were lower during low-flow season and higher during high-flow season than the locally calculated EFRs, even if intra-annual adjustment was included (allocation of low and high flow requirements). Using annual flow quantiles to calculate EFRs is not appropriate for certain types of flow regime. For example, using the Q_{90}_Q_{50} or the Smakhtin method, the calculated EFRs were always lower than the locally defined EFRs of variable rivers (Figure 2.2). The Tennant method did not perform well in tropical case studies because this method was developed for temperate rivers and thus needs to be calibrated for other river types. The flow quantile methods, such as the Smakhtin and Q_{90}_Q_{50} methods, showed that in perennial rivers, as in the Chilean case, there was a higher allocation of EFRs compared to other methods (Figures 2.1, 2.3, 2.4, and 2.5). In variable rivers, the Q_{90}_Q_{50}, the Smakhtin and Tennant methods showed a lower allocation of EFRs during the high-flow season and a higher allocation of EFRs during the low-flow season compared to the locally calculated EFRs (Table 2.5). Similarly, those methods did not seem appropriate for ephemeral and intermittent rivers because they would be flooded during the dry season, which can increase the risk of invasion of exotic species (O'Keeffe 2009). Furthermore, Botter, Basso et al. (2013) agreed with the fact that allocating fixed minimum flows to erratic flow regimes was not appropriate; this is because those flow regimes have a high-flow variability and allocating a fixed minimum flow would be disproportionate to the incoming flows during the low-flow season. Furthermore, flow quantile methods are not flexible enough to be used in global assessments because the allocation of higher flow quantiles than Q_{90} such as Q_{75} and Q_{50}, as suggested in Smakhtin, Revenga et al. (2004), would allocate a flow exceeding the average monthly flow (data not shown).

2.5.4. *Limitations of our study*

The choice of EF methods for our study was limited to hydrological methods because of a lack of data on ecosystem responses to flow alterations for most river basins of the world. This lack of ecohydrological data makes it difficult to determine minimum environmental flow thresholds and tipping points of different freshwater ecosystem across the world. An improved consistent ecohydrological monitoring and forecasting system is required so that a global river classification system can be developed that would account for the sensitivity of the respective aquatic ecosystems to flow modifications (Barnosky, Hadly et al. 2012). To go beyond previous individual unrelated case studies we consistently applied different EF methods across a set of existing case studies located in different climates and freshwater ecosystems regions. Among the 200 existing EF methods, it is difficult to find case studies that quantify the sensitivity of freshwater ecosystems to change in discharge (Poff and Zimmerman 2010). It would be a great improvement if the number of case studies could be increased so that the level of validation could be increased and more accurate algorithms for each ecoregion could be found. For example, a higher allocation of flow might be required in perennial tropical rivers due to their high biodiversity index (Oberdorff, Tedesco et al. 2011) and due to their lower hydrological resilience to climate fluctuation compared to rivers with more variable flow regimes (Botter, Basso et al. 2013). We are aware of the heterogeneity of the case studies in terms of inter-annual variability and for that reason we chose case studies with a minimum of 15 years of hydrological data, which is sufficient to capture inter-annual variability, according to Kennard, Mackay et al. (2010). However, none of the EF methods used in this study explicitly accounted for daily high and low flood pulses, which often drive riparian vegetation (Shafroth, Wilcox et al. 2009).

2.5.5. *Social aspects of environmental flow requirements*

Environmental flow requirements are, in the end, a societal decision which is often made at local scales, and quantification of EFRs depends on the level of protection that is desired by society/policy. However, to develop a global EF method we need a quantification method that can be used in global water hydrological models. We decided to develop a method that reflects a level of ecosystems described as “fair ecological conditions,” as in Smakhtin, Revenga et al. (2004). Including social and political decisions in quantitative assessment is very difficult and beyond the aim of this paper. At the moment, we cannot possibly address this full new research agenda, and we have limited ourselves to the quantification of EFRs as a function of biophysical parameters. However, we acknowledge that there is a need for a more systematic EF method that would link the natural and social science fronts and would create a unifying framework for the assessment and implementation of sustainable EFRs in national water policy (Pahl-Wostl, Arthington et al. 2013). Additional efforts are required to develop a systematic regional environmental flow framework based on multi-disciplinary methods (Poff, Richter et al. 2009, Pahl-Wostl, Arthington et al. 2013). Addressing EFRs, which is part of a proactive management of river basins, is certainly a less costly solution than using reactive solutions such as river restoration measures (Palmer, Reidy Liermann et al. 2008).

2.5.6. *Refining global water assessments*

This study aimed not to refine locally determined EF methods but to identify one or several methods for global application. These new estimates of EFRs will improve global water availability assessments and allow them to better inform other water users. Moreover, expansion of irrigated lands can be carried out in a more sustainable way by accounting for current and future water availability constrained by EFRs. The VMF method estimated that at least 40% of global annual flow should be reserved for environmental flows to keep ecosystems in a fair ecological condition, but that does not necessarily mean that the remaining 60% of the water should be used by other users. It is important to acknowledge that this is a global annual average and that EFRs are highly variable depending on the region and the flow season. Finally, there is no EFR benchmark at a river basin scale. That is why we show in Figure 2.7 a range of annual EFRs at a river basin scale by using a range of the five hydrological EF methods. This approach can guide policy-makers who have to decide for EFRs values in different river basins where ecological and hydrological data are poor and it could be a starting point to implement EFRs at river basin level with "fair" ecological conditions. In future global EF assessments, it will be important to consider the inter-annual variability of flow regimes because EFRs are usually calculated on a long period average (> 20 years) and they might need to be refined for dry years (Hessari, Bruggeman et al. (2012). Regarding the use of ecological datasets, it is worth considering the delay in ecosystem response related to flow events when calculating EFRs (Sun, Yang et al. 2008).

2.6. Conclusion

We tested five different hydrological environmental flow methods for their applicability in global water assessments and found the VMF and Tessmann methods to be valid and easy methods for implementation in global hydrological models. Both methods use a simple algorithm and also take into account intra-annual variability. They improve environmental flow calculations due to their increased time resolution from an annual to monthly basis and the global applicability that this provides. The VMF and Tessmann methods were validated with existing EFR calculations from local case studies and showed good correlations with locally calculated EFRs. Quantile methods such as Smakhtin, Q_{90}_Q_{50}, and Tennant showed some disadvantages in variable flow regimes such as a lower allocation of flow than with locally calculated EFRs and flooding of the river during the dry season. The VMF and Tessmann methods fit many different flow regimes thanks to their algorithm determining low, intermediate, and high-flows; its use in future global water assessment is recommended, especially in the case of variable flow regimes. This validation increases our confidence in using this method in global water assessments. However, EFRs are likely to be adjusted if society wishes to implement a different ecological status for the river. For example, a higher flow allocation might be desired if excellent ecological conditions are required. For that eventuality, we create algorithms that are easily adjusted to societal needs. In the absence of any local calculation of EFRs, using the five hydrological methods can also provide a range of calculated EFRs at global and river basin scale in “fair” ecological conditions. Including EFRs in future global water assessments will improve the estimates of global water boundaries and will enable sustainable scenarios to be produced on the expansion of irrigated land and on the use of water for other users such as the hydropower sector.

Chapter 3

Environmental Flow Requirements deficits at global scale

Freshwater ecosystems have been degraded over the last decades due to intense anthropogenic water extraction and due to climate variability. Components of the natural flow variability have been recognized as preponderant for ecosystems survival. Therefore, Environmental Flow Requirements (EFRs) methods have been developed to maintain healthy rivers and/or to restore river flows. In this study, we used the Variable Monthly Flow (VMF) method to calculate EFRs at global scale. To anticipate a better management of river preservation and/or restoration, it is important to define the origin of the deficit and its respective timing, frequency and magnitude. We used refined spatial (0.5°) and temporal scales (monthly time-step) to define two kinds of environmental flow (EF) deficits: natural deficit when flow does not meet EFRs due to (natural) climate variability and anthropogenic deficit when flow does not meet EFRs due to water extractions. Results show that, on a global annual scale, total EF deficit represents 5% of total global discharge while at regional monthly scale total EF deficit can outpace monthly flow. Natural deficit tends to represent a higher share of total deficit in perennial flow regimes than in intermittent rivers. For example, perennial rivers with low flow alteration such as the Amazon showed an EF deficit ranging from 2 to 12% of total discharge of which natural deficit was responsible for up to 75% of the total deficit. Anthropogenic deficit tends to represent a higher share of total deficit in intermittent flow regimes than in perennial flow regimes due to high water extractions for irrigation. For example, the Indus river has a total deficit representing 130% of total discharge of which 85% are due to anthropogenic extractions. Anthropogenic deficit is therefore exacerbating the effect of natural deficit on freshwater ecosystems especially in intermittent rivers. Globally, the combined deficits represent between 16 to 36 % of total surface withdrawals showing that implementation of EFRs could conflict with about a third of global irrigation extractions from surface waters. Trade-offs between irrigation water use and environment seem to be necessary especially during dry season of intermittent rivers.

Based on: Pastor, A. V., Biemans, H., Franssen W., Gerten D., Kabat, P., and Ludwig, F.,: Environmental flow requirements deficits at global global (in revision).

3.1. Introduction

In the last decades, there was a growing concern about the declines of natural resources including freshwater biodiversity loss (Gleick 2003, WWF/ZSL 2016). Rockström and Karlberg (2010) defined planetary boundaries for human freshwater use at 4000 km^3 $year^{-1}$ meaning that if actual and future water extractions outpace this value, freshwater ecosystems are at risk. More recently, Gerten, Hoff et al. (2013) have re-adjusted the freshwater planetary boundary to a lower threshold, based on a spatially explicit assessment of environmental flow requirements as a subglobal component of this boundary: 2800 km^3 $year^{-1}$ (with an uncertainty range from 1100 to 4500 km^3 $year^{-1}$). Increased human water use has led to the flow alteration of major river basins such as the Indus and the Colorado rivers with 30% overuse of non-renewable water resources (Grafton, Pittock et al. 2013, Wada and Bierkens 2014). Flow alteration is the primary cause of freshwater ecosystems damage (Arthington, Bunn et al. 2006, Poff and Zimmerman 2010) and maintaining rivers close to their natural flow regime is necessary to fulfil ecosystem functions (Acreman and Ferguson 2009). To limit and reduce freshwater ecosystem degradation, river restoration projects have emerged with the aim to restore freshwater ecosystems to acceptable ecological conditions.

To facilitate river restoration projects and to limit water extraction from river systems Environmental Flow Requirements (EFRs) have been defined. EFRs are defined as “the quantity and quality of water required to sustain riverine ecosystems and its human livelihoods” (Brisbane declaration 2007). EFRs have been set as a priority goal in international policy agendas such as in the Sustainable Development Goals (SDGs) and at European scale in the Water Framework Directive (WFD) (Acreman and Ferguson 2009, Steffen, Richardson et al. 2015, Vörösmarty, Hoekstra et al. 2015). Some global assessments started to include EFRs but usually by only including proxies of annual flow (Elliott, Deryng et al. 2014, Bonsch, Popp et al. 2015).

A couple of recent global assessments used the newly improved Variable Monthly Method (VMF) (Pastor, Ludwig et al. 2014) which uses refined temporal scale with previously validated local study cases (Gerten, Hoff et al. 2013, Pastor, Ludwig et al. 2014, Boulay, Bare et al. 2015, Gaupp, Hall et al. 2015, Grill, Lehner et al. 2015, Sadoff 2015, Steffen, Richardson et al. 2015). To maintain rivers in a healthy state, it was shown that EFRs should represent between 20 and 50% of mean annual flow (Smakhtin, Revenga et al. 2004, Pastor, Ludwig et al. 2014). It was also shown that natural flow variability (e.g. high flow and low flow periods) should be maintained to sustain freshwater ecosystem functions (Poff, Allan et al. 2003). However, there is no evidence of how ecosystems respond to water deficit especially in terms of timing, duration and frequency. A recent study introduced the concept of ecodeficit and ecosurplus as to show the impact of dams and reservoirs on seasonal regulation of ecological flow regimes (Vogel, Sieber et al. 2007). Gordon, Peterson et al. (2008) show that ecosystems are very subjective to regime shift and that ecosystem collapse is usually responding with a non-linear response to extreme events. I

At global scale, it is still difficult to quantify how river flow alteration impacts freshwater ecosystems despite few studies correlating freshwater species richness with hydrological parameters (Xenopoulos, Lodge et al. 2005, Comte, Buisson et al. 2013, Davis, O'Grady et al. 2015). New knowledge on macro-ecology of freshwater ecosystems shows evidence of common characteristics between species across eco-regions (Oberdorff, Tedesco et al. 2011, Tisseuil, Cornu et al. 2013). At local scale, there is more evidence that altering natural stream patterns increases the number of exotic species, changes species composition, lowers trophic level etc., but changes in stream habitat remain difficult to quantify due to non-linear responses (Allan and Castillo 2007). Latest studies identified degradation of freshwater ecosystems and low water quality due to drought, and the longer the drought is the lesser specie recovery happens (Lake 2003, Gordon, Peterson et al. 2008). Climate change impacts including lower flows and higher temperature are also likely to affect freshwater ecosystems in the coming decades (Vliet, Ludwig et al. 2013). However, periodic disturbances of flow regime might be beneficial to some freshwater ecosystem species such as eradicating invasive species and maintaining river channels (Suen and Eheart 2006).

In this study we developed the concept of "environmental flow deficit", with the aim to show in time and space the flow that is lacking in a river on an intra-annual variability base (e.g. seasonal water deficit) and on inter-annual base (e.g. supra-seasonal water deficit). We also differentiate environmental deficits due to natural climate variability (natural deficit) and due to anthropogenic water withdrawals (anthropogenic deficit), as it is important to recognize the causes of water deficit in a river to anticipate better water management. Natural deficit is resulting from a lack of water to sustain environment in natural conditions. This last definition can be due to a real hazard drought event causing a hydrological drought in response to precipitation reduction. On one side, if the deficit is due to natural deficit, it might be difficult to supply extra water to ecosystems in case of long droughts. Moreover, natural deficit might be beneficial for increasing the resilience of endemic species (Botter, Basso et al. 2013). On the other side, if deficit is due to anthropogenic purposes such as irrigation, it is more likely that water managers can adapt and release the required water demands between irrigation and ecosystems via specific water regulations. Finally, differentiation of EF deficit will be assessed on two main flow regimes: perennial rivers and intermittent rivers. While perennial rivers are known to have a stable flow all year long, intermittent river are characterized with period of flow cessations (Gordon, McMahon et al. 2004).

Our objectives are: i) to define and differentiate globally the water that is lacking to meet EFRs due to natural climate variability and due to water extractions both in terms of intra-annual and inter-annual variability, ii) to identify refined temporal water deficit in terms of timing, frequency and duration, iii) and finally to identify hot-spots where water deficit is occurring. Finally, the last objective is to cluster river basins in terms of flow regime, flow alteration and sensitivity to EF deficit.

3.2. Material and methods

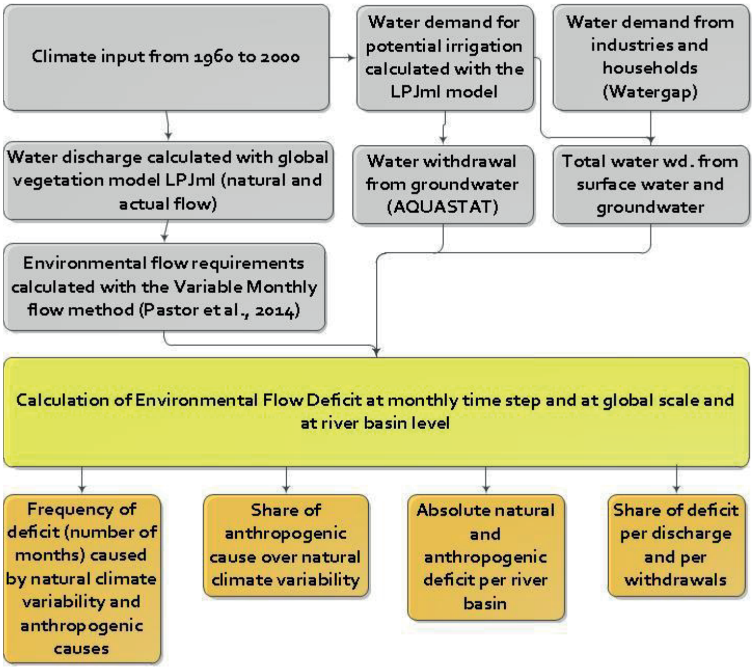

Figure 3.1. Conceptual framework to calculate EFR deficit at global scale

3.2.1. *Model*

We used the global dynamic vegetation and hydrological model LPJmL to simulate natural runoff (without anthropogenic water use) and actual runoff (including anthropogenic water use for irrigation, industries and households). LPJmL was initially developed for simulating carbon cycle at global scale for natural and agricultural vegetation (Sitch, Smith et al. 2003, Gerten, Schaphoff et al. 2004, Bondeau, Smith et al. 2007, Rost, Gerten et al. 2008). It calculates daily water balances at 0.5 degree spatial resolution with flow components laterally routed along the Simulated Topological Network (STN-30) with a constant flow velocity of 1 m/s (Gerten, Schaphoff et al. 2004, Biemans, Haddeland et al. 2011). Runoff occurs when water input exceeds soil water capacity in a 2 layers soil column including the percolation from the second soil column to the adjacent downstream cells except for permafrost (5 soil layers). Land-use is derived from the dataset (FAO 1991), for details on crop repartition see Rost, Gerten et al. (2008). Each cell with crops and pasture can be irrigated and/or rainfed based on the map of irrigated areas (Siebert, Döll et al. 2007).Water is extracted for irrigation and other users from surface water including rivers, dams and reservoirs (Biemans, Haddeland et al. 2011). Irrigation withdrawals can be simulated with 3 conditions: under unlimited water supply (IPOT), here water withdrawals equal water demands for irrigation use and the water that is not available from surface water is supposed to come from non-renewable sources such as groundwater. The second irrigation simulation is based on water availability from rivers, dams and reservoirs (IRES) only, so that if water demand exceeds supply, plant growth will be limited to water supply (Rost, Gerten et al. 2008, Biemans, Haddeland et al. 2011). Third, runoff can also be simulated under natural conditions without irrigation withdrawals (INO) where all crops are rainfed. Water use for industry and households are externally included (Flörke, Eisner et al. 2013). Conveyance losses which represent the volume of water that is lost during transport, and irrigation use efficiency are included in the irrigation calculations to provide gross irrigation demand (Rost, Gerten et al. 2008). Net irrigation requirements are based on evaporation demand and soil water capacity requirements. Potential evapotranspiration (PET) is calculated with the Priestley-Taylor method using soil moisture and rooting depth information, while actual evapotranspiration (ET) is based on PET and potential canopy conductance (calculated with photosynthesis and CO_2 concentrations). In this study we used fixed land use at year 2000 to evaluate the impact of inter-annual climate variability. Irrigation withdrawals were calculated with the IPOT scenario, i.e. assuming that any irrigation requirement can be met (Rost, Gerten et al. 2008, Biemans, Haddeland et al. 2011). Inside a river basin, runoff and EFRs calculation was re-allocated according to discharge repartition (Schewe, Heinke et al. 2014).

We force the model with the input dataset CRU T.S.3.10 (available online at: http://badc.nerc.ac.uk/data/cru/). The input consists of monthly values of precipitation, number of wet days, fraction cloud cover and average temperature for 41 years (1960-2000). The model was first run with a spinup of 1000 years to put carbon and water cycle into equilibrium at year 1960. Then, we ran the model for 41 years (1960-2000) to include inter-annual variability in our analyses.

3.2.2. *Environmental flow method*

In this study, we use the Variable Monthly Flow method (Pastor, Ludwig et al. 2014) to estimate environmental flow requirements at global scale. This method allocates a share of the natural monthly flow to freshwater ecosystems. Each cell has a specific seasonal flow regime (hydrograph) which was divided into three periods: low-flow, high flow and intermediate flow periods. High flows are assigned when the mean monthly flow (MMF) is above 80% of the mean annual flow (MAF), low flows are assigned when the mean monthly flow (MMF) is below 40% of the mean annual flow (MAF), and intermediate flows are assigned when the mean monthly flow (MMF) is between 40% and 80% of the mean annual flow (MAF). We allocate then 30% of the MMF for high flow requirements, 45% of the MMF for intermediate flow requirements and 60% of the MMF for low flow requirements. The share of EFRs for low flows are higher than the rest of the season to protect habitat maintenance and allow good spawning conditions (Pastor, Ludwig et al. 2014). In this study, we define net discharge as the total discharge minus EFRs. Conceptual framework is presented in Figure 3.1.

3.2.3. *Deficit per river basin*

We define two types of environmental flow deficits: the natural deficit and the anthropogenic deficit. Natural deficit represents the flow that is lacking due to natural climate variability only, while, anthropogenic deficit represents the flow that is lacking due to water extractions for irrigation, industry and households. Explicitly, natural deficit happens when monthly flow does not meet EFRs without considering anthropogenic water extractions. By using monthly flow data we define the intra-annual deficit to identify the timing and the duration of the deficit. We also the frequency of natural deficit in terms of number of months and years. To calculate the anthropogenic deficit at global scale we differentiated water withdrawals from surface water withdrawals and groundwater withdrawals. Groundwater withdrawals account for 851 $km^3\ yr^{-1}$ and were spatially defined as a share of total potential withdrawals per country (Siebert, Burke et al. 2010). The EF deficit is calculated on the grid cell spatial scale and aggregated to the river basin. In one case, we calculated the overall EFR deficit where water surplus can compensate EF deficit inside a river basin and in the second case, we calculated the absolute deficit which aggregates only the deficit per river basin and where surpluses of water are not taken into account. EF deficits are defined below:

$$EF\_\mathrm{def}\_nat(m,y) = \sum_{\substack{i=1\\j=1}}^{y,m} (Qnat(m,y) - EFR(m)) \quad [1]$$

$$EF\_\mathrm{def}\_ant(m,y) = \sum_{\substack{i=1\\j=1}}^{y,m} (Qmod(m,y) - EFR(m)) \quad [2]$$

Where EF_def_nat represents the "Natural deficit" or the EF deficit caused by natural climate variability only. EF_def_ant represents the anthropogenic deficit which includes both natural climate variability and anthropogenic water withdrawals where Qnat represents the natural discharge and Qmod represents the actual modified discharge including water withdrawals. y represents the number of years (y=41) to iterate starting from i and m represents the number of months (m=12) to iterate starting from j. Anthropogenic and natural deficit are differentiated by calculating deficit with river runoff minus EFRs with and without human water extractions.

3.2.4. *Sensitivity analyses*

For sensitivity analyses and uncertainty range, we calculated deficit with EFRs plus two standard deviations of the EFRs to have a scenario with high EFRs thresholds (see Shadkam, Ludwig et al. (2016)). The standard deviation of EFRs was calculated over the 41 years' time period. We chose to add two standard deviations to EFRs to test the sensitivity of EFRs thresholds on environmental flow deficit spatially and temporally.

3.2.5. *Classification of river basins*

We use the principal component analysis (PCA) to identify river basins clusters by using 6 hydrological variables. These hydrological variables were chosen to define river flow regimes, level of flow alteration, differentiation between anthropogenic and natural deficit and finally to define duration, timing and frequency of deficit. A PCA of a data matrix extracts the main patterns in the matrix in terms of a matching set of score and loading plots (Wold, Esbensen et al. 1987). We selected the hydrological indicators as input in the PCA when the highest variance was explained. We selected 23 river basins worldwide to represent three different river flow regimes (perennial, intermittent and erratic), two different climates (tropical and temperate) and two levels of flow alteration (below 50% alteration and above 50% of flow alteration).

To characterize flow regime, we used two variables: the baseflow index (BFI) and the Hydrological Variability Index (HVI) variables were defined as below:

$$\mathrm{BFI} = \frac{\mathrm{Q90}}{\mathrm{MAF}} \qquad [3]$$

$$\mathrm{HVI} = \frac{\mathrm{Q25} - \mathrm{Q75}}{\mathrm{Q50}} \qquad [4]$$

Where Q90 represents the flow which is exceeded for 90% of the period of record, MAF represents the mean annual flow, Q25 represents the flow which is exceeded for 25% of the period of record, Q75 represents the flow which is exceeded for 75% of the period of record, and Q50 represents the flow which is exceeded for 50% of the period of record. All our calculations are based on monthly output where volumes of water are in km^3. MOD represents the level of flow modification and is defined as the mean annual actual flow (including anthropogenic water extractions) over the mean annual natural flow. The results of the PCA allowed the definition of four river categories using the baseflow index (BFI) and the MOD variables:

1. Group 1: Perennial flow regime with low flow modification: BFI >= 0.3 and MOD>0.95
2. Group 2: Perennial/intermittent flow regime with low flow modification: 0.15 <= BFI <0.3 and MOD>0.7
3. Group 3: Perennial/intermittent flow regime with moderate flow modification : 0.01 <= BFI <0.15 and MOD>0.7.
4. Group 4 : Perennial/intermittent flow regime with high flow modification: 0.01 <= BFI <0.15 and MOD<=0.7

To perform the PCA, a couple of hydrological indicators were used to define the range of origin, magnitude, timing and frequency of EF deficit:

- Total deficit to discharge

$$DTD(m,y) = \sum_{\substack{i=1 \\ j=1}}^{y,m} \frac{\text{Nat. Deficit }(m,y) + \text{Ant. Deficit }(m,y)}{\text{Qnat }(m,y)} \quad [5]$$

- Natural deficit to discharge

$$DTN(m,y) = \sum_{\substack{i=1 \\ j=1}}^{y,m} \frac{\text{Nat. Deficit }(m,y)}{\text{Qnat }(m,y)} \quad [6]$$

- Anthropogenic deficit to discharge

$$DTA(m,y) = \sum_{\substack{i=1 \\ j=1}}^{y,m} \frac{\text{Ant. Deficit }(m,y)}{\text{Qnat }(m,y)} \quad [7]$$

- Frequency of total deficit – sum of number of months deficit

$$Freq\_tot(y) = \sum_{j=1}^{y} MEF_{def1(y)} + MEF_{def2(y)} \quad [8]$$

- Frequency of natural deficit – sum of number of months deficit

$$Freq\_nat(y) = \sum_{j=1}^{y} MEF_{def1(y)} \quad [9]$$

- Frequency of anthropogenic deficit – sum of number of months deficit

$$Freq\_ant(m,y) = \sum_{\substack{i=1 \\ j=1}}^{y,m} MEF_{def2(m,y)} \quad [10]$$

- Ratio of anthropogenic deficit over natural deficit

$$ATN(m,y) = \sum_{\substack{i=1 \\ j=1}}^{y,m} \frac{\text{Ant. Deficit }(m,y)}{\text{Nat.Deficit }(m,y)} \quad [11]$$

- Ratio of the frequency anthropogenic deficit over natural deficit

$$FATN(m,y) = \sum_{\substack{i=1 \\ j=1}}^{y,m} \frac{\text{Freq\_ant. }(m,y)}{\text{Freq\_nat. }(m,y)} \quad [12]$$

- Ratio of water withdrawals over discharge

$$WTD(m,y) = \sum_{\substack{i=1 \\ j=1}}^{y,m} \frac{\text{Wd }(m,y)}{\text{Qnat }(m,y)} \quad [13]$$

- Ratio of deficit over water withdrawals

$$DTW(m,y) = \sum_{\substack{i=1 \\ j=1}}^{y,m} \frac{\text{EFdef1,2 }(m,y)}{\text{Wd }(m,y)} \quad [14]$$

- Ratio of natural deficit over water withdrawals

$$ATW(m,y) = \sum_{\substack{i=1 \\ j=1}}^{y,m} \frac{EFdef1\ (m,y)}{Wd\ (m,y)} \qquad [15]$$

- Ratio of anthropogenic deficit over water withdrawals coming from surface water

$$ATWs(m,y) = \sum_{\substack{i=1 \\ j=1}}^{y,m} \frac{EFdef1\ (m,y)}{Wd\_surf\ (m,y)} \qquad [16]$$

where MEF represents months with environmental flow deficit and Wd and Wd_surf represent respectively total water withdrawals and water withdrawals coming from surface water only. I,j,m and y are defined in the text above.

3.3. Results

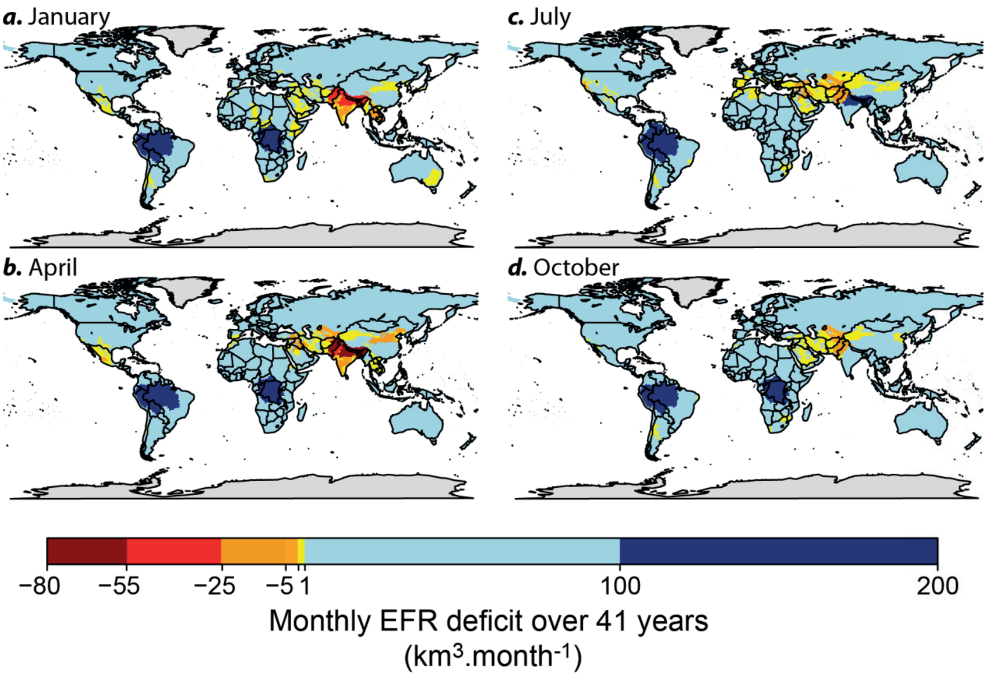

Figure 3.2. Spatial representation of monthly deficit at global scale for four months: Jan, Apr, Jul, Oct averaged over 41 years. Units are in km^3 month^{-1}. Negative values represent deficit of flow (in yellow/orange/red) whereas positive values (in blue) represent surpluses of flow.

Our simulations show that total annual runoff is about 44,000 km^3 per year of which EFRs represent between 18,000 to 28,450 km^3 year^{-1} (40-60% of global runoff) depending on the chosen EFRs threshold. We also noticed that irrigation withdrawals have almost doubled in 41 years from 1,088 km^3 year^{-1} in 1960 to 2,075 km^3 year^{-1} in 2000. The total annual deficit including natural and anthropogenic deficit ranges from 2,321 to-2,706 km^3 yr^{-1} (8-17% of net discharge). The anthropogenic deficit ranges from 1,068 to 1,354 km^3 yr^{-1} and represents about 50% of total annual deficit (Figure 3.2; Table 3.1 and Supplementary Figure A1). We also show that anthropogenic deficit coming from surface water only represents between 260 to 545 km^3 yr^{-1} (about 50% of the total anthropogenic deficit). While total annual deficit is not of global concern, we show here that deficit can be particularly high at specific temporal and regional scales (Figure 3.2). For example, the highest monthly deficits are observed in South and East Asia (up to 80 km^3 month^{-1}) with a duration of at least 5 months (between December and May). Among them, the Indus river shows the maximum annual deficit (200 km^3 per year or 130% of total discharge)). When the deficit outpaces the river flow, withdrawals should come from unrenewable sources such as groundwater. The frequency of the deficit can reach up to 90% of the time (Figure 3.2, Table 3.3 and Supplementary Figure A1) indicating that in the Indus, monthly flows meets EFRs for only 10% of the time. North African and Mediterranean river basins also show large deficits especially in the summer. For example the Euphrates has a deficit of up to 25 km^3 per month and up to 60 km^3 per year (deficit equals 50% of the discharge). In the last case, deficit occurs at least half of the time. Large deficits are also observed in the Western part of the United States such as in the Colorado and the Columbia rivers. Finally, perennial rivers such as the Congo and the Amazon rivers show a very low EF deficit (< 2% of the total discharge) coming mainly from natural deficit and occurring less than 5% of the time.

Table 3.1. Global numbers of discharge, EFRs, deficit and irrigation (km^3 year^{-1}) as a share of surface water and groundwater withdrawals (%)

	Absolute values (km^3 yr^{-1})	Share of Irrigation Wd (%)
Discharge (± sd)	44387±1619	-
EFRs (+ 2sd)	18089 -28450	-
Irrigation withdrawals range (from 1960 to 2000)	1088-2075	-
Total-annual deficit range (from 1960 to 2000)	2321-2706	-
Anthropogenic annual deficit range (from 1960 to 2000)	1068-1353	66-84%
Anthropogenic annual deficit (considering 50% irrigation from groundwater) (from 1960 to 2000)	260-545	16-34%

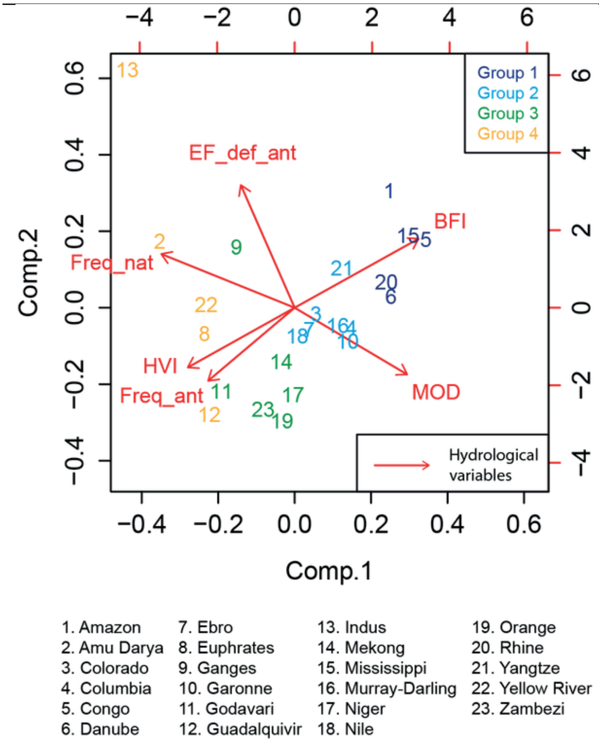

Figure 3.3: Principal Component Analysis of 23 river basin using 6 hydrological variables: EF_def2 (Total deficit), Freq_nat (Natural deficit), Freq_ant (Anthropogenic deficit), BFI (Baseflow index), HVI (hydrological variability index) and MOD (the rate of flow alteration)

A PCA was performed to categorize four groups of river basins (Figure 3.3, Table 3.3). The PCA was defined by 2 components which explain 79% of the variance of the 23 river basins. Six out of 16 hydrological variables were selected when we reached the maximum variance explanation. The first component explains 52% of the variance which is characterized by the BFI and the MOD variables on the positive axis (+0.4) and by the frequency of the natural deficit (Freq_nat), HVI and the frequency of the anthropogenic deficit (Freq_ant) on the negative side of the axis (up to -0.4). The second component is explained by the anthropogenic EF deficit (EF_def_ant) as positive value (up to 0.4) and the Freq_ant and MOD values as negative values (up to -0.2). We notice that river basins groups are following the component 1 with groups starting from the least modified flow (group 1) up to the most modified flow (group 4).

Table 3.2. Hydrological indexes of 23 river basins including discharge, EFRs and natural and anthropogenic deficits

River basin	Discharge ($km^3\ yr^{-1}$)	Mod: ratio of actual flow on natural flow	Baseflow Index (BFI)	Hydrological Variability Index (HVI)	EFR range (low-high; $km^3\ yr^{-1}$)	Share of EFR to discharge (low-high; %)	Water Withdrawals from IRR and OTH ($km^3\ yr^{-1}$)	Total deficit ($km^3\ yr^{-1}$)	Anthrop. deficit ($km^3\ yr^{-1}$)	Frequency total deficit (nb months)	Share of anthropogenic on total deficit (%)	Share of total deficit to Discharge (%)	Share of ant. deficit to discharge (%)	Share of ant. deficit to water Wd. (%)	Share of ant. deficit to surface water Wd. (%)
Amazon	7495 ± 886	1.00	0.47	0.76	2419 - 3690	32-49%	7-8	-125 ± 538	0±0	25	0%	-2%	0%	0%	0%
Amu darya	61 ± 28	0.30	0.08	1.46	21-42	35-68%	86-93	-78 ± 21	-56±15	360	72%	-127%	-91%	-62%	-28%
Colorado	35 ± 22	0.74	0.25	0.84	11-24	31-68%	8-12	-7 ± 4	-3±2	156	48%	-20%	-10%	-34%	-21%
Columbia	171 ± 62	0.94	0.22	0.94	54-98	32-57%	12-16	-12 ± 15	-4±5	84	37%	-7%	-3%	-32%	-20%
Congo	2287 ± 389	1.00	0.54	0.49	722-1141	32-50%	1-2	-37 ± 160	0±0	25	0%	-2%	0%	0%	0%
Danube	207 ± 73	0.99	0.38	0.71	68-118	33-57%	19-21	-13 ± 18	-4±5	54	30%	-6%	-2%	-19%	-2%
Ebro	19 ± 10	0.79	0.19	0.87	6-12	32-65%	4-7	-4 ± 3	-2±2	184	54%	-24%	-13%	-43%	-16%
Euphrates	91 ± 45	0.68	0.02	1.62	27-57	30-63%	52-60	-45 ±19	-29±12	286	65%	-49%	-32%	-52%	-23%
Ganges	1281 ± 32	0.96	0.05	1.69	403-640	31-50%	183-189	-142 ±112	-116±91	227	81%	-11%	-9%	-62%	-35%
Garonne	33 ± 16	0.97	0.27	0.94	11-21	33-63%	1-5	-2 ± 3	-1±1	123	43%	-7%	-3%	-34%	-12%

Table 3.2. (suite) Hydrological indexes of 23 river basins including discharge, EFRs and natural and anthropogenic deficits

River basin	Discharge ($km^3\ yr^{-1}$)	Mod: ratio of actual flow on natural flow	Baseflow Index (BFI)	Hydrological Variability Index (HVI)	EFR range (low-high; $km^3\ yr^{-1}$)	Share of EFR to discharge (low-high; %)	Water Withdrawals from IRR and OTH ($km^3\ yr^{-1}$)	Total deficit ($km^3\ yr^{-1}$)	Anthrop. deficit ($km^3\ yr^{-1}$)	Frequency total deficit (nb months)	Share of anthropogenic on total deficit (%)	Share of total deficit to Discharge (%)	Share of ant. deficit to discharge (%)	Share of ant. deficit to water Wd. (%)	Share of ant. deficit to surface water Wd. (%)
Godavari	171 ± 61	0.92	0.03	1.79	53-99	31-58%	30-36	-29 ± 18	-17±10	285	59%	-17%	-10%	-52%	-29%
Guadalquivir	10 ± 9	0.70	0.01	1.28	3-9	33-91%	3-11	-6 ± 3	-3±1	314	42%	-64%	-27%	-41%	-15%
Indus	155 ± 53	0.22	0.11	1.15	52-94	34-60%	255-263	-201 ± 60	-172±52	461	85%	-130%	-111%	-66%	-37%
Mekong	609 ± 124	0.99	0.05	1.74	190-310	31-51%	17-23	-20 ± 45	-15±33	143	74%	-3%	-2%	-76%	-22%
Mississippi	818 ± 258	0.96	0.51	0.56	264-441	32-54%	67-68	-53 ± 67	-5±7	39	10%	-6%	-1%	-8%	-5%
Murray	109 ± 75	0.92	0.24	0.82	33-80	30-73%	18-21	-15 ±12	-6±5	128	41%	-14%	-6%	-32%	-8%
Niger	500 ± 94	0.99	0.03	1.66	153-249	31-50%	3-9	-10 ± 35	-4±13	95	39%	-2%	-1%	-72%	-10%
Nile	597 ± 125	0.94	0.10	1.45	243-390	41-65%	55-58	-39 ± 62	-23±37	108	59%	-7%	-4%	-41%	-13%
Orange	60 ± 47	0.97	0.04	1.14	21-54	35-89%	1-7	-5 ± 6	-1±2	182	24%	-8%	-2%	-28%	-2%
Rhine	75 ± 33	0.99	0.43	0.65	23-45	31-60%	18-20	-9 ± 9	-5±6	112	60%	-12%	-7%	-27%	-14%
Yangtze	732 ± 184	0.97	0.24	1.01	233-388	32-53%	98-102	-72 ± 66	-42±39	78	59%	-10%	-6%	-42%	-12%
Yellow river	72 ± 24	0.64	0.03	1.52	24-41	33-58%	45-57	-39 ± 17	-29±13	303	75%	-54%	-40%	-57%	-16%
Zambezi	400 ± 41	0.99	0.01	1.92	127-233	32-58%	0-5	-9 ±30	-5±16	151	54%	-2%	-1%	-220%	-17%

Table 3.3. Classification of 23 river basin into four groups according to hydrological indexes. Values are showing range of minimum and maximum of each group.

	Flow (km^3 yr^{-1})	BFI	HVI	Tot. deficit (km^3 yr^{-1})	Tot. Wd. (km^3 yr^{-1})	Tot. Freq. (nb. of months)	Anthrop. to total deficit ratio	DTD	WTD	DTW
Group 1	75 -7495	0.47-0.54	0.49-0.76	9-125	1-67	25-112	0-27%	2-12%	0-25%	0-27%
Group 2	19-732	0.1-0.25	0.82-1.45	2-72	3-100	78-184	37-59%	7-24%	8-31%	32-42%
Group 3	60-1281	0.01-0.05	1.14-1.92	5-142	2-186	95-285	24-81%	3-17	1-19%	28-72%
Group 4	10-155	0.01-0.11	1.15-1.62	6-201	7-259	314-461	42-85%	49-130%	62-168%	41-66%

BFI = Baseflow index
HVI = Hydrological variability index
Tot. deficit = Total deficit
Tot. Wd.= Total withdrawals
Tot. Freq.= Total frequency
Anthrop. To nat. deficit = anthropogenic to natural deficit ratio
DTD = Deficit to Discharge ratio
WTD.= Withdrawals to discharge ratio
DTW= Deficit to withdrawals ratio

In Figure 3.4, we show that localisation of river basin and not correlated with groups found in PCA but rather with flow regime type and level of flow alteration. Group 1 is along the axis of BFI (Amazon, Congo, Mississippi, Rhine, Danube). This group is characterized by rivers with a high BFI (>0.38) and a low HVI (<0.76) (Table 3.2-3.3; Figure 3.5a and supplementary Figure A2a). This group is dominated with perennial rivers of which the deficit is mainly natural (25-45 months out of 492 months) compared to the anthropogenic deficit (0 to 87 months out of 492 months). Group 1 has also the smallest Deficit To Discharge (DTD) ratio (2-12%). Anthropogenic deficit to discharge ratio represents 0 to 7% of the discharge and Deficit to Withdrawals (DTW) reaches up to 27% of withdrawals (e.g. Rhine). For example, the Congo river has the most perennial flow regime (BFI=0.54, HVI=0.49) and shows barely any flow modification except between January and March. Finally, the Congo river is characterized with a low deficit frequency (2 years out of 41) with no anthropogenic deficit.

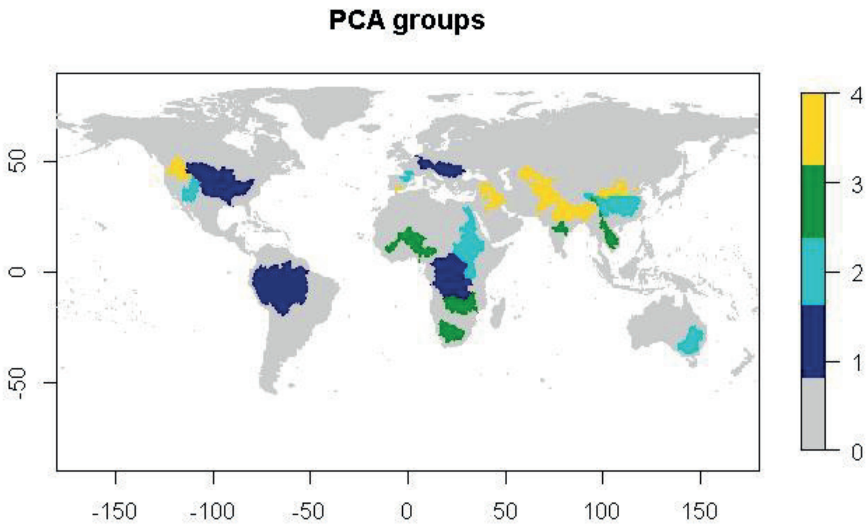

Figure 3.4. Localisation of river basin groups after PCA analysis.

Group 2 is along the axis of MOD variable (Yangtze, Columbia, Garonne, Colorado, Nile). This group is characterized by less stable flows (BFI=0.1-0.27 and HVI=0.74-0.97) (Table 3.2; Figure 3.5b and supplementary Figure A2b). Frequency of deficit is moderate with 78 to 184 months deficit out of 492 months of which 37 to 60% is due to anthropogenic deficit. DTD represents between 7 to 24% (e.g. Ebro) and anthropogenic deficit represents 3-13% of total discharge. Potential affected withdrawals (deficit to withdrawals) represent 34% to 43% of withdrawals and potential affected withdrawals from surface water can reach up to 31%. For example, the total flow of the Columbia river equals 171 km^3 $year^{-1}$ of which 8% are withdrawn for irrigation and other users. Total deficit equals 12 km^3 (7% of total discharge) of which natural deficit represents 63% of total deficit and is occurring between August and February. Anthropogenic deficit equals 3% of total flow and shows a frequency of 31 months out of 492 (between June and October). Potentially affected water withdrawals represent up to 32% (of which 20% affect surface water withdrawals).

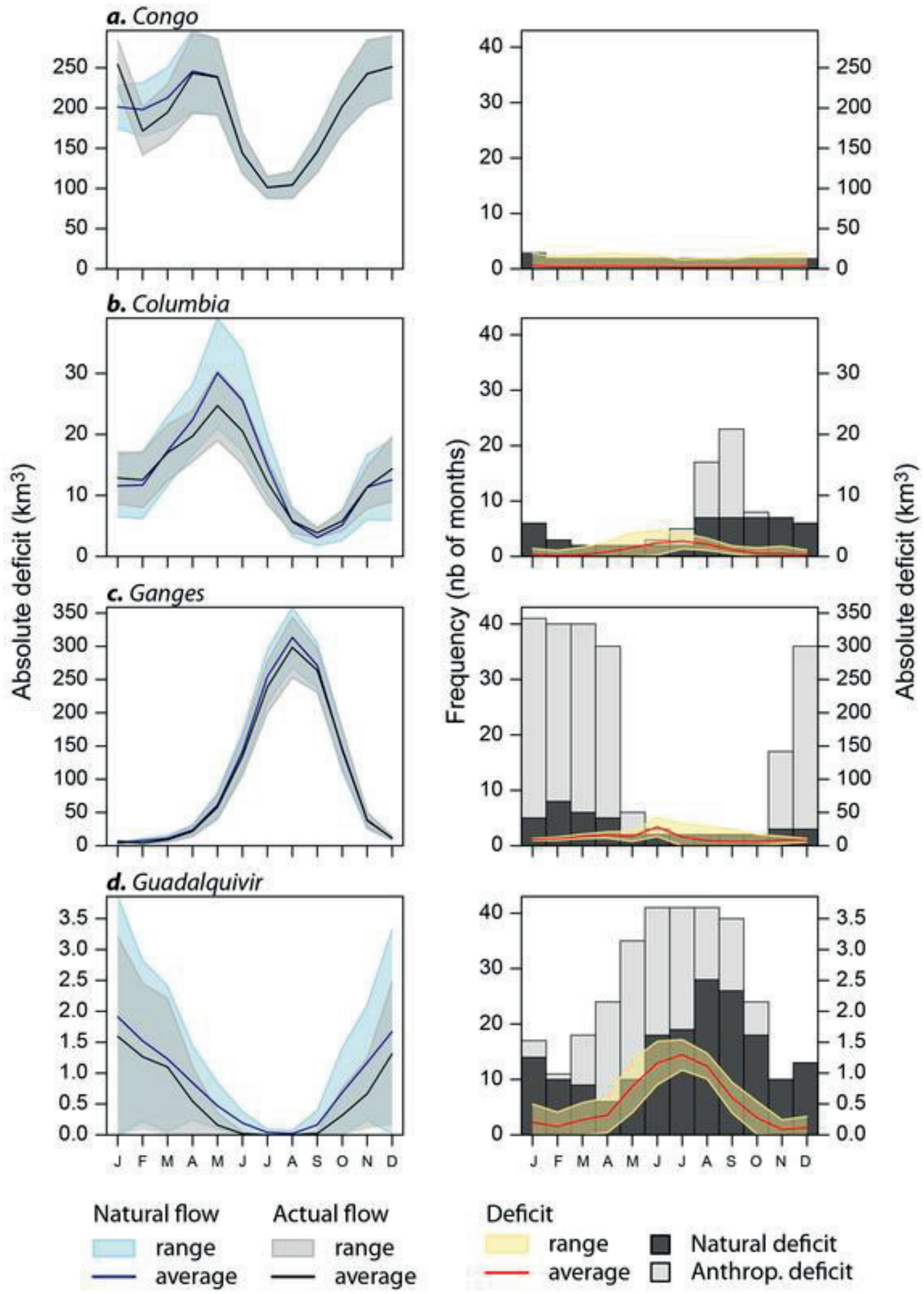

Figure 3.5. Temporal representation of natural and actual flow (left plot) with intra-annual monthly deficit (km^3 $month^{-1}$) and frequency of deficit (number of months) for four river basins representing each group of river basins. Y axis represent absolute flow and deficit have the same scale to enable comparison, Barplots (right plot) represent the frequency of natural deficit (black) and anthropogenic deficit (grey). Total Environmental Deficit (EF) (yellow range-right plot) was shown with positive values for comparison with the flow (blue/grey range – left plot).

Group 3 is in between Freq_ant and MOD variables (Godavari, Niger, Zambezi, Orange, Mekong). This group represents a cluster of river basins with an intermittent to ephemeral flow regime and with a moderate flow modification (BFI=0.01-0.05, HVI=1.14-1.92) (Table 3.2; Figure 3.5c and supplementary Figure A2c). The frequency of the deficit accounts for 95 to 285 months deficit out of 492 months of which 18% to 58% are coming from anthropogenic deficit. DTD ranges between 2 and 17% (e.g. Godavari). Potentially affected withdrawals represent 28 to 72% of total withdrawals (4-66% for withdrawals from surface water only). For example, the Ganges has an intermittent flow regime with a BFI of 0.05 and a HVI of 1.69. Withdrawals represent 186 km^3 (15% of total flow). Total deficit equals 142 km^3 $year^{-1}$ (11% of total flow) and spreads over 227 months (<50% of the time) of which 185 months are due to anthropogenic deficit (81% of total deficit). The duration of the deficit is about 4 months (between November and April) and potentially affected withdrawals represent 62% (of which 26% affect surface water withdrawals).

Group 4 is highly correlated with EF_def_ant (especially the Indus), with Freq_nat (Amu darya) and with HVI and Freq_ant (Yellow river Euphrates and Guadalquivir). This group represents a cluster of river basins with an intermittent to ephemeral flow regime and with a high flow modification (BFI=0.01-0.11, HVI=1.15-1.62, MOD=0.22-0.7) (Table 3.2; Figure 3.5d and supplementary Figure A2d). The frequency of the deficit accounts for 286 to 461 months deficit out of 492 months of which 42% to 85% are coming from anthropogenic deficit. DTD ranges between 49 and 130% (e.g. Indus). Potentially affected withdrawals represent 41 to 66% of total withdrawals (of which 8-36% affect surface water only). For example, the Guadalquivir has an intermittent flow regime with a BFI of 0.01 and a HVI of 1.28. Withdrawals represent 7 km^3 (66% of total flow). Total deficit equals 6 km^3 $year^{-1}$ (64% of total flow) and spreads over 314 months (64% of the time) of which 133 months are due to anthropogenic deficit (42% of total deficit). Duration of the deficit is about 6 months (between May and October) and potentially affected withdrawals represent 41% (of which 8% affect surface water withdrawals).

We show that the higher the BFI is, the lower the frequency of natural deficit is (R^2=0.35) meaning that perennial rivers encounter less deficit caused by climate variability than intermittent rivers. Similarly the BFI is negatively correlated with the frequency of anthropogenic deficit (R^2=0.32) meaning that intermittent rivers encounter more deficit due to anthropogenic sources.

We present one river basin per group in Figure 3.5 showing the magnitude of monthly natural and actual flows (left plot) and the magnitude and frequency of natural and anthropogenic deficits (right plot). The rest of the river basins and respective deficits are shown in supplementary Figure A2.

Group 1 is represented by the Congo river with a low modification of the natural flow (in blue) compared to the actual flow (in grey). On the right plot, we can see that absolute monthly deficit is below 15 km^3 and frequency of the monthly deficit is low (below 5 months over 41). The natural deficit represents 100% of the total deficit (meaning there is no anthropogenic deficit).

Figure 3.5b shows the flow dynamics of the Columbia river representing group 2. In the left plot, we can see that the range of anthropogenic flow was reduced compared to the natural flow and that the actual flow is below the natural flow from April to August. The absolute monthly deficit is about 5 km^3 with summer peaks (in June and July). The frequency of the monthly deficit is less than 10 months over 41 for natural deficit and anthropogenic deficit frequency is mainly occurring between August and September (up to 24 months deficit over 41). Natural deficit frequency is spread over 6 months and anthropogenic deficit is spread over 3 months with a higher share of absolute natural deficit over anthropogenic deficit (63%). Total deficit to withdrawals represents 32% of which 20% are impacting surface water only.

Group 3 is represented by the Ganges showing a high flow modification compared with the actual flow (left plot) and on the right plot, we can see that absolute monthly deficit reaches up to 5 km^3 in June (31-50% of annual discharge). Total frequency deficit is nearly 100% between December and April where anthropogenic deficit frequency represents more than 80% of the total deficit frequency. Total deficit to withdrawals represents 62% of which 35% are impacting surface water only.

Finally, we present group 4 with the Guadalquivir river which shows a lower actual flow than natural flow all year long (left plot). Total monthly deficit can reach up to 20 km^3 in June-July and outpace natural and actual flows in summer months. Total deficit frequency occurs every month with at least 20 months deficit out of 41. Natural frequency deficit is between 10 and 30 months out of 41 with peaks in late summer and anthropogenic deficit frequency is about 10 to 25 months out of 41 occurring mainly between April and September. The Guadalquivir has a total deficit of about 64% of the total discharge. Total deficit to withdrawals represents 66% of which 15% are impacting surface water only.

3.4. Discussion

In this study, we used the improved VMF method to calculate EFRs at a monthly time-step and at a spatial scale of 0.5° (then aggregated to the river basin). This is the first study showing differences between natural and anthropogenic monthly EF deficits including inter-annual variability. We show that total EF deficit represents a low share of the total global runoff (5%). However, at regional scale, we demonstrate that monthly deficit can outpace the available flow (usually during low flow periods) in locations where high irrigation extractions are occurring. Correlations were found between river flow regime and level of flow alteration. For example, tropical perennial flow regimes such as the Amazon and the Congo tend to have low EF deficits with a high proportion coming from natural deficit, while intermittent flow regimes located in dry areas (and usually requiring high irrigation demand) show high absolute deficit with a high proportion coming from anthropogenic deficit and with a frequency of deficit superior to 50%.

Our study shows how to differentiate the origin of the EF deficit (anthropogenic or natural) which allows anticipating a better monitoring including adequate measures for river restoration. Thanks to long term analyses of model data (41 years), we were able to identify how frequent the deficit occurs (inter-annual variability), the timing and the duration of the deficit (intra-annual variability) and on which category of river basins the deficit was more frequent. Natural deficit tends to be higher for intermittent rivers than for perennial rivers especially during low-flow periods. Natural deficit is usually defined as the results from meteorological drought. However natural deficit can also be the result of an overestimation of EFRs simulations or underestimation of flow simulation and might need regional calibration with local data of river flow and ecosystem requirements. In all cases, we could identify natural EF deficit thanks to our inter-annual variability analysis and we provide a global overview analysis yielding classified patterns of EFR transgression (nat. vs. anthr.) which tentatively points to different ways of their possible management/avoidance.

From an ecological point of view, natural deficit can be beneficial or detrimental to freshwater ecosystems. For example, when predictable low flow periods are occurring, it is likely that resilience of endemic species is increased allowing a fast recovery of these last. In some cases, natural deficit can even be required for the removal of invasive species (Lake 2003, Bond, Lake et al. 2008). However, non-predictable deficit with long duration could lead to species extinction due to low provision of refugia for specie recovery and due to consequent abrupt changes in biological community structure and ecosystem processes (Humphries and Baldwin 2003). The recovery of freshwater ecosystems are linked to three kinds of adaptation skills: life history, behavioural and morphological which are used during natural disturbances. However, natural disturbances such as floods and droughts events were shown to be necessary, to a certain extent, to maintain ecological integrity of freshwater ecosystems and can help in the regulation of population size and species diversity (Lytle and Poff 2004).

In the case of anthropogenic deficit, it is likely that its impact would be detrimental to freshwater ecosystems because high human water use can lead to complete flow regime shift reversing ecological integrity (Bunn and Arthington 2002). Anthropogenic deficit can exacerbate the effect of drought and water deficit on freshwater ecosystems by "reducing the resistance and the resilience of species" (Bond, Lake et al. 2008). For example, important components of the flow of the Murray-Darling river such as low flow duration were shown to be heavily modified especially in downstream parts after water abstractions (Maheshwari, Walker et al. 1995). Increase in duration of low-flows can decrease riparian vegetation and can lead to physiological stress such as decreased growth rate, morphological rate and mortality (Poff, Allan et al. 1997). Furthermore, dams and reservoirs can harm freshwater ecosystems via the disruption of the existing movement of water and sediments that provide food and refugia to most of river taxa (Bunn and Arthington 2002).

Our results show that anthropogenic and natural deficits are usually combined and seldom independent. Only in the case of perennial tropical rivers with high discharge such as the Amazon and the Congo, natural deficit explained 100% of the total deficit. Intermittent rivers tend to have a less spread of natural deficit over time than perennial rivers. Natural deficit are usually linked with hydrological droughts which are responses to climatic droughts. According to Keyantash and Dracup (2002), while the response of a hydrological drought (and natural deficit) depends on the location of the climatic drought and on how long the flow requires to reach downstream parts, the anthropogenic deficit is usually timely synchronized with irrigation withdrawals. For example, in the Amazon river, we found that drought years and natural deficit are occurring on the same years (1973, 1982 and 1991) linked to el Niño events (Sheffield and Wood 2012). Whereas, anthropogenic deficits can match extreme droughts due to low precipitation and low water recharge but anthropogenic deficits are usually caused and exacerbated by high irrigation demand in the dry season (Gómez and Blanco 2012).

In this study, we examine the cause of deficit and we define the timing, volume, duration and frequency of the deficit for 23 river basins. This concept of environmental flow deficit is an additional step in the eco-hydrology field by giving key hydrological tools to water managers and freshwater ecologists on when, where and why freshwater ecosystems are at risk. Until now, Xenopoulos et al (2009) have shown that flow alteration could be linked with the loss of freshwater species richness. Similarly, Pracheil, McIntyre et al. (2013) managed to define discharge thresholds to maintain most of specialist species of the Mississippi river. However, more research is needed on "how the trait composition of stream communities varies along geographical and environmental gradients" (Heino, Schmera et al. 2013). Using seasonal and supra-seasonal EF deficit definitions could be helpful for the monitoring of rivers and for the adoption of the most appropriate solution.

We use the VMF method because it is a global EF method with refined spatial and temporal scales which was validated with local study cases (Pastor, Ludwig et al. 2014). Moreover, it was largely implemented and accepted in latest global assessments (Gerten, Hoff et al. 2013, Boulay, Bare et al. 2015, Grill, Lehner et al. 2015, Sadoff 2015, Steffen, Richardson et al. 2015). However, the VMF method provides the same volume of required water every year because it is calculated on a long-term average of natural flow. In this study, we included the natural climate variability component and we show that natural deficit plays a big role in defining environmental flow deficit (about 50% of total deficit). Therefore, it is necessary to estimate EFRs on a minimum of 20 years average to capture inter-annual variability and to avoid under- or overestimation of EFRs. For example, if EFRs were defined by only 5 years average including a dry year such as 2003, the calculated flow could be lower than the required flows and vice versa, if EFRs are overestimated, this could lead to an overestimation of the EF deficit. EFRs can sometimes be adjusted to wet and dry years such as in Richter, Warner et al. (2006). It is also possible to give a range of EFRs to water managers by increasing or decreasing EFRs with its the standard deviation (in this study and in Shadkam, Ludwig et al. (2016)). However, this adaptation is more likely to depend on the local conditions of the river (Pyne and Poff 2016). Alternatively, EFRs can be adapted to the level of flow alteration and to the desired ecological outcome of the river basin community (Acreman, Arthington et al. 2014).

In this study, the anthropogenic deficit represents between 16 and 34% of the irrigation withdrawals coming from surface water only. Nowadays, 40% of the food production comes from irrigated production, thus, this deficit could represent between 6 and 14% of total food production. If up to 14% of the irrigated production would be lost, food security would be at risk if measures to safeguard food access to all are not implemented. Similarly, other studies show that reducing water use for irrigation to preserve ecosystems may lead to water conflicts between users (Poff, Allan et al. 2003, Kingsford and Auld 2005, Richter, Warner et al. 2006). A specific example is shown in Blanco-Gutiérrez, Varela-Ortega et al. (2013) where respecting EFRs for Water Framework Directive in the Guadiana river would cut consequent water use for irrigation and would affect rice growers.

In this study, we consider 41 years of historical flow to evaluate our theoretical concept of deficit in terms of timing, duration and frequency. However, it could be interesting to include climate change, land-use change and future socio-economic scenarios to evaluate the intensity of deficit under global change. In fact, it was shown that climate will tend to be more variable due to intensification of the water cycle and those deficits might increase their pressure on ecosystem degradation in the future (Palmer, Lettenmaier et al. 2009, Davis, O'Grady et al. 2015). For example, a study from Rajagopalan, Nowak et al. (2009) showed that a 20% reduction of the Colorado river flow would imply a tenfold reduction in reservoir storage. Including other parameters than flow alteration in eco-hydrological assessments could also be beneficial such as the inclusion of water temperature (Vliet, Ludwig et al. 2013). In fact, future water temperature were shown to increase the most in US, Europe and China and these last are likely to be exacerbated by decreasing summer low flows in most regions in the coming decades.

To protect rivers, it is necessary to improve environmental river monitoring and anticipate responses to flow changes by choosing appropriate methods (Palmer et al 2009). For example, our results show that part of potentially affected irrigation extractions might be affected by implementing EFRs especially in modified intermittent rivers and that specific flow releases from storage could be applied to preserve ecosystems (Gaupp, Hall et al. 2015). Therefore, when river basin communities decide to implement EFRs, trade-offs between water users should be discussed. For that decision-support models and hydro-economic models could be helpful in the optimization of irrigation water use and EFRs (Acutis, Rinaldi et al. 2009, Blanco-Gutiérrez, Varela-Ortega et al. 2013, Scott, Vicuña et al. 2014). Proactive solutions are necessary to be taken especially when climate change is likely to exacerbate the impacts of deficit on freshwater ecosystems.

While pressure on freshwater ecosystems is increased, regulations on EFRs and on preservation of freshwater resources are growing. This leads to less available water for humans in terms of drinking water, water for irrigation, household and industries. Therefore, solutions to save water locally should be included in future studies and assessments such as increasing green water storage, purifying grey water, reducing export of virtual water, increasing water use efficiency and by desalinating brackish water (Lal 2015). Another option to decrease deficit would be by decreasing irrigation via increasing food imports from water-abundant regions to water-scarce regions and to use more dam regulation (including EFRs releases and specific irrigation use). However, attention should be paid in ensuring return flows to recharge groundwater and downstream rivers in case of over optimizing irrigation use (Ward and Pulido-Velazquez 2008, Scott, Vicuña et al. 2014).

Solutions to restore and/or protect rivers can be targeted to specific group of rivers. For example, our study acknowledges the necessity to identify the cause of EF deficit to take the most appropriate decision. For example, Mediterranean areas are very subjective to both natural and anthropogenic deficits due to long dry seasons combined with high irrigation extractions. Therefore, to tackle anthropogenic deficit, it might be necessary to increase water storage (Gaupp, Hall et al. 2015) and decrease the production of water intensive crops in water scarce areas (Wichelns 2001, Aldaya, Martínez-Santos et al. 2010). The level of intervention to decrease deficit should depends on flow thresholds above which water extractions would lead to ecosystem collapse. Other studies show the importance to estimate the deficit according to its flow regime and its flow alteration. For example, in the Segura river, an intermittent river with high water extractions, four levels of adaptation to drought management were defined: normal, pre-alert, alert, and emergency of which the resulting irrigation withdrawals from surface water were respectively reduced by 100%, 90%, 75% to 50%. This kind of drought management were shown to be successful when groundwater rights were defined (Gómez and Blanco 2012). Finally, in some cases, water shortages were shown to be offset thanks to financial participation of local populations of which the motives was to improve river ecosystem services such as wastewater, natural purification of water, erosion control, habitat for fish and wildlife, and recreation (Loomis, Kent et al. 2000).

3.5. Conclusion

This study presents a new concept on evaluating water deficit for the environment (EF deficit) in terms of duration, timing and frequency. We defined EF deficit at global scale with refined spatial and temporal scales. We identified hot-spots where EFRs were not satisfied in South-Asia, Mediterranean area and in the West coast of US. EF deficit was shown to be rather a regional concern than a global concern. In this study we identified the causes of EF deficit (natural and anthropogenic) which allows being more specific in the choice and the level of intervention of river restoration. For example, a river with a higher anthropogenic deficit the natural deficit is likely to require an agreement between different water users. Identification of the timing, frequency and magnitude of the EF deficit can define the importance of the deficit and its required actions in time. We found correlations between the cause of deficit, the level of flow modification and the type of flow regime. For example, we show that free-flowing rivers such as the Amazon tend to be mainly affected by natural climate variability. Intermittent rivers with moderate to high flow alterations are likely to encounter anthropogenic deficit especially during dry months due to high irrigation extractions. Intermittent rivers showed higher water deficit than perennial rivers due to the exacerbation of natural deficit by anthropogenic deficit. By identifying hot-spots of high deficit, and by labelling where and when natural and anthropogenic deficits are occurring, we examined additional hydrological metrics on how to anticipate and alleviate EF deficits, which could be used to better define improved and targeted solutions. It requires specific targeted actions at regional scale especially in heavily modified rivers where EF deficit outpaces the available flow. Finally, the identification of the origin of the deficit (natural or anthropogenic) including its timing, frequency and magnitude could improve future environmental monitoring of freshwater streams.

Chapter 4

Reconciling irrigated food production with environmental flows in face of SDG agenda

Safeguarding river ecosystems is a precondition for attaining the UN Sustainable Development Goals related to water and the environment (SDGs)1,2, while rigid implementation of such policies may hamper achievement of other goals such as food security and poverty reduction. River ecosystems provide life-supporting functions that depend on maintaining environmental flow requirements (EFRs), but their global quantification remains difficult. Here we establish process-based estimates of EFRs and their violation through human water withdrawals on a global 0.5◦ resolution grid including an uncertainty span, and we quantify the expected loss in food production if water use was to be constrained by EFRs. Our results indicate that 39% of current global irrigation water use (948 km3 yr-1) occurs at the expense of EFRs. 4.6% (13.9%) of global (irrigated) kilocalorie production depend on these volumes, and a ≥10% production loss would occur on roughly half of irrigated cropland if EFRs were to be maintained. Further simulations indicate that a moderate upgrade of irrigation systems could compensate for such losses on a sustainable basis in many regions, which supports implementation of the ambitious and seemingly conflicting SDG agenda.

Based on:

Jägermeyr, J., Pastor, A. V., Biemans, H., Gerten D. (in press) Reconciling irrigated food production with environmental flows in face of SDG agenda. Nature communications.

4.1. Introduction

Global agricultural intensification through ever-increasing resource use is a main driver of current transgressions of ”planetary boundaries”, i.e. critical global and regional levels of anthropogenically influenced earth-system processes such as land use change, biodiversity loss, freshwater use, and nitrogen and phosphorus loads (Steffen, Richardson et al. 2015). Thereby the risk increases that the Earth system is transformed into a post-Holocene state with characteristics that potentially undermine system resilience and human welfare (Steffen, Richardson et al. 2015). Because agricultural production is central to attaining the renewed SDGs, such risks are now acknowledged therein, which commits all countries to a bold and transformative agenda in support of the twin challenge: protection of Earth's life-support system while reducing hunger and poverty (UN 2015). With the human population set to rise to 9 billion by 2050, the implementation of this vision aligned with environmental guardrails requires precautionary policies based on solid quantitative grounds such as formulated in the planetary boundary framework (Griggs, Stafford-Smith et al. 2013). For progress monitoring, a global SDG indicator framework has been developed, but proposed actionable specifications for environment-related indicators remain insufficiently advocated (Griggs, Stafford-Smith et al. 2013, UN 2016).

Freshwater resources, as a core example, are clearly over-exploited and aquatic ecosystems thereby degraded in many regions (Molden 2007, Vorosmarty, McIntyre et al. 2010). Restoration of currently compromised river ecosystems through securing EFRs would thus entail a substantial reduction in irrigated food production, which is the largest global freshwater user, accounting for >70% of human water withdrawals (Siebert and Döll 2010). To quantitatively underpin water targets in the SDG framework (specified below) that bridge sustainable food production, ecosystem maintenance, and water scarcity issues, we here quantify the degree to which present irrigated agriculture contributes to a transgression of EFRs. Using EFRs as an indicator is compatible with the regional planetary boundary for human freshwater use that accounts for the spatial and temporal pattern of local tolerance levels of water use and their transgression, as opposed to the not yet transgressed global boundary (Gerten, Hoff et al. 2013, Steffen, Richardson et al. 2015). In other words, we show how much of irrigated food production would be affected if such policy goals were implemented worldwide in the vein of propositions in the Brisbane Declaration and other aquatic ecosystem policy recommendations (Le Quesne, Kendy et al. 2010, EC 2015). In turn, we assess if more effective farm water management can outweigh associated production 'losses' without compromising the aquatic ecosystems. To approach such analyses at global scale we employ an advanced dynamic global biosphere model that represents natural and agricultural vegetation with associated ecological, hydrological and biogeochemical processes - including river flows, here newly implemented EFR regulations, irrigation, and crop production - in a single internally consistent framework at high spatio-temporal resolution (Jägermeyr, Gerten et al. 2016). The EFRs are defined here as the daily river flow needed to maintain river and delta ecosystem services and, thus, the human livelihoods that rely upon them (Brisbane Declaration 2007). Reflecting methodological uncertainty and varied policies concerning the fraction of river flow which should remain untouched, we apply three differing methods to allocate flow volumes to EFRs (Tessmann 1980, Smakhtin, Revenga et al. 2004, Pastor, Ludwig et al. 2014). Simulations are performed for the time period 1980-2009, with and without consideration of EFRs. In the former case, water withdrawal for irrigation and other purposes (household, industry and livestock, HIL) is disallowed as long as it would tap EFRs. To put irrigation into perspective of total food production, we also illustrate a scenario in the absence of irrigation and highlight an exemplary scenario of moderate irrigation system upgrades (see Methods for details in Annex B).

Table 4.1. Agricultural impacts under different irrigation and flow conservation scenarios. Change in global kcal production and the proportion of affected area (kcal loss ≥ 10%) in the absence of irrigation (1.), with irrigation constrained by environmental flow requirements (EFRs) (2.), and with upgraded irrigation[2] constrained by EFRs (3.) - all compared to the current situation (1980-2009). Also shown are associated changes in irrigation water withdrawal (IWD) and consumption (IWC). Note that kcal production and area affected refer to cropland area, while IWD and IWC refer to the total irrigated area (incl. cash crops, cotton, etc.). Values for 2. and 3. refer to the mean of three EFR methods (with standard deviation in parentheses)[3].

Scenario	Total kcal [% change]	Irrigated kcal [% change]	Total area affected[1] [%]	Irrigated area affected[1] [%]	IWD [%]	IWC [%]
1. No irrigation	-14.7	-44.4	32.5	81.3	-100.0	-100.0
2. Respect EFR	-4.6 (±1.0)	-13.9 (±3.0)	16.6 (±2.3)	51.2 (±5.0)	-39.4 (±7.3)	-33.9 (±6.9)
3. Respect EFR with irrigation upgrade[2]	0.0 (±1.2)	6.1 (±3.6)	12.8 (±2.7)	35.2 (±8.4)	-52.8 (±5.4)	-33.4 (±6.6)

[1] Kcal loss ≥ 10%,
[2] Surface irrigation replaced by sprinkler systems and half of saved consumptive water used to expand irrigation into neighbouring rainfed cropland,
[3] Compare Table S2 for absolute values and respective EFR simulations.

4.2. Results

Our results show that today's human water withdrawals, 2,409 km^3 for irrigation and 1,071 km^3 for HIL, harm many river stretches around the world. Figure 4.1 lays out regions and the degree to which EFRs are currently tapped to sustain the human water demand. EFR alterations reach levels beyond the uncertainty range (given by the three estimation methods applied, see Methods Annex B), and thus indicate severe degradation, especially in Central to South Asia, the Mediterranean region, North America, and in the North China plain. Together with hydrographs highlighting severe EFR alterations at selected river locations, Figure 4.2 shows the ratio of mean annual EFR transgressions (based on pristine discharge) to current mean annual discharge. A dramatic case is the Indus river in Pakistan, where this ratio exceeds 100%, while EFRs are severely altered throughout 11 months per year (Figure

4.2). Yet, we also find alarming EFR alterations along many other rivers such as the Amu Darya, Euphrates, Yellow, Ganges, Murray, and Rio Grande. Global EFR alterations involve a water overuse of 948 $km^3\ yr^{-1}$ for irrigation (equalling 39% of total current irrigation water use) and a further 226 $km^3\ yr^{-1}$ (22%) for HIL (Table 4.1 and Supplementary Table B1; if not indicated otherwise results refer to the mean of three EFR methods). Current food production thus heavily relies on water that would actually be needed to sustain riverine ecosystems. If environmental policies to respect EFRs came into practice (also in regions where irrigated food production currently depends on such) 51% of global irrigated cropland would face kcal production losses ≥10% (see our further simulations, Table 4.1). Amongst intensely irrigated regions, like many Mediterranean countries, North America, and particularly in parts of Central and South Asia, losses would reach >20% at the aggregated level of Food Production Units (FPUs, Figure 4.3b). Total global kcal production would face a 4.6% decline, corresponding to a 13.9% loss of irrigated production (Table 4.1). Note that while kcal production on irrigated land makes up ~33% of total production (confirming earlier estimates (Siebert and Döll 2010)), irrigation water contributes 15% to overall global kcal production, while the remainder is sustained by precipitation (Table 4.1).

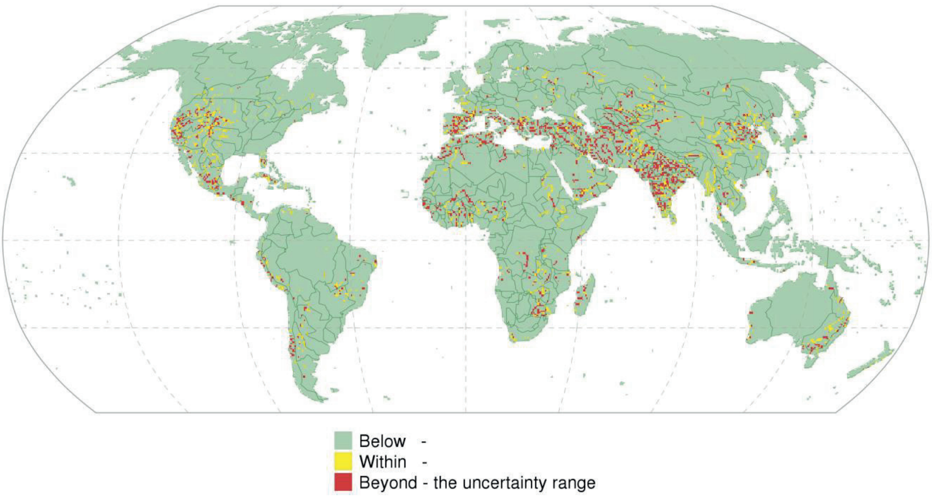

Figure 4.1: Current status of environmental flow alterations. The degree to which EFRs are tapped is expressed as the transgression-to-uncertainty ratio (>5% ”within uncertainty range”, >100% ”beyond uncertainty range”, see Methods), averaged over months with EFR transgressions (1980-2009, 0.5◦ resolution). Borders delineate Food Production Units.

In specific regions, however, the relative contribution of irrigation is much higher, as illustrated in Figure 4.3a. Regions that are simulated to undergo a ≥10% production decline with rigorous implementation of EFRs are currently inhabited by 1.1 billion people, 80% in developing countries (Supplementary Table B3). Since agriculture is at the centre of development and poverty reduction, unambiguous societal impacts are to be expected in default of other adaptation or compensation measures. Case study observations confirm complex difficulties in water re-allocation and infrastructure re-organisation for ecosystem conservancy if environmental flows are tapped already (Le Quesne, Kendy et al. 2010, Hermoso, Pantus et al. 2012, Blanco-Gutiérrez, Varela-Ortega et al. 2013). It is yet a prerequisite to avoid additional and sometimes irreversible degradation of aquatic ecosystems and linked therewith to achieve stable and resilient food production systems needed to ground nested environmental, social, and economic sustainability.

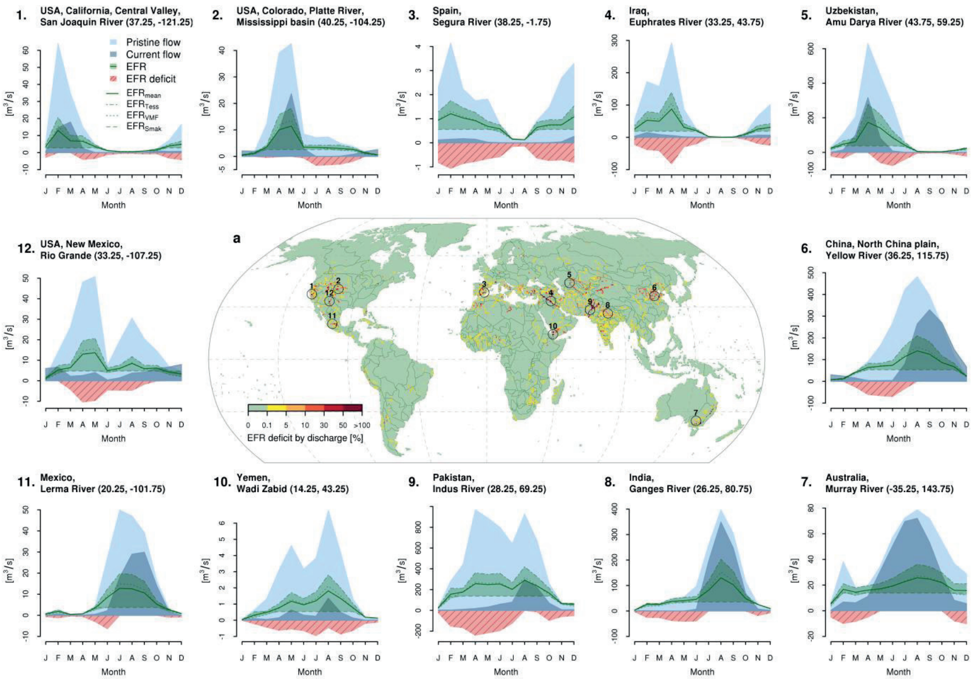

Figure 4.2: Severe alterations of environmental flows at selected river stretches. The map (**a**) illustrates the ratio of mean annual EFR transgressions to mean annual discharge (1980-2009, 0.5◦resolution). Hydrographs (**1.-12.**) highlight exemplarily river stretches with severe EFR alterations, their geographic locations (latitude, longitude coordinates in figure title) are superimposed on map (a). Different EFR methods (mean, Tessmann (Tessmann 1980), VMF (Pastor, Ludwig et al. 2014), Smakhtin (Smakhtin, Revenga et al. 2004), see Methods) are indicated through different line types, highlighted EFR transgressions relate to the mean of EFR methods.

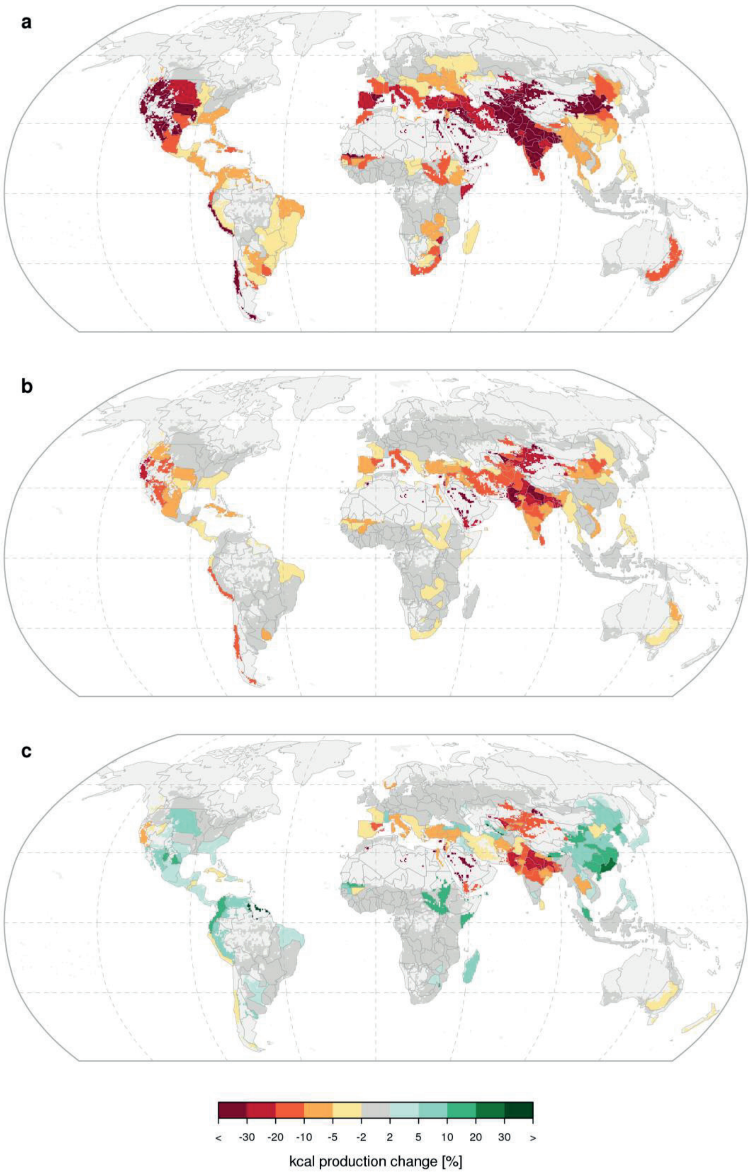

Figure 4.3: Governing environmental flows affects food production. The maps illustrate the change in total (i.e. rainfed and irrigated) kcal production in the absence of irrigation (**a**), with irrigation constrained by EFRs (mean of three EFR methods) (**b**), and with upgraded irrigation (Table 4.1) constrained by EFRs (**c**), aggregated to Food Production Units (1980-2009). Regions with marginal change are shaded (dark grey) and cells without significant cropland fraction (<0.1%) are masked (light grey).

4.3. Discussion

Field-based and global modelling studies indicate that management improvements can advance crop water productivity on a considerable scale (Deng, Shan et al. 2006, Molden 2007, Brauman, Siebert et al. 2013). To be paired with EFR constraints, we here develop an irrigation upgrade scenario as one example out of a spectrum of effective farm water management options (Jägermeyr, Gerten et al. 2016). Our simulations suggest that a transition from surface to sprinkler irrigation systems (using half of the saved consumptive losses for expansion) would suffice – at global level - to outweigh kcal losses associated with a worldwide implementation of EFR policies. Irrigation withdrawals would thereby further decrease to about half the current amount through reductions in conveyance losses and return flows. Irrigation water consumption (withdrawals minus return flows and drainage losses) remains at the same level as under EFR constraints without irrigation improvements (~34% below current value), but with higher shares of productive water consumption (plant transpiration), which reflects the increase in irrigation water productivity (Jägermeyr, Gerten et al. 2015) (Table 4.1). Yet, even under such improved management, 35% of irrigated cropland would remain with a ≥10% kcal loss, mostly in Central and South Asia, which is compensated globally by production gains in other regions (Table 4.1, Figure 4.3c). More ambitious interventions would be needed to minimize local impacts in regions with strong irrigation dependency and significant EFR alterations: for example, combinations of different water management strategies; rainwater management (water harvesting, mulching, conservation tillage) and large-scale irrigation upgrades are associated with sizeable potentials in these regions (Jägermeyr, Gerten et al. 2016).

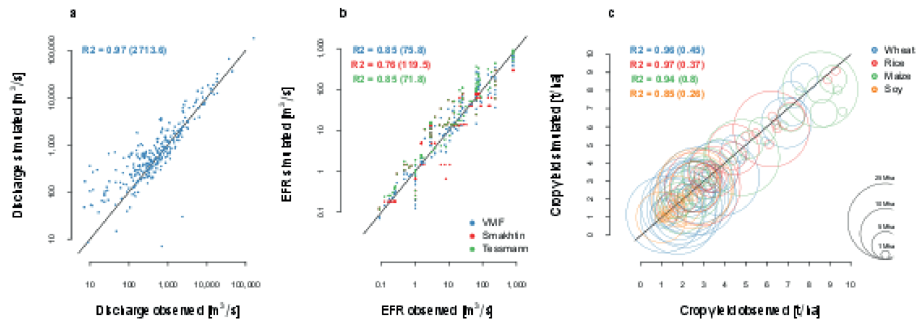

Figure 4.4: Evaluation of LPJmL-simulated key variables. Validation results against observational data are highlighted for mean annual discharge (**a**), environmental flow requirements respectively for three differing calculation methods (**b**), and country-level crop yields (calibrated for management intensity) for main staple crops and the respective top 30 producer countries, chart symbols are scaled by country cropland extent (**c**). The coefficient of determination (*R*2) is shown along with the root mean square error in parenthesis (all data 1980-2009, further details in Methods).

Eventually, incorporating ecological landscape approaches offer additional important merits such as soil fertility optimisation and advanced crop varieties that will further maximise synergies and thus crop water productivity - promising examples have been demonstrated (DeFries, Ellis et al. 2012, Chen, Cui et al. 2014, Rockström, Williams et al. 2016). Overall, the here quantified water management strategy is a showcase to illustrate opportunities to thrive within planetary environmental guardrails (Evers 2015). While not exhaustive, it highlights that farm water management across scales, linked to sustainable environmental flow regulations, would greatly assist the intricate task of such implementations paired with the goals of poverty reduction and agricultural productivity increase as outlined by the SDGs. A number of local EFR implementations prove successful (Le Quesne, Kendy et al. 2010), e.g. Uzbekistan set clear policy targets for water use and savings and was able to reduce its proportion of water resources used , and the 'redline' water policies in China illustrate the integration of national legislations with local institutional frameworks (Liu, Zang et al. 2013). Although the validity of EFRs has become internationally accepted and in many countries provisions are being developed (Declaration 2007, Le Quesne, Kendy et al. 2010, EC 2015), the systematic and comprehensive quantification of EFRs poses methodological, institutional, and financial challenges and is thus still insufficient. Together with often ineffective governance, this explains why existing licenses and policies are not yet being implemented (Le Quesne, Kendy et al. 2010, Biermann, Abbott et al. 2012), although it is clear that EFR assessment and regulation should be a basic requirement of Integrated Water Resource Management, as outlined in the EU Water Framework Directive (EC 2015). That said, the concept has not yet gained the critical influence needed to ensure environmentally sustainable basin management in competition with other water users like agriculture and industry (Poff, Richter et al. 2009). To develop effective national policy mechanisms, comprehensive local and regional assessment and monitoring programmes comprising field data, regional models, and expert judgment are inevitable (Acreman and Ferguson 2009, Poff, Richter et al. 2009, Le Quesne, Kendy et al. 2010). But we also acknowledge the need for more cost-effective, flexible EFR quantifications that, most importantly, conflate the global picture and provide assistance in international decision making, as particularly needed in the process of implementing the ambitious but unspecified SDG water agenda (Figure 4.5).

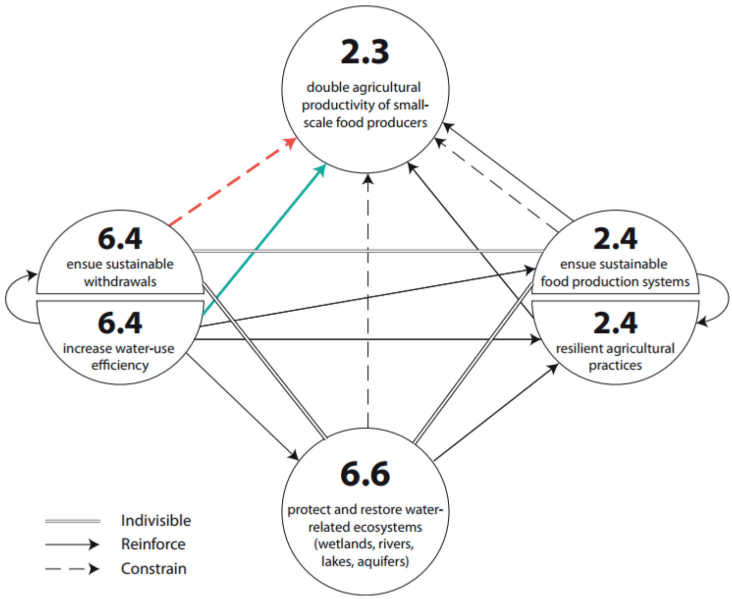

Figure 4.5. Schematic representation of potential conflicts between SDG goals

Through a simplified study (e.g. channel and habitat maintenance floods not considered) with a consistent approach at global level, this study adds quantitative evidence towards that end, providing a process-based quantification of EFRs in dynamic coupling with crop production, while accounting for intra-annual variability and spatial detail. The well-validated calculation approach (Figure 4.4) appears robust as we incorporate three different EFR methods and the dynamic modelling capacity plays out water trade-offs along the river network. Moreover, we assess exemplarily the water-gap, or in other words, the current pressure on the freshwater boundary needed to be overcome in a sustainable farming system and our study is thus in support of the 2030 Agenda for Sustainable Development.

The implementation of the 2030 Agenda clearly requires operative and dedicated governance, but the current lack of established tools and thresholds to quantify related targets form a barrier to translate the agreed principles into concrete action. For example, the indicator for sustainable freshwater withdrawals (6.4.2) was proposed to be directly linked to the EFR concept (UN 2016), but ultimately not stipulated. We here argue that accounting for EFRs is pivotal for attaining sustainable withdrawals (target 6.4) and food production systems (target 2.4), and that the EFR concept may form a basis on which to build operative policy measures, thereby linking the planetary boundaries with the 2030 Agenda. The planetary boundary for human freshwater use defined on behalf of the EFR concept provides a clear and actionable target, but further dimensions also need to be addressed (inaccessible flows, groundwater, pollution).

4.4. Conclusion

Finally, our study highlights that the achievement of sustainable river management in face of internationally stipulated goals for food security and poverty reduction would greatly benefit from integrated strategies that put strong emphasis on adaptation measures through improved farm water management. However, associated opportunities in e.g. rainwater harvesting have not gained required international attention among high-level development policies (Rockström and Falkenmark 2015). Advances in sustainable intensification are coupled to important socio-economic and environmental co-benefits (Rockström, Williams et al. 2016), which become particularly relevant in view of agricultural outlooks suggesting that crop calorie production needs to be increased by >60% in the forthcoming decades to eradicate hunger among the growing human population (FAO 2016). How to achieve this goal against a backdrop of climate change and environmental degradation, while staying within the safe operating space of the Earth system as delineated by the nine planetary boundaries remains one of the grand societal challenges.

Chapter 5

Balancing food security and water for the environment under global change

Freshwater ecosystems are home to the most threatened species on earth. Therefore, attention to the effects of water use on these ecosystems has increased. In this study, we quantify how respecting environmental flow requirements (EFRs), or the water required to sustain freshwater ecosystems, would modify land use, agriculture production, water use and international bilateral trade. We show that, by 2050, climate change and, respecting EFRs, would imply a drastic reduction in water use for irrigation (up to 60%) and a conversion of irrigated to rainfed agriculture area (only fed by rainfall) by 60 Mha (mainly in China and India). To compensate for this reduction in total irrigated agricultural production, international bilateral trade would need increase by 14-16% compared to a business-as-usual scenario with higher exports from water-abundant regions (South America and South-East Asia) to water-scarce regions such as Middle East, North-Africa, China and the Indian sub-continent. Regions with high conversion of irrigated to rainfed area would be more sensitive to climate variability and, thus, less food self-sufficient than today. However, here we show that trade would allow the maintenance of food production globally while sustaining EFRs despite future population growth and climate change.

Based on:

Pastor A.V., Palazzo A., Havlik P., Obersteiner M., Biemans H., Wada Y., Kabat P., Ludwig F.: Balancing food security and water for the environment under global change (in revision)

5.1. Introduction

Freshwater ecosystems are under increasing pressure due to large scale water abstraction for human needs (Loh, Collen et al. 2010, WWF 2014). About 70% of the water abstracted from freshwater systems is used for irrigation and about 40% of our food is produced on irrigated lands (Wada, Wisser et al. 2013). By 2050, without major policy interventions, human water use and expansion of irrigated area are expected to rapidly increase because of population growth and food demand increase (Molden 2007, Sauer, Havlík et al. 2010, Alexandratos and Bruinsma 2012, Cosgrove and Rijsberman 2014, Elliott, Deryng et al. 2014, Gleick and Ajami 2014). In addition, climate change is likely to impair future agriculture production (Weedon, Gomes et al. 2011, Ejaz Qureshi, Hanjra et al. 2013, Porkka, Kummu et al. 2013, Ray, Mueller et al. 2013, Wheeler and von Braun 2013).

Future increasing demands in human water use is likely to increase the pressure on riverine ecosystems (Falkenmark, Rockström et al. 2009). To limit biodiversity loss in riverine ecosystems, Environmental Flow Requirements (EFRs), or the flow required to sustain freshwater ecosystem, have been defined for many river systems around the globe (Tharme 2003, Declaration 2007). Until recently, most methods used to define EFRs were applied to single river basins or estimated with a "rule of thumb" at global scale (Pastor, Ludwig et al. 2014). The lack of consistent and robust methods to properly quantify global EFRs limited assessments to predict future land use and food production. However, the recently developed Pastor et al. (2014) Variable Monthly Flow Method (VMF) was designed with refined spatial and temporal scales to be applied globally (Pastor, Ludwig et al. 2014). Using this method, planetary boundaries of freshwater water resources were estimated at 2800 km^3 yr^{-1}, equivalent to 7% of total runoff, which is lower than in most previous assessments (Gerten, Hoff et al. 2013, Steffen, Richardson et al. 2015). This indicates that less water is available for future agriculture production and only limited expansion of irrigated agriculture is possible unless major investments in water infrastructure are made (Biemans, Haddeland et al. 2011, Jägermeyr, Gerten et al. 2016).

To increase food production, different solutions related to land and water management have been studied such as improved crop management (e.g. shifting to less water intensive crops) and increased food trade from water-abundant to water-scarce areas (de Fraiture and Wichelns 2010, Ercin and Hoekstra 2014, Jägermeyr, Gerten et al. 2016). However, most of these studies often do not integrate all components affecting water management such as climate change, temporal variation in water availability, water demand from other sectors and EFRs (Rosegrant, Cai et al. 2002, Alcamo, Flörke et al. 2007, Shen, Oki et al. 2008, Nelson, Valin et al. 2014, Bonsch, Popp et al. 2015). As a result, developed land use patterns are often unsustainable in terms of water use and availability (Viala 2008, Biemans, Haddeland et al. 2011). Due to the lack of proper assessment frameworks, it is still unclear how global food production would be affected if EFRs were respected.

This paper focuses on finding sustainable solutions to produce sufficient crop products to feed a growing population while at the same time respecting the water needs of the environment. We developed a modelling framework linking water resources, crop production, land use and bilateral international trade optimizing models including EFRs (see Appendix B and Appendix Figure A1). We used existing socio-economic and climate scenarios (see Methods) and we developed four water management policy scenarios (INVEST, EXPLOIT, ENVIRONMENT, ENVIRONMENT+) with ascending level of water restrictions and two trade scenarios (uncontrainsted (UncT) and constrained (ConT)) to evaluate the impact of trade restrictions on crop production.

Here, by applying a robust integrated framework to evaluate future food production, we show that to sustain EFRs (ENVIRONMENT and ENVIRONMENT+ scenarios), we need to increase international trade by about 15% worldwide compared to a standard scenario (INVEST scenario) mainly from water abundant regions to water-scarce regions. We conclude that international trade could alleviate climate change impacts and future water restrictions at a global scale. However, reorganization of food production could imply regional impacts such as a loss of irrigated production in Mediterranean countries and in Asia as well as an increase in rainfed production in Latin America.

5.2. Methods

5.2.1. *Modelling framework*

To assess the impact of implementing EFRs on global water use, future food production strategies and land use change, a modelling framework was developed (Supplementary Figure C1). The framework links the Global Biosphere Management Model (GLOBIOM) with a water availability and demand model (Lunt Potsdam Jena and managed land, LPJmL) including a new module assessing EFRs (see explanation below).

5.2.2. *GLOBIOM model*

GLOBIOM is an economic partial equilibrium model which allocates agricultural crops and commodities based on an endogenous price balance between demand and supply. It includes agriculture, bioenergy and forest modules to optimize land-use allocation(Havlík, Schneider et al. 2011, Havlík, Valin et al. 2014). The model optimizes food and livestock production at a minimum cost under socio-economic and biophysical constrains. The baseline year is 2000 and is recursively dynamic (10 year time-step). The basic spatial unit for food supply is 2 by 2 degrees and food demand is defined at the level of 30 world regions (Supplementary Table C1). Based on information such as current land use, water availability, crop yield and investment cost GLOBIOM simulates change in agriculture area (rainfed and irrigated), forest and other natural areas. Urban areas are static and pasture areas are driven by livestock feed demand. Forest area is partially preserved according to forest management regulations(Havlík, Schneider et al. 2011). Food demand in GLOBIOM is driven by population, per capita income and responses to food prices. Population and income are external inputs into the model. Prices are endogenous to the model and depend on technology, natural resources and consumer preferences.

5.2.3. *Socio-economic scenarios*

Future socio economic development including population, Gross Domestic Product (GPD) and technology change, was based on the Socio-economic Pathway 2 (SSP2)(Kriegler, O'Neill et al. 2012, Samir and Lutz 2014). SSP2 is the middle of the road scenario assuming moderate adaptation and mitigation challenges and a dietary requirement of 3000kc/person/day based on Food and Agriculture Organization (FAO) projections(Kriegler, O'Neill et al. 2012, Samir and Lutz 2014, Fricko, Havlik et al. 2016).

LPJmL model – hydrological model. Water availability was simulated with the Lund-Potsdam-Jena managed land model (LPJmL), which is a global dynamic vegetation model which simulates water and carbon cycles (Gerten, Schaphoff et al. 2004, Bondeau, Smith et al. 2007). The water module was developed with a river routine and the implementation of reservoir operation (Rost, Gerten et al. 2008, Biemans, Haddeland et al. 2011). Water availability was simulated with LPJmL from 2000 to 2050 at a 0.5° x 0.5° spatial resolution. We calculated average monthly water availability for every 10 year time-step from 2000 till 2050 to be used as input into GLOBIOM. The mean monthly runoff estimated by LPJmL was re-distributed according to the average discharge rates in each river basin to have a good spatial representation of water availability within GLOBIOM (Schewe, Heinke et al. 2014). Water availability is aggregated to the Land Unit ID (LUID) with a total of 4845 simulation units within GLOBIOM.

5.2.4. *Climate scenarios*

For the climate change scenarios, LPJmL was driven with the bias-corrected output of two commonly used Global Climate Models (MPI-ESM-LR & HadGCM2-A0) using high future greenhouse gas concentrations (Representative Concentration Pathway, RCP 8.5)(Van Vuuren, Edmonds et al. 2011). Climate forcing data was extracted from the ISIMIP database(Hempel, Frieler et al. 2013, Warszawski, Frieler et al. 2014).

5.2.5. *Water management policy scenarios.*

To assess the impact of EFR restrictions on future land and water use, agricultural production and trade, four policy scenarios for water management were developed:

- The Water Investment Scenario (INV-INVEST) assumes large scale development of irrigation infrastructure and water re-allocation. As a result this scenario assumes that all freshwater within a region can be used and reallocated to optimize irrigation and economic constrains such as meeting future crop demand.

- The Maximum Exploitation Scenario (EXP-EXPLOIT) assumes that all freshwater from rivers and groundwater aquifers can be used up to full depletion at the land unit (2 by 2 degrees) but cannot be reallocated within a region. However, water use for agriculture is constrained by water availability and water demand from other sectors (human consumption, industry and household) at a spatial land unit level and at a temporal monthly time step.
- The Environmental Flow Requirements scenario (ENV-ENVIRONMENT) assumes that water needs to be allocated to the environment first. Further, water use for irrigation is restricted by water demand from other sectors (human consumption, industry and household) at land unit level. EFRs were estimated using the Pastor et al. (2014) VMF method (Pastor, Ludwig et al. 2014).
- The High Environmental Flow requirement scenario (ENV+) is the same as the ENV scenario with the exception that the EFR requirement are 50% higher than the ENV scenario to put a higher priority on the maintenance of ecological status of the freshwater ecosystem over other users.

All water use restriction scenarios were analyzed with climate change (CC) and without climate change (NoCC) (Supplementary Figure C1).

5.2.6. *Description of trade scenarios*

We designed trade scenarios to evaluate how markets (through bilateral trade) compensate water scarcity at local levels due to biophysical limitations, climate change, and reduced water availability for EFRs.

- Constrained Trade (Unc_T): regional bilateral trade flows are fixed to the reference scenario (EXPLOIT without climate change) with SSP2 yield projection and no irrigation use efficiency increase.
- Unconstrained Trade (Con_T): regional bilateral trade flows uses the scenario (EXPLOIT without climate change) with SSP2 yield projection and no irrigation use efficiency increase as a reference but can adapt to different scenarios (INVEST and ENVIRONEMNT) with increasing trade if necessary.

For further details on crop model, EFRs calculation and calibration of water restrictions in the GLOBIOM model, see Appendix Methods. All the analyses and maps have been computed with R studio (https://www.rstudio.com/).

5.3. Results

5.3.1. *General results – trade-off between land-use, water use and trade*

In this study, we expose the trade-offs between land use, water use and trade in order to supply food production by 2050 under 4 scenarios (Figure 5.1, Supplementary Figure B3). We show that agricultural crop production needs to be more than doubled by 2050 (Figure 5.2 and Supplementary Table C4). We show that the impact of EFRs on water use implies a reduction of 40 to 60% of water use which outpaces the impact of climate change on water use (-2%) and the impact of increase of food demand on water use (+19%). For that, we compared 4 scenarios: INVEST (business-as-usual scenario), EXPLOIT (maximal use of local water resources) and ENVIRONMENT and ENVIRONMENT+ scenarios (with moderate and high EFRs restrictions). Under the INVEST scenario, food production would come from a large expansion of agriculture in irrigated and rainfed land, assuming large scale investments in water infrastructure. The INVEST scenario results in the highest water use and the lowest trade use which allow regions to be more food secure than in other scenarios (Table 5.1 and 5.2). Under the EXPLOIT scenario, food production increase would mainly come from an expansion of agriculture rainfed area in Latin America and Mediterranean regions (Figure 5.3). The EXPLOIT scenario results in the most extensive land use with intermediate trade and water use. Finally, ENVIRONMENT and ENVIRONMENT+ scenarios show that to respect EFRs, it is necessary to increase interregional trade in agricultural crop products by 15% compared to INVEST and EXPLOIT scenarios and would imply a large conversion of irrigated to rainfed area in China and India (Figure 5.4-5.5).

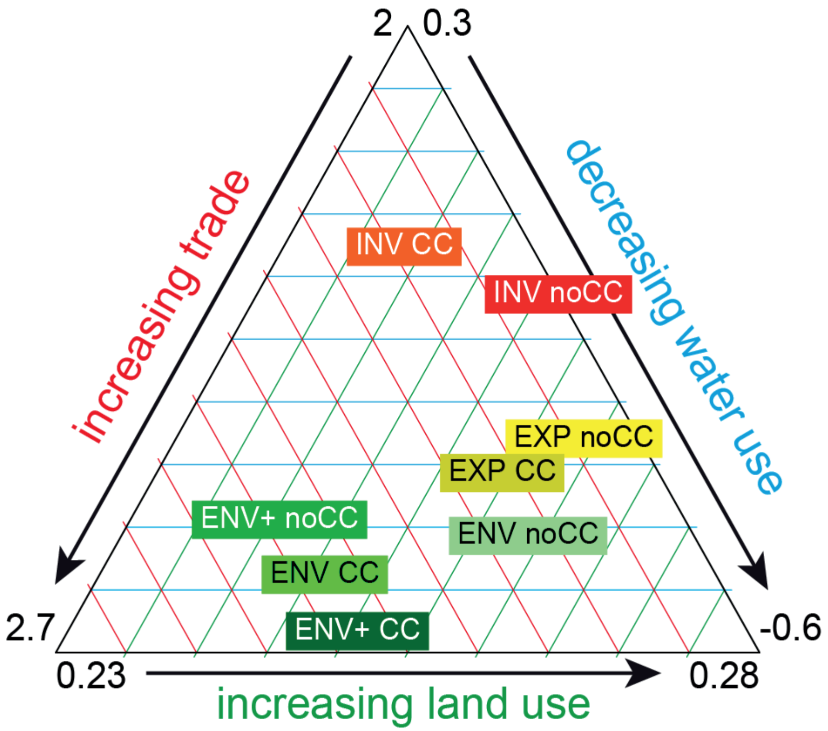

Figure 5.1. Changes in land use, water use and trade under water management scenarios and climate change. Plots showing tradeoffs between land use, water use and trade for different water use restriction scenarios (INVEST (INV), EXPLOIT (EXP), and ENVIRONMENT (ENV)). CC indicates climate change and no CC indicates no climate change at global scale in 2050. Values are calculated as the ratio of the difference between the actual variable in 2050 and the baseline 2000 over the variable of the baseline 2000.

Table 5.1. Water withdrawal for agriculture under climate change, constrained trade and water management scenarios. Units are in km3 yr-1.

Scenarios	noCC[a] Unconstrained Trade	CC[b] Unconstrained Trade	noCC[a] Constrained Trade	CC[b] Constrained Trade
Baseline 2000	2516	2516	2516	2516
INV_2050	2983	2911	2983	2911
EXP_2050	2461	2261	2313	1986
EFR_2050	1774	1561	1636	1371
EFRh_2050	1440	1219	1299	1035

[a] NoCC stands for no climate change,

[b] CC stand for climate change

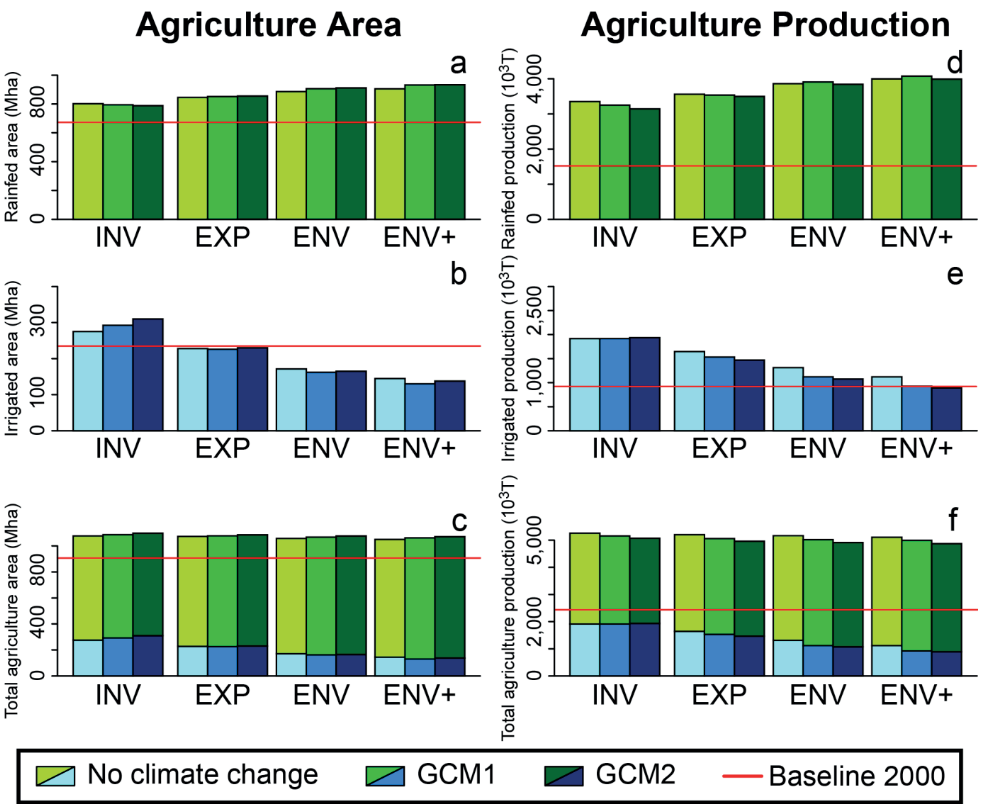

Figure 5.2. Global agriculture area and production under 4 water management scenarios. Total global rainfed area (a), irrigated area (b) and total agricultural area (c) under different water management scenarios and total rainfed (d) and irrigated agricultural production (e) and total agriculture production (f) for four different water management scenarios with increasing restrictions on water availability. Clear blue and green bars represent no climate change, and light color bars represent climate change with GCM1 (MPI_ESM_LR) and darker colored bars represent climate change with GCM2 (hadGCM2_A0). Red line represents the baseline total agriculture area and production in 2000.

5.3.2. *Future water use for irrigation under 4 water management policies*

Implementing EFRs has a large impact on water use for irrigation in 2050 (Table 5.1). In restricted water scenarios (ENVIRONMENT, ENVIRONMENT+), water use would decrease up to 1440 km^3 yr^{-1} without climate change and up to 1219 $km^3 yr^{-1}$ with climate change (up to 50% reduction in water use compared with EXPLOIT scenario in 2050). Under INVEST scenario, water use would be increased by about 500 km3 $km^3 yr^{-1}$ (+10%), assuming use of non-renewable water resources. By limiting water use to local use of surface water and groundwater (EXPLOIT scenario), water use for irrigation would be limited to 2500 $km^3 yr^{-1}$ and would be reduced by 10% in key irrigated areas such as India and China due to adverse impacts of climate change and due to limited access to local water resources (Figure 5.3 and Supplementary Table C3).

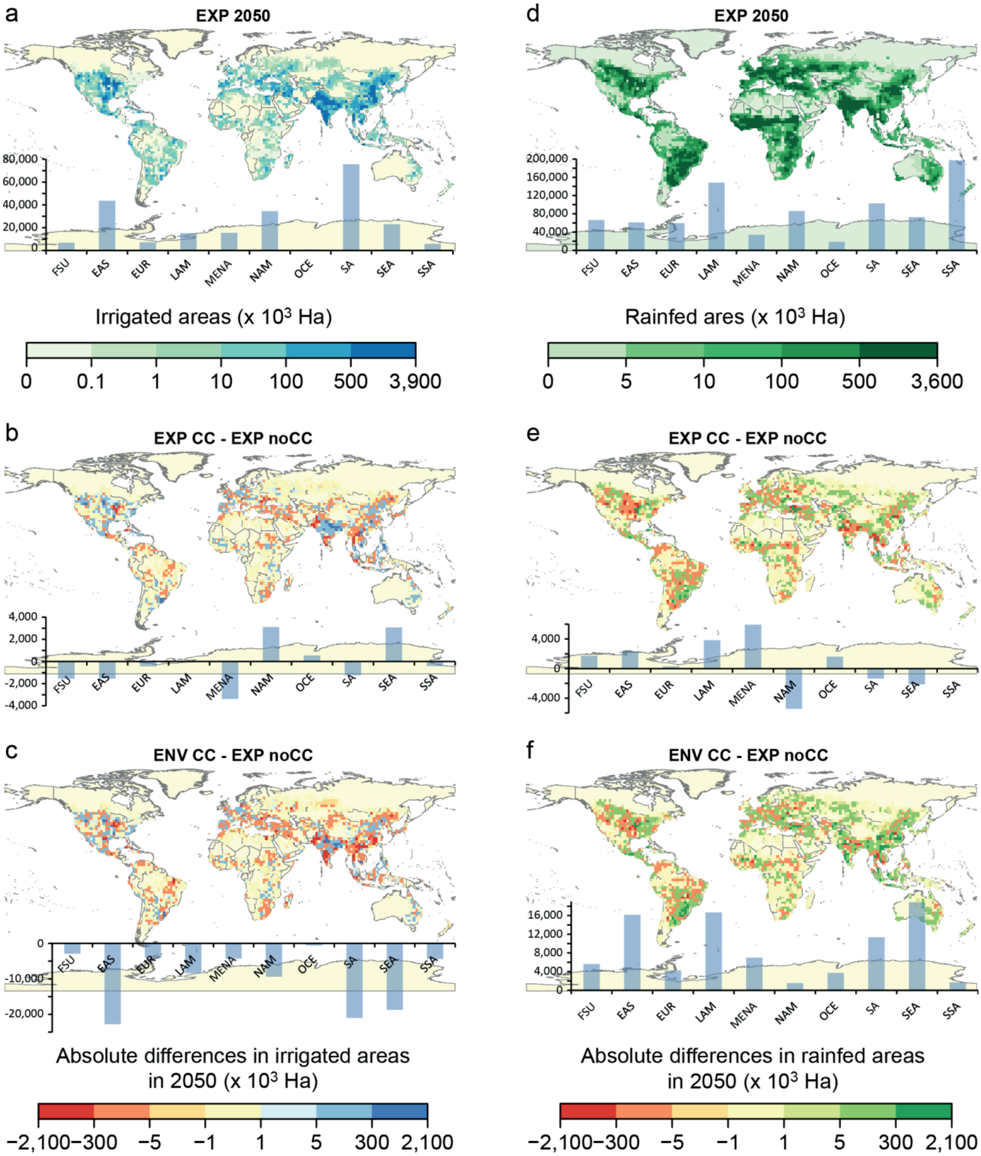

Figure 5.3. Impact of climate change and environmental flow requirements implementation on cropland repartition. Spatial distribution of irrigated (a) and rainfed agricultural (d) area for EXPLOIT scenario in 2050 assuming no climate change. Panels b and f show the impact of climate change on distribution of rainfed and irrigated agricultural area as the difference between the EXPLOIT CC and the EXPLOIT noCC scenarios. Panels c and f show the impact of environmental flows and CC on the distribution of irrigated and rainfed agricultural area as the difference between the ENVIRONMENT CC and the EXPLOIT NoCC scenario.

5.3.3. *Future land use change under water restrictions*

Increasing agriculture crop production under INVEST scenario would be mostly achieved by a 19% expansion of total crop irrigated and rainfed area (Figure 5.3 and Table 5.3) while implementing EFRs (ENVIRONEMNT scenario) would imply a large scale reduction in irrigated areas especially in Asian countries (Figure 5.3). For example, we show that, under the INVEST scenario, irrigated area could expand up to 300 Mha while, while under the EXPLOIT scenario, irrigated area may be reduced to 227 Mha and up to 161 Mha under EFRs restrictions (Figure 5.2, Supplementary Table C3).

5.3.4. *Consequences on future agriculture production*

At the moment, 40% of crop production comes from irrigated area. By 2050 and under EFR restrictions, we show that only 20% of crops could be produced on irrigated land, compared to 38% for the INVEST scenario (Supplementary Table C4). Under EXPLOIT and INVEST scenarios, most of the irrigated production comes at the expense of EFRs, especially in China and India. Under the ENVIRONMENT scenario, irrigated area and production would be greatly reduced in these regions (Figure C3, Supplementary Table C3). For example, in the MENA region and in China, EFRs restrictions in combination with climate change would result in a 50% reduction of irrigated area which represents between 7 to 25Mha compared with the EXPLOIT scenario. While in Europe and Latin America, irrigated area would only decrease by about 35% (Figure 5.3 and Supplementary Table C3).

5.3.5. *Future agriculture conversion*

Our results show that there are two main mechanisms which compensate for the loss of irrigated agriculture (Figure 5.1). The first is through expansion of rainfed area and through conversion from irrigated to rainfed area (Figure 5.2&3, Supplementary Figure C2 and Table C3-4). The second mechanism is by increasing global crop trade (Figure 5.4-5.5; Table 5.2). Surprisingly, the total cropland area remains relatively equal under all scenarios in 2050 (Figure 5.2). Irrigated to rainfed conversion takes place on croplands which would preferably be irrigated (Supplementary Figure C2). However not all irrigated land is suitable for rainfed agriculture such as in China and the Indian sub-continent where we show a reduction of cropland area of about 40MHa. To compensate for this loss of cropland area, we show an expansion of cropland area by up to 20Mha in Latin America, Africa and Russia under the ENVIRONMENT scenario (Figure 5.3, Supplementary Table C3).

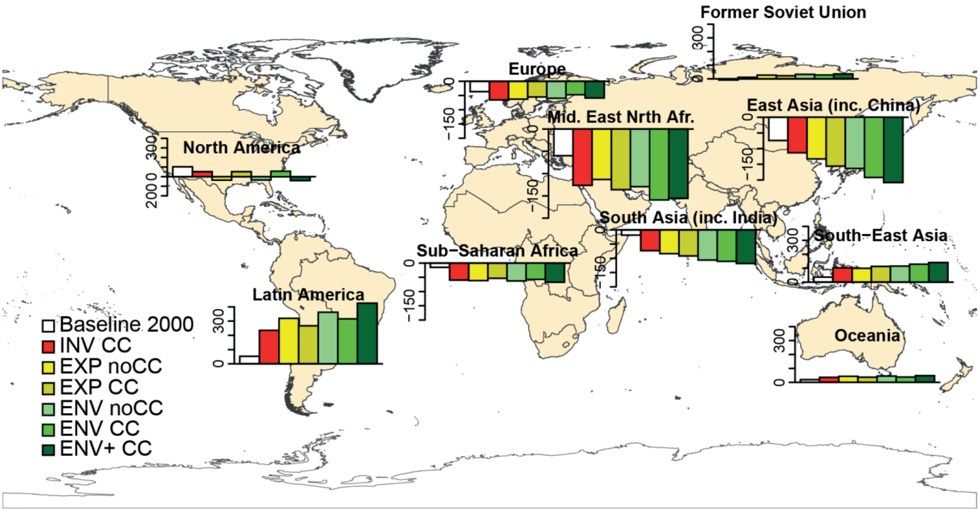

Figure 5.4. Impact of respecting Environmental Flow Requirements and climate change on net trade of agricultural products. The net trade of dry matter agriculture production (10^9g year^{-1}) of agriculture production from 10 global regions for the 2000 Baseline and for different future water management scenarios (INVEST (INV), EXPLOIT (EXP), and ENVIRONMENT (ENV)) is displayed. CC indicates climate change and no CC indicates no climate change. Positive numbers indicate net exports and negative number net imports of agricultural products.

Table 5.2. Agriculture consumption (in million tons) and the percentage of net trade per region. In the second line (values in parenthesis): positive values represent exports and negative values represent imports as a share of agriculture consumption. Consumption is in 1000 metric tons of dry matter per year

Trade		unc_T	unc_T	unc_T	unc_T	unc_T	unc_T	unc_T	unc_T	fix_T	fix_T	fix_T
Climate change		noCC	noCC	noCC	noCC	CC	CC	CC	CC	CC	CC	CC
Water management		INV	EXP	ENV	ENV+	INV	EXP	ENV	ENV+	EXP	ENV	ENV+
Year	2000	2050	2050	2050	2050	2050	2050	2050	2050	2050	2050	2050
World	4323	10221	10098	10049	9971	9900	9693	9610	9534	9533	9059	8781
	(9)	(11)	(12)	(13)	(14)	(11)	(11)	(12)	(13)	(9)	(9)	(9)
CIS	184	308	320	326	329	330	338	339	339	335	335	333
	(-5)	(8)	(9)	(9)	(9)	(3)	(6)	(8)	(8)	(9)	(8)	(9)
EAS	764	1218	1142	1042	951	1171	1093	934	915	1104	846	763
	(-11)	(-12)	(-12)	(-18)	(-29)	(-11)	(-17)	(-26)	(-31)	(-13)	(-15)	(-14)
EUR	363	494	509	518	522	518	525	532	536	514	515	503
	(-11)	(-15)	(-13)	(-12)	(-13)	(-15)	(-12)	(-10)	(-10)	(-11)	(-10)	(-11)
LAM	821	2556	2589	2703	2758	2654	2709	2860	2917	2567	2539	2482
	(6)	(11)	(12)	(13)	(13)	(8)	(9)	(10)	(10)	(9)	(8)	(8)
MNA	125	342	325	271	247	363	333	257	239	324	266	242
	(-92)	(-49)	(-64)	(-153)	(-210)	(-53)	(-69)	(-162)	(-236)	(-52)	(-65)	(-64)
NAM	524	836	839	846	849	676	681	658	653	720	687	660
	(16)	(-4)	(-5)	(-7)	(-7)	(7)	(7)	(8)	(8)	(-1)	(-3)	(-5)
OCE	76	163	164	163	156	177	180	178	171	164	160	159
	(21)	(21)	(20)	(21)	(21)	(17)	(17)	(17)	(17)	(18)	(18)	(17)
SAS	721	2025	1926	1889	1836	1873	1701	1674	1624	1685	1633	1594
	(-3)	(-4)	(-7)	(-8)	(-8)	(-4)	(-6)	(-7)	(-8)	(-3)	(-3)	(-3)
SEA	450	1049	1052	1074	1108	989	1002	1061	1043	984	962	961
	(8)	(9)	(10)	(12)	(12)	(9)	(10)	(11)	(12)	(7)	(7)	(7)
SSA	295	1230	1232	1217	1215	1149	1131	1116	1097	1137	1117	1083
	(-5)	(-5)	(-6)	(-6)	(-7)	(-5)	(-5)	(-5)	(-6)	(-4)	(-3)	(-4)

5.3.6. *Future regional agriculture production shifts*

Respecting EFRs reduces total crop production by 1123Mt (-10 to -30%), primarily in Asia and the MENA region. To compensate for this regional production loss, increased crop trade in combination with higher agricultural crop production in other parts of the world is needed and 30% of the agriculture crop loss would be produced in Latin America, Russia and Europe. As a result, the interregional trade is higher in the ENVIRONMENT scenarios compared to both the INVEST and EXPLOIT scenarios (Figure 5.1-5.45.-5, Table 5.3; Supplementary Figure C4-Table C5).

5.3.7. *International trade by regions*

Results indicate that bilateral trade needs to increase by 5% to compensate for climate change alone, by 10-13% to compensate for EFRs alone and by 17-20% to compensate for combined climate change and EFRs (Figure 5.4; Table 5.2). In general, trade is increasing from water abundant regions such as Latin America and South East Asia to water scarce regions and highly populated regions (Figure 5.4-5.5) especially in China and in the MENA region who need to import more agricultural products under the ENVIRONMENT scenarios (Supplementary Table C5). This increased import is possible due to higher exports from Latin America and South-East Asia (Figure 5.4-5.5). Climate change further increases the imports in agricultural products in China and South Asia (including India). The impact of climate change on global food trade remains lower than implementing EFRs (Figure 5.5).

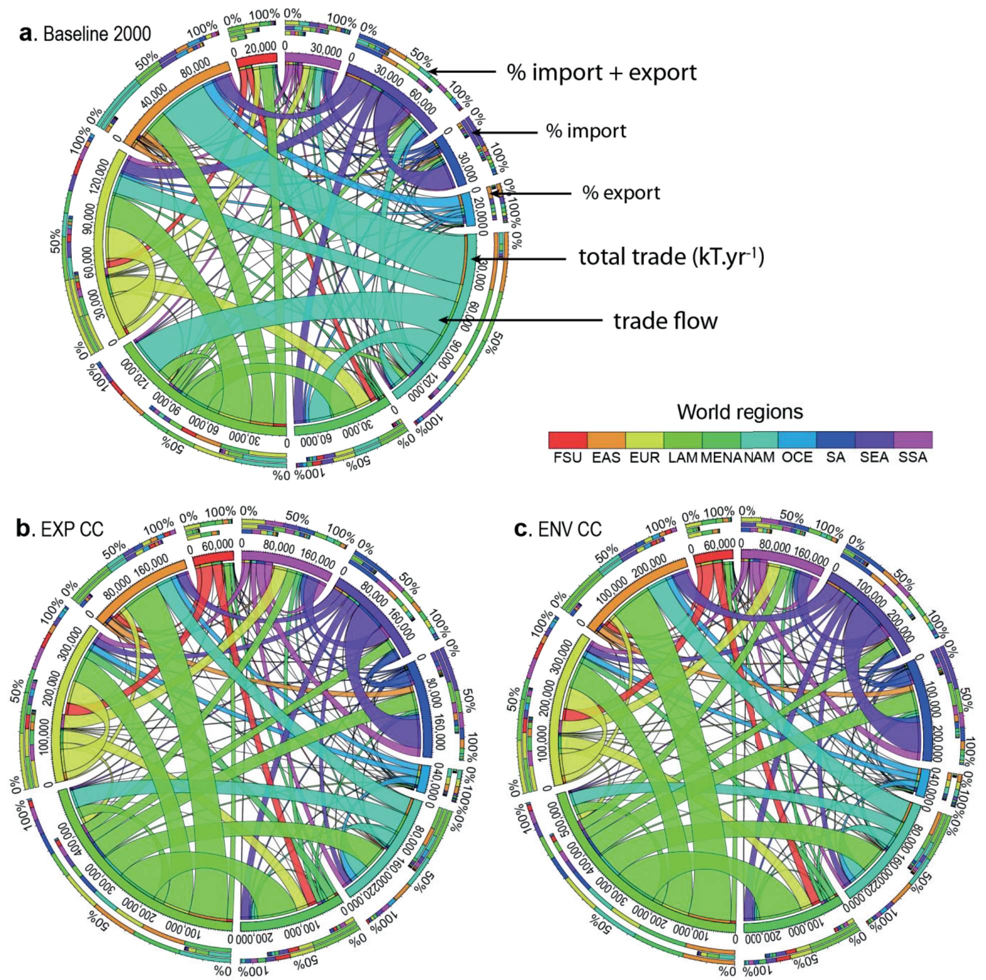

Figure 5.5. Future bilateral trade between regions. Circle flow diagrams showing bilateral agriculture trade between 10 global regions for the year 2000 (a), for the reference scenario (EXP) with climate change in 2050 (b) and the environmental flow scenario (ENV) with climate change 2050 (c). The second circle represents the percentage of agriculture production imported, the third circle represents the percentage of agriculture production exported, and the fourth circle represents the amount of agriculture production traded in kT yr^{-1}.

5.4. Discussion

This study focused on the question of how to meet future crop demand while at the same time respecting the water demands of the environment. Our results indicate that to protect freshwater by meeting EFRs, and supply sufficient food for future generations, irrigated area should be reduced by 30 % relative to the current situation. Interestingly our results indicate that a reduction in irrigated area does not have to result in additional cropland expansion at a global scale. What is needed is a large scale conversion of irrigated to productive rainfed land and increased interregional trade of agricultural products. The combined impact of climate change and EFRs would increase net trade by up to 15% worldwide with a main increase in exports coming from Latin America (+70%) and South-East Asia (+22%) while increases in imports would mainly occur in China (+38%), India (+33%) and MENA countries (+19%).

Our results indicate that increases in trade are necessary to adapt to climate change and allocate more water to the environment. However, trade in agriculture is still limited compared to other commodities because of freight cost and protective laws and regulations. For example, a study from Carole and Ignacio (2015) showed that trade can reduce global agricultural water use. However they show that increased trade also has other consequences such as decreased terrestrial biodiversity and local socio-economical changes. In addition, trade liberalization can increase the environmental impact in countries where environmental production laws are less restrictive. Trade can also be a tricky tool in times of food crisis or drought such as in 2007, increasing food insecurity of poor malnourished people (Suweis, Carr et al. 2015). In our analyses we may underestimate the impact of climate change on food production because our modelling system does not take into account inter-annual variabilities in food production while global warming and climate variability are likely to increase in the coming decades with extreme events(Field, Barros et al. 2014).

Our results show that it is possible to double agricultural production with an increase of cropland by only 20% (145-185 Mha). This assumes a much smaller future yield gap, a rapid update of improved technologies in especially Africa, Asia and Latin America. Tilman et al. (2011) showed similar estimates and found that food production can be doubled with a relatively small expansion of agricultural areas if different adaptation measures are taken to intensify agricultural production. However, a study by Wirsnenius et al. (2010) estimated an agricultural area expansion of 1600 Mha by 2030 which is much higher than our projections. This difference is partly due to less flexible trade-flows between regions and a less flexible land-use change scheme than in our study. Agricultural intensification can also have large impacts on the environment including freshwater ecosystems. Our study explicitly focused on the water quantity aspects of protecting freshwater habitats but this should not come at the expense of the water quality (by using extra nutrients) which can also have a large impact on freshwater biodiversity(Turner and Rabalais 2003).

Our results clearly show the trade-offs between land use, water use and food production versus trade. Especially in Asia and the MENA regions, respecting EFRs reduces national self-sufficiency ratios (Table 5.2). Currently both India and China have developed policies to obtain food security through high self-sufficiency ratios(Yu and Lu 2006). Our analyses show that reducing irrigated water use has large consequences for crop production and that increased imports are necessary to satisfy the demand for agricultural products. While our analyses show that this is possible at a macroeconomic scale, at lower scale this could still have serious consequences for food security(Margulis 2013). It is important that a regime with more agricultural trade is combined with policies guaranteeing food affordability because sufficient production does not guarantee access to food for all.

Finally, this study highlights the trade-offs between land-use, water use and international trade with a focus on respecting EFRs. However, reaching sustainable goals at a global scale remains a challenge, especially in the context of the Water-Food-Energy Nexus context where each component has a target to be respected without compromising environment(Bazilian, Rogner et al. 2011). For example, at regional level it remains a challenge to find trade-offs between water, food and energy such in South-East Asia where conflicts between downstream and upstream water users may exist and where pressure to increase use efficiency of soil and water remain a priority(Rasul 2014).

5.5. **Conclusion**

In conclusion, our results show that it is possible to meet both the global agricultural demand and the water needs of the environment without large scale increases in cropland. However it is necessary to re-allocate agricultural production from water limited to water abundant regions and simultaneously increase trade in agricultural products significantly. Our analyses show that if trade is not allowed to compensate for crop production loss, global agricultural production would reduce and it becomes more difficult to meet the future demand of crop products and sustain environmental flows. Our results show that trade can help to mediate the trade-offs between land and water. Increases in trade and trade liberalization are often mentioned to have negative impacts on the environment and access to food by disadvantaged communities but our results show that an increase in global trade can also help to meet sustainability goals such as satisfying crop demand and guaranteeing sufficient water for the environment.

Chapter 6

Synthesis

6.1. Introduction of synthesis

6.1.1. *Summary of main results*

Freshwater ecosystems are among the most threatened species on Earth before terrestrial and marine ecosystems (Loh, Collen et al. 2010, WWF/ZSL 2016). During the last decades, river degradation due to anthropogenic flow alteration has been widely reported (Foley, DeFries et al. 2005, MEA 2005). It is now generally acknowledged that respecting natural flow variability is essential for optimal ecosystem functioning (Poff, Allan et al. 1997, Arthington, Bunn et al. 2006, Poff and Zimmerman 2010). Natural river flows are altered mainly due to the development of reservoirs and large scale extraction of water for anthropogenic uses. Most of the water is extracted for irrigation and current water demand for agriculture represents about 70% of total water use (FAO 2016). In the coming decades, water demand for food production is predicted to increase and climate change is likely to intensify the water cycle. Therefore, it is necessary to define accurately how much water is available for food and ecosystems at refined time and spatial scales. However, until now, water availability for freshwater ecosystems was often neglected in global integrated assessments and Environmental Flow Requirements (if included) were roughly represented with annual proxies (Foley, DeFries et al. 2005, de Fraiture and Wichelns 2010, Elliott, Deryng et al. 2014). Furthermore, global water stress is still too often addressed at annual time scales while it varies intra-annually (Arnell 2004, Gosling and Arnell 2016).

This thesis aimed at defining: “how we can satisfy current and future water demand for ecosystems and food production under global change ?”. For that, the quantification of water demand for ecosystems to be used in global integrated assessments were refined and the future nexus of land use, water use and food production was addressed (Figure 6.1-6.2-6.3). In chapter 1, I formulated four research questions that were addressed in chapter 2, 3, 4 and 5. The first question focused on designing a refined global Environmental Flow (EF) method with explicit spatial and temporal scales with two requirements: ease of use and high robustness to be implemented in global assessments. The newly designed EF method was compared with four existing hydrological global methods for sensitivity analyses. Additionally, all the global methods were validated using local EFRs assessments based on ecological and hydrological measurements. The newly developed VMF method was the best fitted method for global use and was implemented in chapters 3, 4 and 5 of this thesis and in other global water and environmental assessments (Boulay, Bare et al. 2015, Gaupp, Hall et al. 2015, Steffen, Richardson et al. 2015). In Chapter 3, the aim of the study was to identify where, when and why EFRs were not satisfied globally (Figure 6.2). For that, EF deficit was defined as the flow that is lacking to satisfy freshwater ecosystems. The origin of the deficit was differentiated between (natural) climate variability and anthropogenic water extractions. The magnitude, timing, duration and frequency of the deficit were defined at global and regional scales. Correlations between the river flow regime (perennial to highly seasonal), the climate (tropical, temperate) and the level of flow alteration were found. Perennial flow regimes with low flow modifications were shown to be mainly subject to natural deficit (e.g. Congo). These last were classified with a low level of priority action for river restoration. However, conservation measures should not be neglected especially in tropical perennial rivers which contain the largest endemic populations of freshwater species worldwide (e.g. Amazon) (Oberdorff, Tedesco et al. 2011). Highly seasonal rivers with a lower base flow index and with high flow modifications were found to have higher anthropogenic deficits than deficits coming from climate variability (e.g. Indus, Ganges). In this case, priority of action is high and solutions to restore freshwaters encompass finding trade-offs between different water sectors (environment and irrigation). In chapter 4, EFRs was defined as the water user with the first priority (before anthropogenic demand) in the global hydrological and vegetation model (LPJmL). Global water use for irrigation and food production was quantified with EFRs implementation. Results show that the main SDGs goals on food security (SDG 2.3), sustainable agriculture production (SDG 2.4), sustainable water withdrawals (6.4) and water conservation (SDG 6.6) would be conflicting under EFRs implementation (Figure 4.5; Figure 6.2). Finally, the upgrade of irrigation techniques and use efficiency was tested and showed large alleviation in food insecurity. In chapter 5, an integrated modelling framework combining hydrology, land-use and trade to optimize future water use under global change was used (Figure 6.1-6.3). Future emission scenarios to simulate impact of climate change and future socio-economic scenarios were used. The optimization model was supplemented with water constrains on local water availability including EFRs. This study shows that it is possible to sustain both food security and water demand for ecosystems under global change if there is large scale reduction in irrigated agriculture in combination with increased international food trade. The main results of this thesis are summarized in Table 6.1. In the next sections the main results of the thesis are discussed in a broader context and an outlook to future research is given. This study is also placed in a broad scientific context and in the perspective of future contribution to water management and policy makers.

Table 6.1. Summary of main research questions and findings

Research question	Methods	Results
Q1: How can we represent EFRs at global scale ? (Ch 2)	Criteria of the newly developed Variable Monthly Flow (VMF) method: - Follow natural flow variability - Ease of use and applicable to all GHMs - Comparison with other global methods - Tested and validated with local study cases (located in different ecoregions using different flow regimes)	- VMF monthly requirements are calculated by using 30 to 60% of monthly natural flow (depending on flow seasonality) - On an annual scale, EFRs calculated with 5 EF methods represent between 20 and 50% of global runoff
Q2: Where, when and why EFRs are not met globally and what does that imply? (Ch 3)	EF deficit was defined as the flow that does not satisfy EFRs including: - The origin of the deficit: natural or anthropogenic ? - The timing of the deficit: which season ? - The magnitude: how big is the deficit compared to available flow ? - The frequency: how often does the deficit occur ?	- 50% of the EF deficit is caused by anthropogenic issues - Hot-spots of anthropogenic EF deficits are Asia, Mediterranean areas and West coast of US - Perennial flow regimes have usually a low deficit mainly caused by natural deficits - Highly variable and modified rivers usually have high seasonal deficits due to anthropogenic water extractions - Anthropogenic deficit usually exacerbates the natural deficit
Q3: What if EFRs were set as the first water user (before irrigation demand) ? (Ch 4)	Water demand for ecosystems was set as the first priority (before irrigation) in the river routing and irrigation scheme of LPJmL.	- 5-10% of global food production loss (5% calorie loss) - 30% loss of irrigated production (14% calorie loss) - Areas facing high challenges in meeting EFRs and food demand: north of Indian peninsula, Mediterranean countries and West Coast of US Increasing irrigation use efficiency by 50% can compensate food production loss globally and remain a local challenge especially in Asia and in the Mediterranean basin
Q4: How to satisfy both food security and water demand for ecosystems under global change ? (Ch 5)	Modelling framework developed to integrate hydrological, land use and economic optimization models (GLOBIOM-LPJmL)	Meeting future water demand for food and freshwater ecosystems is possible by: - Increasing land expansion by 100 Mha (mainly rainfed areas in Latin America and Russia) - Reducing irrigation water use by 40 to 60% - Reversing 60Mha of irrigated to rainfed area (especially in Asia) Increasing food trade by 15% compared to a business-as-usual scenario with food export mainly coming from Latin America to South and East of Asia

The discussion of the main results of the thesis is divided in five parts: EFR methods, modelling framework for integrating water in global assessments, scientific contribution, policy impact and future outlook. A last paragraph is devoted to the main conclusions of the thesis.

6.2. EFR modelling and data

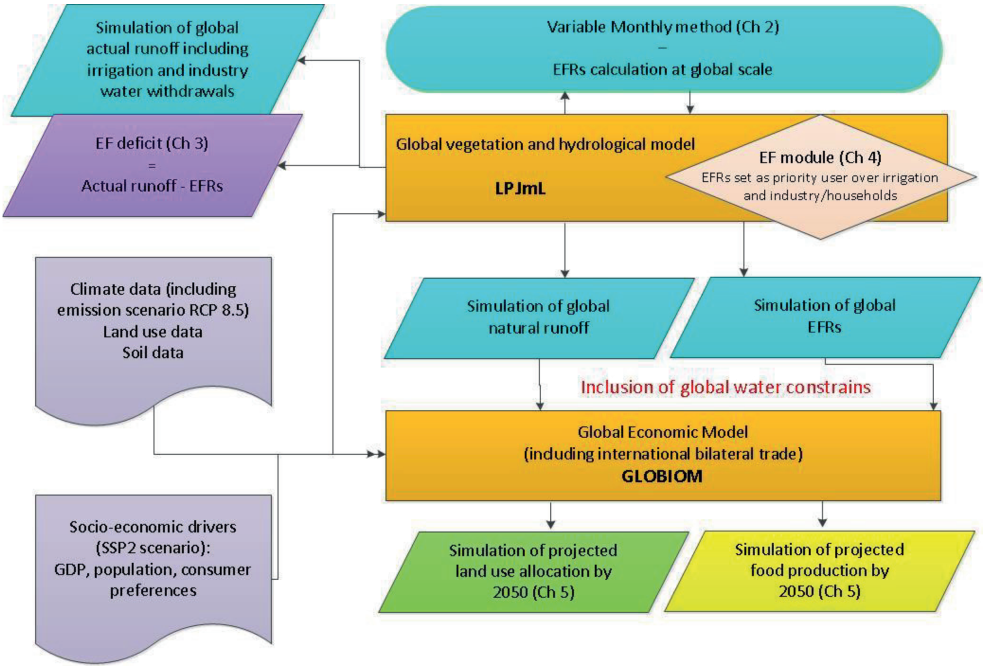

Figure 6.1. Schematic representation of model linkages with integration of spatial and temporal input databases

6.2.1. *VMF method in global context in its application*

At the beginning of the 21st century, global land-use assessments did not take water availability for ecosystems into account (Foley, DeFries et al. 2005, Molden 2007, de Fraiture and Wichelns 2010). The first global assessment taking into account EFRs was Smakhtin et al. (2004) who defined net water availability as the difference between total annual runoff and EFRs. Smakhtin et al. used quantiles and annual proxies to define EFRs. However, irrigation and ecosystem demands are highly seasonal and improving temporal resolution of global water resources was necessary. Hanasaki et al. (2008) and Hoekstra et al. (2011) improved temporal resolution of global EF methods by using monthly proxies. However, the rules defined by Hanasaki et al. (2008) only allocated between 10 and 30% of mean monthly runoff which is likely to be too low for sustaining EFRs, while Hoekstra et al. (2011) used 80% of monthly flow allocation to the environment which seems to be high and unrealistic (especially in rivers with high irrigation extractions). Therefore, in chapter 2, the VMF method was developed and tested with the advantages to be robust and easily applicable to any large scale hydrological or water resources model. Its validation with local study cases worldwide and its comparison with other EF methods makes it unique and shows that the method is reliable for a wide range of climates and river regimes. In Chapter 2, a gap was bridged between local and global scales of the eco-hydrology by using local study cases for serving a global purpose. Thanks to the new estimates of global freshwater boundaries with the VMF (Pastor, Ludwig et al. 2014), a couple of global studies are now able to accurately quantify water resources at global scale in terms of water footprint and availability (Gerten, Hoff et al. 2013, Boulay, Bare et al. 2015, Gaupp, Hall et al. 2015, Sadoff 2015).

6.2.2. *Improved representation of water deficit at global and regional scales*

Until recently, integrated global assessments often neglected water availability and EFRs (if included) were defined using annual proxies (Molden 2007, de Fraiture and Wichelns 2010). One of the applications of using accurate estimates of EFRs is to precisely quantify water resources in terms of availability and demand at global and regional scales. In chapter 2 and 3, the temporal component of EF methods by defining the seasonality (intra-annual variability) and the frequency of the EF deficit (inter-annual variability) were incremented and hot-spots of where, when and why EFRs were not satisfied are presented. In chapter 3, defining the origin of EF deficits (anthropogenic or natural) improved the identification of water scarcity with the aim to improve the type and level of intervention to restore and/or protect rivers. Furthermore, knowledge on the frequency of the deficit (in terms of inter-annual variability), on the type of river flow regime (perennial, intermittent) and on the level of flow alteration due to water extraction for humans was given. The frequency of deficits allows defining the level of importance for future interventions and the level of flow alteration gives information on how much water users will need to compromise between different users. While a free-flowing river encountering sporadic natural deficits might not require intervention because natural deficits might be beneficial for ecosystem functioning, intermittent rivers with high flow alterations are likely to need attention on river restoration and/or preservation. For example, an intermittent river with high flow alteration usually shows a high anthropogenic deficit such as the Colorado river and finding trade-offs between water users is here fundamental. Defining water stress is not only about defining where water is lacking for humans, but is also about identifying where, when, and why water is lacking for human and freshwater ecosystems. Defining the origin of deficit, timing, frequency and magnitude were shown as primordial to characterize water deficit (ch. 3).

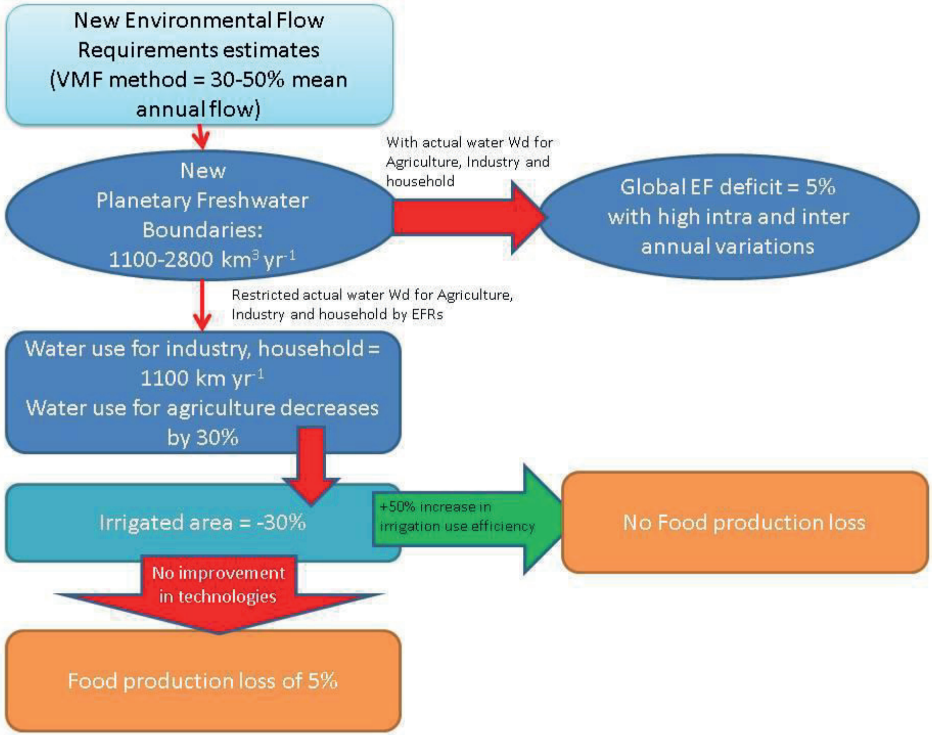

Figure 6.2. Synthesis of thesis results from chapter 2,3,4

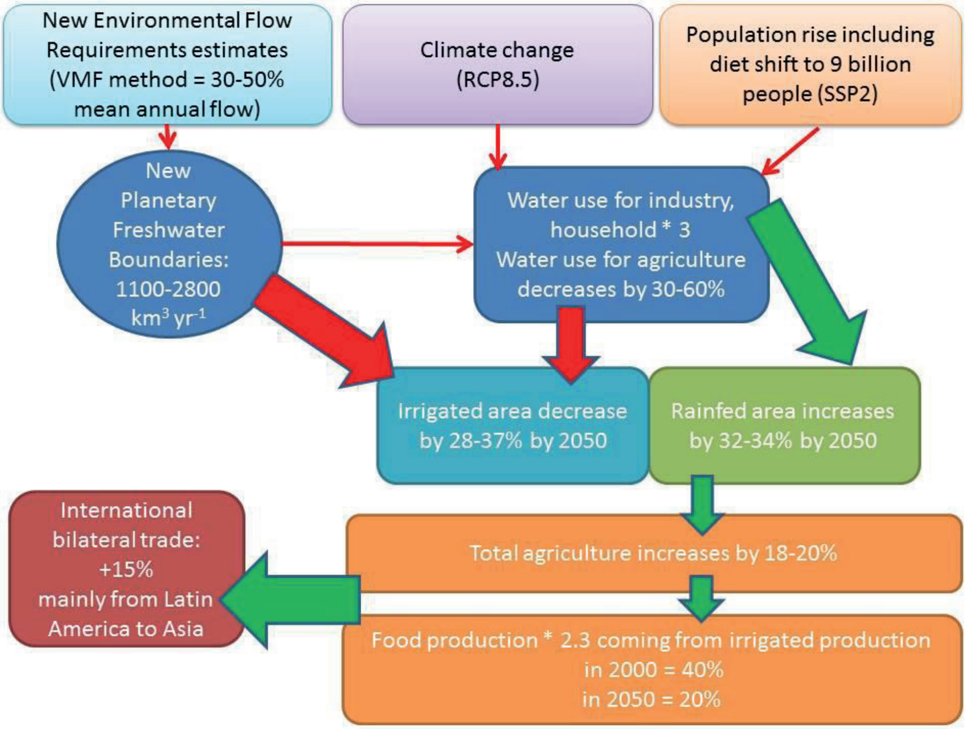

Figure 6.3. Impact of including EFRs restrictions into future global agriculture assessment (Ch 5). Green arrows represent an increase to the following variable and red arrows represent a decrease to the following variable.

6.2.3. *Limitations and uncertainties of modelling EFRs*

Although the VMF method was recently acknowledged and used in several global integrated assessments (Gaupp, Hall et al. 2015, Steffen, Richardson et al. 2015), there are several limitations in the use of this method. First, the VMF method was validated with only 11 field experiments and has the potential to be improved and calibrated with more study cases. However, finding additional study cases is not easy especially in arid and tropical areas where there is a lack of data. Most of the study cases using eco-hydrological relationships are found in temperate climates where EFRs rules were developed such as in US and in Europe (Pastor, Ludwig et al. 2014). Second, the VMF method was designed to sustain freshwater ecosystem globally in “fair” ecological conditions such as in Smakhtin et al. (2004). The reason to choose this level of desired ecological conditions was that half of the rivers are already fragmented and altered and it would be nearly impossible to restore rivers to pristine conditions. However, government and local water managers might decide to set higher EFRs thresholds to attain “good” ecological conditions such as in the Water Framework Directive. In this case, the VMF thresholds (percentage of monthly flow) might be increased and calibrated with local ecosystems requirements. For example, in the study of Shadkam, Ludwig et al. (2016), the VMF thresholds were increased by adding two standard deviations of the flow after hand calibration to meet EFRs defined for Urmia lake and to refill the lake.

6.2.4. *Eco-hydrological relationship impact on EF methods*

In a broader context, none of the existing EF methods including the VMF method is able to predict to what extent the implementation of EFRs will result in improved ecosystems functioning and increased biodiversity. There is a need for more knowledge and research on the relationship between flow thresholds and species survival and ecosystem recovery. One way to improve eco-hydrological relationship is to analyze ecosystem responses to flow changes. However, ecosystem responses are usually not linear (Poff and Zimmerman 2010) and freshwater ecosystems responses are very variable according to their climate, flow regime and level of flow alteration (ch 3). Therefore, guidelines on relevant hydrological parameters can be found in chapter 3 which could help future monitoring of freshwater species. Furthermore, local EF methods such as habitat simulation models (e.g. Phabsim) are often calibrated with the water requirements of only one species per river. However, to define accurate eco-hydrological relationships it is necessary to consider the entire river ecosystem (e.g. fish, riparian vegetation and amphibians) including different trophic levels.

6.2.5. *Inter-annual variability*

Finally, some limitations in the use of the VMF method was found in ch 3, 4 and 5 due to its fixed inter-annual variability. The VMF method is calculated with an average of 15 to 20 years of natural flow, therefore, when it is applied to extreme dry years, EFRs calculated with VMF might be higher than the actual available flow. On the other side, EFRs calculated with VMF might be underestimated in wet years which might be years with big floods which are required for channel formation and sediment flushing (ch 2). Therefore, evaluating inter-annual EF deficit with a fixed inter-annual EF method might be limiting. To adjust EFRs to inter-annual variability, the monitoring of water flow on a daily or monthly basis could be done to compensate any EF deficit with a buffer. The allocation of a higher flow in the next period could therefore be created. To adapt the VMF method to wet years, it might be necessary to include inter-annual floods in the method such as with the DRIFT method (Arthington, Rall et al. 2003). Adjusting EFRs to floods is usually required for maintaining sediments flushing and channel formation (Mathews and Richter 2007).

6.2.6. *Adaptation to ecoregions*

The VMF method could be adapted to ecoregions depending on their respective ecosystem resilience and species diversity. For example, tropical freshwater ecosystems are known to have high freshwater biodiversity and might be resilient to sporadic events however the increase of abrupt changes might hamper ecosystem functioning because tropical rivers encompass the highest diversity of endemic species on earth (Abell, Thieme et al. 2008). On the other side, arid rivers with high flow alteration might be more sensitive to EF deficit due to the exacerbation of their natural deficit by anthropogenic water extractions. In practical terms, EF methods can be supplemented with additional rules on reservoir flow release for species survival especially in dry years (Gaupp, Hall et al. 2015). However conflicts might occur if water for humans is lacking especially during dry periods.

6.3. Environmental flow requirements in the context of water, land-use and food nexus

6.3.1. *Global hydrological modeling - LPJmL model*

Due to the complexity of the study to integrate various disciplines such as hydrology, socio-economy, agronomy and climate, various models such as global vegetation and hydrological models and socio-economic models were combined. For that, I used LPJmL because it already integrates a crop model and a hydrological model including feedbacks such as the effect of CO_2 fertilization on vegetation and/or the calculation of return flows. The last research questions of this study required the development of an extra-module (EF model) within LPJmL (ch 4) and required the linkage of LPJmL with the economic model GLOBIOM (ch 5).

Temporal and spatial representation of EFRs and its respective integration in LPJmL

To answer Q1, Q2 and Q3, it was necessary to develop an improved EF method with refined spatial and temporal scales (ch 2, 3) which could be integrated in the river routing scheme of LPJmL. The complexity of this study was to decide and design an adequate EF method with high robustness and easiness of use. However, some temporal adjustments had to be tackled during the modeling exercise because the VMF method was designed at a monthly time-step and LPJmL has a river routine scheme with a daily time-step. Therefore, the VMF method was adapted to daily flows and a buffer was created so that if the daily flow does not meet the daily EFRs threshold, EFRs can be satisfied in the 5 following days (ch 4). For this study, LPJmL was selected because of its integrated and dynamic representation of carbon and water cycle including an irrigation scheme. To our knowledge, EFRs were never addressed dynamically and included in the river and irrigation modules of any GHM. However, LPJmL has some limitations in terms of global hydrology such as the representation of seasonality of the flow in terms of timing and magnitude (Biemans, Haddeland et al. 2011, Biemans, Speelman et al. 2013). This can partly explained by the assumption of constant flow velocity worldwide at 1m s^{-1}. Furthermore, LPJmL does not account dynamically for groundwater storage and recharge. Withdrawals from groundwater are calculated as the difference between the potential irrigation demand (IPOT run) and the actual irrigation withdrawals (IRES run – withdrawn from surface water from dams and reservoirs). Therefore, in ch 3 and 5, the spatial groundwater database from Siebert, Burke et al. (2010) was used. Groundwater withdrawals are here defined as the share of irrigation withdrawals per grid cell. Recent assessments could be helpful for the implementation of groundwater storage, recharge and release such as in the PCR model which includes groundwater dynamics (Wada, van Beek et al. 2010, Gleeson, Wada et al. 2012, Famiglietti 2014). One of the consequences of not representing groundwater dynamically is that simulation of baseflow might be underestimated and estimation of low-flow requirements might be below actual freshwater ecosystem requirements. This calculation can be exacerbated if simulation of flow seasonality (peak flow timing) is not accurate enough so that flow allocation for EFRs does not match freshwater ecosystem requirements. For example, if low-flows are calculated 1 or 2 months earlier than the actual timing, the calculated EFRs might not meet the real ecosystem demand. Despite these limitations, LPJmL was chosen because of its integrated representation of hydrology, irrigation scheme and crop development and it was the best model to answer our research questions.

6.3.2. *Integration of bilateral trade in integrated assessments and its limitations*

Land-use optimization models

The last decade, the use of land-use optimization models have been widely used for the allocation of the most suitable crops to the most suitable areas in terms of biophysical characteristics, demand/supply of the product and socio-economics dynamics (Agarwal, Green et al. 2002). The advantage of land-use models compared to biophysical models such as crop and hydrological models is that they optimize crop allocation and can shift between land and crop choice. Furthermore, these models are suitable to answer food security challenge via the use of international trade. These models are also used to simulate future climate change, socio-economic and bioenergy scenarios (Prins, Stehfest et al. 2010, Smith, Gregory et al. 2010, Lampe, Willenbockel et al. 2014). In this study, I used the partial equilibrium (PE) model Global Biosphere Management model (GLOBIOM) to simulate future land-use under climate change, socio-economic change and water restriction scenarios. GLOBIOM encompasses 200,000 simulations units with supply-side features. Production functions are modeled by means of biophysical processes. Land competition and expansion are simulated with functions of demand side. GLOBIOM selects a type of land use intensification (from extensive to intensive-irrigated) and maximizes the profit of each activity (forestry, bioenergy and agriculture). Crop yields are generated at the grid cell level based on the Erosion Productivity Impact Calculator (EPIC) model which uses soil type, slope, altitude and climate information (Williams, Jones et al. 1989, Liu, Williams et al. 2007). Crop yields vary with their respective management system (subsistence, extensive, intensive, and irrigated) and location. Crop water demand including irrigation demand is also calculated for each crop, each spatial unit and for each management via EPIC algorithms.

Spatial challenge

To resolve the last research question the economic model GLOBIOM was fed with new inputs of monthly water availability and EFRs to create additional constrains on future food production and cropland allocation. However, inconsistencies in the spatial resolution of both LPJmL and GLOBIOM models exist. For example, LPJmL has a spatial resolution of 0.5 deg. (67,420 cells) and spatial units of GLOBIOM are smaller than 0.5 deg. and are unevenly distributed (200,000 units). Therefore, a spatial unit of 2 by 2 deg. was used for several reasons: first, it would have been unrealistic to downscale water availability to smaller scales than 0.5 deg., second, the computational simulation time was too high for this purpose (considering the integration of monthly time-step in GLOBIOM) and third, GLOBIOM does not include a river routing with dams and reservoir storage which was partly solved by aggregating water availability at a spatial resolution of 2 deg. (approx. 200km by 200lm). As a consequence, on one hand, the new scale aggregation reduces computation time and problem complexity, on the other hand, assuming homogenous biophysical and economical constrains within a spatial unit might disregard local constrains. For example, some re-allocation of crops might not be feasible on certain slopes and soil type which is here neglected.

Temporal challenge

First, while crop water demand are calculated annually in the GLOBIOM model, irrigation water demand are calculated on a monthly basis in LPJmL. However, as shown in chapters 3 and 4, seasonal water demand for irrigation withdrawals highly conflict with ecosystem requirements. Therefore, a new algorithm was developed to compare monthly water availability with monthly irrigation demand in GLOBIOM in ch 5. The assumption was that irrigated crop yield was lowered to its respective rainfed crop yield when water was not available for irrigation. On one hand, this assumption is valid only if soil water capacity is high enough to support future crop development. On the other hand, lack of irrigated water does not necessarily lead to a decreased yield especially if the chosen crop is drought-resistant such as sorghum. Finally, lack of water can also lead to crop failure which was not considered in this study. However results show that if water restrictions are high for a couple of decades (scenario with high EFRs restrictions in ch 5), the recursive dynamics of the model GLOBIOM re-allocate low water intensive crops in water scarce areas and increase food import from water-abundant regions to water-scarce areas. This last trends were also acknowledged in a couple of other studies (de Fraiture and Wichelns 2010, Biewald, Rolinski et al. 2014).

Second, GLOBIOM is an optimization model starting from a baseline year of 2000 with a time-step of ten years. While the impact of climate variability and extreme years on future crop production is acknowledged (Iizumi and Ramankutty 2016), extreme events are likely to increase with climate change and might negatively impact food security (Ray, Mueller et al. 2013, Iizumi and Ramankutty 2016). Therefore, including monthly intra and inter-annual variability was complex. For that, averaging monthly water demand and availability around the required years of GLOBIOM (2000, 2010, 2020, 2030, 2040, 2050) was necessary. By using averages of climate and hydrological inputs, the impact of climate variability on crop production and its respective re-allocation (e.g. drought of 2003) is here underestimated. For example, droughts can create market shocks due to lower food production and can lead to food insecurity.

Crop representation in LPJmL and GLOBIOM

While GLOBIOM uses modules from the EPIC model for simulating crop development and water demand including 18 crops (representing 70% of total crop harvested area), LPJmL simulates agriculture production using 12 Crop Functional Types (CFTs) also representing between 70 and 80% of total crop harvested area. Each CFT encompasses a group of crops such as C3 cereals for wheat, sorghum etc. and C4 crops such as maize. However CFTs do not directly match GLOBIOM-EPIC crops. In principle, using GLOBIOM crop production and its respective water demand would be the most practical; however, these data were only available at an annual scale. Therefore, crop water demand from GLOBIOM was calibrated with the crop calendar of LPJmL over each river basin. The calibration at such scale can lead to the underestimation and overestimation of specific crop water demand and the timing of the irrigation might not be adequate in some locations. For example, by simulating earlier irrigation demand than required, the selection of less water intensive crops might be chosen even if in reality water was available. Reversely, if irrigation timing is delayed, water availability might be overestimated and a crop that is not adequate to a specific crop calendar might be selected.

Socio-economic limitations

In chapter 5, the role of trade in food security and in safeguarding water for nature was highlighted. Similar to this study, MacDonald, Brauman et al. (2015) showed that 1/5th of global land and water use for irrigation is used for food exports and this share is likely to increase with increased water restrictions. Increased food trade can be used as a water saving solution in countries with high irrigation demand relying on high food import. However, production will need to be increased in water-abundant countries at the expense of other ecosystems (e.g. forest). In MacDonald et al. study, trade was shown to represent 26% of gross agriculture production and 20% of gross produced calories (of which 50% are from wheat); 20% of total area is also devoted to export with a high share going from Americas to East Asia. Therefore, when considering land and water use, environmental burdens are shifted to exporting countries (e.g. deforestation in South America) especially if the resources are produced from one single region.

Finally, changes in future land use simulated by GLOBIOM and by any optimization model might not always be realistic. For example, in countries where trade represents a high share of food consumption, local government might be reluctant to increase the share of food import to maintain food-secure people. For example, in the case of a sudden economic crisis, food price increase might prevent countries relying on food imports to buy additional food. Therefore, some governments are likely to adopt self-sufficient food regulations over environmental protection. For example, in Sri Lanka, producing food for self-sufficiency would imply 69% increase in water use and 29% increase in fertilizer use (Davis, Gephart et al. 2016). Finally, increase in trade might increase reallocation of food production which is not always possible because adopting new cropping systems is not simple in terms of market and available technology. Farmers and people have their own economic and social values implying specific diets and in some cases, food import might be preferred over crop shifts or diet shifts. For example, some studies in Africa show that diet composition has increased the share of imported western cereals such as wheat, maize and rice over local production of traditional drought-resistant species such as sorghum and millet (Kennedy and Reardon 1994). Furthermore, changing cropping systems would require new investment in machinery for seedling and harvesting, which might be an important limiting factor.

6.4. Scientific contribution

The scientific contribution of this study encompasses two parts. First, an effort was made on improving methods to estimate EFRs at global scale and second, the newly developed VMF method has been integrated in global land use and water assessments for food impacts studies (ch. 4 and 5) and/or to evaluate accurate water footprints and water scarcity evaluations worldwide (Boulay, Bare et al. 2015, Gaupp, Hall et al. 2015).

6.4.1. *Improved estimates of large scale environmental flow requirements*

This study shows the improvement in the representation of EF algorithms at global scale in term of refined temporal scale, validity with local study cases and easiness of use. Contributions to the global hydrology and eco-hydrology fields were made with the development of a new robust method to calculate EFRs with refined spatial and temporal scales (the VMF method, ch 2). The VMF method was compared with two existing methods using annual proxies of available flow (Tennant and Smakhtin) and with one method using monthly proxies (Tessman). In chapter 2, the Q90_Q50 method was designed by using annual quantiles but in this chapter, it was demonstrated that the use of annual quantile methods (e.g. Smakhtin and Q90_Q50) are overestimating EFRs of perennial rivers and underestimating EFRs of intermittent rivers. The only global method that used monthly proxies based on study cases was found in Hoekstra, Mekonnen et al. (2012), however, their study was based on the presumptive method developed by Richter, Davis et al. (2012) which only used four temperate study cases. Furthermore, Richter et al. only allows 20% of monthly flow to be extracted by humans which seems unrealistic in a world containing more than 50% of fragmented rivers and more than 6000 dams (Lehner, Liermann et al. 2011). The EF method that was the closest to the VMF method was the Tessman method (1980) because it allocates different shares of flow according to the seasonality however the Tessman method defines seasonality with different algorithms and does not allow irrigation withdrawals during low-flow season when irrigation demand is the highest. Finally, the VMF method showed the best performance with local study cases.

An additional contribution was made in the eco-hydrological field by bridging spatial scales from local study cases to global application of EFRs. Local study cases were shown to be very useful for the validation of global simplified EF methods. Finally, a global dataset calculating EFRs in "fair ecological conditions" at a monthly time step was made available for use. The dataset provides ratio of monthly flow and absolute values of EFRs at a spatial resolution of 0.5 deg and is already used in global food and water assessments.

6.4.2. *Including Environmental Flow Requirements in integrated global assessments*

In times of increasing water demand for humans and food, defining water availability for freshwater ecosystems with accurate timing and refined spatial scales was necessary to anticipate future water stress for humans and ecosystems. Therefore, the VMF method was developed with the aim to better define planetary boundaries for freshwater ecosystems at refined spatial and temporal scales. By improving the temporal dimension of EF methods, we are now able to compare water extractions for irrigation and freshwater ecosystems on a seasonal base. One of the major applications of the VMF method was, thus, to calculate EF deficits and/or EFR transgressions at global scale. While Steffen et al. (2015), Gerten et al. (2013) and Sadoff (2015) studies used the VMF method to show water stress and EFRs transgressions due to irrigation extractions on a monthly basis, (Gaupp, Hall et al. 2015) used the VMF to evaluate how storage could help in meeting water demand for humans without jeopardizing environmental flow requirements. Until now, global water studies were rarely including EFRs including intra and inter-annual variability. In chapter 3, the use of the VMF method contributed to the definition of the origin, magnitude, timing, duration and frequency of the deficit at global scale. The new findings highlighted the origin of the deficit (anthropogenic or natural) and the inter-annual deficit at specific hot-spots of the globe. These findings give new guidelines to water managers on which level and type of intervention to adopt to preserve river ecosystems. Finally, the development of the VMF method has not only enriched global water assessments but also local studies such as the study from Shadkam, Ludwig et al. (2016) where an adaptation of the VMF method was implemented to refill the Urmia lake. Finally, in case of nonexistence of local ecological data at watershed level, the VMF method can also be implemented until local hydrological and ecological data are collected.

In chapter 4, a contribution to the global hydrological field was made by developing a new EF module in the global vegetation and hydrological model LPJmL. In this study, the aim was to set EFRs as the first water user before irrigation demand so that the implied food calorie loss due to EFRs could be assessed. EFRs were included in the river routine of LPJmL (technical details in Ch 4). The implementation of EFRs at such refined scale allows the routing of EFRs along the river network to takes into account upstream/downstream components. Hence, more water is available downstream for both ecosystems and irrigation users. On the other side, this implies that less water is available upstream for irrigation purposes and storage. In practice, this modeling exercise still requires the consideration of socio-economic factors for implementation. Finally, a scenario with 50% increase in irrigation use efficiency was run (considering improved water harvesting techniques) and showed that globally, both actual food production and EFRs could be sustained. However, in the study, future changes in climate conditions and population growth were not included.

One of the main scientific contributions of this thesis was the development of a framework including EFRs restrictions in an economic optimization land-use model (Ch 5) including global change. This framework allows for the assessment of how saving water for the environment would affect future land use and food production patterns. This framework allows for the answering of new questions in terms of impact of water use by the environment on land use and food production. Despite multi-datasets coupling and technical challenges, this study reveals a strong signal in the use of trade increase to compensate first for the increase in water demand for future food production, second for the increase in water use restrictions (set for EFRs) and finally, for the increase in water cycle intensification due to climate change. It is the first global study that included EFRs at such spatial and temporal scale with scenarios of climate change (RCP8.5) and socio-economics changes (SSP2) which shows that to sustain food security and EFRs under global 15% increase in traded food commodities would be required compared to a business-as usual scenario. The biggest bilateral food trade increase is likely to occur from Latin America to Asia and setting future EFRs restrictions would also imply a drastic reduction of irrigated area in China and India.

6.5. Contribution to water management and policy makers

Thanks to the improved representation of spatial and temporal resolutions of the VMF method and its implementation in global integrated assessments for food, water and land use (Sadoff 2015, Steffen, Richardson et al. 2015), policymakers can design improved solutions concerning future food and water security for human and ecosystems. This thesis highlights three main contributions to water management and policymakers. First, this study highlights large scale EF deficits due to human water extractions, second that future SDGs on food and water security are conflicting and finally, that more trade is needed to protect freshwater ecosystems while guaranteeing food security.

The first finding is that human water extractions are causing large scale EF deficits (Ch 3; (Gerten, Hoff et al. 2013)). In Chapter 3, EF deficit is highlighted with refined spatial and temporal scales including the definition of its origin, timing and frequency for major river basins. This new scientific knowledge can help international policymakers in deciding where river restoration projects should occur and can evaluate the level of priority of action. Policymakers and water managers can adapt the solutions according to the origin of the deficit, its level of flow alteration and to the type of flow regime.

The second finding addresses the conflicting SDGs such as SDG 2 and 6 (achieving food security vs. safeguarding water access to all (including humans and ecosystems) (Griggs, Stafford-Smith et al.

2013, UN 2016). In chapter 4, constrained interactions between sustainable withdrawals (6.4) and agricultural productivity (2.3) were assessed, and in particular, the reinforcing interaction of water productivity increases (6.4) with food production (2.3) were assessed and in turn also with sustainable withdrawals. Sustainable withdrawals (6.4) are indivisibly linked with the target to protect and restore water-related ecosystems (6.6) and also indivisibly linked with sustainable food production systems (2.4). Chapter 4 concludes on the requirement to improve agricultural water productivity with strong reinforcing links to various other targets - a ‘nexus target’ (Figure 4.5). Hotspots of food loss per capita might be prioritized in international policy agenda such as India and China which rely on a relatively high share of irrigated agriculture production.

Third, one of the main findings of the thesis is addressed in chapter 5 where the use of international bilateral trade was found to be a requirement to maintain food and water security for humans and ecosystems. On one hand, it is a positive outcome to show that food security can be sustained by 2050. On the other hand, this study shows that many regional adaptations will be required to reach these targets. For example, a considerable increase in crop yield will be required and water intensive crop production might need to be reallocated to water-abundant regions. Therefore, there is a big technological and social challenge to be overcome in the coming decades and these last, should come with adequate international regulations. At global scale, results show that improved implementation of EFR policies will be required and that the VMF method can be used as a first order estimate of water needs for the environment and could facilitate implementing EFRs in new policy frameworks. At regional level, the use of economic model to optimize water resources for irrigation and EFRs might be useful (Blanco-Gutiérrez, Varela-Ortega et al. 2013).

6.6. Outlook (and future research)

On the research agenda, despite recent improvements in collecting global freshwater databases, harmonization of hydrological and ecological data at global scale would be necessary for future EFRs implementation (Abell, Thieme et al. 2008, Oberdorff, Tedesco et al. 2011, Tisseuil, Cornu et al. 2013). For that, additional field data collection of hydrology and freshwater ecology of rivers is required and the inclusion of water quality in future EFRs calculations is essential (Vliet, Ludwig et al. 2013). These data collections could contribute to increase our understanding of eco-hydrological relationships and could improve algorithms of EFR calculations.

Then, improving inter-annual representation of EFRs would be necessary to evaluation future impact of extreme events on food production and ecosystems. For example, specific rules for allocating EFRs during dry years might be required so that humans and freshwater ecosystems could be sustained (Qureshi, Connor et al. 2007).

Finally, it would be interesting to evaluate the role of dams and reservoirs in maintaining EFRs especially with future intensification of the water cycle with climate change such as in Gaupp, Hall et al. (2015). Additionally, including the level of river regulation and fragmentation in EFRs algorithms would be an asset for sustaining certain river branches and their respective ecosystems (Grill, Lehner et al. 2015).

6.7. Conclusion

The main question of this study was "How to sustain water for food and ecosystems under global change?". To address this question, an improved spatial and temporal environmental flow method was developed. This new VMF method is simple and robust enough so it can be implemented in any global hydrological or integrated model and it was also validated with data from local study cases. The use of this new method allowed the comparison of seasonal water demand for irrigation and water demand for freshwater ecosystems. Results showed that when irrigation withdrawals are prioritized over EFRs, not enough water is available to sustain ecosystems especially in the Mediterranean regions, in Asia and the West coast of the US. Results indicated that EF deficits are not only part of a regional problem but have global implications. For example, highly modified rivers such as the Indus has a EF deficit of 130% of available flow while free-flowing rivers such as Congo show 2% deficit mainly caused by natural climate variability. If EFRs would be prioritized over water extractions for irrigation this would result in a 30% reduction of irrigated food production and 5% of total global food production leading to more food insecurity. The final step focused on assessing the impact of implementing EFR regulations on future land use allocation and food trade. The results showed that limiting water use in agriculture to sustain EFRs would result in higher food production in water-abundant regions such as Brazil to compensate for food production loss from irrigated areas in East and South Asia. Finally, this thesis shows the need to integrate multi-sectoral water users in global assessments to achieve sustainable goals such as safeguarding full access to food and water for humans and ecosystems.

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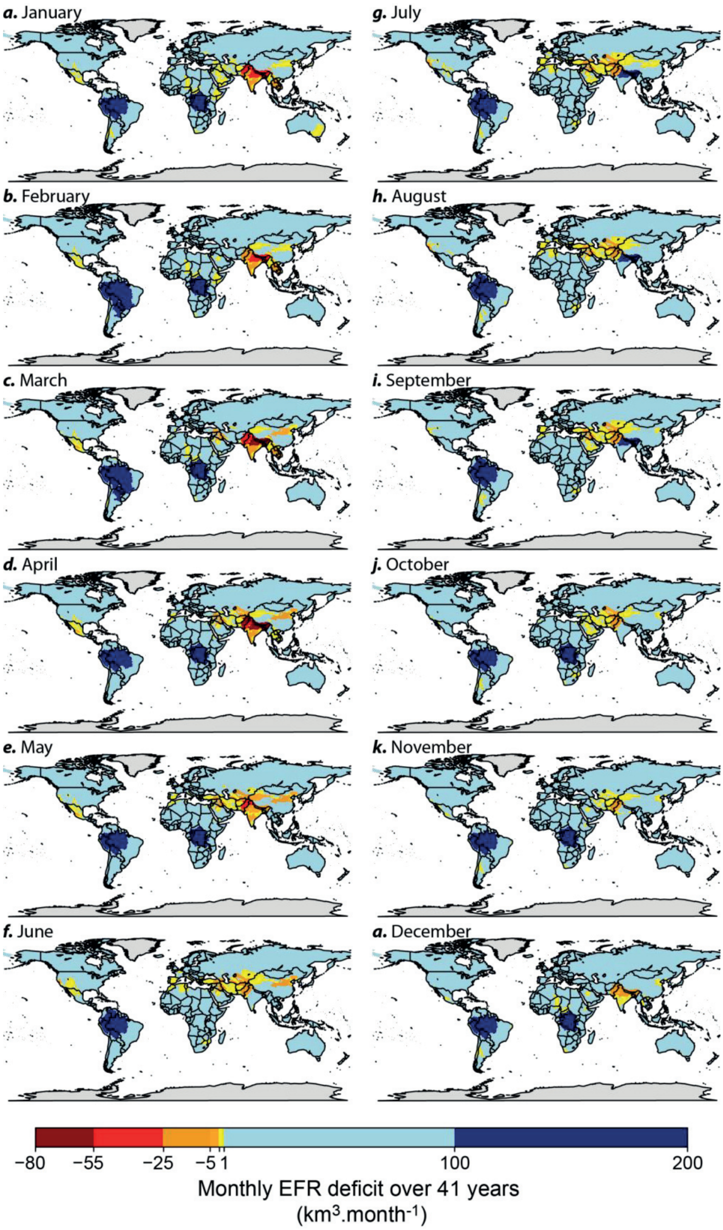
Annex A

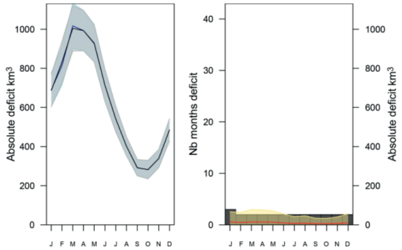
Additional information for chapter 3

Supplementary Figure A1. Spatial representation of monthly deficit at global scale for 12 months. Units are in km^3 month^{-1}.



Supplementary Figure A2. Temporal representation of natural and actual flow (left plot) with intra-annual monthly deficit (km^3month-1) and frequency of deficit (number of months) for four river basins representing each group of river basins. Barplots represent the frequency of natural deficit (black) and anthropogenic deficit (grey).

a. Group 1



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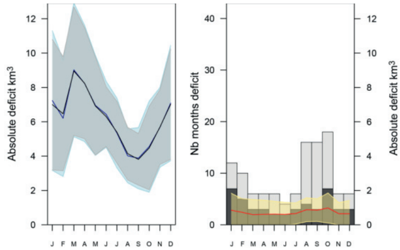

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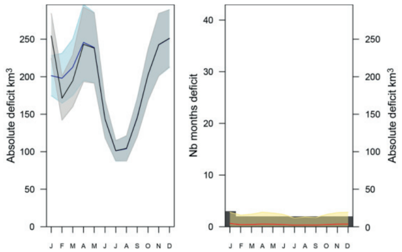
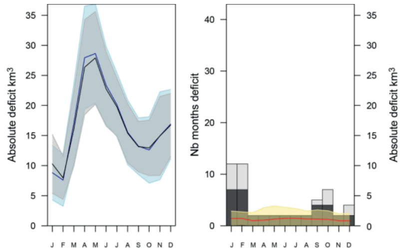
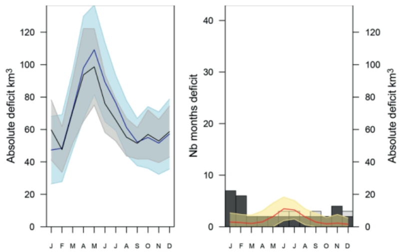


Congo

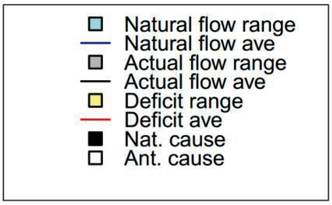


Danube

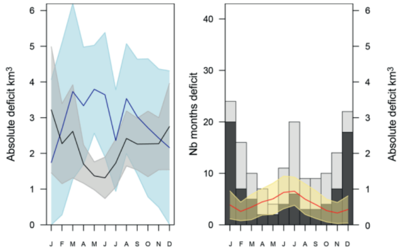


Mississippi

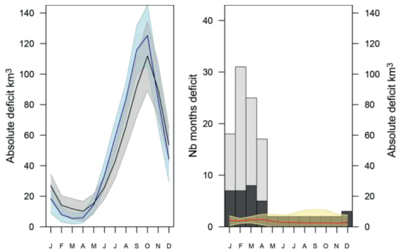
Legend



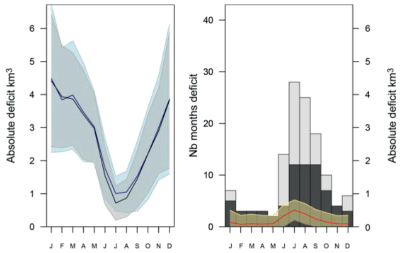
b. Group 2



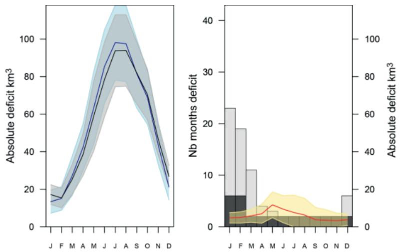
Colorado



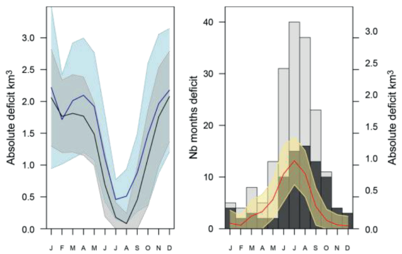
Nile



Garonne



Yangtze



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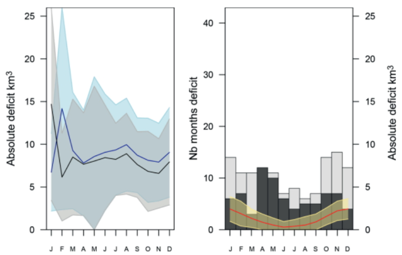
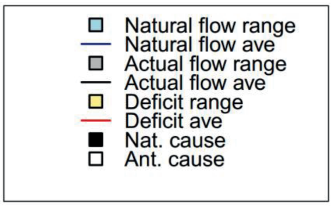
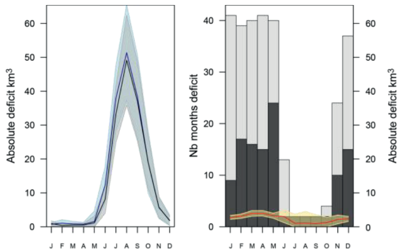


Murray

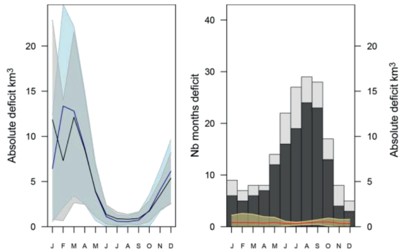
Legend



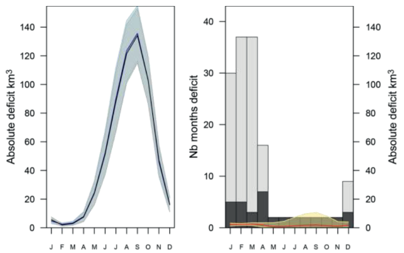
c. Group 3



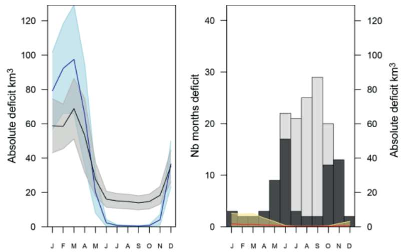
Godavari



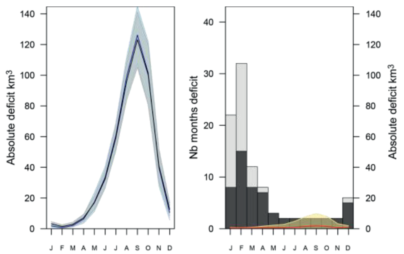
Orange



Mekong

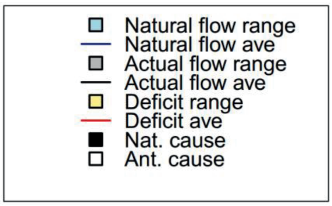


Zambezi

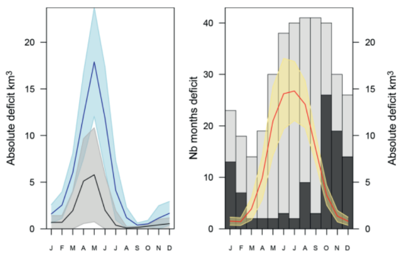


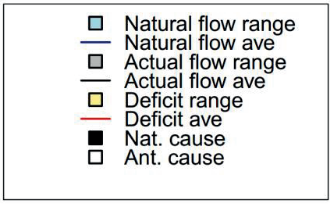
Niger

Legend



d. Group 4





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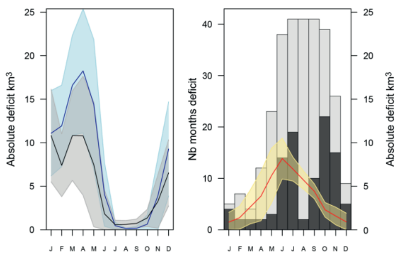
Amu darya



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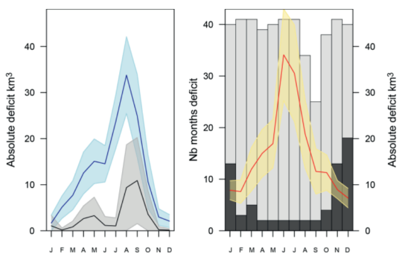

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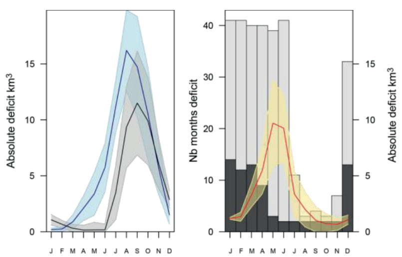


Yellow river

Annex B

Additional information for chapter 4

Methods

Environmental flow requirement objectives.

We estimate EFRs, irrigation demand and withdrawals, and crop calorie production with a biosphere model that simulates these processes daily, as an intrinsic part of natural and managed ecosystem dynamics. We use the concept of EFRs to allocate maximum allowed monthly water withdrawals, expressed as a percentage of 'pristine' undisturbed mean monthly river flow (determined globally for each 0.5◦ grid cell from a simulation without considering human land use, water infrastructure, and water withdrawals; forced with climate data of the simulation period 1980-2009; see below). We include three hydrological EFR estimation methods to depict an uncertainty range, which reflects methodological differences and which can be interpreted as the outcome of different environmental policies. Based on a simulation considering current agricultural patterns, reservoir management, and multi-sectoral human water withdrawals (see Model and simulation protocol), this uncertainty range is also used to classify river segments according to the current status of transgression of EFRs, i.e. the sub-global freshwater use boundary1 (Figure 1). The EFR calculation methods aim at reaching a 'fair' ecological status, which is a conservative assumption as this status still can be characterised by disturbed biota, loss or reduction in spatial distribution of sensitive species, and occurrence of alien species2. The Tessmann method3 and the Variable Monthly Flow method4 account for seasonal EFR variation by distinguishing high-, intermediate-, and low-flow regimes based on different proportions of mean monthly river flow (MF) and mean annual flow (AF) of long-term average 'pristine' conditions (Table S1). To protect habitat maintenance and essential flow variability, supporting the 'natural flow paradigm'5, for each month different flow volumes are allocated to EFRs with respect to the flow regime (Table S1). The Smakhtin et al.2 method assumes static EFRs throughout the year, but allocates two components therefor, a minimum base flow (exceeded 90% of the time, ***Q*90**) and a percentage of AF depending on mean seasonal river flow variability. For rivers with stable seasonal flow and thus high ***Q*90** values relative to AF (***Q*90** > 30% AF), only the baseflow is allocated. In case of higher flow variability (***Q*90** can go down to zero for intermittent rivers), fractions of AF are allocated 2

additionally (Table S1). The Smakhtin et al. method provides by definition a seasonally constant EFR target, in case pristine discharge is lower than the *EFRSmak* target, we set it to the value of pristine flow. Overall, such conceptually simple "per-cent-of-flow" approaches are commonly used proxies for EFR estimates, as applicable at large scales, and they can provide a high degree of protection for natural flow variability5.

Illustration of pressure on the freshwater boundary.

The status of the freshwater boundary displayed in Figure 1 is based on the proportion of EFR transgressions (*EFRtran*) and the EFR uncertainty (range of EFR estimates from three methods as defined above), calculated for each month and grid cell (*EFRdef*). *EFRdef* is shown as the average over months in which both, pristine river discharge and current *EFRtran* are $\geq 0{:}1$ $m\mathbf{3}{=}s$, respectively, throughout the simulation period 1980-2009. $EFRtran = max(EFR - discharge; 0)$ is calculated as the mean of the three EFR methods. The map in Figure 2a illustrates the ratio of mean annual *EFRtran* to mean annual discharge. See Figure S1a for the sum of annual *EFRtran* in million m**3**, and Figure S1b for the average number of months in which at least one of three methods indicates *EFRtran* $\geq 0{:}1$ $m\mathbf{3}{=}s$.

Model and simulation protocol.

The LPJmL model globally represents biogeochemical land surface processes, simulating daily water fluxes in direct coupling with the establishment, growth, and productivity of major natural and agricultural plant types at 0.5◦ resolution. Crop production is represented by 12 specified crop functional types, irrigated or rainfed. Spatially explicit data on cropland extent and the mechanistic representation of irrigation systems is described in6. Carbon assimilated through photosynthesis is allocated to harvestable storage organs (e.g. cereal grain) and three other pools (roots, leafs, stems).

Table B1. Definition of hydrological seasons and respective EFR allocations. Mean monthly flow (MF), mean annual flow (AF) refer to pristine river flows, *Q***90** defines the baseflow that is on average exceeded 90% of the time (simulated under 1980-2009 climate but in the absence of human water flow and landuse alterations). The Tessmann and the variable monthly flow method (VMF) account for seasonal EFR variation, while the Smakhtin et al. method allocates a seasonally static EFR estimation.

EFR method	**Flow regime classification**			**Environmental flow requirements**		
	low-flow	high-flow	low-flow	intermediate-flow	high-flow	
	Tessmann3	MF ≤ 40% AF	MF > AF	100% MF	40% AF	40%MF
	VMF4	MF ≤ 40% AF	MF > 80%AF 60%	MF	45% MF	30% MF
	Smakhtin et al.			$Q90$; if $Q90 > 30\%AF$ $Q90 + 7\%AF$; if $Q90 \leq 30\%AF$ $Q90 + 15\%AF$; if $Q90 \leq 20\%AF$ $Q90 + 20\%AF$; otherwise		

Sowing dates are calculated based on climate and crop type, but fixed during the simulation period after 1980. In tropical regions that exhibit predominant precipitation seasonality, sowing dates on irrigated land are forced to occur in the dry season. Land use patterns are held constant at year 2005. For all simulations, LPJmL is forced with the climate input data and spin-up protocol as described in7 for the time period 1980-2009. A simulation omitting irrigation is performed based on the same land use patterns, but under rainfed conditions only. Otherwise water withdrawals for irrigation and for household, industry and livestock are simulated to be constrained by local availability of renewable freshwater, including a representation of dams and reservoirs (with EFRs release regime, yet channel and habitat maintenance floods not considered). Surface and subsurface runoff are accumulated along the river network and subsequently available for downstream reuse. In this study there is no implicit assumption about contributions from fossil groundwater and water diversions, which are expected to amount to ~20% of global irrigation water requirements8. Based on these well-validated streamflow estimates (Figure 4), EFRs are calculated as described above. In the ”respect EFR” simulation, total water withdrawal is temporally restricted as long as it would tap EFRs. For each above-defined EFR method we perform an individual model run, but results presented throughout the text refer to the mean of the three simulations and the standard deviation is associated in Table 1 (individual results are shown in Table S2). Additionally we simulate a scenario of moderate irrigation system upgrade in which surface irrigation systems are assumed to be replaced by sprinkler systems (except paddy rice) and half of saved consumptive ’losses’ are available to expand irrigation into neighbouring rainfed cropland (total cropland area remains constant)7. Since observed efficiency improvements do not necessarily result in lower water withdrawals (farmers often expand irrigation or use higher value crops, instead of losing water allocations)9, we allocate half of saved consumptive water to irrigation expansion (if rainfed cropland is available in the same grid cell). Note that return flows are not considered savable losses throughout this study as they might be accessible for downstream users.

Table B2. Global sums of food production and water abstractions for different management scenarios. Global sums of kcal production, area affected (kcal loss > 10%), irrigation water withdrawal (IWD) and consumption (IWC), and withdrawal and consumption for household, industry, and livestock (HIL WD and HIL WC) are shown for the following scenarios: current situation (1.), in the absence of irrigation (2.), with irrigation constrained by EFRs (3., for 3 methods respectively), and with upgraded irrigation constrained by EFRs (4., for 3 methods respectively), details in Methods. The simulation period is 1980-2009.

Total production	Total production	Irrigated production	Total area affected	Irrig. area affected	IWD	IWC	HIL WD	HI WC
	[1013*kcal*]	[1013*kcal*]	[*Mha*]	[*Mha*]	[*km*3]	[*km*3]	[*km*3]	[*km*3]
1. Today	740.0	244.2	0.0	0.0	2409.3	1254.7	1070.5	192.8
2. No irrigation	631.6	135.8	262.1	157.2	0.0	0.0	1090.8	196.5
3. Respect EFR								
Tessmann	698.1	202.3	152.7	110.3	1258.8	730.7	779.7	138.4
VMF	712.6	216.7	116.2	93.3	1570.1	891.9	909.4	163.3
Smakhtin et al.	707.6	211.7	131.6	93.8	1555.3	868.2	843.9	146.2
4. Respect EFR with irrigation upgrade								
Tessmann	730.5	249.4	124.9	85.0	987.5	741.2	774.8	137.2
VMF	747.7	266.6	80.9	53.0	1222.0	896.5	903.9	161.9
Smakhtin et al.	742.6	261.5	102.8	64.7	1201.8	871.1	840.1	145.2

Model validation.

The validation of LPJmL-simulated key variables for the time period 1980 to 2009 is highlighted in Figure 4. Uncalibrated LPJmL discharge simulations are compared with the latest mean annual discharge observations from GRDC (Global Runoff Data Centre) stations10. EFR simulations are validated against hydro-ecological data from 12 ecologically, hydrologically, and climatically different local study cases (details in4), suggesting they capture a sufficiently broad range of environmental settings if applied to the global scale. Country-level crop yield observations are obtained from FAO Stat[11]. Simulated global kcal production of $7{:}8 \ast 10^{15}$ kcal in 5 year 2006 is ~18% short of reported values ($9{:}5 \ast 10^{15}$ kcal[12]), because LPJmL cannot account for multi-cropping systems.

Code availability.

The LPJmL code that supports the findings of the study are available from the corresponding author upon request.

Table B3. Number of people affected by EFR regulation. Listed is the number of people inhabiting food production units (FPU) with a decrease in total kcal production between 5% and 25% if EFRs were respected (mean of 3 methods).

Kcal decline at FPU level	People affected [million]	Fraction of global population [%]
5%	1,700	29.8
10%	1,070	18.7
15%	810	14.2
20%	630	11.1
25%	300	5.3

Annex C

Additional information for chapter 5

Appendix Table C1. Micro and mega regions of GLOBIOM

Macro region	Micro region	Country
Former soviet union (USSR) - (FSU)	Former_USSR	Armenia, Azerbaijan, Belarus, Georgia, Kazakhstan, Kyrgyzstan, MoldovaRep, RussianFed, Tajikistan, Turkmenistan, Ukraine, Uzbekistan
East of Asia (EAS)- Planned Asia and China (PAC) - (CHI)	ChinaReg	China
	JapanReg	Japan
	SouthKorea	KoreaRep
Europe (EUR)	EU_Baltic	Estonia, Latvia, Lithuania
	EU_CentralEast	Bulgaria, CzechRep, Hungary, Poland, Romania, Slovakia, Slovenia
	EU_MidWest	Austria, Belgium, France, Germany, Luxembourg, Netherlands
	EU_North	Denmark, Finland, Ireland, Sweden, UK
	EU_South	Cyprus, Greece, Italy, Malta, Portugal, Spain
	RCEU	Albania, Bosnia, Croatia, Macedonia, Serbia-Montenegro
	ROWE	Norway, Switzerland
Latin America and the Caribbean (LAM) - (LAC)	BrazilReg	Brazil
	MexicoReg	Mexico
	RCAM	Belize, Costa Rica, Cuba, Dominican Republic, El Salvador, Guadeloupe, Guatemala, Haiti, Honduras, Jamaica, Nicaragua, Panama, Trinidad and Tobago
	RSAM	Argentina, Bolivia, Chile, Colombia, Ecuador, Guyana, Paraguay, Peru, Suriname, Uruguay, Venezuela
Middle-East and North-Africa (MNA) - (MENA)	MidEastNorthAfr	Algeria, Bahrain, Egypt, Iran, Iraq, Israel, Jordan, Kuwait, Lebanon, Libya, Morocco, Oman, Qatar, Saudi Arabia, Syria, Tunisia, United Arab Emirates, Yemen
	TurkeyReg	Turkey
North America (NAM) - (NA)	CanadaReg	Canada
	USAReg	USA
Pacific OECD (OCE) - (OECD) Oceania	ANZ	Australia, New Zealand
	Pacific_Islands	Papua New Guin, Vanuatu
South Asia (SA) - (IND)	IndiaReg	India

	RSAS	Bangladesh, Bhutan, Nepal, Pakistan, SriLanka
South-East Asia (SEA) - Other Pacific Asia (OPA)	RSEA_ OPA	BruneiDarsm, Indonesia, Malaysia, Myanmar, Philippines, Thailand, TimorLeste
	RSEA_ PAC	Cambodia, KoreaDPRp, Laos, Mongolia, VietNam
Sub-Saharan Africa (SSA)	Congo Basin	Cameroon, CentAfrRep, CongoDemR, CongoRep, EqGuinea, Gabon
	Eastern Af	Burundi, Ethiopia, Kenya, Rwanda, Tanzania, Uganda
	SouthA frReg	SouthAfrica, SouthernAf, Angola, Botswana, Lesotho, Madagascar, Malawi, Mozambique, Namibia, Swaziland, Zambia, Zimbabwe
	Wester nAf	Benin, BurkinaFaso, Chad, CotedIvoire, Eritrea, Gambia, Ghana, Guinea, GuineaBissau, Liberia, Mali, Mauritania, Niger, Nigeria, Senegal, SierraLeone, Sudan, Togo

Appendix Table C2. Table of input data and scenarios

Variables	Scenarios
Climate input	MPI_ESM_LR (GCM1)
	hadGCM2_A0 (GCM2)
Climate change	No climate change (noCC)
	Climate change (CC)
Water management	INVEST (INV): Business as usual with water restriction at regional level,
	EXPLOIT (EXP): Water restriction from water availability at LUID level
	Environmental Flow Requirements (ENV): Water restriction from water availability at LUID level including EFRs
	Environmental Flow Requirements high (ENV+): Water restriction from water availability at LUID level including 1.5 * EFRs
International trading	Unconstrained trade (UncT): Unconstrained optimized international trading to meet food requirement
	Constrained trade (CstT): Bilateral international trading fixed to the exploit scenario

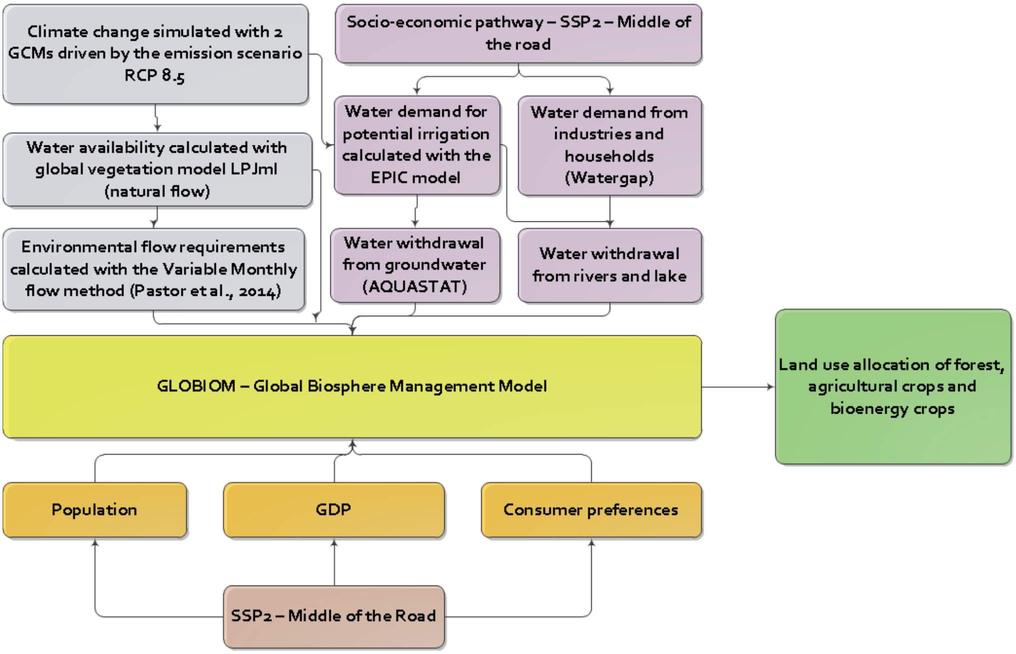
Appendix Table C3. Total agriculture area in million hectares (Mha) for 10 mega regions with percentage of irrigated area (2nd line) with 4 water management scenarios (INV, EXP, ENV, ENV+) and climate change scenarios (noCC and CC).

Climate change		noCC	noCC	noCC	noCC	CC	CC	CC	CC
Water management	-	INV	EXP	ENV	ENV+	INV	EXP	ENV	ENV+
Year	2000	2050	2050	2050	2050	2050	2050	2050	2050
World	**907**	**1,077**	**1,072**	**1,060**	**1,052**	**1,092**	**1,081**	**1,072**	**1,066**
% irrigated area	**(26)**	**(26)**	**(21)**	**(16)**	**(14)**	**(28)**	**(21)**	**(15)**	**(13)**
CIS: Russia +	78	73	73	74	74	73	73	74	74
% irrigated area	(5)	(4)	(9)	(7)	(5)	(6)	(8)	(5)	(3)
EAS: China +	124	110	104	97	93	111	105	97	96
% irrigated area	(49)	(61)	(43)	(31)	(24)	(57)	(39)	(24)	(21)
EUR: Europe	65	66	67	67	67	65	67	67	67
% irrigated area	(10)	(11)	(11)	(9)	(8)	(13)	(11)	(8)	(5)
LAM: Latin America	96	158	163	167	168	165	170	173	173
% irrigated area	(12)	(10)	(9)	(7)	(6)	(10)	(9)	(6)	(4)
MNA: Middle-east and North Africa	44	49	49	47	46	53	52	50	49
% irrigated area	(38)	(40)	(31)	(25)	(22)	(35)	(25)	(18)	(16)
NAM: North America	115	120	120	120	119	116	118	117	117
% irrigated area	(29)	(28)	(29)	(27)	(26)	(36)	(33)	(30)	(27)
OCE: Oceania	19	19	19	20	20	21	22	23	24
% irrigated area	(4)	(5)	(4)	(3)	(2)	(7)	(5)	(3)	(2)
SAS: India +	181	185	178	172	168	190	174	170	168
% irrigated area	(42)	(53)	(41)	(34)	(30)	(60)	(44)	(36)	(33)
SEA: South-East Asia	70	97	95	95	95	94	97	99	98
% irrigated area	(29)	(24)	(24)	(16)	(13)	(27)	(24)	(12)	(5)
SSA: Sub-Saharan Africa	115	201	203	202	201	203	203	202	201
% irrigated area	(4)	(3)	(3)	(2)	(1)	(4)	(3)	(2)	(1)

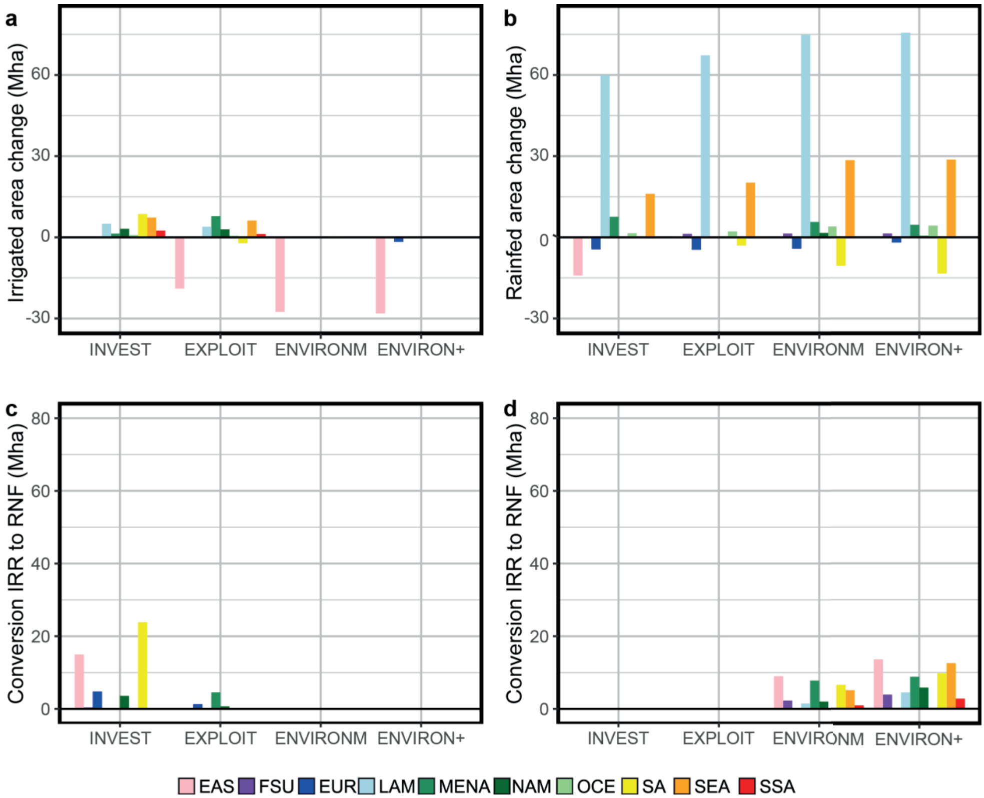
Appendix Table C4. Global agriculture production averaged over 2 GCMs per region in 2000 and 2050 including climate change (CC), water management (REG, EXP, ENV, ENV+) and international trading (unconstrained trade (unc_T) and constrained trade (fix_T)). The first line per region indicates the agriculture production and the second line is the percentage of food produced on irrigated area. Production is in 1000 metric tons of dry matter per year.

Trade		unc_T	unc_T	unc_T	unc_T	unc_T	unc_T	unc_T	unc_T	fix_T	fix_T	fix_T
Climate change		noCC	noCC	noCC	noCC	CC	CC	CC	CC	CC	CC	CC
Water management		INV	EXP	ENV	ENV+	INV	EXP	ENV	ENV+	EXP	ENV	ENV+
Year	2000	2050	2050	2050	2050	2050	2050	2050	2050	2050	2050	2050
World	**4323**	**10221**	**10098**	**10049**	**9971**	**9900**	**9693**	**9610**	**9534**	**9533**	**9059**	**8781**
% irrigated production	**(38)**	**(32)**	**(29)**	**(25)**	**(22)**	**(35)**	**(29)**	**(21)**	**(18)**	**(30)**	**(23)**	**(19)**
CIS: Russia +	184	308	320	326	329	330	338	339	339	335	335	333
% irrigated production	(6)	(4)	(12)	(10)	(8)	(7)	(11)	(8)	(4)	(12)	(8)	(4)
EAS: China +	764	1218	1142	1042	951	1171	1093	934	915	1104	846	763
% irrigated production	(53)	(58)	(56)	(43)	(34)	(64)	(54)	(35)	(32)	(54)	(38)	(35)
EUR: Europe	363	494	509	518	522	518	525	532	536	514	515	503
% irrigated production	(14)	(15)	(13)	(10)	(9)	(15)	(13)	(9)	(6)	(12)	(8)	(7)
LAM: Latin America	821	2556	2589	2703	2758	2654	2709	2860	2917	2567	2539	2482
% irrigated production	(24)	(9)	(8)	(7)	(6)	(9)	(8)	(7)	(4)	(8)	(6)	(5)
MNA: Middle-east and North Africa	125	342	325	271	247	363	333	257	239	324	266	242
% irrigated production	(62)	(69)	(64)	(50)	(43)	(66)	(58)	(40)	(36)	(59)	(43)	(37)
NAM: North America	524	836	839	846	849	676	681	658	653	720	687	660
% irrigated production	(45)	(52)	(54)	(51)	(50)	(53)	(51)	(45)	(43)	(54)	(49)	(47)
OCE: Oceania	76	163	164	163	156	177	180	178	171	164	160	159
% irrigated production	(35)	(26)	(24)	(21)	(18)	(20)	(25)	(18)	(13)	(26)	(18)	(15)
SAS: India +	721	2025	1926	1889	1836	1873	1701	1674	1624	1685	1633	1594
% irrigated production	(67)	(63)	(55)	(53)	(49)	(71)	(57)	(52)	(49)	(58)	(52)	(49)
SEA: South-East Asia	450	1049	1052	1074	1108	989	1002	1061	1043	984	962	961
% irrigated production	(28)	(14)	(14)	(9)	(8)	(30)	(26)	(14)	(6)	(26)	(14)	(6)
SSA: Sub-Saharan Africa	295	1230	1232	1217	1215	1149	1131	1116	1097	1137	1117	1083
% irrigated production	(12)	(9)	(10)	(7)	(7)	(10)	(9)	(6)	(2)	(10)	(6)	(2)

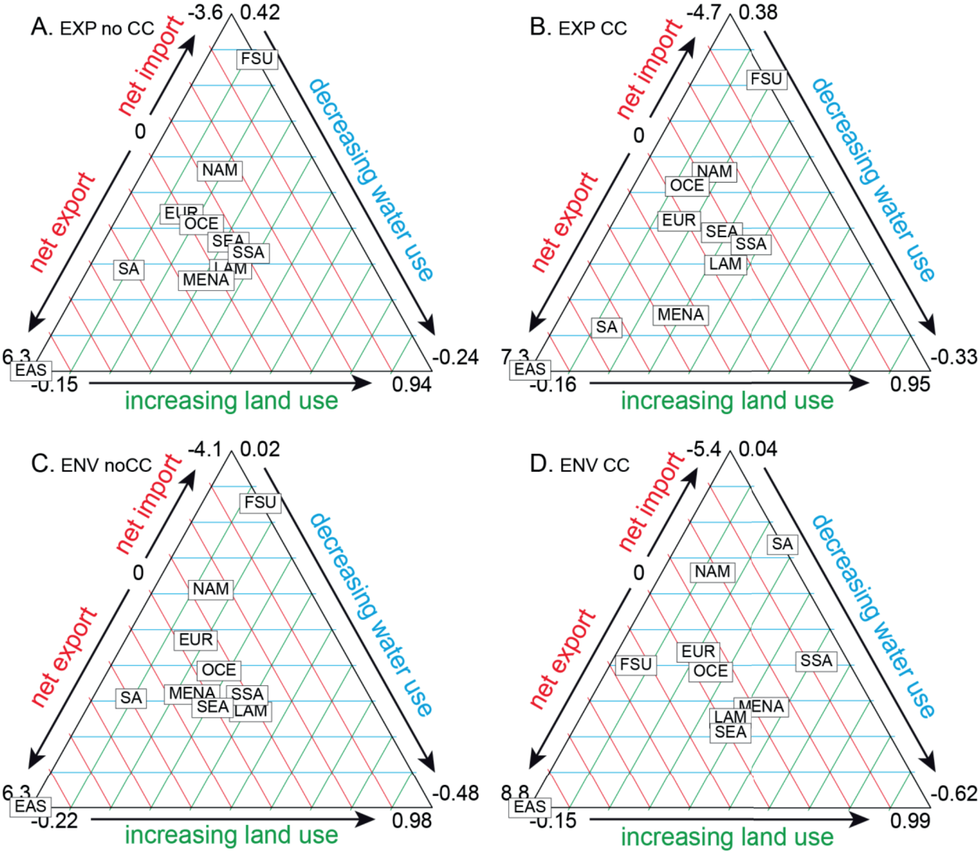
Appendix Figure C1. Conceptual framework of including water dynamics and climate scenarios into the GLOBIOM model.



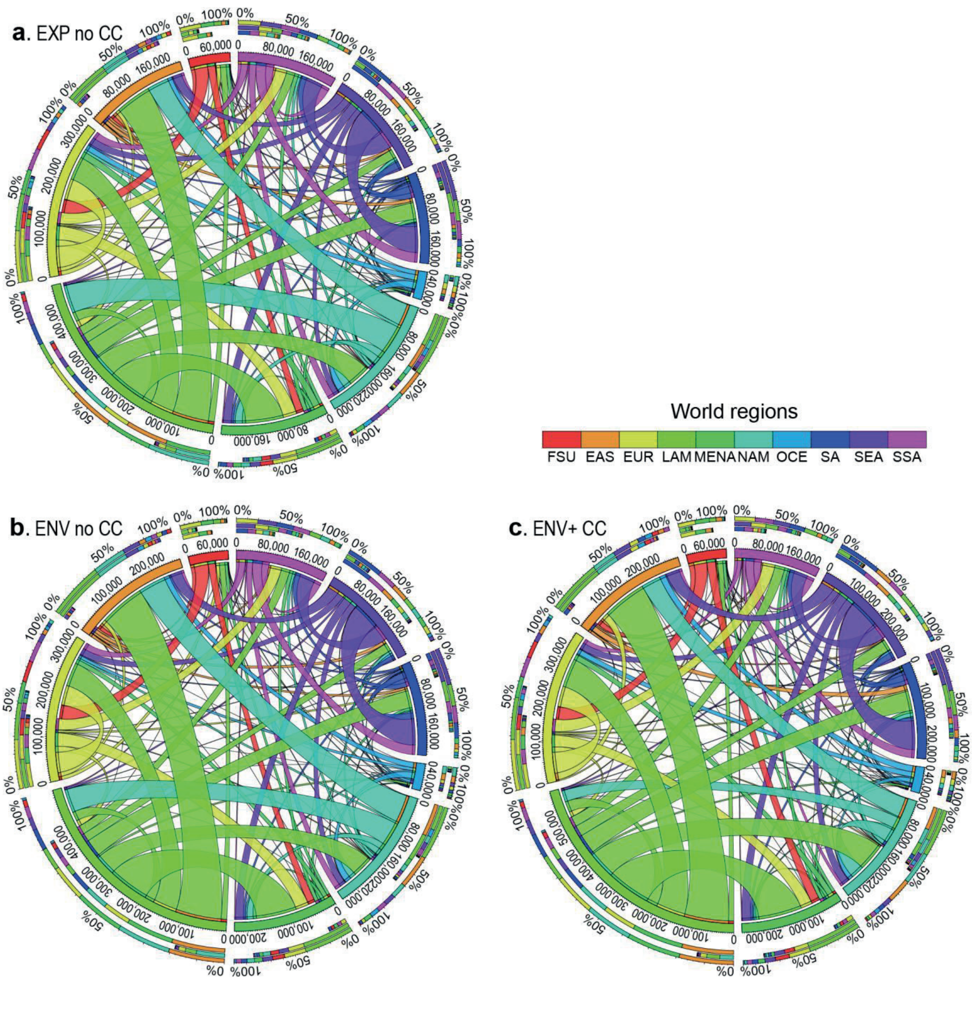
Appendix Figure C2. Spatial distribution of expansion of irrigated area, expansion of rainfed area, conversion from rainfed to irrigated and conversion from irrigated to rainfed areas in million hectares (Mha) for 10 world regions and for 4 water management scenarios (INV, EXP, ENV, ENV+).



Appendix Figure C3. Regional changes in trade, water and land use according to water management scenarios. Plots representing trade-off between 10 world regions among 3 variables trade, water use and land use for 4 water management scenarios (EXP no CC (a), EXP CC (b), ENV noCC (c) and ENV CC (d)).



Appendix Figure C4. Bilateral trade flow between regions under climate change and EFRs restrictions. Circle flow diagrams showing bilateral agriculture trade between 10 global regions for the EXP noCC (a), for the ENV noCC without climate change in 2050 (b) and the environmental flow scenario (ENV+ CC) with climate change 2050 (c). The second circle represents the percentage of agriculture production imported, the third circle represents the percentage of agriculture production exported, and the fourth circle represents the amount of agriculture production traded in kT yr-1.



Appendix C1. Appendix methods

Description of crop yield calculation

Yields of 18 different crops are estimated using the Environmental Policy Integrated Climate (EPIC) model and are adjusted according to the country scale GDP (Williams, Jones et al. 1989, Liu, Williams et al. 2007). Crop yields vary according to different cropping systems (such as intensive rainfed, extensive rainfed, and irrigated system) and according to land-use changes via the re-allocation of individual crops to more or less productive fields. Future crop yield projections were based on SSP2 yields with econometric calibration including technical progress, GDP and CO_2 fertilization (Herrero, Havlik et al. 2014).

Description of Environmental Flow Requirements (EFRs) calculations

Environmental flow requirements (EFRs) were estimated using the Pastor et al. (2014) Variable Monthly Flow (VMF) method. The VMF method follows the natural variability of river discharge by adjusting the EFR requirements according to the flow season. The VMF method was designed to improve the protection of freshwater ecosystems during low-flow seasons. In the VMF method, the EFR is set to 60% of the mean monthly flow during the dry (low-flow) season and 30% during the wet (high-flow) season. So, in the simulations where the VMF method for EFR is implemented 40% of the river water is available to other users during the dry season and 70% in the wet season. The VMF method was previously validated with 11 local case studies, where EFRs were calculated based on local ecological and hydrological parameters (Pastor et al. 2014). For the simulations, EFRs were calculated based on simulated natural run-off of the previous 15 years. For the climate change scenarios, EFRs were re-calculated every 10 years so it was assumed that in the future EFRs will be adapted based on new flow regimes.

Calibration of annual irrigation demand

To include water use for irrigation within GLOBIOM builds on the work presented in Sauer, Havlík et al. (2010), by defining spatially explicit irrigation demand, irrigation source, and seasonality of water, as well as examining the impacts of climate change. GLOBIOM calibrates spatially explicit water demand for irrigation, Irrigated Water Demand (*IWD)*, in the initial year 2000 using the irrigated cropland area dataset available from SPAM (You and Wood 2006), EPIC estimates of the crop irrigation water requirements, to match the FAO AQUASTAT statistics for water withdrawn for irrigation available from AQUASTAT (FAO 2015). For this study, simulations from the GLOBIOM model were adjusted from an annual to a monthly time step in order to account for the seasonality of water availability and demand.

Calibration of monthly irrigation demand to seasonality

The annual irrigated water demand estimated by EPIC was rescaled to a monthly time step using a coefficient of seasonal irrigation (CSI) defined for every grid cell. CSI is based on the monthly irrigated water withdrawal from LPJmL using Equation (1).

$$CSI(c, m) = \sum_{i=1}^{m} \left(\frac{mid(c,m)}{aid(c,a)}\right) \quad (1)$$

Where c, is the cell of the LPJmL model, *m*, the month and *mid*, monthly irrigation demand and *aid* is, annual irrigation demand. For simulations of the impacts of climate change, the annual water irrigated water requirements were estimated by EPIC, which considers the potential crop yields taking into account the local climate (Williams, Jones et al. 1989, Liu, Williams et al. 2007).

Representation of water sink in GLOBIOM

In this study, we divided the irrigation water demand into three categories: irrigation sourced by surface water (SWD), irrigation sourced by groundwater (GWD), and irrigation sourced by non-renewable sources (NR). We used the spatially explicit map of irrigated areas sourced from groundwater from Siebert, Burke et al. (2010) to determine the share of IWD sourced by groundwater (Equation 2). Non-renewable withdrawals were calculated as the water deficit that cannot be fulfilled by surface water or groundwater in 2000. The amount of water withdrawal coming from groundwater and nonrenewable sources is assumed to remain constant over time.

$$IWD_{m,lu} = SWD_{m,lu} + GWD_{m,lu} + NR_{m,lu} \quad (2)$$

To determine the irrigation sourced by surface water, we determined the surface water available, under the assumption that agriculture is the residual user of water, behind industry and households, and in certain scenarios, the environment.

Biophysical and economical water scarcity

In the simulations, the biophysical scarcity at the pixel level as well as the economic scarcity of the water price from the water supply curve take into account the growing demand for surface water from the other sectors as well as impacts from climate including the change in the quantity of surface water available (WA) and in change in the spatially explicit water demand for irrigation (*IWD)*. To capture the scarcity cost of water, GLOBIOM uses a supply function of the total volume of water withdrawn (the regional-level *IWD*) and a marginal price, which increases as water becomes scarce, in addition to the regional, crop, and pixel specific irrigation costs per hectare, developed by Sauer et al. 2010. Future water consumption from industry and households was based on Flörke, Kynast et al. (2013). Additionally, environmental flow requirements are added to some of the scenarios over the time period and further restrict the water available for agriculture.

Summary

Freshwater ecosystems are among the most threatened ecosystems on Earth. At the same time, water demand for food is projected to increase with projected increase in population and diet shift putting part of the population under pressure in terms of food security. These projections are likely to be exacerbated by climate change. Over the past decades, irrigated areas have nearly tripled to meet actual human food requirements. Today, 40% of food production comes from irrigated production and about 30% from irrigated areas. This increasing share of irrigated production has come at the expense of freshwater ecosystems and river health. About half of the rivers have been fragmented and altered via the constructions of dams and reservoirs and via diversion of river flow to irrigated fields. Furthermore, water demand for industry, household and hydropower is predicted to increase and competition between water sectors will intensify. Under actual water competition, water availability for freshwater ecosystems has often been neglected.

Over the past decade, awareness was given to define planetary boundaries for natural resources especially freshwater ones. While irrigation withdrawals and industries and household withdrawals already reach respectively about 2600 km^3 yr^{-1} and 1000 km^3 yr^{-1}, planetary boundaries for freshwater have been defined to 4000 km^3 yr^{-1}. With the expected rise in water demand for food and industries, freshwater boundaries are likely to be exceeded in the coming decades and it is urgent to define global water availability and demand with accurate time and spatial resolutions. More specifically, it is necessary to develop a method that enables the calculation of water demand for freshwater ecosystems known as "Environment Flow Requirements" (EFRs). EFRs were often neglected in global assessments and/or defined with annual proxies.

The overall objectives of this thesis were to redefine global water demand for freshwater ecosystems (EFRs) and set these last as a priority in global integrated assessments. For that, it was necessary to design a robust methodology that can be easily implemented in Global Hydrological Models (GHMs) and in global integrated assessments.

In chapter 2, existing global and local Environmental Flow (EF) methods were reviewed. Three methods were selected among existing global methods, including the Smakhtin method, which is based on a combination of annual quantiles and proxies of annual flow, the Tennant method, which is based on annual proxies of flow, and the Tessman method, which is based on monthly proxies of flow. Two other methods were designed for this study: the Variable Monthly Flow (VMF) method, which is based on the allocation of the percentage of monthly flow to the environment and the Q90_Q50 method, which is based on the allocation of flow quantiles. These methods were compared with 11 local case studies from different ecoregions, for which EFRs have been defined locally with ecological and hydrological data collection. The VMF method showed the best performance against

local case studies and demonstrated easiness of use and validation with different flow regime types. Among the five global EF methods, EFRs represent 20 to 50% of mean annual flow to maintain EFRs in "fair" ecological conditions.

In chapter 3, the concept of "Environmental Flow (EF) deficit" was designed. It represents the lacking flow to meet EFRs. EF deficit was defined on a monthly basis at 0.5 deg. The originality of this study is that the origin of the deficit was characterized by the natural deficit and the anthropogenic deficit. Natural deficit is defined when EFRs are not met due to natural climate variability and anthropogenic deficit is defined when EFRs are not met due to water extractions for irrigation or other users. The frequency, timing and magnitude of each deficit were also calculated at global scale. The EF deficit was also studied for 23 river basins, which are located in different ecoregions, and it was shown that flow regime type, origin of deficit, magnitude of deficit and level of flow alteration were correlated. Perennial rivers such as the Congo River showed only natural deficit while very altered river such as the Godavari river showed high respective natural and anthropogenic deficit.

In chapter 4, we set EFRs as a priority user in the global vegetation model LPJmL. It was shown that to sustain EFRs in "fair" ecological conditions, irrigation water use should be reduced by 30%, which would lead to 30% less food coming from irrigated area and a total of 5% loss in food production. Calorie loss per capita was really high in developing countries where population density is high such as in South-East Asia. This loss in food production can however be compensated by an increase of 50% in irrigation use efficiency.

In chapter 5, we used an economic optimization model (GLOBIOM) to study future global change including different constrains of EFRs. It was shown that, under future climate change (RCP 8.5) and socio-economic development (SSP2), international trade should be increased by 15% to compensate for EFRs implementations compared to a business-as-usual scenario. The positive outcome is that it was demonstrated that food and water security for humans and ecosystems can be sustained with three levees: use of trade (+15%), conversion of irrigated land to rainfed land (60Mha) in South Asia and expansion of rainfed land into natural area in Latin America.

In the chapter 6, we reviewed and analyzed each chapter as an ensemble. The new development of the VMF method is acknowledged thanks to its application in all chapters of this thesis and in many other global assessments. Among them, two studies redefined the freshwater planetary boundaries at 2,800 $km^3\ yr^{-1}$ which is lower than previous estimates defined by Rockstrom et al. (2009). This thesis allowed the inclusion of EFRs in global integrated assessments with refined temporal and spatial scales and water demand for ecosystems are now recognized and acknowledged. The limitations of the VMF method are also discussed such as its weakness to be compatible with inter-annual studies considering extreme events such as floods and droughts. Further data collection on eco-hydrological relationships should be organized and harmonized at global scale to further improve EFRs at global

scale. Characterization of EF deficit with differentiation of the anthropogenic and natural deficit can be used as a tool to prioritise actions in terms of river restoration/protection. In face of meeting future SDGs, we highlighted the complexity in meeting food and water security for humans and ecosystems. Competition between different water sectors already exist and require local, regional and international consensus to satisfy all water users while safeguarding water availability for freshwater ecosystems. For that, future improvement in agriculture and water management is fundamental to provide future sustainable water access to humanity.

Samenvatting

Zoetwater ecosystemen zijn een van de meest bedreigde soorten ecosystemen op aarde. Tegelijkertijd staan water en voedsel-zekerheid onder druk, vooral met het oog op toekomstige sociaal-economische ontwikkelingen (populatie groei , dieet verschuiving) en de intensifering van de water cyclus als het gevolg van klimaatverandering. De oppervlakte van geïrrigeerd landbouwgrond is de laatste decennia verdrievoudigd om te kunnen voldoen aan de de menselijke voedsel consumptie. Op het moment komt 40% van de voedselproductie van geïrrigeerde productieketens en nog eens 30% van geïrrigeerde gebieden. De toename van irrigatie voor voedselproductie gaat ten koste van zoetwater ecosystemen en de kwaliteit van rivieren. De helft van alle rivieren wereldwijd zijn gefragmenteerd en/of gewijzigd door de aanleg van dammen en waterreservoirs en door het verleggen van rivierbeddingen om landbouwgrond van water te voorzien. De verwachting is dat de vraag naar water voor gebruik in industrie, huishoudens en waterkrachtcentrales de komende jaren alleen maar zal toenemen. Als een gevolg, zal de concurrentie voor water tussen watersectoren groter worden. In het verleden is gebleken dat wanneer er concurrentie voor watergebruik is, het belang van water voor zoetwater ecosystemen vaak wordt genegeerd.

Het laatste decennium zijn er “planetary boundaries” bepaald voor het gebruik van natuurlijke hulpbronnen, vooral voor zoetwater. Het watergebruik voor landbouwirrigatie en industrie en huishoudens liggen momenteel respectievelijk op ongeveer 2600 km3 yr-1 en 1000 km3 yr-1. De planetaire grenzen voor het gebruik van zoetwater zijn vastgesteld op 4000 km3 yr-1. Met de verwachte groei in de vraag naar water door de landbouw en industriële sectoren, dreigen deze planetaire grenzen al te worden overschreden in de komende decennia. Het is daarom noodzakelijk om de verandering van de wereldwijde beschikbaarheid van water en vraag naar water te definiëren op zowel ruimtelijke als temporele schaal met een nauwkeurige resolutie. In het bijzonder is het noodzakelijk om een methode te ontwikkelen waarmee de waterbehoefte van zoetwater-ecosystemen, de zogenaamde "Environment Flow Requirements" (EFRs), kunnen worden berekend.

De algemene doelstelling van dit proefschrift is om de wereldwijde waterbehoefte van zoetwater ecosystemen (EFRs) te herdefiniëren en deze als prioriteit mee te nemen in mondiale ”Integrated Assessment Models” (IAMs). Om dit te doen, was het noodzakelijk om een robuste methode te ontwikkelen die geïmplementeerd kan worden in “Global Hydrological Models” (GHMs) en IAMs .

In hoofdstuk 2 wordt een overzicht gegeven van bestaande methodes voor het bepalen van mondiale en lokale “Environmental Flow” (EF). Drie methodes voor het bepalen van de mondiale EF worden besproken: de “Smakhtin” methode, gebaseerd op een combinatie van jaarlijkse kwantielen en een schatting van de jaarlijkse waterstroom, de “Tennant” methode, gebaseerd op jaarlijkse schattingen van de waterstroom en de “Tessman” methode, gebaseerd op maandelijkse schattingen van de

waterstroom. Twee nieuwe methodes werden voor deze studie ontwikkeld: de “Variable Monthly Flow” (VMF) methode, gebaseerd op de allocatie van het percentage van de maandelijkse stroom naar het milieu en de “Q90_Q50” methode gebaseerd op de allocatie van stroom kwantielen. Deze methodes worden in hoofdstuk 2 vergeleken met 11 lokale studies, die EFRs baseren op lokale ecologische en hydrologische data, uitgevoerd in verschillende ecoregios van de wereld. De VMF methode benaderde de resultaten van de lokale studies het beste en was gemakkelijk te gebruiken en te valideren voor verschillende soorten waterstroming regimes. De vijf mondiale EF methodes laten zien dat dat, om zoetwater ecosystemen in een redelijke ecologische staat te houden, EFRs rond de 20-50% liggen van de gemiddelde jaarlijkse waterstroom.

In hoofdstuk 3 wordt het concept “Environmental Flow (EF) tekort” besproken. Dit is het tekort in de waterstroom om aan de EFRs te voldoen. EF tekorten werden gedefinieerd op maandelijkse basis met een resolutie van 0.5 graden. Het vernieuwende aan deze studie is dat er onderscheid wordt gemaakt tussen natuurlijke en antropogene EF tekorten. Er is sprake van een natuurlijk EF tekort als er niet aan de EFRs wordt voldaan vanwege natuurlijke klimaat variabiliteit. Een antropogeen EF tekort ontstaat als er niet aan EFRs kan worden voldaan wegens watergebruik voor irrigatie van landbouwgronden of andere vormen van watergebruik door mensen. De frequentie, het moment en de omvang van elk EF tekort werden ook bepaald op mondiale schaal. Ook werd het EF tekort bekeken voor 23 specifieke rivieren in verschillende ecoregios. Het type stromingsregiem, de afkomst van het EF tekort en de omvang van de tekorten en de mate waarin rivierbeddingen waren aangepast bleken met elkaar te correleren. Sterk aangepaste rivieren, zoals de Godavari rivier, bleken een sterk natuurlijk en antropogeen tekort te hebben, terwijl overblijvende rivieren, zoals de Congo rivier, alleen een sterk natuurlijk tekort bleken te hebben.

In hoofdstuk 4 bespreken we het effect van EFRs als deze als prioriteit voor watergebruik worden gesteld in het mondiale vegetatie model LPJmL. De resultaten laten zien dat, om zeker te zijn van redelijke ecologische condities, watergebruik voor irrigatie van landbouwgronden zou moeten worden teruggebracht met 30%. Deze verlaging zou leiden tot een afname van 30% van de voedselopbrengsten van geïrrigeerde landbouwgrond en een verlies van 5% van de totale wereldwijde voedselproductie. Het model laat ook een hoge afname zien in het aantal calorieën per hoofd van de bevolking in ontwikkelingslanden met een hoge bevolkingsdichtheid, zoals Zuidoost Azië. Echter, het is mogelijk om het verlies in de voedselproductie op te vangen door een toename van de irrigatie efficiëntie van 50%.

In hoofdstuk 5 gebruiken we een economisch optimalisatie model (GLOBIOM) om verschillende beperkingen van EFRs te bestuderen in combinatie met de effecten van toekomstige wereldwijde veranderingen. De resultaten laten zien dat met de geprojecteerde klimaatverandering (RCP 8.5) en sociaal-economische ontwikkelingen (SSP2), de internationale handel met 15% zou moeten worden verhoogd om te compenseren voor de implementatie van EFRs. De positieve uitkomst is dat voedsel- en water- zekerheid voor mensen en ecosystemen kunnen worden ondersteund met drie maatregelen: gebruik van handel (+15%), omzetten van geïrrigeerde landbouwgronden naar landbouwgrond gebaseerd op regenval (60 Mha) in Zuid Azië en uitbereiding van landbouwgronden zonder irrigatie naar natuurlijke gebieden in Latijns America.

In hoofdstuk 6 herzien en analyseren we de resultaten van alle hoofdstukken gezamenlijk. De ontwikkeling van de VMF methode wordt erkend vanwege zijn toepassing in alle hoofdstukken van dit proefschrift en in vele andere wereldwijde assessments. Twee studies hebben de "planetary boundaries" van zoetwater gebruik opnieuw vastgesteld op 2800 km^3 yr^{-1}, wat een veel lagere waarde is dan de eerdere schattingen zoals gedefinieerd door Rockstrom et al. (2009). Dit proefschrift maakt het mogelijk om EFRs op te nemen in mondiale "Integrated Assessment Models" met hoge temporele en ruimtelijke resolutie. Verder heeft dit proefschrift het belang van water voor zoetwater ecosystemen een herkenbaar en erkenbaar probleem gemaakt. De tekortkomingen van de VMF methode, zoals de zwakke compatibiliteit met inter-jaarlijkse studies die extreme gebeurtenissen zoals overstromingen en droogte beschouwen, worden uitgebreid besproken. Om de schattingen van EFRs op mondiale schaal verder te verbeteren is het nodig om wereldwijd meer gegevens over eco-hydrologische relaties te verzamelen, vergelijkbaar te maken en te categoriseren. De karakterisering van EF tekorten en het onderscheid tussen antropogene en natuurlijke tekorten kan worden gebruikt als een instrument om prioriteiten te stellen op het gebied van rivier restauratie en/of beschermings maatregelen. Met het oog op de toekomstige SDGs, hebben we de complexe problematiek beschreven omtrent voedsel en water veiligheid voor zowel de mens als ecosystemen. De concurrentie tussen verschillende watersectoren heeft altijd bestaan en vereist lokale, regionale en internationale consensus om alle watergebruikers tevreden te stellen en tegelijkertijd het behoud van water voor zoetwater ecosystemen veilig te stellen. Ten slotte, zijn, om te kunnen voorzien in de water behoefte van de mens op een duurzame manier, toekomstige verbetering in landbouw en waterbeheer essentieel.

Résumé

Les écosystèmes d'eau douce contiennent les espèces les plus menacées de la planète. Parallèlement, les demandes en eau pour l'alimentation vont augmenter linéairement avec la croissance de la population et les changements de régimes alimentaires mettent en péril la sécurité alimentaire mondiale. Durant les dernières décennies, les surfaces en terres irriguées ont presque triplé pour satisfaire les besoins croissants de l'alimentation humaine. Aujourd'hui, 40% de l'alimentation humaine provient de la production agricole irriguée, laquelle recouvre 30% des surfaces agricoles. Cette croissance continue de la production des terres irriguées s'est développée au détriment des écosystèmes d'eau douce et au détriment de la qualité des rivières. Presque la moitié des rivières du monde ont été fragmentées et détruites via la construction de barrages et de réservoirs et via la déviation du débit des rivières vers les champs irrigués. De plus, il faut envisager que les demandes croissance en eau pour l'industrie, les foyers et les centrales hydro-électriques vont augmenter. La compétition entre les secteurs d'activités utilisant l'eau va donc s'intensifier et face à cette pression croissante, les disponibilités en eau pour les écosystèmes d'eau douce sont souvent négligées.

Durant la dernière décennie, la définition « des limites planétaires » concernant l'utilisation des ressources naturelles est devenue critique, notamment pour les ressources en eau. Alors que l'utilisation de l'eau du secteur agricole et industriel atteint respectivement 2600 km^3 par an et 1000 km^3 par an, les limites planétaires pour l'eau douce ont été définies à 4000 km^3 par an. Avec une projection croissante de demande en eau, le seuil des limites planétaires en eau douce est menace d'être dépassé. Il est donc urgent de redéfinir les disponibilités et les demandes mondiales en eau avec des échelles spatiales et temporelles fines. Il est aussi essential de développer une méthode permettant le calcul des demandes en eau pour les écosystèmes d'eau douce nomme : le « débit réservé ». Ce dernier a souvent été négligé dans les évaluations intégrées et/ou souvent défini seulement avec des pourcentages annuels du débit.

Les objectifs de cette thèse sont de redéfinir les demandes mondiales en eau pour les écosystèmes d'eau douce et de leur donner une priorité dans les évaluations intégrées mondiales. Pour cela, il était indispensable de développer une méthode à la fois solide et facilement applicable dans les modèles hydrologiques à échelle mondiale et dans les évaluations intégrées à échelle mondiale

Dans le chapitre 2, une étude bibliographique des méthodes du débit réserve est effectuée. Trois méthodes seront sélectionnées parmi les méthodes existantes : la méthode Smakthin basée sur les quantiles annuels et pourcentage du débit annuel, la méthode Tennant basée sur les pourcentages du débit annuel et la méthode Tessmann basée sur le pourcentage mensuel du débit. Deux autres méthodes sont conceptualisées : la «Variable Monthly Flow » méthode ou la méthode au « débit mensuel variable » basée sur le pourcentage du débit mensuel et la méthode du Q90_Q50 basée sur des quantiles du débit annuel. Ces méthodes ont été comparées à onze cas d'études locaux situés dans différentes écorégions du monde dont les débits réserves ont été définis avec la collecte de données écologiques et hydrologiques. La méthode VMF a révélé la meilleure performance par rapport aux cas d'études locaux et elle a démontré sa facilité d'utilisation et de validation avec différents types de régime d'écoulement. Parmi les cinq méthodes globales, le débit réservé représente 20 à 50% de l'écoulement annuel moyen pour maintenir les écosystèmes d'eau douce dans des conditions écologiques "acceptables".

Dans le chapitre 3, le concept de «déficit du débit environnemental » ou « Environnemental Flow (EF) déficit » a été conçu. Il représente le manque de débit pour répondre au débit réservé. Le déficit de débit réserve a été calculé mensuellement avec une dimension spatiale de 0,5 degré. L'originalité de cette étude est que l'origine du déficit a été caractérisée par le déficit anthropique et naturel. Le déficit naturel est défini lorsque le débit réservé n'est pas satisfait en raison de la variabilité naturelle du climat et le déficit anthropique est défini lorsque le débit anthropique n'est pas suffisant pour l'irrigation. La fréquence, la durée, le timing et la magnitude de chaque déficit ont également été calculés à échelle mondiale. Le déficit du débit réservé a également été étudié pour 23 bassins situés dans différentes écorégions et il a été démontré que le type de régime d'écoulement, l'origine du déficit, l'ampleur du déficit et le niveau d'altération de l'écoulement étaient corrélés. Les rivières stables comme le fleuve Congo ont montré seulement un déficit naturel alors que les rivières très dégradées comme la rivière Godavari ont montré un haut déficit naturel et anthropique.

Dans le chapitre 4, nous avons défini le débit réserve comme un utilisateur prioritaire dans le modèle de végétation globale LPJmL. Il a été démontré afin de maintenir le débit réservé dans des conditions écologiques "acceptables" que l'utilisation de l'eau pour l'irrigation devrait être réduite de 30%, ce qui entraînerait une baisse de 30% de moins de nourriture provenant des terres irriguées et une perte totale de production alimentaire de 5%. Il en résulterait une perte de calories très élevée par habitant dans les pays en développement. Cette perte de production alimentaire peut toutefois être compensée par une augmentation de 50% de l'efficacité de l'utilisation de l'irrigation.

Dans le chapitre 5, nous avons utilisé un modèle d'optimisation économique (GLOBIOM) afin d'étudier les changements futurs à échelle mondiale, y compris les restrictions en eau par le débit réservé. Il est démontré que, dans le cadre du futur changement climatique (RCP 8.5) et du développement socioéconomique (scenario SSP2), l'utilisation du commerce devrait être augmentée de 15% pour compenser les mises en œuvre du débit réservé par rapport à un scénario normalisé. Le résultat positif et probant est le suivant : la sécurité alimentaire et hydrique pour les humains et les écosystèmes peut être maintenue sous trois conditions: l'utilisation du commerce (+ 15%), la conversion des terres irriguées en terres pluviales (60Mha) en Asie du Sud et l'expansion des terres pluviales notamment en Amérique latine.

Dans le chapitre 6, nous avons examiné et analysé chaque chapitre sous forme d'ensemble. Le nouveau développement de la méthode VMF est reconnu grâce à son application dans tous les chapitres de cette thèse et ainsi que dans de nombreuses autres évaluations scientifiques. Parmi celles-ci, deux études ont redéfini les limites planétaires d'eau douce à 2800 km^3 par an, ce qui est inférieur aux estimations précédentes définies par Rockstrom et al. (2009). Cette thèse a permis l'inclusion du débit réservé dans les évaluations mondiales intégrées avec des échelles spatiales et temporelles fine et la demande en eau pour les écosystèmes est désormais reconnue. Les limites de la méthode VMF sont également discutées, comme notamment sa faiblesse pour être compatible avec les études interannuelles tels les événements extrêmes incluant inondations et sécheresses. Il faudrait organiser et harmoniser la collecte de données sur les relations éco-hydrologiques à l'échelle mondiale afin d'améliorer les méthodes de débit réservé. La caractérisation du déficit avec la différenciation du déficit anthropique et naturel peut servir d'outil pour définir le niveau de priorité d'action en termes de restauration/protection des rivières. Dans le cadre de l'agenda des futurs SDG, nous avons souligné la complexité de satisfaire la sécurité alimentaire et hydrique pour les humains et les écosystèmes. La concurrence entre les différents secteurs de l'eau existe déjà et nécessite un consensus local, régional et international afin de satisfaire tous les utilisateurs d'eau tout en préservant la disponibilité de l'eau pour les écosystèmes d'eau douce. Pour cela, l'amélioration future de l'agriculture et de la gestion de l'eau est fondamentale, elle assurera un accès durable et pérenne à l'humanité.

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Curriculum Vitae

Amandine V. Pastor was born on August 22nd 1983 in Perpignan (France). She studied Agriculture Engineering for 5 years at EIP (Toulouse, France) and during that time she worked in farms, universities and institutes worldwide (France, Canada, Australia, Spain). In 2006, she came to Wageningen to do a double Master degree in Plant sciences (in Natural resources management). During her Master thesis and internship, she went to Mexico; first to study the impact of leguminous species and organic fertilizers on maize yield and soil conservation, then she came back to study sediments loads in rivers of Michoacán and to assess the impact of land use on these last.

After her graduation, Amandine worked for the ICARDA as a project manager of the OASIS project to assess the impact of soil and water conservations measures on soil erosion in Morocco. After that, she work for the Farming System Ecology group for nearly 2 years as a research assistant on soil fertility aspects and on giving lectures about nutrient management and irrigation. Finally, she joined the Earth System Science (now called Water Systems and Global Change group) to start her PhD as an AIO with an NWO grant which was part of a call on managing and modelling global water resources in the planetary boundary context. During her thesis, Amandine went to spent scientific journey at the PIK institute to work with LPJmL and collaborate with respective colleagues, she attended many conferences around environmental flows and in 2014 she won the YSSP grant to spend 3 months at IIASA to work on including water restrictions in GLOBIOM model. In 2015, we was employed by IIASA for one year to continue her YSSP research and delivered her thesis on February 2017. Since then, she started her post-doc at IRD (France) and at cE3c (Portugal) to develop and apply narrative scenario for the MASCC project (Mediterranean agricultural soil conservation under global change).

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Sense Education Certificate

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Wageningen, 30 May 2017

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Prof. dr. Huub Rijnaarts

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The SENSE Research School declares that **Ms Amandine Pastor** has successfully fulfilled all requirements of the Educational PhD Programme of SENSE with a work load of 39.5 EC, including the following activities:

SENSE PhD Courses

- Introduction to R for statistical analysis (2010)
- Environmental research in context (2012)
- Research in context activity: 'Reviewing the intergovernmental panel on climate change (IPCC) review together with Netherlands environmental assessment agency Dutch delegation on Chapter 23 Europe' (2013)

Other PhD and Advanced MSc Courses

- Competence assessment, Wageningen University (2011)
- Introduction to C Programming for Embedded Systems, Personal Track Safety training centre (2011)
- Scientific writing, Wageningen University (2012)
- Voice matters, Wageningen University (2013)
- Interpersonal communication, Wageningen University (2013)
- Impact2C summer school, Max Planck Institute for Psycholinguistics (2013)
- Techniques for writing and presenting scientific papers, Wageningen University (2013)

External training at a foreign research institute

- Exchange with Potsdam Institute for Climate Impact Research (PIK), Potsdam, Germany (2013)
- Young Scientists Summer Program (YSSP), International Institute for Applied Systems Analysis (IIASA), Laxenburg, Austria (2014)

Management and Didactic Skills Training

- Assisting practicals in the BSc course 'System Earth - analysis at regional and global scales' (2011-2012)
- Teaching and assisting practicals in the MSc course 'Introduction to global change' (2013-2014)

Selection of Oral Presentations

- *Including environmental flow requirements in large river basins.* 2nd International Conference on World's Large Rivers, 21-25 July 2014, Manaus, Brazil
- *How to expand irrigated land in a sustainable way?* European Geosciences Union General Assembly (EGU2015), 12-17 April 2015, Vienna, Austria
- *Environmental flow deficit at global scale – implication on irrigated agriculture.* European Geosciences Union General Assembly (EGU2016), 17-22 April 2016, Vienna, Austria

SENSE Coordinator PhD Education

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Chapter's photo (by Amandine V. Pastor):

Chapter 1	Morning run, Rhine, Wageningen, NL (February 2015)
Chapter 2	Sunset at Rhenen lake, Rhenen, NL (May 2016)
Chapter 3	Promenade près du Llech, Espira, Fr (January 2015)
Chapter 4	Spree River, Berlin, Ge (March 2015)
Chapter 5	Danube (metro Alte Donau), Vienna, AT (November 2015)
Chapter 6	When Rio Negro meets the Amazon, from the Sky (July 2014)
Back cover	A delicate system, by Amandine V. Pastor (December 2014)