Identifying downstream gains from local losses

A new set of methods for tracking water reuse across river basins

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by

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SUMMARY

Downstream reuse of previously withdrawn water resources is a common phenomenon across river basins worldwide, particularly those with (semi-)arid climate conditions and intensive water resources development. Water reuse often occurs unplanned, remains undetected, and as a result is insufficiently considered in water saving attempts, water allocation strategies, and water rights and pricing systems. This has led to a long list of ineffective and counter-productive introductions of Water Saving Technologies and Practices (WSTPs), with significant economic, social, and environmental consequences. Awareness of indirect water reuse has increased in recent years, both in integrated river basin management as well as in the irrigation sector, which has traditionally been focused on enhancing irrigation efficiencies. However, accounting for water reuse in decision-making has remained limited due to problematic terminology, scarcity of data, and a general lack of methods and tools for explicit assessment of water reuse across hydrological systems.

This research aims to address these problems by developing a coherent set of methods for spatiotemporal evaluation of water reuse. This dissertation presents and demonstrates an appropriate framework of concepts and indicators, as well as a number of complementary procedures for quantifying these indicators based on innovative data sources and newly developed algorithms.

As a first step, the key hydrological processes associated with water reuse in a river basin are evaluated. A sound accounting framework is proposed based on the concept of hydrological fractions, tailored to the specific flows that are important to distinguish in water reuse assessments: consumptive and non-consumptive use, recoverable and non-recoverable flows, discharge to surface and groundwater, and managed (anthropogenic) and non-managed (natural) return flows. This framework is used to analytically explore the dynamics of water reuse systems and demonstrate impacts of reuse on recoverable flows and water saving opportunities across spatial scales.

In order to build on previously developed methods for evaluating water reuse, existing indicators are reviewed to identify their strengths and limitations. Three main categories exist, each with their own potential value to decision makers: indicators characterizing (i) an entire system of water users, (ii) a single water user based on its dependency on upstream return flows, and (iii) a single user based on downstream reuse of its non-consumed water. Generally, the more explicitly an indicator incorporates quantitative water flows and the more dimensions of reuse it addresses, the more actionable it is to policy makers. However, as indicators get more complex and ambitious, data requirements increase as well. Even for a well-gauged basin, the Arkansas Basin in the USA, it is found data scarcity limits a comprehensive assessment of water reuse, and some of the theoretically most informative indicators can simply not be quantified.

With the availability of the required hydrological data identified as a main shortcoming, alternatives to conventional measurement techniques are needed. These should particularly allow for partitioning consumptive and non-consumptive water use. To this end, the potential of satellite-derived data products to evaluate spatiotemporal dynamics of water availability and water use is investigated. For the Red River Basin in China and Vietnam, different public-domain actual evapotranspiration (ET_{act}) products are reviewed and evaluated in terms of consistency with measured streamflow and remotely-sensed rainfall. Since the individual ET_{act} datasets are based on fundamentally different algorithms, the case is made for constructing an ensemble product that makes use of their complementary qualities over heterogeneous terrains. Integration of this new ET_{act} product with satellite-derived data on precipitation and land uses results in robust water accounts for the yearly and multi-annual time scales. In addition, monthly storage changes can successfully be estimated by solely relying on remotely-sensed rainfall and ET_{act}. It is thus concluded that monthly, global satellite-derived ET_{act} products in the public domain provide a promising basis for stand-alone hydrological analyses, as well as for feeding and constraining simulation models. Moreover, they contain information on spatial water consumption that is essential for the subsequent steps towards quantifying water reuse.

The distinction between consumed and non-consumed fractions of a water withdrawal is the fundament of the proposed framework for water reuse analysis. Readily available ET_{act} products do not differentiate between sources of water, i.e. whether ET_{act} depends on *blue* (ET_{blue}) or *green* water (ET_{green}), and withdrawal data are often not available. A novel method to separate ET_{blue} from ET_{green} and calculate total blue water supply (Q_w), based on Budyko Theory, is presented and demonstrated for the Indus Basin Irrigation System (IBIS) in Pakistan. Overall, ET_{act} in IBIS is found to depend for 76% on applied irrigation water. The resulting ET_{blue} maps and Q_w estimates are used to compute Consumed Fractions (*CF*) for each canal command area, which range between 0.38 and 0.66. It turns out that most command areas rely substantially on water not diverted at the main canal head. The relatively low *CF* values and the fact that long-term canal supplies largely suffice to sustain ET_{blue}, indicate that extensive reuse of non-consumed flows occurs *within* CCAs. At the same time, the notably higher *CF* of the entire IBIS means that reuse of non-consumed water *between* CCAs is also considerable. Although the IBIS is generally not regarded as efficient, it is thus in fact well-adapted to (informally) reuse irrigation return flows.

To explicitly consider hydrological linkages between water users and avoid premature conclusions on reuse, non-consumed water needs to be tracked to determine whether it is actually recovered downstream. Such an analysis is imperative to anticipate impacts of WSTP implementation on downstream water users. In order to track and visualize spatial interactions and potential trade-offs, satellite-derived data need to be integrated with a water resources model that can incorporate multiple water management and allocation scenarios. A modification of the water quality tracing module in the Water Evaluation And Planning (WEAP)

model is developed and tested to track return flows of individual users and their downstream reuse for agricultural, domestic or natural purposes in the Segura River Basin in Spain. Based on the proposed framework of hydrological fractions and water reuse indicators, water users in the Segura Basin are described in terms of the scope for WSTP introduction and their vulnerability to upstream efficiency increases. In addition, as water demand calculations are part of the WEAP-VirtualTracer (WEAP-VT) model, impacts of changing flows on water shortages across the basin are also presented.

Chapters 2 - 5 of this dissertation each discuss methods for addressing a key component of a water reuse analysis in the basin context: (i) routines for extracting the basic spatiotemporal input data from satellite-derived data products, (ii) an algorithm for estimating (non-)consumed fractions at different scales, (iii) a water resources model tailored to tracking non-consumed flows and reuse, and (iv) meaningful indicators to inform the development, implementation, monitoring, and evaluation of technological and political interventions. Integrating these complementary methods can support water managers to take well-informed, cost-effective decisions, and reduce unintended, harmful trade-offs.

Since very few scientific studies have been performed until now that focus explicitly on quantifying indirect water reuse, there is considerable scope for future research to further explicate the opportunities and limitations of the innovative methods proposed in this dissertation, as well as to explore alternatives. In particular, the following topics are recommended for future research efforts: (i) determination of strengths and weaknesses of individual ET_{act} algorithms in different conditions and optimal ways for integration into ensemble products, (ii) obtaining a better understanding of the behavior of the Budyko Hypothesis across spatial and temporal scales in human-dominated water systems, and (iii) exploring alternative methods for estimating CF based on remote sensing, geographical information systems, and hydrological models. Scientific and technological progress could ultimately allow for an integrated model of green and blue water, constrained and fed by satellite-derived information, which addresses water reuse by simulating interactions between different types of water users through both surface water and groundwater processes.

This research deliberately takes a quantitative hydrology approach to understand interlinkages within existing water (re)use systems, and identify hydrological implications of WSTP implementation and accompanying policies that would have been otherwise overlooked. However, any attempt to incorporate such information into political, economic or technological decisions, should acknowledge that drivers and consequences of water reuse transcend disciplines and can be agronomical, economic, chemical, social, and political. As a result, consultation of a wide range of expertise is needed for effective management decisions, and multidisciplinary research efforts are required.

SAMENVATTING

Benedenstrooms hergebruik van water dat eerder onttrokken was uit een oppervlakte- of grondwaterlichaam is een veelvoorkomend fenomeen in stroomgebieden wereldwijd. Dit geldt vooral voor gebieden waar een (semi-)aride klimaat gepaard gaat met een hoge mate van ontwikkeling qua watergebruik en daaraan gerelateerde infrastructuur. Waterhergebruik vindt doorgaans informeel plaats, wordt niet expliciet waargenomen of gemonitord, en is daardoor vaak nauwelijks in acht genomen in pogingen tot waterbesparing, procedures van waterverdeling, en systemen die waterrechten reguleren. Als gevolg hiervan werken technologieën en praktijken om water te besparen (WSTPs) veelal onvoldoende, of zelfs averechts, met alle economische, sociale en ecologische gevolgen van dien.

De bewustwording van het vóórkomen van indirect waterhergebruik neemt de laatste jaren toe, zowel binnen integraal waterbeheer als in de geïrrigeerde landbouw, een sector die zich traditioneel sterk richt op het lokaal verhogen van irrigatie-efficiëntie. Desalniettemin wordt er in de praktijk nog onvoldoende rekening mee gehouden door besluitvormers in de landbouw- en watersector. Dit komt voort uit een foutief en verwarrend gebruik van terminologie, beperkte databeschikbaarheid, en een algeheel gebrek aan technische methoden om waterhergebruik in hydrologische systemen expliciet vast te stellen en kwantitatief te benaderen.

Dit onderzoek beoogt om deze problemen aan te pakken door een samenhangende set methoden te ontwikkelen om waterhergebruik te analyseren, zowel ruimtelijk als in de tijd. Daartoe beschrijft en test dit proefschrift een raamwerk van concepten en indicatoren, alsook een aantal complementaire procedures om deze indicatoren te kwantificeren met gebruik van innovatieve databronnen en nieuw ontwikkelde algoritmes.

De eerste stap is om de relevante hydrologische processen te identificeren en adequaat te beschrijven. Een conceptueel raamwerk op basis van de theorie van hydrologische fracties (*hydrological fractions*) wordt voorgesteld, specifiek gericht op de waterstromen die onderscheiden moeten worden in analyses van waterhergebruik: consumptief en niet-consumptief gebruik, herwinbare en niet-herwinbare stromen, afwatering op oppervlakte- en grondwater, en beheersbare en niet-beheersbare retourstromen. Met behulp van dit raamwerk wordt de dynamiek van typische waterhergebruikssystemen verkend en worden de effecten van hergebruik op waterbesparingsmogelijkheden gedemonstreerd.

Om voort te kunnen bouwen op eerder ontwikkelde methodes voor het in kaart brengen van waterhergebruik, worden bestaande hergebruiksindicatoren besproken met betrekking tot hun sterke punten en beperkingen. Drie hoofdcategorieën zijn geïdentificeerd: indicatoren die (i) een systeem van watergebruikers in zijn geheel beschrijven, (ii) een individuele gebruiker

beschrijven op basis van zijn afhankelijkheid van drainagewater van bovenstroomse gebruikers, en (iii) een individuele gebruiker beschrijven op basis van benedenstrooms hergebruik van zijn drainagewater. In het algemeen geldt: des te explicieter een indicator gebruik maakt van kwantitatieve informatie van de relevante waterstromen en des te meer dimensies van waterhergebruik worden meegenomen, des te bruikbaarder de indicator is voor beleidsmakers. Echter, meer ambitieuze en complexe indicatoren zijn ook data-intensiever. Zelfs voor een goed gemonitord stroomgebied als de Arkansas Rivier in de VS blijkt dataschaarste een probleem voor een grondige hergebruiksanalyse; enkele van de (in theorie) meest informatieve indicatoren kunnen simpelweg niet worden gekwantificeerd.

Dit betekent dat alternatieven dienen te worden gezocht voor het meten van de relevante waterstromen in het veld. Het gaat hierbij met name om het onderscheid tussen consumptief en niet-consumptief watergebruik, wat de basis vormt van de in dit proefschrift voorgestelde aanpak. Hiertoe is de potentie van satellietdata onderzocht om de dynamiek van waterbeschikbaarheid en -gebruik in ruimte en tijd te detecteren. Voor het stroomgebied van de Rode Rivier in China en Vietnam zijn vijf openbaar beschikbare datasets van actuele verdamping (ET_{act}) bestudeerd. De consistentie van deze ET_{act} producten met gemeten rivierafvoeren en satellietafgeleide regenvalproducten is geanalyseerd. Aangezien de ET_{act} datasets gebaseerd zijn op fundamenteel verschillende algoritmes, kan door het slim construeren van een ensemble product gebruik worden gemaakt van de complementariteit van deze benaderingen. Het nieuwe ET_{act} product dat op deze manier voor de Rode Rivier is samengesteld, is gecombineerd met ruimtelijke regenval- en landgebruiksdata om robuuste *water accounts* op te stellen voor jaarlijkse en meerjaarlijkse tijdschalen. Daarnaast is het mogelijk gebleken om maandelijkse veranderingen in waterberging te modelleren door alleen gebruik te maken van regenval en ET_{act} uit satellietgebaseerde dataproducten. De conclusie van deze analyse is dat maandelijkse satellietafgeleide ET_{act} datasets op globale schaal een veelbelovende basis vormen voor op zich zelf staande hydrologische analyses, alsook voor het voeden en kalibreren van simulatiemodellen. Bovendien is de ruimtelijke informatie die zij bevatten over consumptief watergebruik van groot belang voor de vervolgstappen richting het kwantificeren van waterhergebruik.

Om de consumptieve fractie (CF) van een wateronttrekking te bepalen, is het nodig om binnen de totale ET_{act} te differentiëren tussen de verdamping die afhangt van de onttrekking (*blauw* water) en de verdamping die afhangt van regenval (*groen* water). Daarnaast doet zich met betrekking tot de noemer van de CF ratio het probleem voor dat onttrekkingsgegevens vaak niet beschikbaar zijn. Een nieuwe aanpak voor het kwantificeren van zowel de teller als de noemer van de CF, op basis van de Budyko theorie, is beschreven en gedemonstreerd voor het Indus Basin Irrigatiesysteem (IBIS) in Pakistan. Met gebruik van deze methode is vastgesteld dat ET_{act} in het IBIS voor gemiddeld 76% afhangt van irrigatiewater (ET_{blue}). De kaarten van ET_{blue} en de berekende totale aanvoer van blauw water (Q_w) zijn gebruikt om de CF van elk beregend gebied op hoofdkanaalniveau (CCA) vast te

stellen. Hieruit blijkt dat de *CF* in IBIS op *CCA*-schaal varieert tussen 0.38 en 0.66, waarbij de meeste gebieden in substantiële mate afhankelijk zijn van water dat niet als oppervlaktewater door het hoofdkanaal wordt aangevoerd. Aangezien de *CF* waarden relatief laag zijn en de debieten in de hoofdkanalen over het algemeen voldoende zijn om ET_{blue} in stand te houden, is de conclusie dat aanzienlijk hergebruik van irrigatie water plaatsvindt *binnen* de CCA's. Tegelijkertijd betekent de duidelijk hogere *CF* van het totale IBIS dat hergebruik van water *tussen* CCA's zeker niet kan worden verwaarloosd. Ondanks dat het IBIS over het algemeen niet wordt gezien als een efficiënt irrigatiesysteem, blijkt hieruit dat (informeel) hergebruik van drainagewater intensief plaatsvindt.

Om de hydrologische verbindingen tussen watergebruikers expliciet mee te nemen en te voorkomen dat hier foutieve aannames over worden gedaan, is het nodig om retourstromen te traceren en vast te stellen in hoeverre benedenstrooms hergebruik daadwerkelijk plaatsvindt. Een dergelijke analyse is essentieel om te kunnen voorzien of de implementatie van WSTPs nadelige gevolgen kan hebben voor benedenstroomse watergebruikers. Om de ruimtelijke interactie en mogelijke wisselwerkingen tussen gebruikers in kaart te brengen, dienen satellietafgeleide data te worden geïntegreerd met een simulatiemodel dat meerdere waterbeheersscenario's en allocatiestrategieën kan analyseren. Een aangepaste versie van de waterkwaliteitsmodule in het Water Evaluation And Planning (WEAP) model is ontwikkeld en getest om retourstromen van individuele gebruikers te traceren en hun benedenstrooms hergebruik voor agrarische, huishoudelijke en natuurlijke doelen inzichtelijk te maken. Voor het stroomgebied van de Segura rivier in Spanje zijn de belangrijkste watergebruikers beschreven in termen van kansen betreffende WSTP-implementatie, en hun kwetsbaarheden t.a.v. veranderingen in de efficiëntie van bovenstroomse gebruikers. Hiervoor is het in Hoofdstuk 2 voorgestelde raamwerk van hydrologische fracties en waterhergebruiksindicatoren gebruikt. Aangezien berekeningen van watervraag eveneens onderdeel uitmaken van het ontwikkelde WEAP-VirtualTracer (WEAP-VT) model, zijn ook de effecten op watertekorten in het systeem voor verschillende scenario's gekwantificeerd.

In elk van de hoofdstukken 2 t/m 5 van dit proefschrift worden methodes belicht die betrekking hebben op een belangrijk aspect van een waterhergebruiksanalyse in de context van een stroomgebied: (i) routines om tijdreeksen van de benodigde ruimtelijke invoergegevens te verkrijgen uit satellietafgeleide dataproducten, (ii) een algoritme om (niet)-consumptieve fracties van watergebruik op verschillende schalen te kwantificeren, (iii) een simulatiemodel toegespitst op het traceren van retourstromen en hergebruik, en (iv) informatieve indicatoren die de ontwikkeling, implementatie, monitoring en evaluatie van technologische en beleidsmatige maatregelen kunnen ondersteunen. De integratie van deze complementaire methodieken kan waterbeheerders helpen om geïnformeerde en kosteneffectieve beslissingen te nemen, en ongewenste schadelijke wisselwerkingen tussen watergebruikers te beperken.

Aangezien tot nu toe zeer weinig wetenschappelijke studies zijn uitgevoerd die zich nadrukkelijk richten op het kwantificeren van indirect waterhergebruik, is de verwachting dat er door toekomstig onderzoek nog veel te winnen valt t.a.v. het uitdiepen van de mogelijkheden en beperkingen van de in dit proefschrift voorgestelde methoden. De volgende onderwerpen hebben hiertoe naar verwachting de meeste potentie: (i) vaststellen van sterke en zwakke punten van individuele ET_{act} algoritmen in verschillende omstandigheden en optimale manieren om deze te integreren in ensemble producten, (ii) het beter begrijpen van de validiteit van de Budyko Hypothese op verschillende ruimtelijke en temporele schalen in door de mens gedomineerde watersystemen, en (iii) het verkennen van alternatieve methoden om CF te schatten op basis van satelliet remote sensing, geografische informatiesystemen en hydrologische modellen. Uiteindelijk kunnen wetenschappelijke en technologische vooruitgang leiden tot een geïntegreerd model van groen en blauw water, waarin satellietafgeleide informatie wordt geassimileerd, dat waterhergebruik traceert door de interacties tussen verschillende soorten watergebruikers via zowel oppervlakte- als grondwater te simuleren.

Dit onderzoek richt zich bewust op methoden die hun grondslag vinden in de kwantitatieve hydrologie. Zo kan een systeem van water(her)gebruikers op een fundamentele wijze worden beschreven, op grond van de hydrologische connecties tussen individuele gebruikers en de mogelijke gevolgen van lokale WSTPs voor waterbeschikbaarheid elders. Het is echter belangrijk om hierbij aan te tekenen dat, om deze inzichten te verwerken in beleidsmatige, economische of technologische ingrepen, het multidisciplinaire karakter van indirect waterhergebruik dient te worden meegenomen. Zowel oorzaken als gevolgen van waterhergebruik overstijgen individuele disciplines en kunnen agronomisch, economisch, chemisch, sociaal en politiek van aard zijn. Het is derhalve vereist om kennis uit al deze domeinen te raadplegen en multidisciplinair vervolgonderzoek te doen om te komen tot effectieve beslissingen.

1 INTRODUCTION

1.1 Research context

1.1.1 *Saving water – the solution to the water crisis?*

Water is vital to sustaining human life and natural ecosystems, and a fundamental resource in the production of food, energy and goods in the manufacturing industry. In many parts of the world, economic development and population growth increasingly exert pressure on the finite amount of water that is available. Shifting precipitation patterns and more severe and prolonged drought episodes, inflicted by climate change, further exacerbate the mismatch between supply and demand (Immerzeel et al., 2020). Impacts are social, economic as well as ecological, and include (transboundary) political conflict, loss of livelihoods, food insecurity, economic damage, and ecosystem deterioration (de Graaf et al., 2019; Oki and Quiocho, 2020; Petersen-Perlman et al., 2017; World Bank, 2016).

As a result, "saving water" is nowadays a major objective of water pricing systems, allocation strategies, investments to build or modernize infrastructure, and public awareness campaigns targeted at the general public. Such political and technological interventions, however, only save water effectively if they actually make a quantity of water available to an alternative water use. This water use is often located downstream from the intervention and should have been identified as suffering from water stress or threatened by expected future supply deficiencies. Its services should be regarded as valuable to society, with benefits such as crop cultivation, industrial production, household water supply, or in the form of hydrological ecosystem services (Simons et al., 2017).

Although the concept of water saving may seem straight-forward at first glance, it has been frequently discussed in scientific literature starting in the mid-1990s (Seckler, 1996; Willardson et al., 1994). Many research articles have since then contributed to an extensive knowledge base of both conceptual explorations (e.g. Haie and Keller, 2014; Perry, 2011; Zhang et al., 2019) and empirical analyses (e.g. Lecina et al., 2010a; Pfeiffer and Lin, 2014; Wang et al., 2020) of the preconditions for truly achieving water savings. This continuing interest of the scientific community is due to the long list of practical cases where measures intended to save water have proven ineffective, or even counter-productive. Most of these cases involve irrigated agriculture, typically concerned with large managed water flows, and thus associated with a scope for water saving that is often assumed significant. Perez Blanco et al. (2019) reviewed the impact of 224 individual cases of implementation of Water Saving Technologies and Practices (WSTPs), distributed

around the globe, aimed at maintaining agricultural production while stabilizing or decreasing water consumption. Worryingly, an opposite effect of increased water consumption was found to be reported in 83% of the reviewed studies, and downstream water availability decreased in 69% of them.

Apparently, actually saving water is more complicated than it initially may seem. As discussed in the following sections, the challenges of realizing water savings in practice primarily result from a lack of incentives for water users to discharge locally-saved water back to the hydrological system, as well as from the disregard for existing downstream water reuse by water managers, engineers, and financing bodies of WSTPs.

1.1.2 *Water reuse definitions*

The non-consumed flow from a water user enters a network of hydrological flow paths and may ultimately be captured at another location. In this dissertation, water reuse is defined as the downstream use of non-consumed water that was previously withdrawn upstream. It therefore explicitly relates to *blue* water (Falkenmark and Rockström, 2006), and effects of rainfall are excluded from the concept. Reuse can take place through artificial withdrawals from surface or groundwater for purposes such as agricultural and landscape irrigation, industrial processes, domestic use, and aquaculture. Another form of water reuse is the dependency of natural systems on upstream non-consumed water for delivering valuable ecosystem services. Downstream (*indirect*) reuse mostly occurs unplanned and does not necessarily involve a treatment process. It is a phenomenon that occurs across a river basin, facilitated by both natural and manmade pathways, currently mostly invisible to water managers, and therefore largely disregarded in national statistics and policy mechanisms (Ait-Mouheb et al., 2020; Drewes et al., 2017). Recent studies raise awareness of complex networks of water reuse between users of varying nature, occurring through surface water as well as groundwater resources (e.g. Grogan et al., 2017; Thebo et al., 2017).

This dissertation sets out to identify and explore methods that support spatiotemporal assessment of the indirect, and particularly unplanned, water reuse as defined above. It should be noted that the term water reuse is also discussed in scientific literature in other contexts. Most notably, it is increasingly proposed as a non-conventional freshwater source to alleviate water scarcity (Dingemans et al., 2020). From that perspective, water reuse should be considered a WSTP in its own right. It is then by definition planned, with regulated non-consumed flows transported through hydraulic infrastructure, often involving treatment to *reclaim* water for on-site recycling or downstream reuse. A well-known example is the water reuse strategy outlined in Egypt's National Water Resources Plan (MWRI, 2005). Similar to other interventions affecting non-consumed flow, direct water reuse (or water *recycling*) should be carefully assessed against potential harmful impacts downstream, e.g. whether water is already reused informally downstream,

or playing a major role in maintaining environmental flows during parts of the year (Ait-Mouheb et al., 2020).

1.1.3 *Implementing Water Saving Technologies and Practices: motives and pitfalls*

Implementation of WSTPs involves local changes in technology and management practices. Here, "local" refers to any entity which has a degree of control over its consumption of water, which can be, for example, a field, irrigation scheme, factory, or town. Such entities are always located in the larger context of a river basin and typically interact with upstream and downstream users through their water management decisions. However, the individual water user, operating at this local level, is often not incentivized to take its role in the larger network into account.

A water user generally aims to maximize the benefits obtained from a given volume of water that is withdrawn. The quantity of water that does not contribute to a user's primary purpose is considered "lost", and therefore needs to be reduced as much as possible. There are two pathways on which a water user can embark to achieve this goal: (i) increasing production (of crop, energy, consumer goods, etc.) per unit of water consumed, and (ii) reducing the portion of abstracted water that is discharged in the production process and thus not contributes to the water user's purpose. Irrigated agriculture is typically targeted by WSTPs, due to the sector's economic and nutritional importance, as well as the sheer magnitude of hydrological flows involved. Various studies have taken stock of the different WSTPs that can be considered in agriculture, and can all be accommodated in the two aforementioned categories (e.g. Ahmad et al., 2007; Giordano et al., 2017; Van Opstal et al., 2020).

The first pathway is commonly referred to as improving *water productivity*, expressed for example in kg per m^3 or $ per m^3 of water consumed (Molden et al., 2010). An increase in productivity provides a water user with the opportunity to increase overall benefits while stabilizing consumptive use of water. In agriculture, for example, the production side of the equation can be enhanced by practices such as improving fertilization, laser land levelling, or switching to less water-intensive crop varieties. The denominator can be decreased by reducing the volume of water consumed as an unintended result of the production process, such as bare soil or open water evaporation. Conversion from surface irrigation to sprinkler or drip systems is often promoted to achieve this. In practice, however, overall consumption often does not change as water then shifts from non-beneficial to beneficial consumption (FAO, 2017; Jägermeyr et al., 2015).

The second pathway puts water that was previously not consumed, also referred to as return flow, drainage, or wastewater, to local beneficial use. Examples include fixing leaky supply and conveyance systems or developing additional storage facilities on-site. Figure 1.1 illustrates the typical consumptive and non-consumptive flows associated with irrigated agriculture. Increasing water consumption relative to supply is especially a common objective of the introduction of hi-tech irrigation systems, and is referred to as improving *classical irrigation*

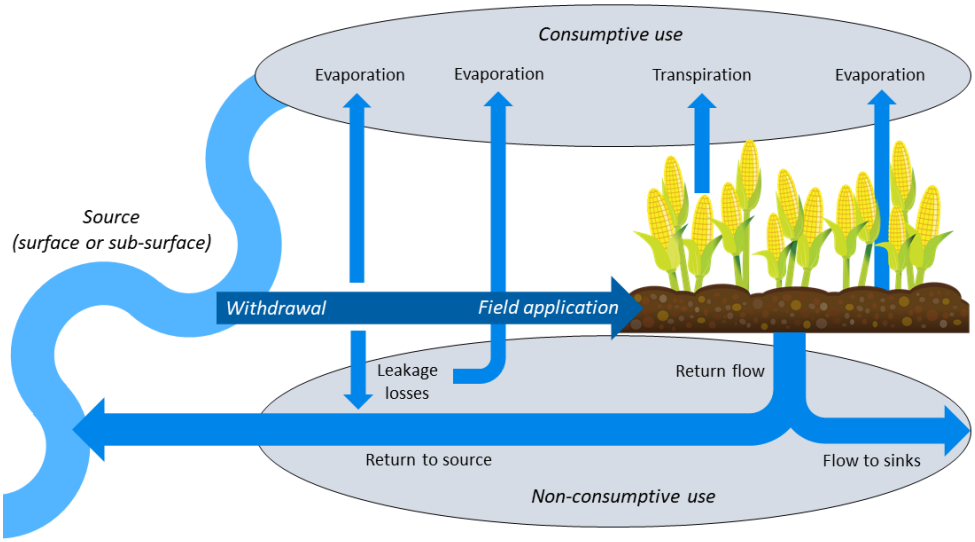

Figure 1.1. Consumptive and non-consumptive water use in irrigated agriculture (based on Richter et al., 2017).

efficiency (Jensen, 2007). However, by retaining and consuming water that previously returned to surface or groundwater resources, water supply to downstream users can be negatively affected. As a result, efficiency-enhancing measures can only effectively save water on a larger scale under very specific preconditions related to hydrology and governance. Matters become even more complicated in systems where low efficiencies are in fact beneficial in certain ways. For example, agricultural ecosystems in arid climates are prone to soil salinization, which needs to be mitigated through application of excess irrigation water. Also, in conjunctive use systems such as the Indus Basin Irrigation System in Pakistan (Chapter 4), return flows caused by low irrigation efficiencies in fact help to replenish aquifers and avoid systematic overexploitation (Qureshi, 2018).

It is in most cases an economic decision for a farmer to implement WSTPs, expecting a return on investment through increased yields, higher crop quality, or the possibility to grow higher-value crops (Perry et al., 2009). Water saving as such, however, is generally not in their interest (e.g. Ortega-Reig et al., 2017). The past decades have seen the implementation of significant investments by International Financial Institutions (IFI's) and government subsidy programs to support agricultural water users with the introduction of WSTPs, aiming to modernize what has been considered a highly wasteful sector (e.g. Lopez-Gunn et al., 2012; Rodriguez Díaz et al., 2012; Salman et al., 2020). Particularly traditional systems relying on surface irrigation and earthen canals have been targeted, with overall efficiencies commonly measured to be under 50% (Bos and Nugteren, 1990). In these projects, the big question often remains unanswered: what happens with the non-consumed water? Since no new underground seas are being formed, a major

part of global irrigation return flows is likely already reused. This existing indirect water reuse, combined with the tendency of rational water users to beneficially consume the water that is freed-up locally, is a root cause of WSTPs failing in practice (FAO, 2017). To ensure that WSTP implementation does not lead to increasing consumption and decreasing water availability at basin level, but also to avoid the pitfall of wrongfully generalizing beneficial reuse of all return flows (Lankford et al., 2020), the baseline situation and larger hydrological context need to be thoroughly understood prior to intervening.

1.1.4 *Accounting for indirect reuse in water policy and regulations*

Achieving water savings is an important objective of actions taken by basin water authorities and other public institutions involved in water management, with the ambition to allocate additional water to demands that are currently not met. Both aforementioned categories of WSTPs (productivity and efficiency) can theoretically lead to basin-scale water saving, but only if other water users, including sites of environmental importance, actually gain access to an increased supply. It is therefore essential for policies accompanying WSTPs to prevent upstream users from maximizing their own benefits with the "extra" water that is made available, if their return flows are currently reused to the benefit of others. Depending on the hydrological setting, they should be sufficiently motivated to reduce abstraction from the water source, decrease their consumptive use, and/or sustain a certain minimum level of return flow.

Designing and implementing these policies and regulations is proving a challenging task (e.g. Fei et al., 2021; Zhang et al., 2020). Capping evapotranspiration is one of the policy options showing promising results in selected case studies, although currently not yet widely practiced (Richter et al., 2017). An example is the Turpan Oasis in Northwestern China, where natural lakes are reported to recover after several years of operations (World Bank, 2017). Still, the numerous instances of ineffective WSTPs show that water policy makers and managers often fail to complement improved technology and practices with effective regulations. A notorious example is the implementation of the Murray-Darling Basin Plan in Australia, which includes mechanisms of purchasing agricultural water entitlements by the government and subsidizing hi-tech irrigation infrastructure, to transfer the assumed water savings to environmental use. However, evidence points at significantly increased water withdrawals by participating farmers, and previously assumed streamflow increases are called into question (Wheeler et al., 2020; Williams and Grafton, 2019a).

When trying to understand the scope for saving water, planning for improved infrastructure and management practices, and developing effective water allocation and monitoring mechanisms, policy makers need to satisfy the high demand for information associated with each of these processes. For example, after a basin authority has determined that a high-priority wetland experiences water stress (i.e. demand exceeds supply), they decide that options need to be explored

to save water elsewhere in the basin for alleviating this stress. This requires knowledge of the network of upstream water users affecting supply to the wetland, such as farms and urban areas. The basin authority needs to understand how much water is withdrawn, how much of it leaves the system through consumption, and how much is discharged and ultimately reused by the wetland. In addition, complex scale effects and trade-offs upstream of the wetland need to be accounted for. As discussed in Chapter 2, assessment of these water flows is currently greatly inhibited by a lack of data and tools, and further complicated by invalid or ambiguous terminology and analytical frameworks. It is tempting to avoid complexity and assume that a higher efficiency is good by definition; an assumption only valid if no downstream water reuse occurs (Frederiksen and Allen, 2011). As a result, implementation of upstream WSTPs might simply reallocate water from downstream to upstream users, thus depriving our hypothetical wetland from water even further.[1]

The aspects of water reuse that need to be considered by water managers are not just related to quantitative flows. Water quality degradation is a well-known effect of subsequent instances of water use, with its severity depending on the type of water sources, nature of use, and type of water treatment method, if any (Dingemans et al., 2020). Especially unplanned reuse can pose risks to human health as well as the environment, as it is often not explicitly incorporated in policies and guidelines. Common situations where water quality is particularly of concern are when municipal wastewater effluent is discharged to rivers or aquifers and informally reused downstream, or when progressive irrigation cycles lead to increased concentration of salts and agrochemicals in the applied water (e.g. Barnes, 2014; Beard et al., 2019; Drewes et al., 2017).

Clearly, water managers would benefit greatly from a better understanding of water reuse, in order to ensure that WSTP introduction and water allocation strategies achieve the intended objectives and all relevant implications and trade-offs are properly taken into account. In short, information on water reuse is a prerequisite to (i) determine whether truly "saving water" is in fact feasible, (ii) identify water users to be targeted, (iii) determine what type of WSTPs could be effective in the local context, (iv) support development of policy mechanisms to complement these WSTPs for optimizing chances of success, (v) monitor the effectiveness of existing regulations, and (vi) shed light on water quality risks caused by unplanned reuse.

[1] It can also be a deliberate choice to reallocate downstream water to upstream users, e.g. justified on an economic basis. Still, this requires a thorough understanding of water reuse, to assess implications of upstream flow alteration by WSTP introduction, identify affected downstream users, and potentially decide on compensation measures.

1.1.5 *The potential of satellite remote sensing for analyzing water reuse*

A robust assessment of indirect water reuse first and foremost requires quantitative data on the water balance, and how it is influenced by withdrawal, consumption, and discharge of water from and to the hydrological system. The sources used for obtaining these data need to be flexible regarding spatial scales and should allow for monitoring of the relevant flows with regular time intervals.

Remote sensing with satellite-based sensors, both passive and active, provides a data stream that has proven promising in satisfying these preconditions. In addition, it provides low-cost measurements over extensive areas and is an alternative to implementation of elaborate field measurement campaigns (thus avoiding common challenges related to terrain accessibility and/or political sensitivity). Because of these characteristics, earth observation is increasingly deployed in water resources assessments (UNESCO, 2021). Physical water-related parameters potentially relevant to evaluating water reuse, such as rainfall, actual evapotranspiration, and soil moisture, are available in the public domain from various satellite-derived data products for the entire globe (Karimi and Bastiaanssen, 2015). Post-processing and interpretation steps are still required to translate these data into information products and tools targeted at water managers.

The Water Accounting Plus (WA+) framework, in various forms, is often used for integrating different remotely sensed data layers and presenting derived indicators to inform water management authorities (e.g. Delavar et al., 2020; Hunink et al., 2019; Karimi et al., 2013a). WA+ is usually implemented on the basin scale and disaggregated for different types of water use, yielding important information on sector-level consumptive use and overall scope for water resources development. Water reuse, however, remains implicitly included in the basin-level outputs, as connectivity and water flows between users are not explicitly considered.

Part of this dissertation investigates the potential of satellite remote sensing to satisfy the knowledge gaps associated with the dynamics and patterns of water reuse across a river basin. In this manner, it can be seen as complementary to WA+ and other basin-level water accounting approaches. The connection of satellite-derived information to simulation models is also explored, to evaluate direct interactions between water users and allow for analyzing impacts of different water management strategies.

1.2 Research scope and objective

From Section 1.1, it was established that there exists an urgent need for a set of methods that allows for the assessment of indirect, unplanned water reuse in river basins. To address the identified knowledge gaps, the methodology should include an analytical framework of indicators, as well as the input data and algorithms required to apply this framework in practice. Ultimately, this should support water managers to take well-informed, cost-effective decisions and reduce unintended,

harmful trade-offs. The fundamental concepts should apply to all types of water use. However, special attention needs to go out to methods applicable to irrigated agriculture; the sector typically targeted by WSTPs and with the greatest potential for reuse of its non-consumed flows.

The overall objective of this research is thus *to develop a coherent set of methods for spatiotemporal evaluation of water reuse across river basins, encompassing concepts, indicators, input data, and algorithms.* This dissertation aims to provide the scientific community with a set of promising methods to further explore in different geographical settings, tailor to particular conditions, and improve with new data and according to future insights.

The following research questions are addressed:

1. What are the key hydrological processes associated with water reuse in a river basin among users of varying nature, and how should these be described in a sound accounting framework?
2. What are the knowledge and data gaps related to existing methods for evaluating non-consumed water and its downstream reuse?
3. What is the potential of satellite-derived data products to evaluate spatiotemporal dynamics of water availability and water use?
4. Can the consumptive and non-consumptive portions of water use be quantified based on satellite remote sensing data?
5. How can spatial interactions and trade-offs between water users be tracked and visualized to support more effective water resources management?

1.3 Outline of the dissertation

The research questions are discussed in four main chapters. Chapter 2 addresses research questions 1 and 2. Here, the existing body of work on the analysis of water reuse processes in river basins is reviewed, and a framework is presented for assessment of the relevant hydrological flows. This chapter also discusses existing water reuse indicators, and reviews them in terms of information provided as well as their applicability in practice. A concise case study analysis, the Arkansas River Basin in the USA, is presented to demonstrate the insights gained by application of these indicators to a real-life situation. The crucial variables to address in water reuse assessments are presented and key knowledge and data gaps are identified.

Chapter 3 addresses the third research question and focuses on the use of satellite-derived data to evaluate river basin hydrology. The chapter focuses on Global-scale Satellite-derived Data Products (GSDPs) on actual evapotranspiration and precipitation, which have recently become available in the public domain. By investigating the case study of the Red River Basin in Vietnam and China, the chapter evaluates the accuracy of state-of-the-art GSDPs and evaluates the opportunities and limitations associated with integrating GSDPs to describe sub-annual hydrological dynamics and constrain or calibrate hydrological models.

Chapter 4 builds on the preceding chapters by demonstrating an approach to use GSDPs for quantifying a key input parameter in water reuse assessments: the Consumed Fraction (*CF*) of the volume of water supplied to irrigated agriculture. Research question 4 is answered by presenting a method based on Budyko Theory, which is demonstrated in the context of the Indus Basin Irrigation System (IBIS) in Pakistan. The variability of *CF* and non-consumed flows is investigated between canal command areas and across spatial scales. Results are discussed in relation to dynamics of water reuse in the IBIS.

In Chapter 5, spatial interactions between water users are quantified by integrating a VirtualTracer module into a water resources model, focusing on a case study of the Segura River Basin in Spain. Based on the insights gained from Chapter 2, key water reuse indicators are formulated and quantified for sub-annual as well as multi-annual periods, and at the user, sectoral, and basin levels. By varying Consumed Fraction values at the user level, different water management scenarios are simulated to analyze the scope for obtaining benefits related to basin-scale water saving, relieving water stress, and mitigating overexploitation of water resources.

Finally, Chapter 6 synthesizes the main research findings and emphasizes the interlinkages between the methods presented in the previous chapters. This chapter discusses the potential implications for water management and provides recommendations for future research.

2

WATER REUSE IN RIVER BASINS WITH MULTIPLE USERS: A LITERATURE REVIEW

Unraveling the interaction between water users in a river basin is essential for sound water resources management, particularly in a context of increasing water scarcity and the need to save water. While most attention from managers and decision makers goes to allocation and withdrawals of surface water resources, reuse of non-consumed water gets only marginal attention despite the potentially significant volumes. Consequently, gross mistakes are often made in claims of water saving. This chapter reviews the methods and indicators that aim to translate geographically explicit data to meaningful information on the cascade of water reuse across a river basin. First a conceptual representation of processes surrounding water withdrawals and associated definitions is discussed, followed by a section on connectivity between individual withdrawals and the complex dynamics arising from dependencies and tradeoffs within a river basin. The current state-of-the-art in categorizing basin hydrological flows is summarized and its applicability to a water reuse system is explored. The core of the chapter focuses on a selection and demonstration of existing indicators developed for assessing water reuse and its impacts. It is concluded that although several methods for analyses of water reuse and non-consumed flows have been developed, multiple crucial aspects of water reuse are excluded from existing indicators. Moreover, a proven methodology for obtaining crucial quantitative information on non-consumed flows is currently lacking. Future studies should aim at spatiotemporal tracking of the non-consumed portion of water withdrawals and showing the dependency of multiple water users on such flows to water policy makers.

Chapter based on: Simons, G.W.H., Bastiaanssen W.G.M., and Immerzeel W.W., 2015. Water reuse in river basins with multiple users: a literature review. Journal of Hydrology vol. 522, pp. 558-571; doi10.1016/j.jhydrol.2015.01.016

2.1 Introduction

Water scarcity is regarded as one of the world's biggest challenges (FAO, 2012; UN-Water, 2012). Growing water scarcity increases the need for effective management of water resources, with sustainable access to water expected to be a priority in the post-2015 Sustainable Development Goals (SDG's) under development by the United Nations (Griggs et al., 2013; UN-Water, 2013). Factors such as population growth and changing diets influence demand, while climate change is expected to impact regional availability of renewable water resources (Falkenmark, 2013; Oki and Kanae, 2006). Semi-arid and arid areas are particularly vulnerable to water scarcity due to limited replenishment of available surface freshwater from precipitation, triggering groundwater overexploitation (Döll et al., 2012; Konikow, 2011; Wada and Bierkens, 2014).

The river basin, containing a variety of water users requiring access to a share of the available inflow, is the natural unit for developing strategies to cope with water scarcity. Decisions need to be taken based on the integrated hydrological, economic, and environmental systems. However, in practice, development of infrastructure in river basins to capture sufficient water for satisfying local demand often takes place up to, and beyond, the point where commitments can no longer be met by natural flows. These commitments include agreed water quota to downstream users and sustaining certain environmental flow levels. As a result, the phenomenon of basin closure (Molle et al., 2010; Seckler, 1996) is the reality in many river basins, with famous examples such as the Yellow River (Yang and Jia, 2008), Krishna (Venot et al., 2008a), and Jordan River Basins (Venot et al., 2008b).

The growing complexity of the network of water users in many basins has led to elaborate discussion of appropriate methods and terminology to describe and evaluate water use. The desire exists to have standardized indicators to communicate complex hydrological information generated by the scientific community to water policy makers, facilitating comparisons between individual water users and river basins, as well as monitoring progress towards policy goals. However, ambiguous definitions and disagreement on proper applications of indicators have resulted in a range of examples of erroneous and often misleading interpretations of the water balance (Frederiksen and Allen, 2011; Perry, 2007). The discussion is strongly connected to the issue of scale and is in particular associated with accounting for water that is withdrawn, but not consumed. The extent to which this water in fact constitutes a resource for downstream water users is the crucial question. Non-consumed water may become available for withdrawal by downstream users through natural and artificial pathways. However, whether reuse of this water occurs is often unknown, while such information is essential for predicting basin-wide implications of locally altered flows. Total water saving potential at basin level is often overestimated due to the disregard of downstream water reuse (Molle and Turral, 2004).

Systems for regulating and evaluating water management are traditionally based on water withdrawals only. Consequently, water saving studies generally focus on

analyzing the magnitude of water withdrawals, which may overestimate the full impact on downstream water users as reuse is ignored by definition. Examples of water right systems based on withdrawals are the Chinese Water Withdrawal Permit System (World Bank, 2012) and the Australian national water accounting system (BOM, 2012). AQUASTAT, the global information system on water and agriculture of the Food and Agriculture Organization of the United Nations (FAO) is arguably the most comprehensive data source on water use that is available, but is also focused on withdrawals rather than the distinction between consumed and non-consumed water. Flow valuation concepts provide interesting opportunities for basing water allocation on the value generated by a water particle along its full flow path (Seyam et al., 2002), but should not neglect the downstream values generated by non-consumed flows.

Chapagain and Tickner (2012) described how consideration of consumed flows rather than withdrawals provides valuable insights in the pressure on water resources; illustrating the need to go beyond water withdrawals when regulating water permits, particularly in water-scarce areas. Over-exploited basins have the undesirable situation that evapotranspiration (comprising both landscape ET and incremental ET as a result of irrigation) exceeds precipitation, and that the shortage of water is supplemented from the surface water storage system and (un)saturated soil water zone. Reduction of this excessive consumptive use will automatically restore streamflow (e.g. Bastiaanssen et al., 2008). Thevs et al. (2014) investigated the discrepancies between water consumption and water withdrawal quota for the overexploited Aksu-Tarim Basin, China. Shifting from withdrawal allocations to water consumption management is a measure that is advocated by the World Bank (2012), Wu et al., (2014) and Zhong et al. (2009) in the context of the Hai Basin. This general notion is supported by Hoekstra (2013) who advocated restrictions of water consumption through “blue water footprint caps”, proposing a value of 20% of natural runoff as a rule of thumb.

Managing non-consumed flows provides another way of adapting water management to water-scarce conditions. Examples of intervention strategies targeted at non-consumed water are wastewater treatment, water retention, and reuse of drainage water for irrigation. From an economic point of view, not consuming withdrawn water can have positive externalities that need to be addressed in water pricing systems (Macdonald et al., 2005; Taylor et al., 2014). Certain countries include return flow obligations as part of their water right systems, and thus explicitly recognize the need to quantify non-consumed water flow and reuse. The basin-wide effectiveness of managing non-consumed water depends strongly on where, relative to the hydrological system of the basin, it is implemented. Delineating water management zones can be helpful to outline appropriate management strategies for different locations in a river basin. The concept of hydronomic zones (Molden et al., 2009, 2001b) is a method of catchment zonation primarily based on the potential for reuse of non-consumed water from an area, including the impact of water quality loss due to pollution or salinity. It is

helpful as an initial tool to provide contextual information, but more detailed information on quantitative flows is needed for proper management application.

A framework for assessing water use based on consumed and non-consumed water, demands a set of tools for basin-wide categorization and quantification of these flows. Remotely-sensed ET mapping by means of surface energy balances has developed rapidly, and spatially discrete ET maps can be used to describe consumed flows (e.g. Anderson et al., 2012; Karimi and Bastiaanssen, 2015). However, substantially less attention is paid to identifying the non-consumed portion of water withdrawal, distinguishing between recoverable and non-recoverable water, and the downstream users that may be relying on recovering return flows.

This chapter explores the processes associated with water reuse in a river basin among users of varying nature and reviews existing methods for directly or indirectly describing non-consumed water, recoverable flow and/or water reuse. Selected indicators are demonstrated through application to the example case of the Arkansas Basin in Colorado, USA. Based on relevant literature, existing research gaps are identified regarding the development of a basin-level framework to assess the fate of non-consumed flow in a cascade of multiple water users.

2.2 Definition of water (re)use - flows and processes

2.2.1 *Definitions of a single water user*

The gross inflow available to a water user consists of the sum of artificially withdrawn and naturally supplied water (Perry, 2011). Two principal types of water users can be distinguished, based on the extent to which they are dependent on natural and artificial water supply. The flow processes associated with these categories of water users, and therefore the options for management interventions, are fundamentally different. Figure 2.1 gives a schematic overview of the typical hydrological flows at water users relying on (1) water withdrawals, and (2) natural inflow. Some of the flows depicted in Figure 2.1 are managed, others are manageable, and some are non-manageable (Karimi et al., 2013a).

Type 1 water users depend on groundwater withdrawals and/or surface water withdrawals, for example with the purpose of domestic use or irrigation in the dry season. Desalinization and inter-basin transfers are other forms of anthropogenic water supply. Type 2 comprises natural systems such as wetlands, lagoons, aquatic ecosystems, groundwater-dependent ecosystems, as well as agriculture that is entirely rainfed. Naturally supplied water is mostly precipitation, but can also include groundwater seepage, interflow, and inundations. A combination of both types, thus a mixture of naturally and humanly governed inflow, is occurring for example for irrigation under conditions of erratic rainfall, or a combination of rainfall and controlled inundations for certain wetlands. The concepts presented in this chapter are focused on users under (or approaching) Type 1 conditions, thus depending on “blue” water (Falkenmark and Rockström, 2006), and associated

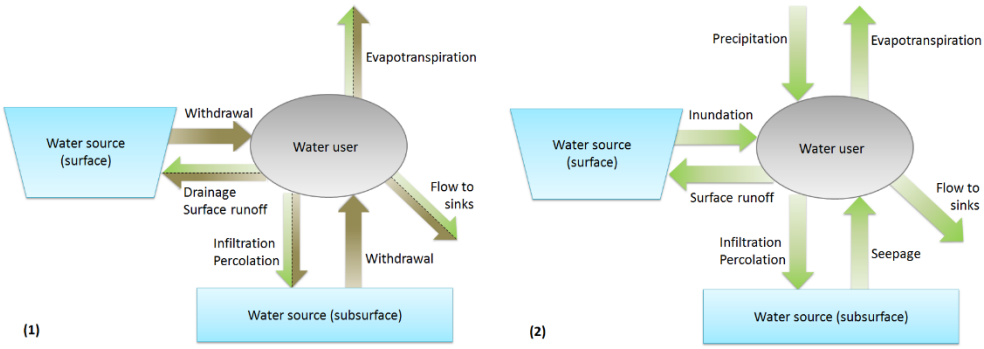

Figure 2.1. Typical hydrological flows associated with water users dependent on (1) water withdrawal and (2) natural inflow. Green arrows indicate flows governed by natural processes; brown arrows indicate flows managed by humans.

with a direct and potentially significant human influence on the hydrological cycle. Mitigation activities to manage supply and demand have the biggest potential for this type of users.

The following equations are used to describe the basic categorization of flow processes to distinguish between consumed and non-consumed water (Frederiksen and Allen, 2011; Perry, 2007):

$$Q_w = Q_c + Q_{nc} \tag{2.1}$$

$$Q_{nc} = Q_r + Q_{nr} \tag{2.2}$$

where Q_w is water withdrawn from surface water or groundwater, Q_c is consumed water, Q_{nc} is non-consumed water, Q_r is recoverable water, and Q_{nr} is non-recoverable water.

Consumed water is defined as the water that is removed from surface water or groundwater systems and that is no longer available for downstream users. It consists mainly of evapotranspiration, but in specific situations also includes water incorporated in agricultural or industrial products and drinking water for humans and livestock. Recoverable water is withdrawn but feeds back into the hydrological system and is available for capture and reuse downstream. Non-recoverable water flows towards deep aquifers that are unprofitable to exploit, oceans, or other saline bodies, and is therefore unavailable for downstream reuse. An especially complex issue is the potential deterioration of water quality by point source and nonpoint source pollution. Whether pollution levels indeed cause water to become non-recoverable depends on water quality requirements of the downstream users. Non-recoverable water from a water user may in turn cause other water bodies to become non-recoverable. Additional factors that may cause water to become non-recoverable are salinized soils or heating in industrial processes. All water that is not consumed in the process of withdrawal, is denoted by the term non-consumed

water, which is synonymous to return flow. A full glossary of all water balance terms is provided in Appendix A.

Ratios between consumed and non-consumed water are typically determined by the nature of the user. Agricultural water withdrawals are known to have large proportions of non-consumed flows with irrigation efficiencies (consumed water divided by withdrawals) typically between 30% and 70% (Bos and Nugteren, 1990; Brouwer et al., 1989; Perry, 2007). The irrigation efficiency of sprinkler and drip systems can be as high as 70 to 95%. Irrigation technique, drainage infrastructure, crop type, soil and topography all affect the irrigation efficiency. Water withdrawals for livestock are typically largely consumed, as cattle drinking water. On the global scale, consumptive use in livestock is an agricultural water use of secondary importance (Wada, 2013) although it should be noted that the livestock sector is a principal water user in countries like Botswana (FAO, 2006). Industrial water consumption varies greatly depending on the type of industry, the nature of the water supply, technological processes, and climatic conditions, but is usually an insignificant fraction of water intake. A primary application is cooling water for thermal and nuclear power stations. Other significant industrial water users are the chemical, metallurgy, and paper industries (Shiklomanov, 2000). Domestic water withdrawals are made by municipal services and private homes. Consumptive losses occur from evaporation of the water used by municipalities for plants, streets, recreation zones, and personal gardens, with drinking water for private homes being insignificant (Shiklomanov, 2000). Other noteworthy, largely non-consumptive, sectors are hydropower generation, mining, and fisheries.

2.2.2 *Hydrological connectivity between multiple water users and its impacts*

The concept of hydrological connectivity relates to the ease with which water can move across the landscape in different ways (Lexartza-Artza and Wainwright, 2009), and is determined by factors such as topography, geology, soil type, presence of water ways and hydraulic infrastructure. In developed areas, non-consumed water is generally transported by either sewerage systems or agricultural surface drainage systems, ultimately to end up in a river where a next user may tap into. Subsurface drainage removes excess water through conduits, deep open drains and wells, feeding streams through piped outlets. Water infiltrated through the soil profile into the groundwater recharges the aquifer, and this water may again contribute to the river base flow or be put to (often ecological) use in downstream groundwater seepage zones. Non-consumed water that was initially recoverable may become non-recoverable along its flow path, for example due to injection with pollutants or leakage of water through faults systems to deep geological formations. Similarly, non-recoverable water may become recoverable due to dilution by rainstorms.

Hydrological connectivity is a broad term for which many different definitions have been developed over the years (Bracken et al., 2013). In the context of reuse, it is important to not only account for the spatial aspects of connectivity but to also

acknowledge the temporal component. It is relevant to know whether a user relies on water directly from a river, recoverable water from upstream users delayed through canals and drains, or recaptured groundwater from an aquifer. Among these cases there is a substantial difference in timing of water delivery to the downstream user, up to several orders of magnitude. Depending on regional climatic and hydrological conditions, shifts in the existing timing of water supply may be detrimental to the purpose of the downstream user (King, 2008; Lankford, 2006).

The recoverable flow from a water user may be reused once or multiple times by downstream water users. A schematic impression of a cascading system of water users within a river basin is presented in . The figure depicts three water users connected by either natural waterways such as a network of streams and/or aquifers, or artificial flow paths such as canals, subsurface drains and sewerage. To satisfy its water demand, the downstream water user is dependent on the non-consumed flow from the upstream user. Decreasing non-consumed flow from A and B by reducing withdrawals and/or increasing consumption, as commonly happens when irrigation technology or management is improved (Contor and Taylor, 2013), may be detrimental to water availability for the respective downstream users. Assuming C as the final user before water leaves the basin and flows out into the sea, saving water here will actually free up water. However, it could be inappropriate to consider all outflow from C as available for further development, as a certain level of reserved flow may exist that provides ecological benefits or prevents saltwater intrusion. It should be noted that is simplified to illustrate the concept of water reuse and hydrological connections. In reality, B and C will likely have other sources of inflow in addition to non-consumed flow from A, and there could be a portion of non-recoverable outflow from each user.

The above description shows that one should be very careful when identifying water savings and water losses. Knowledge of water reuse is a necessity for proper decision-making. On the scale of the river basin, water can only be truly saved by reducing consumptive use or return flows that are not reused (Allen et al., 2005; Seckler et al., 2003). Various authors even warn for an increase in basin-wide water consumption when local water conservation measures disregard the hydrological setting of water users (Ahmad et al., 2013; Ward and Pulido-Velazquez, 2008). Non-consumed flows that are reused downstream should only be reduced when somehow a more valuable purpose is designated to the upstream user. This approach to water reuse is now widely acknowledged, and has triggered a questioning of the efficiency concepts so often utilized in irrigation accounting (Jensen, 2007; Perry, 2007; Seckler et al., 2003). Optimizing the ratio between consumed and diverted water (e.g., an increase from 40% to 50% for A in Figure 2) may be desirable at the local level, but it is crucial to realize the implications for downstream water users. This demonstrates the need to quantitatively express basin-wide reuse processes.

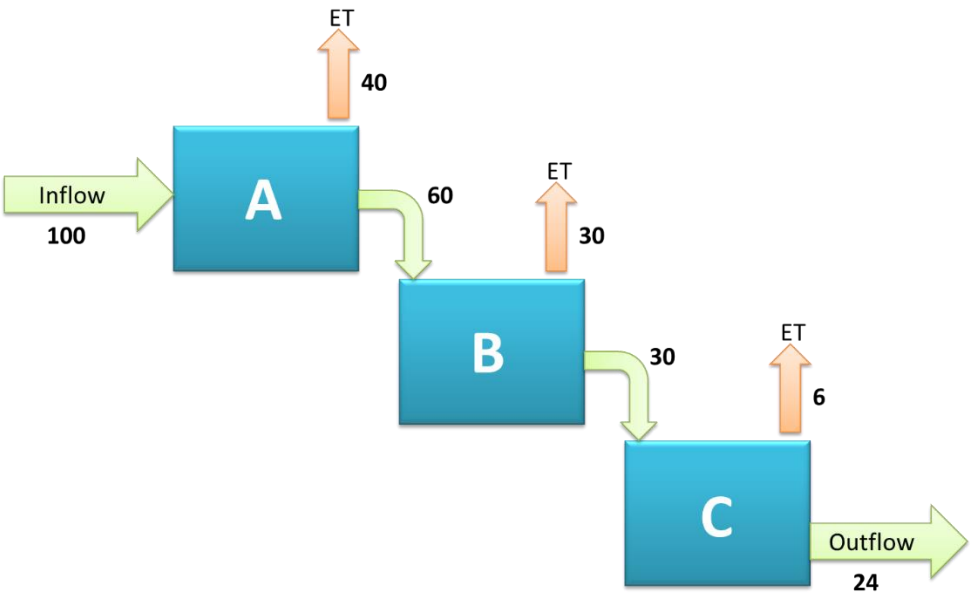

Figure 2.2. Hypothetical cascade of different types of water users (A, B, C) in a river basin that all have a different Consumed Fraction.

It should be noted that benefits other than water saving could be achieved by increased efficiencies, such as improved upstream production, reduced pumping costs (although e.g. switching from surface irrigation to pressured systems may in fact increase energy costs), increase in-stream flow and ecological health, and improved downstream water quality (Clemmens et al., 2008; Gleick et al., 2011). Elaborate discussion of such co-benefits is outside the scope of this chapter.

Molden and Bos (2005) explored how systems of cascading water users typically develop when water demand exceeds supply, triggering a response of construction of hydraulic infrastructure to facilitate reliable water delivery to and drainage from the water users under stress. Water scarcity is conventionally regarded as the main driver for water reuse, however the variability of water supply is also an important factor (Hermanowicz, 2006). The reuse of agricultural drainage water is one of several non-conventional sources of water to alleviate water scarcity in arid countries (Qadir et al., 2007). It is a popular way of optimizing the usage of the available water supply, and at the same time disposing of drainage water. A famous example is Egypt's Nile Delta, where reuse of drainage water reduces the irrigation water requirements by 20% (Barnes, 2014). Serial Biological Concentration, effectively applied in California and Australia (Ayars, 2007), provides a framework for integrated water and salt management by sequential reuse of drainage water on successively more salt-tolerant crops. Engineering projects such as reservoir storage and conjunctive surface and groundwater use can potentially increase the portion of non-consumed water that can be reused. The same effect can be achieved by treatment of industrial water, if technologically and economically feasible.

An increased complexity and intensity of the network of water use and reuse means that changes in quantity, quality and timing of flows will have greater implications in closing river basins than in basins where water is abundant. Natural and anthropogenic reuse thus ought to be quantitatively understood.

2.3 Analytical description of (non-)consumed water and reuse

The use of hydrological fractions is widely acknowledged as a comprehensive and objective way of quantifying all inflows and outflows associated with water withdrawals (Allen et al., 2005; Karimi et al., 2013a; Perry, 1999; Willardson et al., 1994). The concept consists solely of quantitative terms that are consistent with the fundamental principles of hydrology. Water use indicators can be defined from the basic fractions (e.g. Pereira et al., 2012). This section presents an analytical framework for a quantitative analysis of (non-)consumptive use based on hydrological fractions. The framework holds for indirect water reuse, where non-consumed flows are discharged back into the system. To illustrate the impact of multiple reuse cycles on the overall scope for water saving, the simplified cascade approach from is used below in several example calculations.

Building on equations 1 and 2, the following hydrological fractions can be defined:

$$CF = \frac{Q_c}{Q_w} \tag{2.3}$$

$$NCF = \frac{Q_{nc}}{Q_w} \tag{2.4}$$

$$RF = \frac{Q_r}{Q_w} \tag{2.5}$$

$$NRF = \frac{Q_{nr}}{Q_w} \tag{2.6}$$

where *CF* is the consumed fraction, *NCF* is the non-consumed fraction, *RF* is the recoverable fraction and *NRF* is the non-recoverable fraction.

Based on the recoverable fractions of upstream users, it is possible to determine the recoverable flow that arrives at a certain location in a cascade of interconnected water users. We derive the following equation for a hypothetical system in which all of Q_{nc} is reused, with three different *RF* values occurring in the system:

$$Q_r = Q_{w\_init} * {RF_x}^{n_x} * {RF_y}^{n_y} * {RF_z}^{n_z} \tag{2.7}$$

where Q_{w_init} is the withdrawal by the first user in the cascade, RF_x, RF_y and RF_z are different values of the Recoverable Fraction, and n_x, n_y and n_z are the number of

users in the system with RF_x, RF_y and RF_z, respectively. For the simple case of Figure 2.2 , Equation 2.7 can be solved to compute recoverable flow at user C as follows:

$$Q_r = 100 * 0.6^1 * 100 * 0.5^1 = 30.$$

The concept of Equation 2.7 can be adjusted for the amount of different *RF* values in a cascade. In the case all water users have the same *RF*, or in the case of local recycling, the equation amounts to:

$$Q_r = \frac{(Q_{w_{init}} RF_x)^n}{Q_{w_{init}}{}^{n-1}} \tag{2.8}$$

or:

$$Q_r = Q_{w_{init}} RF^n \tag{2.9}$$

Based on a Q_{w_init} of 100 units, Figure 2.3 explores how Equations 2.7 and 2.9 dictate that Q_r decreases as the number of withdrawals increases along a flow path. For demonstration purposes, the system is simplified to consist of users with a single *RF*. Four scenarios were selected (0.9, 0.75, 0.5 and 0.25). These values can be seen as representative of different types of water users. An *RF* of 0.9 is typically representative of an industrial water user where most withdrawals return to the hydrological system (Wada et al., 2011). An *RF* of 0.75 is a plausible value for domestic withdrawals, as not all households are connected to a sewage system and people and animals consume water by respiration. An *RF* of 0.5 typically holds for the irrigation sector, and an *RF* of 0.25 could be found in greenhouses where return flow is small (and sometimes even 0 when all water is recycled internally). Figure 2.3 demonstrates that after 5 to 6 reuse cycles, hardly any recoverable water will be left in a chain of water users with an *RF* of 0.5 or lower.

To put the portion of recoverable water into perspective of total non-consumed water, it is meaningful to express Q_r as a fraction of Q_{nc} as follows:

$$RE = \frac{Q_r}{Q_{nc}} \tag{2.10}$$

$$RE = \frac{RF}{NCF} \tag{2.11}$$

with *RE* named Return Flow Efficiency by King (2008) and Recycling Efficiency by Wallace and Gregory (2002). For the sake of consistency with other terminology used in this research, we propose to utilize the term Reuse Efficiency for *RE*. An *RE* of 1 means that *RF* = *NCF* and all non-consumed water is recoverable downstream.

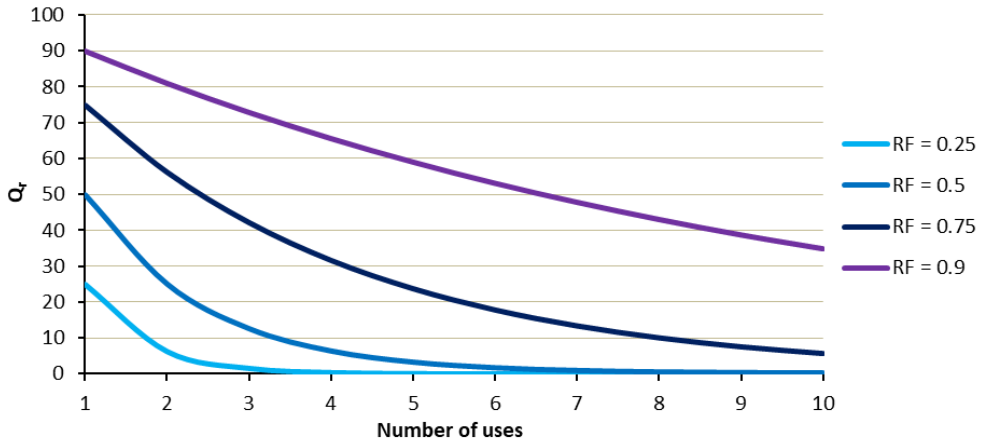

Figure 2.3. Decrease of recoverable flow (Q_r) with rate of reuse, for different Recoverable Fraction (*RF*) values.

Irrespective of whether the water user is primarily consumptive or non-consumptive, a high value for *RE* is a desirable situation.

Figure 2.4 displays the *CF* of a cascade of water users for different values of RF and *RE*. The *CF* value of the individual water users was chosen to be 0.4, with 40% described as a "reasonable" value for scheme irrigation efficiency (Brouwer et al., 1989). A range of *RE* values is given, with the corresponding *RF* under the given *CF* of 0.4. The figure, modeled after an earlier analysis performed by Wallace and Gregory (2002) shows that the system-scale *CF* value does increase with water reuse, but that the recoverable fraction is key in determining the extent to which *CF* increases with scale. Depending on *RF*, a maximum value for *CF* is approached after roughly a number of 2 to 6 uses. Even when *RF* is high, water must be reused a substantial number of times before a value of 70-80% can be achieved.

Further sub-division of non-consumed water into more specific fractions provides additional value for identifying water management options. King (2008) performed several analyses that highlighted the role of groundwater recharge through deep percolation for downstream reuse. Due to the significant differences in processes that govern transport of recoverable water to and through the groundwater as opposed to surface water, added value lies in the distinction of two separate fractions:

$$RF_{sw} = \frac{Q_{r_{sw}}}{Q_r} \tag{2.12}$$

and

$$RF_{gw} = \frac{Q_{r_{gw}}}{Q_r} \tag{2.13}$$

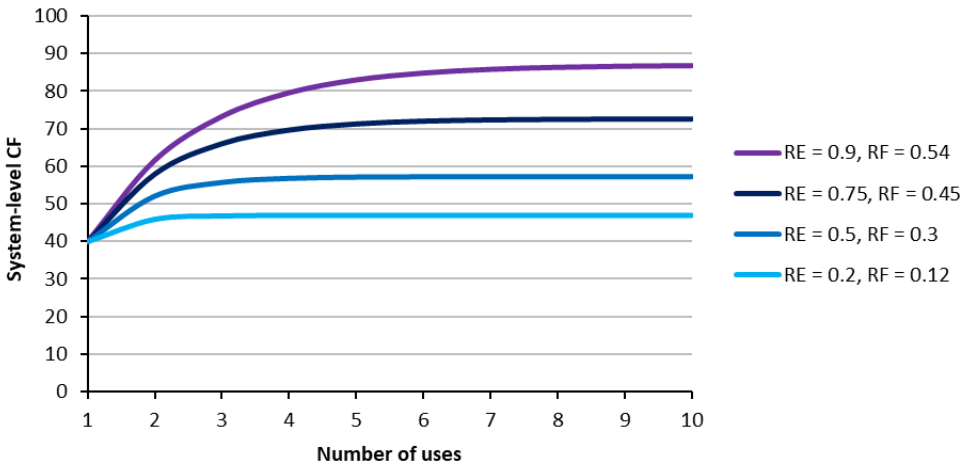

Figure 2.4. Increase of system Consumed Fraction (*CF*) with rate of reuse, for different Reuse Efficiency (*RE*) values and the corresponding Recoverable Fraction (*RF*). The *CF* of individual users is assumed at 0.4.

where Q_{rsw} and Q_{rgw} are the portions of recoverable water that contribute to surface water and groundwater recharge respectively, RF_{sw} is the fraction of recoverable water feeding into surface water and RF_{gw} is the fraction of recoverable water contributing to groundwater recharge. High values for RF_{gw} may indicate a more complex reuse system and greater uncertainty of the time scale associated with recharge, transport and downstream recovery.

Even when focusing solely on deliberate withdrawals, return flow can be discharged through both anthropogenic and natural pathways. Knowledge on whether recoverable flow is driven by natural or artificial processes gives insight in the opportunities for spatiotemporal management of this flow. This differentiation can be described as follows:

$$Q_r = Q_{r_a} + Q_{r_n} \tag{2.14}$$

$$RF_a = \frac{Q_{r_a}}{Q_r} \tag{2.15}$$

$$RF_n = \frac{Q_{r_n}}{Q_r} \tag{2.16}$$

where Q_{ra} is the flow governed by man-made infrastructure such as canals, drains and sewerage, and Q_{rn} is the flow discharged through natural processes such as unmanaged surface runoff, infiltration, and percolation. RF_a and RF_n are the anthropogenic and natural fractions respectively. A high RF_a value indicates more direct opportunities for management interventions.

An overview of all relevant hydrological fractions is given in Figure 2.5.

Whether non-consumed water is reused by a downstream user depends on many factors, such as the profitability of recovery (e.g., pumping from a deep aquifer), the time frame in which the water arrives downstream the geographic location of streams and aquifers, and the quality levels and composition of water. These factors are highly dependent on local conditions and impede standardized definitions of recoverable and non-recoverable water. One should therefore be cautious in determining the recoverable and non-recoverable fractions in the early stages of an analysis. This approach is in line with Lankford (2012), who stated that the general assumption of the world's recoverable fraction being actually reused downstream is in fact an oversimplification.

The presented framework of fractions for analyzing non-consumed and recoverable flows deliberately avoids subjective terms such as the distinction between beneficial and non-beneficial water consumption, and productivity of consumed water. As shown by Boelens and Vos (2012) and Frederiksen et al. (2012), views on what is regarded as beneficial or productive water consumption will differ among the various stakeholders in a river basin, implying that this should be left out of a basic framework for physical accounting. Similarly, the effect of pollutants on water reuse such as incorporated in the concept of effective efficiency Haie and Keller (2008) are left out of the basic definitions as put forward in this chapter. In this way, no dependence on the type of the pollutant, or the nature of downstream water reuse, is introduced in the basic concept (Haie, 2008; Perry, 2008).

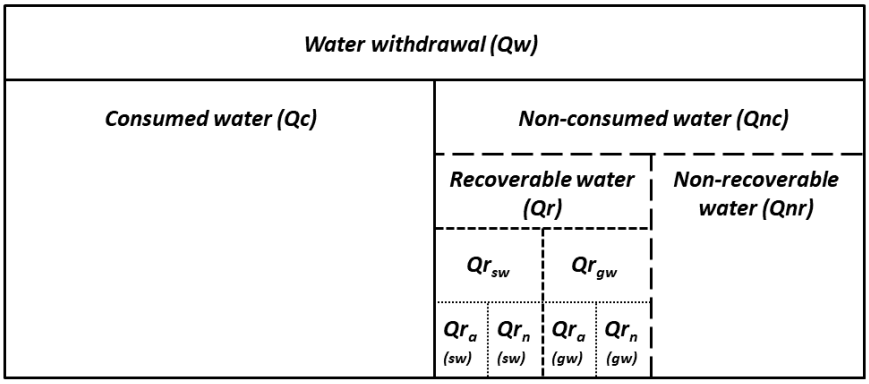

Figure 2.5. Categorization of the relevant flows for assessing water reuse. $Q_{r_{sw}}$ and $Q_{r_{gw}}$ are volumes of recoverable water recharging surface water and groundwater, respectively; Q_{r_a} and Q_{r_n} are the respective volumes of recoverable water discharged by anthropogenic and natural processes.

2.4 Selection and demonstration of relevant methods and indicators

This section explores a number of selected literature studies that present and apply methods and indicators to assess water reuse and/or its impacts. Although their approaches differ, these studies all express the interaction between different water users by means of a certain methodology, and thus go further than only accounting for conditions at the level of a single user. The applicability of the indicators is demonstrated for the water reuse system in the Arkansas Basin, Colorado. A synthesis of the review is given in Table 2.1.

2.4.1 *Water reuse indicators from literature*

Basin-level water accounting with multiple users

The International Water Management Institute (IWMI) developed a framework of irrigation accounting based on the concept of hydrological fractions (Molden et al., 2001a). More recently, this framework has been expanded to fit the requirements of a basin-wide analysis of a variety of water users of different natures (Karimi et al. (2013a), *www.wateraccounting.org*). Examples of published water accounting studies are abundant, e.g. (Bastiaanssen and Chandrapala, 2003; Harrington et al., 2009; Karimi et al., 2013b; Karimov et al., 2012; Shilpakar et al., 2011). As in water accounting hydrological fractions are quantified on the (sub)basin scale; an entire water use and reuse network is reflected in the values of these indicators.

For example, Karimi et al. (2013b) used water accounting for the Indus Basin to determine that 20% of water withdrawals is recoverable for reuse, computing *RF* for the entire basin at 0.2. Furthermore, they estimated that on average water is reused 4 times in the basin based on the discrepancy between computed basin-level *CF* in agriculture and literature values for field-scale irrigation efficiency. The basin *CF* itself, alternatively named Depleted Fraction or Basin Efficiency (Seckler et al., 2003), indirectly holds information on the occurrence of water reuse (as illustrated in Figure 4). El-Agha et al. (2011) studied how drainage water reuse in the Nile Delta irrigation schemes is reflected in system-level *CF* values. For several branch canal command areas, monthly system *CF* values are substantially affected by the reuse of drainage water. Correctly interpreting high system *CF* values is however a complex exercise, as these are also typically found for unsustainable systems.

Water reuse as a function of scale

Hafeez et al. (2007) conducted a study primarily aimed at quantifying water reuse at different spatial scales in a rice-based irrigation system in the Philippines. Their goal was to test the hypothesis of scale-dependent efficiencies due to water reuse, using the water accounting approach on different spatial scales within an irrigated rice system. The authors integrated an extensive set of field measurements of surface water flows, pumping records, and groundwater depths, and remotely sensed ET to perform a multi-scale water accounting study. Linear regression was applied, and the rate of reuse was expressed in m^3 per additional unit of surface

area under consideration. Based on measurements of reuse of pumped groundwater and surface water through check dams, reuse of surface water and reuse of groundwater were evaluated separately. The reuse of surface water was found to increase linearly with 4.6 x $10^6 m^3$ per 1000 ha, with the farmers using pumps for either complete or supplemental irrigation causing an increase in (re)use of water through pumping by 1.3 x $10^6 m^3$ per 1000 ha.

Expressing this approach in a linear regression equation yields:

$$y_{sw} = Q_{w_{sw},ha}\, x + B_{sw} \tag{2.17}$$

$$y_{gw} = Q_{w_{gw},ha}\, x + B_{gw} \tag{2.18}$$

where y_{sw} and y_{gw} are the volumes of reused surface water and groundwater respectively, $Q_{w_{sw,ha}}$ comprises use of water through check dams per ha, $Q_{w_{gw\_ha}}$ represents groundwater withdrawals per ha, and B a residual term close to 0. As percolation is found to be higher than groundwater withdrawals, all groundwater withdrawals are envisaged as reuse.

The concept of plotting reuse against area could potentially be upscaled to the river basin level. A steeper slope would then be expected for closing river basins. The slope of the resulting curve could thus serve as an indicator of water reuse, although the extent to which meaningful relations can be found for heterogeneous basins remains open for further research.

Water Reuse Index

The Water Reuse Index was developed by Vörösmarty et al. (2000) and was adopted by the United Nations in their World Water Development Reports and the SEEAW water accounting framework (UN, 2012). The Water Reuse Index at a location (*x,y*) is computed by dividing the aggregate of upstream water (domestic, industrial, and agricultural) withdrawals $Q_{w_{upstream}}$ by the mean annual surface and subsurface runoff (Q_{xy}) at that location:

$$WRI_{xy} = \frac{\sum Q_{w_{upstream}}}{Q_{xy}} \tag{2.19}$$

As such, the index reflects the consecutive times that water is withdrawn during its passage downstream. The Water Reuse Index can be computed for any point in a basin and typically increases toward the basin mouth, representing a progressive increase in reuse of runoff. A high value at the basin level is an indication of intensive competition for water resources among users. However, whether the upstream users are primarily consumptive or non-consumptive is not taken into consideration.

Vörösmarty et al. (2005) showed how plots of *WRI* against downstream distance indicate locations where a river encounters significant withdrawals, represented by high *WRI* values, and impacts of little water use and presence of runoff and tributary inputs in the form of low *WRI* values. Near the outflow point of the Nile River, a *WRI* of approximately 1 was found under mean annual flow conditions, indicating that the accumulated upstream water withdrawals are almost equal to mean annual discharge. They also evaluated the sensitivity of *WRI* to climate variability. For a river such as the Orange River, located entirely in a semi-arid region, a dramatic reaction to drought conditions was found, with *WRI* rising an order of magnitude.

Return flow ratio

In the Aqueduct Global Risk Atlas of the World Resources Institute, the term Return Flow Ratio (*RFR*) is calculated for a catchment, user, or location, as the amount of upstream non-consumed water divided by the available surface water (Gassert et al., 2013). The *RFR* global base maps were computed using average values of available blue water (available surface water minus upstream consumptive use) over the period 1950 - 2008 and are publicly available[2]. Although it is mainly discussed in the context of potential water quality risks, *RFR* is essentially a quantitative term, indicating dependency on water that was previously applied upstream but not consumed. In generalized terms, *RFR* is defined as:

$$RFR_{x,y} = \frac{\sum Q_{nc_{us}}}{Q_{sw_{xy}}} \tag{2.20}$$

where $Q_{nc_{us}}$ is upstream non-consumed water and $Q_{sw_{xy}}$ is surface runoff at the location, user, or catchment under consideration. Note that the difference between eq. (2.16) and eq. (2.20) is the accumulated consumptive use in case groundwater can be ignored.

The *RFR* map for the state of California shows a general southward increasing gradient of *RFR*, along a combination of natural waterways (e.g., the Sacramento River) and reservoirs, canals and aqueducts associated with large artificial projects such as the California State Water Project and Central Valley Project. Very high values of >80% occur at major urban centers such as Los Angeles, San Jose, and San Francisco, indicating a high dependency of these areas on water that was previously used upstream.

It should be noted that the term Return Flow Ratio is also used to evaluating irrigation and drainage systems; in this context it is defined as the amount of water appearing in a drain divided by water supply to the scheme (Masashi et al., 2013). This definition is equal to the Non-Consumed Fraction.

[2] (*http://www.wri.org/resources/maps/aqueduct-water-risk-atlas*, retrieved 01-Jul-2015)

Degree of return flow reuse

Chinh (2012) defined a set of indicators for a reuse system, particularly in the context of irrigation and drainage. However, their concepts apply in principle to different scales and different water users. The degree of return flow reuse is a parameter that indicates the fraction of recoverable water that is actually reused in the catchment. For specific irrigated conditions with clearly defined source and reuse schemes, and both an internal and external drain that collect drainage water that is potentially reused, it is defined as follows:

$$DRR = \frac{x_D Q_{P,D} + x_D x_E Q_{P,E}}{D_{cs}} \tag{2.21}$$

where x_D is the mixing ratio between surface drainage from a source scheme and total flow into a drain, x_E is the mixing ratio of catchment drainage water with external water sources, $Q_{P,D}$ and $Q_{P,E}$ are pumped volumes by internal and external reuse stations respectively, and D_{cs} is surface drainage from the source scheme.

The conceptual model can apply to a generalized water reuse system, irrespective of type of upstream water use (the "source scheme"), downstream water use ("reuse scheme"), or pathway between them ("drain"). In case of multiple instances of reuse, a sequence of mixing ratios can be included. The concept acknowledges the presence of different destinations of the recoverable flow, allowing for a distinction between surface water and groundwater as follows:

$$DRR = \frac{x_{sw}\, Q_{w\_sw_{ds}} + x_{gw}\, Q_{w\_gw_{ds}}}{Q_{nc}} \tag{2.22}$$

where x_{sw} is the mixing ratio of non-consumed water from a user with surface water, $Q_{w\_sw_{ds}}$ is surface water withdrawal downstream, x_{gw} is the mixing ratio of non-consumed water with groundwater, $Q_{w\_gw_{ds}}$ is groundwater withdrawal downstream, and Q_{nc} is the non-consumed water from the user under consideration.

Reuse Dependency

The fraction of gross inflow to a water user that is dependent on reuse of upstream non-consumed water is expressed as the reuse dependency *RD*, originally termed "dependency of reuse schemes" Chinh (2012). Consistent with the terminology used in this chapter, *RD* is expressed as:

$$RD = \frac{DRR_{us}\, Q_{nc_{us}}}{P + Q_w} \tag{2.23}$$

or alternatively:

$$RD = \frac{x_{us}\, Q_w}{P + Q_w} \tag{2.24}$$

where x_{us} is the mixing ratio of upstream users of non-consumed water with the water source, and P is supply of precipitation to the user. The reuse dependency relates the portion of withdrawal that is provided by non-consumed recoverable water to the gross inflow. Reuse dependency may increase in case of either a higher mixing ratio, an increase in withdrawals, or a decrease of rainfall. Similar to *DRR*, this indicator gives a direct assessment of water reuse, but requires a large amount of input data.

Water Saving Efficiency

Törnqvist and Jarsjö (2012) applied a calibrated distributed hydrological model to quantify the effect of reuse of return flows on potential for basin-wide water savings in the Amu Darya and Syr Darya basins, Central Asia. They found that the basin-scale water savings are approximately 60% lower than corresponding on-farm reductions in irrigation water application, since water is reused and, hence, return flows decrease when less water is applied.

To express this effectiveness of water saving measures in an indicator, the Water Saving Efficiency (*WSE*) is introduced. *WSE* is defined as the ratio between the increase in river discharge, and the reduction in on-farm irrigation water application that caused this increase in inflow. Or, in more general terms:

$$WSE = \frac{Q_{sw\_downstream_{new}} - Q_{sw\_downstream_{old}}}{Q_{w_{old}} - Q_{w_{new}}} \tag{2.25}$$

where $Q_{sw\_downstream_{new}}$ is surface runoff at a certain downstream point after implementation of the water saving measure, $Q_{sw\_downstream_{old}}$ is downstream runoff before implementation, Q_{wold} is water withdrawal before implementation, and Q_{wnew} is water withdrawal after implementation.

Thus, a *WSE* of 1 would indicate that no water reuse occurs before the drainage water returns to its source. In an illustrative example of applying *WSE*, Törnqvist and Jarsjö (2012) find the largest differences between the downstream part of the Amu Darya basin (~0.8) and the upstream part of the Syr Darya basin (~0.15). In terms of water savings, it would therefore be much more efficient to implement improvements in the irrigation system in the Amu Darya delta.

The WSE concept offers a parameter that can be mapped continuously in space and which provides a direct indication of the effectiveness of water saving measures. Thereby, it informs on the extent to which users downstream of a certain water use depend on recapturing its non-consumed flow.

Table 2.1. Review of selected concepts and indicators for assessing water reuse.

Concept	Key references	Definition	Input data required	Main advantages	Main limitations
Basin-level recoverable fraction (*RF*)	(Karimi et al., 2013a, 2013b)	The portion of water withdrawals that is not consumed and can be recovered for reuse downstream	Water withdrawals, recoverable water	Gives insight into basin-level scope for water reuse and enables basin inter-comparisons	Results relate to a black box situation: no information is presented on what happens within
Basin-level consumed fraction / depleted fraction (*CF*)	(El-Agha et al., 2011)	The portion of system inflow that is consumed	Consumed water, rainfall, water supply from outside the system	Gives insight into basin-level consumption, sustainability and enables basin inter-comparisons	Ambiguous meaning of high values: an efficient system, or an unsustainable system?
Linear regression of water reuse to scale: withdrawals per ha	(Hafeez et al., 2007)	The amount of water that is reused per additional unit of surface area	Surface water withdrawals, deep percolation, groundwater pumping, surface area per user	Disaggrega-tion of spatial units allows for assessing the effect of water reuse on system-level efficiency	Questionable applicability to hetero-geneous systems, requires a lot of input data
Water reuse index (*WRI*)	(UN, 2012; Vörösmarty et al., 2005, 2000)	A measure of the number of times water is withdrawn consecutively during its passage downstream	Surface and shallow aquifer runoff, upstream water withdrawals	Requires little input data	Does not distinguish between consumed and non-consumed water
Return flow ratio (*RFR*)	(Gassert et al., 2013)	The portion of available water previously used and discharged upstream as wastewater	Surface runoff, upstream non-consumed water	Distinguishes consumed and non-consumed water from withdrawals, requires little input data	Does not include a distinction between recoverable and non-recoverable water

Table 2.1. (Continued).

Concept	Key references	Definition	Input data required	Main advantages	Main limitations
Degree of return flow reuse (*DRR*)	(Chinh, 2012)	The fraction of drainage water that is reused in the catchment	Non-consumed water, flow in external water sources, downstream withdrawals	A direct description of downstream dependency on a user's non-consumed water	Requires a lot of input data
Reuse dependency (*RD*)	(Chinh, 2012)	The fraction of the water supply of reuse areas that is covered by drainage reuse.	Degree of return flow reuse, upstream non-consumed water, gross inflow	A direct description of water reuse by a certain user	Requires a lot of input data
Water saving efficiency (*WSE*)	(Törnqvist and Jarsjö, 2012)	The ratio between the increase in river discharge and reduction in on-farm irrigation water application	Withdrawals, downstream water supply, future withdrawals, predicted downstream water supply	Assesses the effectiveness of water saving measures	Requires hydrological modeling of future conditions, introducing uncertainties
Downstream-ness	(van Oel et al., 2011, 2009)	Ratio between upstream area and the total area of the river basin	Upstream surface area, total basin surface area	Can be applied to many basin properties, little input data needed	The basic concept does not include quantitative flows

Downstreamness

The concept of downstreamness, introduced by van Oel et al. (2009), defines a function in a river basin based on the area of its upstream catchment. In this way, downstreamness is valuable in raising awareness of the spatial context of water supply to a location, and in evaluating a certain location based on its upstream commitments. The approach allows for studying the process of closure in the sub-basin upstream from any point in a river basin, thus providing a framework for analysis of spatial hydrological flows at the level of individual water users or other geographical units.

The downstreamness (D_x) of a location is defined as the ratio between the area of upstream catchment and total basin surface area. Thus, with increasing D_x, larger

natural runoff is expected. Measured or modeled surface runoff at a certain location can be compared to the expected linear relation between D_x and runoff, with a substantial deviation from this line being an indication of basin closure in the catchment of the measuring location. Closure may be caused by storage in reservoirs or by upstream water withdrawals for consumptive use.

The downstreamness of a function in the basin (e.g., water availability or water use) is defined as the downstreamness-weighted integral of that function divided by its regular integral. For example, the comparison between D_x of storage capacity and D_x of actual stored volume was proposed by van Oel et al. (2011) as an indicator of closure of the (sub-)basin that supplies the location under consideration. An analysis of downstreamness is useful to evaluate water reuse and the vulnerability of a type of water use. Taken from van Oel et al. (2011), for a basin with n geographical units:

$$D_{wd} = \frac{\sum_{x=1}^{n} WD_x D_x}{\sum_{x=1}^{n} D_x} \tag{2.26}$$

where D_x = downstreamness of water demand at location x; and WD_x = water demand of location x. D_{wd} is a measure of how far downstream water demand in the basin is located on average and can therefore be viewed as a proxy for water reuse. When a high value for D_{wd} is found for a type of water use, this could indicate a larger dependence on recoverable water from upstream users.

2.4.2 *Example application: the Arkansas Basin*

To demonstrate the type of information provided by the selected water reuse indicators, we have computed their values for the Arkansas Basin in Colorado, USA. Data from the Draft Basin Implementation Plan (DBIP, WestWater, 2014) was used as the basis for this analysis. The DBIP lists the different users that are present in the Arkansas River Basin, with their main water sources, specific withdrawals for different years, and typical return flows for the agricultural and industrial users. For demonstration purposes, our analysis focuses on users that are at least partly consumptive and are connected through withdrawing from and discharging to the main stem of the Arkansas River. Some other, minor users exist in the area that rely on groundwater pumping, however regulations prescribe that their return flows recharge the same aquifer rather than discharge to the surface water system (WestWater, 2014). Interbasin water supply projects are disregarded in our analysis. It is beyond the scope of this chapter to examine the hydrological conditions in the Arkansas Basin in detail, as the aim of this exercise is merely to illustrate the applicability of water reuse indicators.

Table 2.2 presents the relevant properties of the selected water users, as well as river flows (available water) and indicator values. All figures in Table 2.2 are valid for 2010, an average year in terms of rainfall (WestWater, 2014). Withdrawals and return flows of agricultural and industrial users were taken from the DBIP. For

municipal water users, return flow values are not included in the DBIP, and an *NCF* of 40% was assumed based on typical municipal return flows in the Colorado River basin (Cohen and Martin, 2011). It is assumed that on a yearly time scale, all return flows from the listed users re-enter the Arkansas River at the point of withdrawal, and that they are entirely recoverable for downstream users (*RE* = 1). Available water presented in Table 2.2 was measured by the upstream gauge nearest to the respective user. Gauged tributaries, intermediate withdrawals and return flows were used to provide river flow estimates for users lacking a flow gauge directly upstream. Downstreamness was computed based on sub-basin delineation derived from SRTM elevation data (USGS, 2004). *RD* was calculated relative to withdrawals only (excluding precipitation), and thus indicates the dependency of withdrawals on return flows in an average year. Of the indicators discussed in this chapter, WSE could not be computed as outputs from simulation models are not available. Similarly, water reuse could not be examined as a function of scale, as the surface area of water users is unknown. All other indicators discussed in the previous section are listed in Table 2.2.

Jointly, the reuse indicators provide an insight into the water use cascade along the Arkansas River. The overall recoverable fraction of the system is 0.44. With a D_{wd} of municipal water use of 12.6% and a D_{wd} of agricultural water use of 42.7%, agricultural users are generally located downstream from municipal users. D_x describes the geographical position of each user relative to the basin area. *WRI* and *RFR* generally increase with D_x, as would be expected for most water reuse systems. As most major water users in the area have a similar *NCF*, a large increase in *WRI* often corresponds with a similarly large increase in *RFR*. For the final nine users in the cascade, the sum of upstream water withdrawals exceeds water availability (*WRI* > 1). For the final five users, the sum of upstream return flow exceeds water availability (*RFR* > 1). *RD* values show that the final five users in the cascade rely on return flow for more than 50% of their water withdrawals, thus providing a more direct indication of actual reuse than *WRI* and *RFR*. All three indicators logically rise quickly directly downstream of the discharge of a large volume of return flow, in particular when this coincides with a decrease of river flow (e.g., at Las Animas Consolidated). *DRR* values show that the return flow of all users is being reused in its entirety within the system. The high variance in withdrawal volumes means that a major part of this reuse does not necessarily take place at the subsequent user. The *DRR* of 0 for the Buffalo Canal should be interpreted with caution, as the Colorado-Kansas border was taken as the downstream boundary of the DBIP. River flow has been substantially reduced at this point, but water users are likely still relying on withdrawals from the Arkansas River across the state border.

Table 2.2. Water users in the Arkansas River system and associated values of water reuse indicators (2010). The sequence of listing corresponds with the situation of water users along the main river. *Mun* = municipal, *Ind* = industrial, *Agr* = agricultural.

Water user	Type	Available water (hm³)	Q_w (hm³)	Q_{nc} (hm³)	Q_c (hm³)	*NCF* (-)	*WRI* (-)	*RFR* (-)	*DRR* (-)	*RD* (-)	D_x (-)
City of Salida	*Mun*	335.5	3.7	1.5	2.2	0.40	0.01	0.00	1.00	0.00	0.04
Canon City	*Mun*	608.1	7.2	2.9	4.3	0.40	0.01	0.00	1.00	0.00	0.11
City of Florence + *CF*&I steel	*Mun / Ind*	603.8	60.7	47.3	13.4	0.78	0.02	0.01	1.00	0.01	0.13
Bessemer Ditch	*Agr*	637.7	84.5	36.3	48.2	0.43	0.11	0.08	1.00	0.08	0.16
City of Pueblo	*Mun*	589.5	34.2	13.7	20.5	0.40	0.26	0.15	1.00	0.14	0.16
Comanche Power Plant	*Ind*	688.7	13.1	2.1	11.0	0.16	0.28	0.15	1.00	0.13	0.17
Colorado Canal	*Agr*	732.7	84.7	36.4	48.3	0.43	0.28	0.14	1.00	0.12	0.25
Rocky Ford Highline	*Agr*	707.8	108.7	46.7	61.9	0.43	0.41	0.20	1.00	0.17	0.32
Oxford Farmer's Ditch	*Agr*	645.9	40.0	17.2	22.8	0.43	0.61	0.29	1.00	0.22	0.33
Otero Canal	*Agr*	623.1	8.1	3.5	4.6	0.43	0.70	0.33	1.00	0.25	0.34
Catlin Canal	*Agr*	629.7	118.3	50.9	67.4	0.43	0.71	0.33	1.00	0.25	0.38
Holbrook Canal	*Agr*	562.3	60.1	25.8	34.2	0.43	1.00	0.46	1.00	0.31	0.39
Rocky Ford Ditch	*Agr*	528.0	27.1	11.7	15.5	0.43	1.18	0.54	1.00	0.35	0.39
Fort Lyon Storage Canal	*Agr*	512.6	65.9	28.3	37.5	0.43	1.27	0.58	1.00	0.36	0.39
Fort Lyon Canal	*Agr*	475.0	270.1	116	154	0.43	1.51	0.68	1.00	0.40	0.42
Las Animas Consolidated	*Agr*	321.0	36.3	15.6	20.7	0.43	3.07	1.37	1.00	0.62	0.43
Fort Bent	*Agr*	252.9	23.4	10.1	13.4	0.43	4.04	1.80	1.00	0.75	0.66
Amity Canal	*Agr*	242.8	136.7	58.8	77.9	0.43	4.31	1.92	0.99	0.76	0.67
Lamar Canal	*Agr*	69.1	64.6	27.8	36.8	0.43	17.1	7.60	0.99	1.00	0.67

Table 2.2. (Continued).

Water user	Type	Available water (hm^3)	Q_w (hm^3)	Q_{nc} (hm^3)	Q_c (hm^3)	*NCF* (-)	*WRI* (-)	*RFR* (-)	*DRR* (-)	*RD* (-)	D_x (-)
Buffalo Canal	*Agr*	32.2	31.7	13.6	18.1	0.43	38.7	17.2	0.00	1.00	0.83
System-scale indicators											
Recoverable Fraction (-)											0.44
D_{wd} municipal (%)											12.6
D_{wd} agricultural (%)											42.7

2.5 Discussion

This chapter demonstrates that a systematic, fractions-based approach is instrumental in categorizing basin flows and identifying recoverable water and water "losses". For expressing the occurrence of water reuse in a way that is meaningful for water management decision-making, a methodology is required that considers multiple dimensions to water reuse in a river basin context. Relevant dimensions include the volume of non-consumed water, fraction of recoverable water, spatial hydrological connectivity, travel time, water quality degradation towards the mouth of the river, and hydrological location of a water user.

The reviewed water reuse indicators can be roughly divided into three classes. An overview is provided in Table 2.3. The first class (A) of indicators regards the entity of interest (a system of multiple users, often a (sub-)basin) essentially as a black box. A single value is produced for a delineated geographical area. Examples are the basin-level assessments of the Recoverable Fraction and Consumed Fraction in water accounting, and the withdrawals per hectare as determined by linear

Table 2.3. Classification of reuse indicators.

Class	Description	Indicator
A	Indicators producing a single value for a delineated geographical area with multiple water users	Basin-level Recoverable Fraction (-) Basin-level Consumed Fraction (-) Withdrawals per hectare (m^3/ha)
B	Indicators defining a water user based on upstream flow processes	Water Reuse Index (-) Return Flow Ratio (-) Reuse Dependency (-) Downstreamness (-)
C	Indicators defining a water user based on downstream reuse of its non-consumed water	Degree of Return Flow Reuse (-) Water Saving Efficiency (-)

regression of water reuse on surface area. These concepts provide an indication of the impact of water reuse occurring within the area and time frame under consideration. They are particularly helpful in estimating, for example, the system-level sustainability of water management and the regional potential for water saving. However, no information is presented on local flows and interactions between water users within the study area.

Class B of reuse indicators relates a water user at a particular location in a basin to upstream flow processes, and directly or indirectly describe its dependency on reusing non-consumed water. Downstreamness, *WRI*, *RFR* and *RD* are examples of this approach. D_x of a single water user does not include quantitative flows and its interpretation relies largely on assumptions. This concept is more valuable when applied to more generalized water use properties of a basin, such as storage capacity or a specific type of water demand. The *RFR* is more directly related to reuse than *WRI*, as it accounts for upstream non-consumed water rather than total water withdrawals. Especially when highly consumptive users are present upstream, these two indicators will yield substantially different results. *RD* is the most direct assessment of water reuse, as here actual withdrawals are included rather than total available water. Class B indicators are especially useful when aiming to identify upstream water competition and possibly basin closure, and to assess the vulnerability of a water user to changes in upstream conditions. As their input data requirements are highly different, which indicator to use will largely depend on the information that is available.

Class C of reuse indicators define a water user based on the downstream reuse of its non-consumed water. Examples are *DRR* and *WSE*. These indicators give an indication of the likely effects of changes in flows. A user with a high *DRR* value (and thus a low *WSE*) plays an important role in the water reuse cascade and should therefore not be a target of water saving measures. The value of these indicators lies in their direct link to water management interventions. They can be used to determine, or supplement, a distinction of different water management zones in a river basin. Drawbacks could be the large amount of input data needed to quantify *DRR* and the extra uncertainties introduced in *WSE* due to the need for simulation modeling.

The reviewed indicators offer a range of options for investigating water reuse on a variety of spatial scales, including the individual water user. Input information on a high spatial resolution to feed such analyses is increasingly available. However, our assessment of water reuse systems indicates that not only the spatial dimension, but also the temporal dimension of flow is relevant. It is a striking observation that none of the reviewed indicators integrate time-specific flows into their definitions. Defining recoverable flow as a function of time is a necessity for a better understanding of water reuse systems. Also, no distinction between surface water and groundwater flow is made in the original definitions. This lack of information disregards the significant difference in both space and time between connectivity through the surface water system and the groundwater system. Other

relevant information, such as the recoverable portion of total return flow, and the distinction between anthropogenic and natural flows, is equally excluded from the indicators.

An important note is that, even if all relevant dimensions would be accounted for in the reuse indicators, the Arkansas River case study shows that the required input data is not available from a comprehensive management plan for a basin in one of the most data-abundant areas of the world. Indicators *DRR* and RD potentially hold the most direct information on water reuse, but the availability of input data is limiting. A number of important assumptions need to be made when assessing the Arkansas River reuse system, and as such our simple demonstration is exemplary for most basin-wide water use and water allocation studies. Typical assumptions include an equal *NCF* for all users of a similar nature, and the assumption that recoverable volumes equal non-consumed volumes (RE = 1). Such simplifications, forced by a lack of data, prohibit a thorough assessment of water reuse, and thus of water losses and potential for water savings. This problem requires the development of a method that integrates an analysis of connectivity between water users and computation of the relevant hydrological fractions.

2.6 Concluding remarks

As pressure on global water resources increases and more river basins approach a state of closure, there is an undeniable need for effective management of the finite amount of water available in a basin during the hydrological year. Local water saving measures do not work without an understanding of downstream impacts. There is a major pitfall in rushing to conclusions by applying subjective performance indicators at an early stage in water management analyses. When modifying hydrological fractions amounts to a redistribution of a fixed volume of water rather than true water savings, the question is whether the upstream advantages compensate the previous benefits of non-consumed flows now reduced. Quoting Contor and Taylor (2013): "Any proposal to improve irrigation technology or management must be accompanied by careful water budget analysis of the present-condition fate of the non-consumed fraction of applied irrigation water, and of the human and ecosystems made of the current waste stream."

Although the importance of data on water reuse for achieving goals on basin-scale water resources planning is now generally acknowledged, little work has been done with a primary focus on mapping the relevant dimensions of water reuse. It is argued that an improved analysis of water reuse will be helpful to understand existing interactions and localizing potential water supply issues, constructing sound basin water accounts, identifying appropriate water management strategies for different locations, and predicting effects of future interventions. Ultimately, this can support development of successful water allocation policies and water rights systems, both in terms of withdrawal permits and return flow obligations. Consistent use of terminology and definitions is essential to avoid

misunderstanding of the water balance and subsequent adverse effects of interventions.

The need exists for a hydrologically consistent approach to express water reuse, strongly rooted in the concepts of consumptive use, non-consumptive use, and hydrological connectivity. The key parameter to track is the non-consumed portion of a water withdrawal. In case this water is indeed recoverable for downstream users, many services and benefits are potentially obtained from this water that is initially 'lost'. The indicators reviewed in this chapter need to be complemented with an assessment of both the spatial and temporal dimension of the recoverable flow. There is a lack of geographical methods to quantify these recovery processes on a monthly and annual time frame. This is relevant for ungauged, poorly gauged, and gauged basins because return flow cannot be measured in a straight-forward manner. Future studies should aim at tracking the non-consumed fraction of water withdrawals and showing the dependency of multiple water users on these flows to water policy makers.

3

INTEGRATING GLOBAL SATELLITE-DERIVED DATA PRODUCTS AS A PRE-ANALYSIS FOR HYDROLOGICAL MODELLING STUDIES: A CASE STUDY FOR THE RED RIVER BASIN, VIETNAM

With changes in weather patterns and intensifying anthropogenic water use, there is an increasing need for spatiotemporal information on water fluxes and stocks in river basins. The assortment of satellite-derived open-access information sources on rainfall (P) and land use/land cover (LULC) is currently being expanded with the application of actual evapotranspiration (ET_{act}) algorithms on the global scale. This chapter demonstrates how global remotely sensed P and ET_{act} datasets can be merged to examine hydrological processes such as storage changes and streamflow prior to applying a numerical simulation model. The study area is the Red River Basin in China in Vietnam, a generally challenging basin for remotely-sensed information due to frequent cloud cover. Over this region, several satellite-based P and ET_{act} products are compared, and performance is evaluated using rain gauge records and longer-term averaged streamflow. A method is presented for fusing multiple satellite-derived ET_{act} estimates to generate an ensemble product that may be less susceptible, on a global basis, to errors in individual modeling approaches. Subsequently, monthly satellite-derived rainfall and ET_{act} are combined to assess the water balance for individual sub-catchments and types of land use, defined using a global land use classification improved based on auxiliary satellite data. It is found that a combination of TRMM rainfall and the ensemble ET_{act} product is consistent with streamflow records in both space and time. It is concluded that monthly storage changes, multi-annual streamflow and water yield per LULC type in the Red River Basin can be successfully assessed based on currently available global satellite-derived products.

Chapter based on: Simons, G.W.H., Bastiaanssen W.G.M., Ngo, L.A., Hain, C.R., Anderson, M. and Senay, G.B., 2016. Integrating Global Satellite-Derived Data Products as a Pre-Analysis for Hydrological Modelling Studies: A Case Study for the Red River Basin. Remote Sensing, 8, 279; doi:10.3390/rs8040279

3.1 Introduction

Global surface and groundwater resources are under increasing pressure from human water use and climate change (Gleeson et al., 2012; Haddeland et al., 2014; Wada and Bierkens, 2014). Well-informed decision making on water management is essential for coping with tensions between water availability and water demand. This requires a feasible methodology for quantifying the current state of water resources in terms of hydrological flows and connectivity, as well as indicators of water use and reuse (Simons et al., 2015). Once reasonable estimates of these quantities have been established, simulation models can be used to examine the predicted consequences of different scenarios related to policy adjustments, climate change, land use modifications, etc. (e.g. Droogers and Bouma, 2014).

The fundamental components of the water balance that need to be quantified include precipitated water, consumed water, water withdrawals, and non-consumed water with varying definitions and sub-classifications to be found in widely used water assessment frameworks such as Water Footprint (Hoekstra et al., 2011), Water Accounting Plus (Karimi et al., 2013a), and SEEA-Water (United Nations, 2012), among others. Relating precipitation and/or withdrawals to consumptive use through evapotranspiration provides a basis for an assessment of weekly or monthly surplus (i.e. groundwater recharge, drainage, surface runoff dynamics) or deficit (i.e. irrigation, inundation, return flows and their reuse). The role of soil water storage changes is essential at smaller time scales and should get sufficient attention (Ahmad and Bastiaanssen, 2003).

Satellite-derived datasets have been increasingly put to use in the field of water resources management at a range of different spatiotemporal scales. They provide valuable information in poorly gauged or inhospitable areas and transcend political borders. By now, methodologies for deriving precipitation (P) and actual evapotranspiration (ET_{act}) from remotely-sensed data are well-established (Anderson et al., 2012; Karimi and Bastiaanssen, 2015; Kidd and Levizzani, 2011). For purposes of water accounting, identification of management options and relating water consumption to services and benefits, it is desirable to relate the quantified flows to types of land use and land cover (LULC) within a river basin or, ideally, to individual water users. This facilitates a description of water users in a river basin in terms of their dependency on water from different sources, as well as the extent to which they "produce" water for potential downstream reuse (Simons et al., 2015).

A number of global-scale satellite-derived data products (GSDPs) for P, ET_{act} and LULC are available. Many of these are already in the public domain or soon to be released, which makes them a valuable and easily accessible resource for water management researchers, consultants and policy makers. Scientific literature provides a substantial body of review work on these products and their fundamental algorithms. Open-access rainfall GSDPs are extensively evaluated in scientific literature for a variety of geographical areas across the globe, e.g. (Asadullah et al., 2008; Hessels, 2015; Khandu et al., 2015; J. Liu et al., 2015; Stisen

and Sandholt, 2010; Toté et al., 2015). Existing GSDPs on LULC and their validation are discussed for example by Mora et al. (Mora et al., 2014) and Tsendbazar et al. (Tsendbazar et al., 2014). Conversely, global-scale ET_{act} products based on remote sensing are relatively new. A wealth of literature on satellite-based techniques for quantifying ET_{act} is available (Kalma et al., 2008) and the basic algorithms are well-documented (Anderson et al., 2011; Chen et al., 2013; Guerschman et al., 2009; Mu et al., 2013; Senay et al., 2013). Many institutions are now taking the next step by developing and distributing operational evapotranspiration products for the globe at spatial resolutions of ≤ 5 km. ET_{act} GSDPs provide independent datasets for calibrating hydrological models and land surface models. Comparative analyses of ET_{act} models applied on the continental to global scales have recently come available and typically intercompare two individual satellite-derived *ETact* products for specific regions (Alemu et al., 2014; Hu et al., 2015; Hu and Jia, 2015; Velpuri et al., 2013; Yilmaz et al., 2014), some also including ET_{act} outputs from global hydrological models and land surface models (Trambauer et al., 2014). Comprehensive evaluations of a larger number of satellite-derived ET_{act} estimates, in the style of the many *P* assessments that are available, have so far only sparsely been conducted (Bhattarai et al., 2016; Singh and Senay, 2015). This is related to the limited availability of these products in the public domain up to now, which is currently changing rapidly.

Some recent papers have focused on integrating rainfall, ET_{act} and LULC GSDPs and their combined potential for assessments of water resources. Bastiaanssen et al. (Bastiaanssen et al., 2014) successfully computed the annual water balance of the Nile basin, including net withdrawals. Wang-Erlandsson et al. (2016) demonstrated how global *P* and ET_{act} time series can be used to compute the storage capacity of the root zone. The integrated use of satellite-derived *P* and ET_{act} is a reality check on a pixel-by-pixel basis and an opportunity to check data quality that goes beyond the comparison with individual rain gauges or eddy covariance towers, which both cover very limited areas. If quality is found to be satisfactory, such data can be integrated in hydrological modeling procedures on the regional and global scale. In addition, Hain et al. (Hain et al., 2015) demonstrated how ET_{act} retrieved from energy balance can be combined with an inferred local water balance to diagnose ancillary sources and sinks of moisture across landscapes, e.g. due to intensive irrigation or agricultural drainage, or access to shallow water tables.

The aims of this chapter are to (i) demonstrate how integrating satellite-derived *P*, ET_{act} and LULC maps constitutes an important pre-analysis in the first stages of hydrological modeling; (ii) show that consistency between hydrological variables is a way to evaluate and intercompare individual earth observation products, with a focus on five new global ET_{act} products; and (iii) evaluate the suitability of global satellite-derived data products for assessing water resources in a basin with challenging conditions for remote sensing. We present our case in the context of the transboundary Red River Basin in Southeast Asia, traditionally a problematic region for remote sensing because of weather patterns, but also a basin with

pressing water management issues where limited international data sharing hampers a comprehensive understanding of basin water use and hydrology.

3.2 Materials and methods

3.2.1 *Study area*

The Red River Basin (Figure 3.1) can be roughly divided in an upstream half situated in the province of Yunnan in southern China and a downstream half in northern Vietnam, with a portion of less than 1% located in Lao PDR. Its surface area is approximately 164,000 km². The Red River has two main tributaries: Đà River (Lixian in Chinese) and Lô River (Panlong). The upstream part of the basin is largely forested, mountainous and sparsely populated. The delta of the Red River, downstream of the confluence of the three major branches, is a densely populated area of great importance to Vietnam for its agricultural productivity and economic activity.

Annual rainfall varies substantially across the Red River Basin, with values between 700 mm and 3000 mm found based on long-term station time series (Le et al., 2012; Li et al., 2008), while even local annual averages of over 4,000 mm/year are reported (Diep et al., 2007). Approximately 80% of the rainfall occurs from May to October, which comprises the wet season for both the Vietnamese and the Chinese portion of the basin (Zhongyan, 2012). The variability of river discharge in space and time, as well as population growth, lead to substantial challenges related to

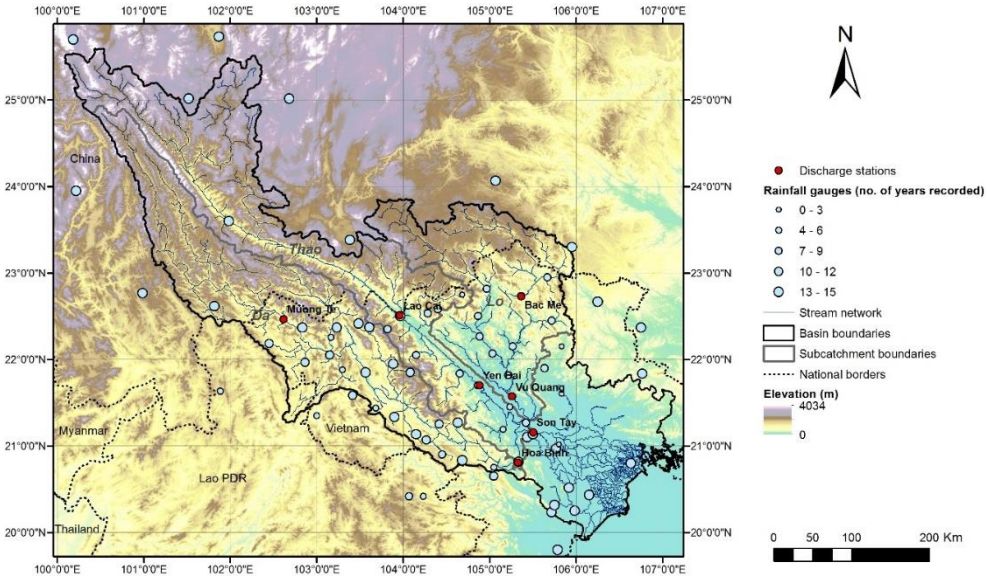

Figure 3.1. Topography of the Red River Basin, with its network of rivers and irrigation canals, and locations of flow gauges and rainfall stations used in this study.

flood control and water stress, particularly in Vietnamese territories (Kattelus et al., 2014). Water management options across the basin have increased with the construction of five large multi-purpose reservoirs in the Vietnamese Đà River and tributaries of the Lô River, as well as many smaller hydropower dams in both China and Vietnam. However, this has also increased the need for spatiotemporal data on water availability to support reservoir management (Castelletti et al., 2012).

At the tail end of the basin, the Red River Delta has seen many centuries of human water management, from the construction of hydraulic works for protection from floodwaters to the support of irrigation by avoiding inflow of brackish water and enhancing land drainage, making use of tidal influences if possible. Three zones can be distinguished: the lowlands, midlands, and highlands, based on their elevation relative to the water table (Devienne, 2006). The spatial distribution of water resources across the Delta is unequal, with some areas approaching the minimum level of water availability required to "sustain life and agricultural production" (Luu et al., 2010). Most of the surface area of the Delta is characterized by rice paddies for a major part of the year. Typically, two rice seasons are observed, an irrigation-dependent spring season and a rainfall-dependent summer season (Klapetek et al., 2010). If irrigation water availability allows, farmers grow a third "dry" crop such as vegetables or maize during the October-February period, particularly in the highlands and midlands. Reuse of drainage water within irrigation schemes is substantial (Chinh, 2012). Still, non-consumed irrigation water is one of the main sources of aquifer recharge, and thus of industrial and domestic water supply (Bui et al., 2011). The outflow from the complex stream network of the Red River Delta into the Gulf of Tonkin occurs through nine different outlets (Vu et al., 2014).

3.2.2 *Land use / Land cover*

The current application requires an accurate and recent LULC map covering the study area with sufficient spatial detail and distinguishing between classes relevant for the nature of water use, including a class for irrigated cropland. An overview of existing global LULC maps is provided by Mora et al. (2014), with spatial resolutions ranging from mid-resolution (300 - 500m) to lower resolution (≥ 1km) products. In addition, the first high-resolution Landsat-based global LULC products are now also available (Chen et al., 2015; Gong et al., 2013). The number of classes of the available LULC maps varies from 9 to 37, and years of coverage from 1992 to 2012. Based on the criteria mentioned above, in particular Globcover 2009 (Defourny et al., 2009) and GLCNMO2008 (Tateishi et al., 2014) were identified as potentially suitable inputs to this study.

Accuracies of global LULC products were previously found to be in the range of 69-87% (Karimi and Bastiaanssen, 2015). Ongoing initiatives such as the Global Observation for Forest Cover and Land Dynamics (GOFC/GOLD) of ESA seek to enhance the quality of global LULC products. In the meantime, auxiliary satellite images from the public domain are helpful to enhance LULC maps for a specific region. We adopted an approach of deriving an optimized LULC map for the Red

River Basin derived from a combination of existing LULC GSDPs and time series of freely available MODerate resolution Imaging Spectroradiometer (MODIS) satellite images, a proven methodology for improving the accuracy of LULC maps (Vuolo and Atzberger, 2014; Yu et al., 2013). Regional-scale improvement of global land cover products, incorporating auxiliary data and a priori knowledge, leads to more accurate and actionable water accounting information.

The 300 meter GlobCover 2009 map was taken as the basis for the new LULC map. Although the spatial distribution of forested and shrubland classes seems in accordance with expert knowledge, the original Globcover 2009 product largely contains rainfed cropland pixels for the Red River Delta. This is erroneous when viewed against the abundant presence of irrigation infrastructure. However, distinguishing between rainfed and irrigated cropland is not straight-forward, as the wet season is likely rainfed in both classes, with water coming from rainfall or seasonal floods recession (Johnston et al., 2012). The main distinctive feature between single-season, exclusively rainfed crops and multi-cropped areas with at least one irrigated cycle is therefore the occurrence of a winter and/or spring crop (Nguyen et al., 2015).

According to the GlobCover 2009 validation report (Bontemps et al., 2011), irrigated pixels are regularly misclassified as other agricultural classes. Therefore, to correct the GlobCover 2009 agricultural classes, first all cells containing >50% cropland were merged into a single cropland class. MODIS NDVI values within the merged cropland class during the spring season were decisive in distinguishing irrigated from rainfed agriculture. Pixels covered by clouds, as indicated by the MODIS pixel reliability layer, were omitted from this analysis. No gap filling of individual images was performed, in order to only include pixels directly sensed by MODIS with sufficient quality. An average NDVI of at least 0.55 in the months March to May was used as a criterion for identifying irrigation, in accordance with the typical Red River Delta spring cropping cycle. A different cropping calendar was identified from NDVI time series analyses for the northern parts of the basin, with a pronounced peak during January. For this reason, a second precondition of a minimum NDVI of at least 0.55 in January was introduced to account for irrigation in the upstream portion of the basin. The underlying assumption is that an NDVI of 0.55 for cropland in the Red River Basin cannot be achieved in January or March-May by relying solely on rainwater.

In addition to the correction of the GlobCover 2009 cropland classification, a visual assessment of the original map against high-resolution satellite imagery indicated an underestimation of urban area in the Red River basin. It was observed that the urban land use class of GLCNMO2008 is more realistic and these cells were therefore introduced to represent built-up area in the improved LULC map. As a final step, isolated pixels were filtered out using a GIS focal majority filter.

MODIS NDVI time series of three major classes in the final LULC map are displayed in Figure 3.2. While some noise is apparent due to the different cloud masks applied to each of the individual images, distinct temporal patterns can be identified. The

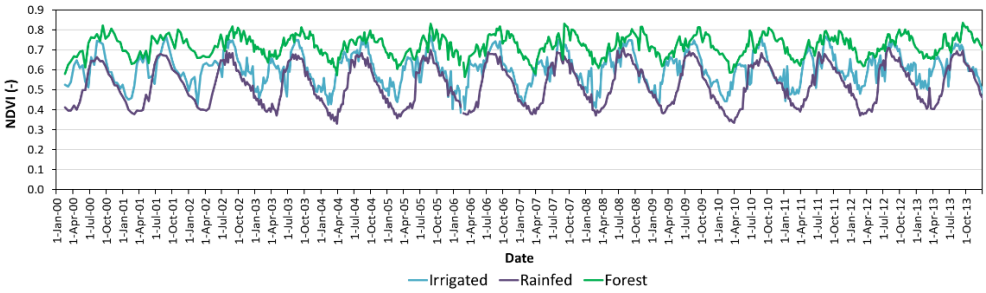

Figure 3.2. NDVI time series of three main LULC classes in 2000-2014 based on the MOD13Q2 and MYD13Q2 products. "Forest" contains all forest types in Globcover 2009.

second annual cropping season in the irrigated class is clearly visible when compared to the rainfed cropland. A third, less pronounced peak of irrigated NDVI values can be observed in the winter months. Year-to-year differences of winter and spring crop NDVI are illustrative of varying water availability. As is to be expected, average NDVI of the merged forested class remains relatively stable and high (> 0.5) throughout the entire year.

The enhanced LULC map is depicted in Figure 3.3. Visual comparison with the IWMI map of irrigation in Asia[3] shows similar spatial distribution of rainfed and irrigated land. As the Red River Delta has been the focus area of most previous studies, availability of validation data is largely limited to this area. The modifications to the original GlobCover 2009 yield a total irrigated area of 869,029 ha in the 10 provinces that make up the Red River Delta administrative region. Previous studies report irrigated acreages between 670,000 ha and 850,000 ha, although the corresponding spatial and temporal scope is not always specified (Castelletti et al., 2012; Fontenelle, 2001; Luu et al., 2010; Nguyen, 2011; Turral et al., 2002). However, Nguyen et al. (2015) reported 1,180,000 ha of double-cropped rice in 2007-2011 based on Advanced Synthetic Aperture Radar (ASAR) data, so some uncertainty persists. Overall, the new LULC map corresponds well with information available from other sources and suffices for the current purpose.

3.2.3 *Rainfall*

A spatially distributed monthly rainfall product is required which covers the Red River Basin for the last 10 - 15 years. Existing rainfall GSDPs with over 10 years of data in the period 2000 to present and a spatial resolution of ≤ 0.25 degree were downloaded and evaluated: the Tropical Rainfall Measurement Mission monthly best estimate (TRMM 3B43 v7), the global rainfall estimate based on the CPC MORPHing technique (CMORPH) and the Climate Hazards Group InfraRed Precipitation with Station dataset (CHIRPS v1.8). Since no readily available

[3] (*http://waterdata.iwmi.org/irra/*, retrieved 19-Nov-2015)

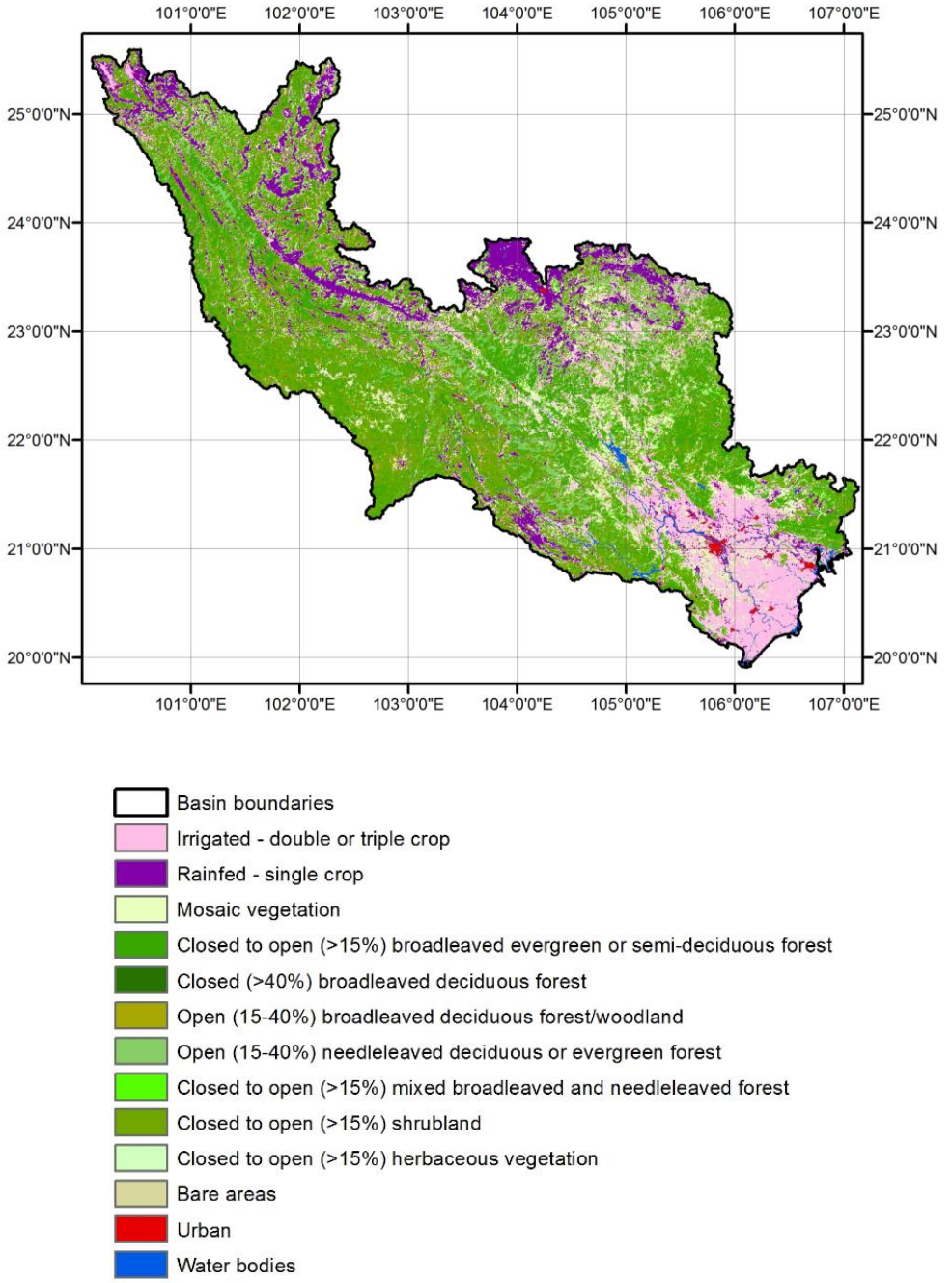

Figure 3.3. Land use / land cover in the Red River Basin.

CMORPH monthly product exists, three-hourly data were aggregated to obtain monthly values.

Table 3.1 presents the main characteristics of the rainfall GSDPs evaluated for the Red River Basin.

In order to select the most accurate rainfall product for the target basin, the performance of each of the GSDPs was assessed by means of ground observations. Daily rainfall station data were purchased from the Vietnamese National Center for Hydro-Meteorological Forecasting (NCHMF) and downloaded from the NOAA Global Summary of the Day (GSOD) database, as distributed by the National Climatic Data Center (NCDC). In total, multiple years of rainfall data for 76 gauges were available for GSDP validation. Figure 3.1 indicates the location and amount of data available for each station. A full list of all rain gauges can be found in Table B.1. Data from 62% of these stations are not provided in the public domain and are therefore particularly suitable for validation, since the TRMM and CHIRPS algorithms incorporate a calibration procedure based on open-access rainfall gauge measurements. Nevertheless, it was decided to also include public GSOD data in this validation exercise as otherwise no validation data from Chinese territories would be available.

Figure 3.4 shows plots of satellite-derived monthly rainfall data against rain gauge measurements. Of the evaluated products, the TRMM regression line is closest to 1:1 correspondence line, followed by CHIRPS and CMORPH, respectively. A few outliers are visible, where high gauged rainfall amounts do not correspond with satellite-derived estimates. These were all recorded at the Bac Quang station. It is unclear whether this signifies an issue with the measurement station or the GSDPs. Either way, as these 10 points only comprise a minor portion of the total number of monthly rainfall amounts (10,368), their impact on further analyses is negligible.

Table 3.1. Evaluated rainfall GSDPs for the Red River Basin. The basin-wide mean rainfall (μ) and year-to-year standard deviation (σ) are reported for the overlapping period (2003 – 2014). April - September and October - March rainfall statistics are listed separately to reflect the regional seasonality of rainfall.

Product	Temporal coverage	Original / applied resolution	Key references	Annual *P* (mm)		Apr-Sep *P* (mm)		Oct-Mar *P* (mm)	
				μ	σ	μ	σ	μ	σ
TRMM 3B43 v7	01/1998 - 10/2015	0.25° / 25 km	(Huffman and Bolvin, 2014)	1,546	122	1,302	70	244	61
CHIRPS v1.8	01/1981 - present	0.05° / 5 km	(Funk et al., 2014)	1,403	115	1,223	88	180	40
CMORPH	12/2002 - present	0.25° / 25 km	(Joyce et al., 2004)	1,169	173	1,071	151	99	40

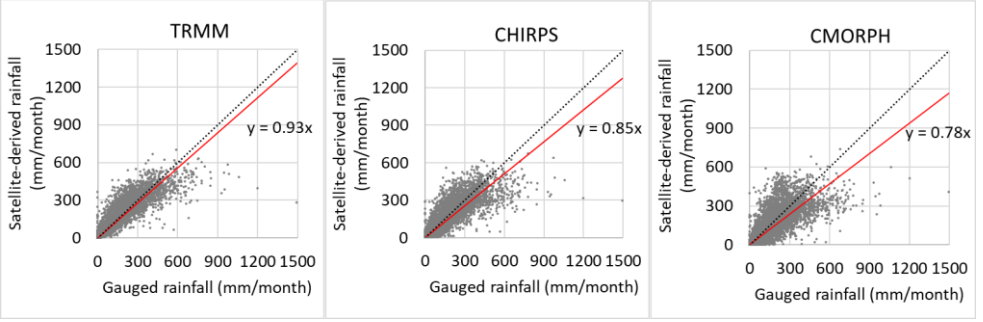

Figure 3.4. Comparison of monthly rain gauge data with three satellite products for 2003 – 2014. The dashed line indicates a 1:1 correspondence and the red line gives the linear regression best fit with 0 intercept.

The error in monthly rainfall estimates for each of the products is further evaluated in Figure 3.5. With -5.83 mm, CHIRPS has a slightly lower error than TRMM, while the mean error of CMORPH monthly rainfall estimates are furthest from measured values. It is interesting to note that, although the CHIRPS mean error is lower than the TRMM mean error, the standard deviation of the CHIRPS error is higher as a result of the amount of months with large error values. Table 3.2 lists a number of other commonly used validation statistics. These indicate a favorable performance of TRMM in terms of the relationship between measured and estimated values (r), the relative mean absolute error (RMAE), and the predictive power of the algorithm relative to the gauged mean (Nash-Sutcliffe coefficient).

Based on the findings discussed above, TRMM was identified as the most suitable GSDP for describing monthly rainfall in the Red River basin. This is in line with earlier findings that TRMM is the most favorable option for satellite-derived rainfall on the monthly scale in an area in southern China (T. Liu et al., 2015), and a successful application of TRMM precipitation in a modeling study in central Vietnam (Le et al., 2014). Apparently, for the Red River Basin, the higher spatial resolution of the CHIRPS product does not lead to a more accurate assessment of

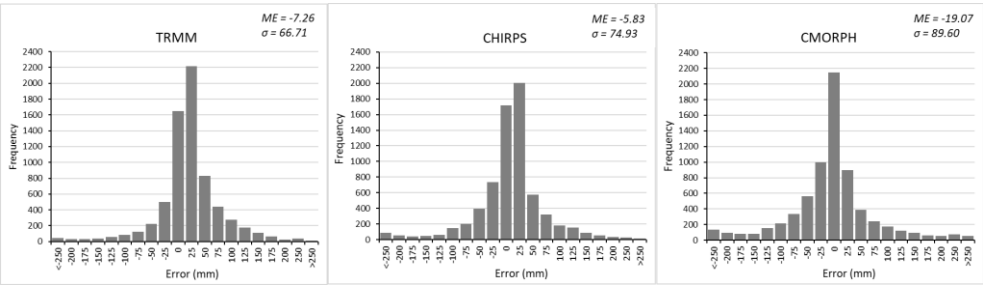

Figure 3.5. Frequency distributions of the error in monthly rainfall. Indicated are the mean error (ME) and standard deviation (σ) in mm.

Table 3.2. Pairwise validation statistics for three satellite products (S) based on all available station records (G) for the period 2003 – 2014.

Indicator	Formula	CHIRPS	TRMM	CMORPH
Pearson correlation coefficient r (-)	$r = \frac{\sum(G-\bar{G})(S-\bar{S})}{\sqrt{\sum(G-\bar{G})^2}\sqrt{\sum(S-\bar{S})^2}}$	0.851	0.884	0.786
Relative Mean Absolute Error RMAE (-)	$RMAE = \frac{1}{n}\frac{\sum\lvert(S-G)\rvert}{\bar{G}}$	0.327	0.296	0.427
Nash-Sutcliffe Model Efficiency Coefficient NS (-)	$NS = 1 - \frac{\sum(S-G)^2}{\sum(G-\bar{G})^2}$	0.721	0.777	0.585
Bias (-)	$Bias = \frac{\sum(S)}{\sum(G)}$	0.956	1.055	0.857

rainfall when compared to the point scale. It should be noted that some of the GSOD stations used for validation may also have been part of the TRMM and CHIRPS algorithms, whereas CMORPH is uncorrected for station values.

Bias-correction of rainfall GSDPs is often performed based on ground observations. However, special attention should go to the issue of scale when comparing point measurements of rainfall gauges to coarse pixels (Cheema and Bastiaanssen, 2012). Naturally, a 25km pixel can be quite heterogeneous e.g. in terms of topography, and different rainfall rates may occur over short distances within a grid cell. Vernimmen et al. (2012) discuss in detail how the presence of multiple ground stations within a grid cell enhances opportunities for validation. In the Red River Basin, five TRMM pixels were identified containing two rainfall stations (Figure B.1). The records of these gauges were averaged per pixel and plotted against TRMM values. This resulted in a slope of the fitted line of 0.97 (Figure B.2). This increase relative to 0.93 (Figure 3.4) indicates that performance of TRMM seems satisfactory in terms of representing intra-pixel variability. Although the sample size is insufficient to draw any definitive conclusions, this brief analysis does not provide a reason for assuming that a point-based bias correction would improve the 25km TRMM rainfall estimate.

3.2.4 *Actual evapotranspiration*

Available ET_{act} products

While the network of rain gauges in the Red River Basin is sufficient to arrive at a well-informed choice of an optimal GSDP for precipitation, this is unfortunately not the case for evapotranspiration. No network of ET_{act} measurements is available for the Red River Basin, limiting the foundation for selecting a single ET_{act} GSDP. We therefore take an ensemble approach to defining ET_{act} across the basin, combining information from multiple GSDPs.

In this study, five ET_{act} products were evaluated with a coverage of the Red River Basin at a spatial resolution of ≤ 5 km with a time series of over 10 years: the MODIS

Global Terrestrial Evapotranspiration Product (MOD16, (Mu et al., 2013)), the Operational Simplified Surface Energy Balance (SSEBop, (Senay et al., 2013)), the revised Surface Energy Balance System (SEBS, (Chen et al., 2013)), CSIRO MODIS Reflectance Scaling actual ET (CMRSET, (Guerschman et al., 2009)), and the Atmosphere-Land Exchange Inverse (ALEXI) water and energy budget model (Anderson et al., 2007). Although these products all use MODIS satellite data to some extent, their fundamental modeling strategies are markedly different. SSEBop and SEBS rely on MODIS land surface temperature (LST) data for determination of the latent heat flux. ALEXI uses a similar approach but integrates a range of different spaceborne data sources. CMRSET combines a vegetation index for estimating photosynthetic activity with shortwave infrared reflections to estimate vegetation water content and presence of standing water. MOD16 follows the Penman-Monteith logic and relies on visible and near-infrared data to account fo Leaf Area Index (LAI) variability. The latter is currently the only global product that has been tested and reviewed in a substantial number of scientific articles (Hu et al., 2015). For a detailed description of each of the ET_{act} algorithms, the reader is referred to the citations listed in Table 3.3.

ALEXI is the only model for which no preprocessed monthly product was available. Therefore, weekly values were aggregated to monthly maps, with ET_{act} during weeks overlapping two months being proportionally divided over these months. Maps of annually averaged ET_{act} for the Red River Basin in 2003 - 2012 retrieved from the five aforementioned methods can be found in Figure C.1.

Table 3.3 lists basin-averaged ET_{act} according to the individual products. Annual average ET_{act} in 2003 - 2012 falls within a range of 268 mm, with SSEBop on the low end and SEBS on the high end of the values. It is interesting to note that the standard deviation of seasonal sums in the dry season is higher than in the wet season for all products. This reflects the different ways in which the algorithms simulate evapotranspiration under stressed conditions; during the rainy season, ET_{act} will likely equal potential evapotranspiration (ET_{pot}) most of the time. None of the retrieved annual ET_{act} amounts conflict with reported values for reference evapotranspiration in the Red River Basin (Le et al., 2012), or with the basin annual average ET_{pot} of 1,306 mm according to a 1 km global dataset on long-term average monthly ET_{pot} distributed by CGIAR (Zomer et al., 2008).

Karimi and Bastiaanssen (2015) report a mean absolute percentage error of 5.4% for remote sensing-based ET_{act} estimations. However, the range of values in Table 3.3 indicates that algorithms developed for the global scale yield substantially different outlooks on the Red River Basin water balance. This is also visible when comparing the spatial patterns in Figure C.1. Specific locations where ET_{act} values of the different products correspond or contradict can be observed in Figure D.1, where a spatial depiction of the coefficient of variation (CV) in annual average ET_{act} is provided per pixel. The highest CV values are observed in areas with high elevation along some of the subbasin boundaries, where especially SEBS deviates from the other GSDPs (see Figure A3). A high CV is also found in the coastal zone,

Table 3.3. Characteristics of different ET_{act} products for the Red River Basin. The basin-wide mean ET_{act} (μ) and inter-annual standard deviation (σ) are reported for the 2003 – 2012 period. Temporal coverages indicate the time series of each product that were available for this study.

Product	Temporal coverage	Original / applied resolution	Key references	Annual ET_{act} 2003-2012 (mm)		Apr-Sep ET_{act} (mm)		Oct-Mar ET_{act} (mm)	
				μ	σ	μ	σ	μ	σ
MOD16 (A2)	Jan 2000 – Dec 2015	926 m / 1 km	(Hu et al., 2015; Mu et al., 2013)	1,009	21	626	11	383	10
SSEBop	Jan 2003 – Dec 2013	1 km / 1 km	(Savoca et al., 2013; Senay et al., 2013)	886	37	614	11	273	25
SEBS	Jan 2001 – Dec 2013	5 km / 5 km	(Chen et al., 2013)	1,154	65	725	22	429	50
CMRSET	Jan 2000 – Dec 2012	0.05°/ 5 km	(Guerschman et al., 2009)	960	47	565	23	396	27
ALEXI	Jan 2003 – Dec 2014	0.05°/ 5 km	(Anderson et al., 2015, 2007)	1,104	33	709	13	395	26

possibly caused by differing methodologies for dealing with standing water, or differences in applied land/water masks.

Examining the monthly variability of ET_{act} for different LULC classes against a priori knowledge is a way to further evaluate the five models. Figure 3.6 shows how monthly ET_{act} varies for three major land use types: irrigated cropland, rainfed cropland and the merged forested classes. In general, the different products agree reasonably well in terms of temporal patterns in monthly ET_{act}, and no clear discrepancies are observed in relation to known monthly rainfall patterns. The least temporal variation is observed in CMRSET, and the highest in SEBS followed by SSEBop. Rainfed agriculture has generally the lowest ET_{act} of these three LULC classes, according to all products. It is found that all models compute a reduction in the difference between the rainfed and irrigated classes as the wet season progresses. This is to be expected to a certain extent, as rainfed crops will have access to sufficient water during this period. The difference remains the largest in SSEBop ET_{act}, whereas almost full convergence of the rainfed and irrigated CMRSET curves occurs from July onwards. MOD16 is the only model that predicts ET_{act} to be highest for the forest class throughout the year. ALEXI and CMRSET predict a very similar time series for the forested and irrigated classes, which may seem surprising as the physical conditions of these ecosystems are rather different. However, both forest and irrigated crops have access to ancillary moisture unavailable to rainfed crops (the forests due to deeper rooting depths), and with the current information it is difficult to determine which of the five temporal curves for these LULC types are most realistic. Despite the differences between products,

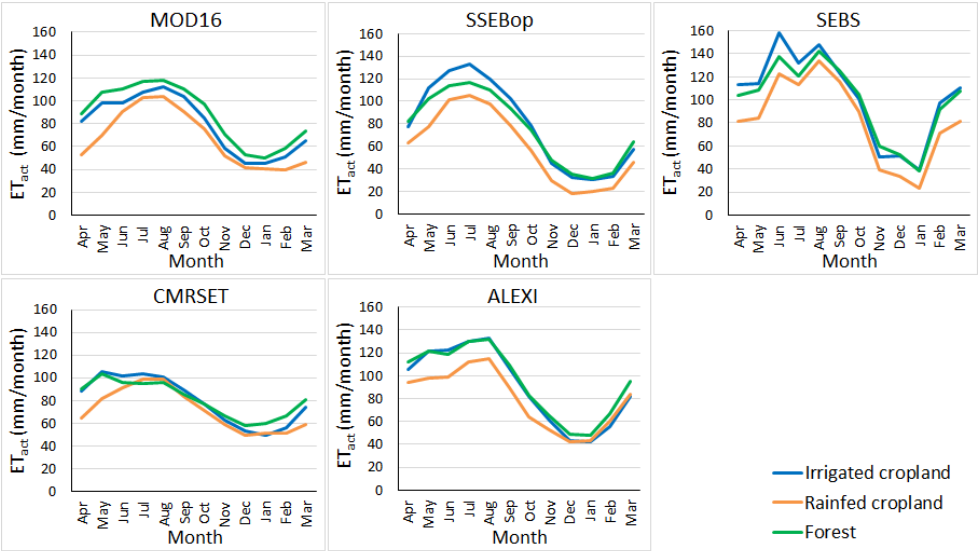

Figure 3.6. Average monthly ET_{act} for three land use / land cover types.

Figure 3.6 does not provide sufficient basis for excluding any of the ET_{act} models from further analyses.

Solving the water balance to evaluate ET_{act}

As the information available for the Red River Basin is insufficient to verify the quality of the ET_{act} products independently from rainfall, TRMM data were used in combination with streamflow records to check the closure of the water balance:

$$Q = P - ET_{act} - \Delta S, \tag{3.1}$$

where Q is measured river discharge and ΔS is the change in catchment storage. Fundamental hydrological principles and the law of mass conservation dictate that, over a number of hydrological years, the rainfall surplus (P - ET_{act}) should equal Q at the downstream end of a catchment. In this study, the storage change over a period of 10 years is assumed to be negligibly small. Time series of daily river discharge were purchased from the NCHMF for the hydrological stations indicated in Figure 3.1. Metadata of these stations are provided in Table E.1. Using SRTM elevation data, upstream catchments were derived for each of the available measurement stations. Bac Me, Mường Tè and Lào Cai are located in mountainous areas and all have a catchment located for the largest part in China. Hòa Bình, Yên Bái and Vũ Quang are located at downstream points in the Đà, Thao and Lô subbasins respectively. The Sơn Tây station is located after of the confluence of the Đà, Thao and Lô River. No streamflow time series is measured downstream from Sơn Tây (Duc et al., 2011; Vu et al., 2014).

First, a preliminary check of the reliability of these Q data was performed by checking the consistency of temporal patterns between upstream and downstream stations in the same river branch (Figure E.1). Although the upstream and downstream stations in the Đà and Lô basins follow approximately the same pattern, the time series for the Thao River are quite different. For the years 2004 - 2005 hardly any runoff seems to be generated in the largely forested area of 14,000 km^2 between Lào Cai and Yên Bái, while in 2003 - 2004 Q measurements downstream are even lower than upstream (in other words, net consumption seems to occur), which is impossible given the size and dominant LULC types of the area. As the Yên Bái discharge curve corresponds well with temporal patterns observed at other stations, it was decided to eliminate Lào Cai from further analyses. Averaged over the overlapping period of records, Yên Bái, Vũ Quang and Hòa Bình, the three downstream stations in the subbasins, measure 92.8% of the total runoff at Sơn Tây. This is according to expectations, with the remaining 7.2% to be generated in the small intermediate area. In short, the analysis of streamflow records yields sufficient confidence in all available measurement stations, with the exception of Lào Cai.

It was decided to use long-term streamflow at one downstream gauging station to assess the area-averaged ET_{act}. Sơn Tây is the obvious choice, as it is located downstream of the confluence of the main tributaries and upstream of the Red River Delta, the main area of water demand. ET_{act} upstream from Sơn Tây was compared against TRMM rainfall and measured streamflow in Table 3.4. Hydrological years were defined from April 1st until March 31st of the subsequent calendar year, in order to include one full wet and dry season. Using this precipitation and streamflow dataset, SSEBop shows the best performance over this basin in terms of accordance with the laws of mass conservation, overestimating P minus Q by only 3.4%. For all other ET_{act} products, values are found to exceed P minus Q with a range of 14.0% (CMRSET) to 34.3% (SEBS).

It is important to realize that the aforementioned differences between P minus Q and ET_{act} are not only a product of uncertainties in satellite-derived P and ET_{act}. A variety of factors cause a potentially significant uncertainty in streamflow records, with errors of 10 - 20% not uncommon for single observations (Di Baldassarre and Montanari, 2009; McMillan et al., 2012; Pelletier, 1988). In the Red River Basin, local stage-discharge relations may become outdated after a number of years, depending on geology, in-stream sand mining and changes in erosion-sedimentation patterns due to reservoir construction. Specifically for the Sơn Tây gauging station, an error of 10-15% in streamflow values was reported in 2014 (Vu et al., 2014). Since the SSEBop retrieval of ET_{act} falls well within this range of accuracy, we assume that it represents the upstream conditions most accurately in terms of absolute ET_{act}. Still, the outcomes of such assessments should be regarded as comparative analyses, rather than absolute validation exercises.

Table 3.4. TRMM rainfall (*P*), measured streamflow at Sơn Tây (*Q*) and ET_{act} from each of the products for the overlapping period of hydrological years. Only the area upstream of the gauging station has been considered.

Hydr. year	*P* (mm)	*Q* (mm)	*P - Q* (mm)	ET_{act} (mm)				
				MOD16	*SSEBop*	*SEBS*	*ALEXI*	*CMRSET*
2003/2004	1,401	604	797	1,023	822	1,084	1,110	946
2004/2005	1,590	703	887	984	861	1,145	1,059	912
2005/2006	1,452	701	751	1,023	836	1,066	1,094	916
2006/2007	1,519	656	863	1,007	827	1,078	1,125	919
2007/2008	1,615	761	855	1,018	883	1,197	1,029	870
2008/2009	1,793	949	844	1,017	885	1,128	1,068	1,007
2009/2010	1,386	675	712	1,007	821	1,025	1,124	979
2010/2011	1,483	595	888	970	917	1,168	1,066	986
2011/2012	1,424	532	892	1,006	892	1,166	1,152	1,001
Average	**1,518**	**686**	**832**	**1,006**	**860**	**1,117**	**1,092**	**949**

Construction of an ensemble ET_{act} product

While *P* minus *Q* comparisons provide a means for assessing general reasonability of ET_{act} retrievals at basin scales, they provide no information about the relative model accuracy in spatially distributing ET_{act}. Each of the algorithms incorporates different inputs, procedures and assumptions, leading to substantial differences in spatial patterns between models, which can be viewed in Figures A3 and A4. Previous studies demonstrated that the performance of a certain ET_{act} algorithm is dependent on factors such as LULC type, climate and the presence of mountains (Alemu et al., 2014; Bhattarai et al., 2016; Hu et al., 2015; Singh and Senay, 2015; Velpuri et al., 2013), meaning that the accuracy of ET_{act} predictions will vary across a basin. An ensemble approach was taken toward generating "best-guess" maps of ET_{act} in the Red River Basin, under the assumption that spatial errors between related yet differing mapping approaches will tend to cancel in the ensemble average. A superior performance of different ET_{act} ensemble products with respect to individual algorithms was previously observed for the Nile Basin (Hofste, 2014), where flux towers were available for validation.

To identify models that are spatially most similar, spatial patterns were analyzed in terms of the Pearson correlation coefficient (r) at the pixel level (Table 3.5). A minimum value of 0.5 was assumed to represent a sufficiently strong spatial correlation to warrant combination in an ensemble ET_{act} product. It was found that the correlation between all pair-wise combinations of ALEXI, MOD16 and SSEBop was above this threshold, whereas CMRSET and SEBS do not achieve this level of correlation with any of the products. Pixel values of monthly ET_{act} for ALEXI,

Table 3.5. Pearson correlation coefficient of annually averaged ET_{act} pixel values in the entire Red River basin.

	ALEXI	CMRSET	MOD16	SEBS	SSEBop
ALEXI		0.249	0.679	0.181	0.714
CMRSET	0.249		0.095	0.408	0.419
MOD16	0.679	0.095		0.111	0.539
SEBS	0.181	0.408	0.111		0.383
SSEBop	0.714	0.419	0.539	0.383	

MOD16 and SSEBop were scaled around 1 (the average for each product upstream of Sơn Tây) and the resulting maps were averaged to create a relative ET_{act} map for each month. Finally, these relative values were multiplied with the SSEBop ET_{act} Sơn Tây catchment average. In this way, a final monthly ET_{act} product was constructed that is congruent with the basin water balance inferred from P minus Q, as well as with the spatial patterns predicted by the majority of the available ET_{act} GSDPs. The resulting annual ensemble ET_{act} for the Red River Basin is presented in Figure 3.7.

3.3 Results

In this section, the ensemble-averaged ET_{act} is used to study the water budget of the Red River basin. Long-term rainfall surplus is examined to determine the net production and consumption of water resources across the basin, in wet vs. dry seasons, and per LULC class. Subsequently, monthly runoff patterns are investigated for each subcatchment and storage changes are expressed as a function of rainfall surplus.

3.3.1 *Rainfall surplus*

Rainfall surplus (P_{sur}) is the total water budget available for generating surface runoff, replenishing aquifers, or recharging soil moisture stores. The partitioning of P_{sur} among different hydrological processes depends on factors such as soil type, slope, and intensity of precipitation. For multi-annual time scales on which ΔS can be neglected, P_{sur} equals the water yield ($P - ET_{act} - \Delta S$), the comprehensive term that is transported downstream through surface and subsurface pathways to constitute river flow.

Figure 3.8 presents the rainfall surplus in the Red River Basin for 2003 - 2012. From this map it can be concluded that the Red River Basin in a sense is an atypical river basin, with the upstream part generating relatively little runoff. Particularly the forested areas of the northern portion of the basin have a low P_{sur} over this ten-year period. Rainfall is lower here than in other parts of the basin, and forests likely grow deep roots to tap into aquifers. The highest P_{sur} occurs in the central part of the basin, a transitional area between the low-lying southeast and the mountainous

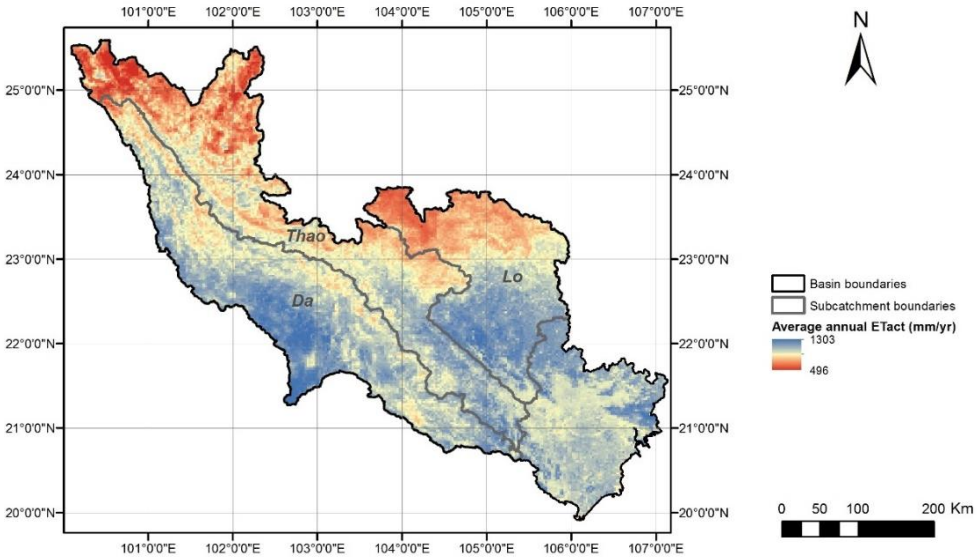

Figure 3.7. Annual ET_{act} averaged in the Red River Basin for the period 2003 - 2012.

north, with peak values of up to 1,300 mm/yr. From the perspective of transboundary water management, it is interesting to note that the majority of the average annual P_{sur} occurs in Vietnamese territories (825 mm, or ~73,000 km^3), while only 390 mm (~30,000 km^3) is produced in China.

Figure 3.8 shows that the irrigated Red River Delta on average does not consume water on the annual scale. This, however, is not the case when examining the irrigated spring rice season. Figure 3.9a shows how P_{sur} becomes negative due to water withdrawals during February-April 2010, when a net water consumption of up to 100 mm is observed in the delta. In general, a negative P_{sur} can be partially related to changes of water storage in the unsaturated zone, but a negative value during elongated periods is indicative of withdrawals. During the rainy summer season, P_{sur} is high in the entire basin (Figure 3.9b). Within the delta, P_{sur} is observed to be highest in the western part, where drainage is the most challenging due to the low relative altitude in relation to the water level (Devienne, 2006).

To evaluate water consumers and producers in the Red River Basin, the spatially distributed P_{sur} assessment was coupled with the improved LULC map (Figure 3.3). Table 3.6 provides an overview of water consumption and production by the different LULC classes in the Red River Basin. It is found that, on average, there is no net water-consuming LULC class on the annual scale. The largest amount of water in the Red River basin is produced by the extensive forest and shrubland ecosystems (an annual total of 62.3 km^3). In total, 102.6 km^3 (or 621 mm/yr) of water is produced on average per year, which can be viewed as an estimation of the total outflow of the complex stream network of the Red River Delta.

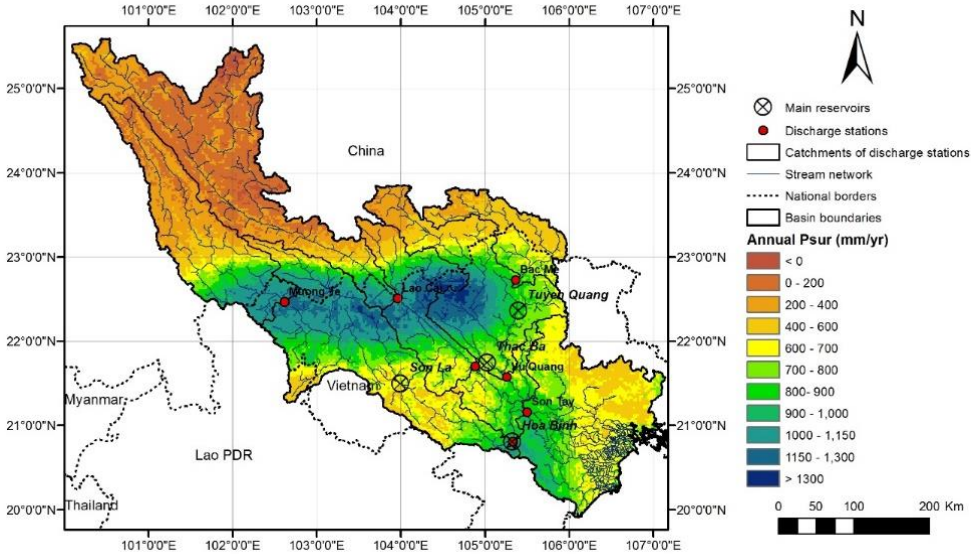

Figure 3.8. Annual average rainfall surplus in the Red River Basin for the period 2003 – 2012. Also indicated are the largest dams that provide opportunities for water storage. Hòa Bình and Thac Ba dam were operational during this entire period, Tuyen Quang and Son La dam were commissioned in 2008 and 2011, respectively.

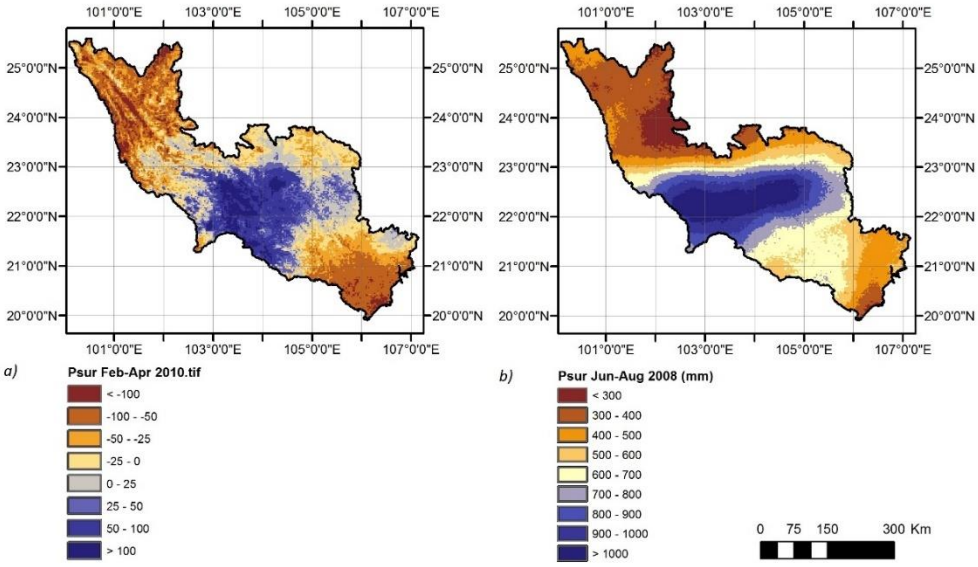

Figure 3.9. Rainfall surplus during February 2010 - April 2010 (a) and June - August 2008 (b), respectively the driest spring rice season and the wettest three-monthly period in the time series under consideration.

Table 3.6. Overview of consumptive use and water production (P - ET_{act}) per LULC class in the Red River Basin for the period 2003-2012.

LULC class	Area (km^2)	P (mm/yr)	P (km^3/yr)	ET_{act} (mm/yr)	ET_{act} (km^3/yr)	μ P_{sur} (mm/yr)	σ P_{sur} (mm/yr)	P - ET_{act} (km^3/yr)
Irrigated - double or triple crop	22,656	1,592	36.1	890	20.2	701	256	15.9
Rainfed - single crop	18,899	1,175	22.2	737	13.9	438	261	8.3
Mosaic vegetation (<50% cropland)	20,926	1,586	33.2	887	18.6	700	305	14.6
Closed to open broadleaved evergreen or semi-deciduous forest	32,431	1,602	51.9	9497	30.8	653	345	21.2
Closed broadleaved deciduous forest	1,817	1,534	2.8	860	1.6	674	364	1.2
Open broadleaved deciduous forest/woodland	4,282	1,411	6.0	881	3.8	531	349	2.3
Open needleleaved deciduous or evergreen forest	11,948	1,478	17.7	858	10.2	620	348	7.4
Closed to open mixed broadleaved and needleleaved forest	3,460	1,328	4.6	839	2.9	488	299	1.7
Closed to open shrubland	46,406	1,523	70.7	913	42.4	609	338	28.3
Closed to open herbaceous vegetation	397	1,633	0.6	958	0.4	675	181	0.3
Urban areas	594	1,618	1.0	853	0.5	766	168	0.5
Total	**165,178**		**249.1**		**146.4**			**102.6**

One of the most striking findings from this analysis is that a relatively large amount of water is produced by areas classified as irrigated cropland, while the opposite is found for the single-cropped rainfed class. Although this may be counterintuitive, it is caused by the geographical concentration of single crop agriculture in areas with a relatively low annual rainfall (+/- 1,000 - 1,300 mm). It is observed that the areas equipped with irrigation infrastructure (particularly the delta) are generally receiving more rainfall from the Tonkin sea during the rainy season than the zones dominated by rainfed agriculture further land inwards. Therefore, the observed

higher ET_{act} in double- or triple-cropped systems (890 mm/yr vs. 737 mm/yr) does not lead to a lower rainfall surplus compared to single crop agriculture. This very high summer rainfall in the delta is a known phenomenon, and the different tributaries and canals essentially serve as drainage canals during this period (Kono and Tuan, 1995).

3.3.2 *Runoff response patterns and storage changes*

When considering time scales of a single year or smaller, the change in storage ΔS becomes an essential component of the water balance. By relating the measured Q from different gauging stations to upstream P_{sur}, it is possible to e.g. identify the locations within a river basin where most streamflow originates, and the time periods when water stores in the soil profile and aquifers are replenished.

For different sections of the Red River Basin, measured streamflow and satellite-derived P_{sur} are compared in Figure 3.10. In the rainy season, streamflow from the catchments of all available stations typically lags behind the increase in P_{sur} by 1 to2 months, while the decline in both parameters around September occurs simultaneously. This is likely caused by water storage in aquifers and the soil profile, occurring up to the point of saturation after which all P_{sur} will be discharged as surface runoff. River discharge in parts of the Red River Basin is largely managed, as several large man-made reservoirs are present aimed at flood buffering and hydropower generation (IMRR, 2011). Dry season flow is highest at Hòa Bình and Vũ Quang, where artificial storage capacity in the upstream catchments is largest.

Table 3.7 presents the long-term Q/P_{sur} values for each of the catchments. Some values deviate substantially from 100%, which indicates that the 2003-2012 ΔS term may not be negligible for these areas. The low 10-year average of 80.9% for Mường Tè can be explained by the construction of several dams in the Chinese part of the Đà basin. In previous work (van de Giesen et al., 2015), at least nine hydropower reservoirs were identified that were commissioned in the years 2007-2009. The filling of these reservoirs in the preceding years has caused an average Q/P_{sur} of 56.1% until March 2007, whereas for April 2007 till September 2012 a value of 99% is found, indicating an almost perfect closure of the water balance by satellite-derived P_{sur}. The total volume of water stored in the new Chinese reservoirs in Đà River and its tributaries between April 2003 and March 2007 is estimated at 22.7 km^3. Another interesting finding is that annual Q/P_{sur} values for the Hòa Bình catchment continuously exceed 100%, whereas the opposite is observed for the adjacent Yên Bái catchment. In combination with the satisfactory agreement between P, ET_{act} and Q data in other catchments, and in the absence of any notable interbasin transfers, this phenomenon may be partly explained by groundwater flow from the Thao basin to the Đà basin.

To compare monthly Q and P_{sur}, Table 3.7 lists the slope and R^2 obtained from linear regression between both variables. For the entire gauged portion of the Red River Basin (upstream of Sơn Tây), 43% of all rainfall surplus is converted to surface

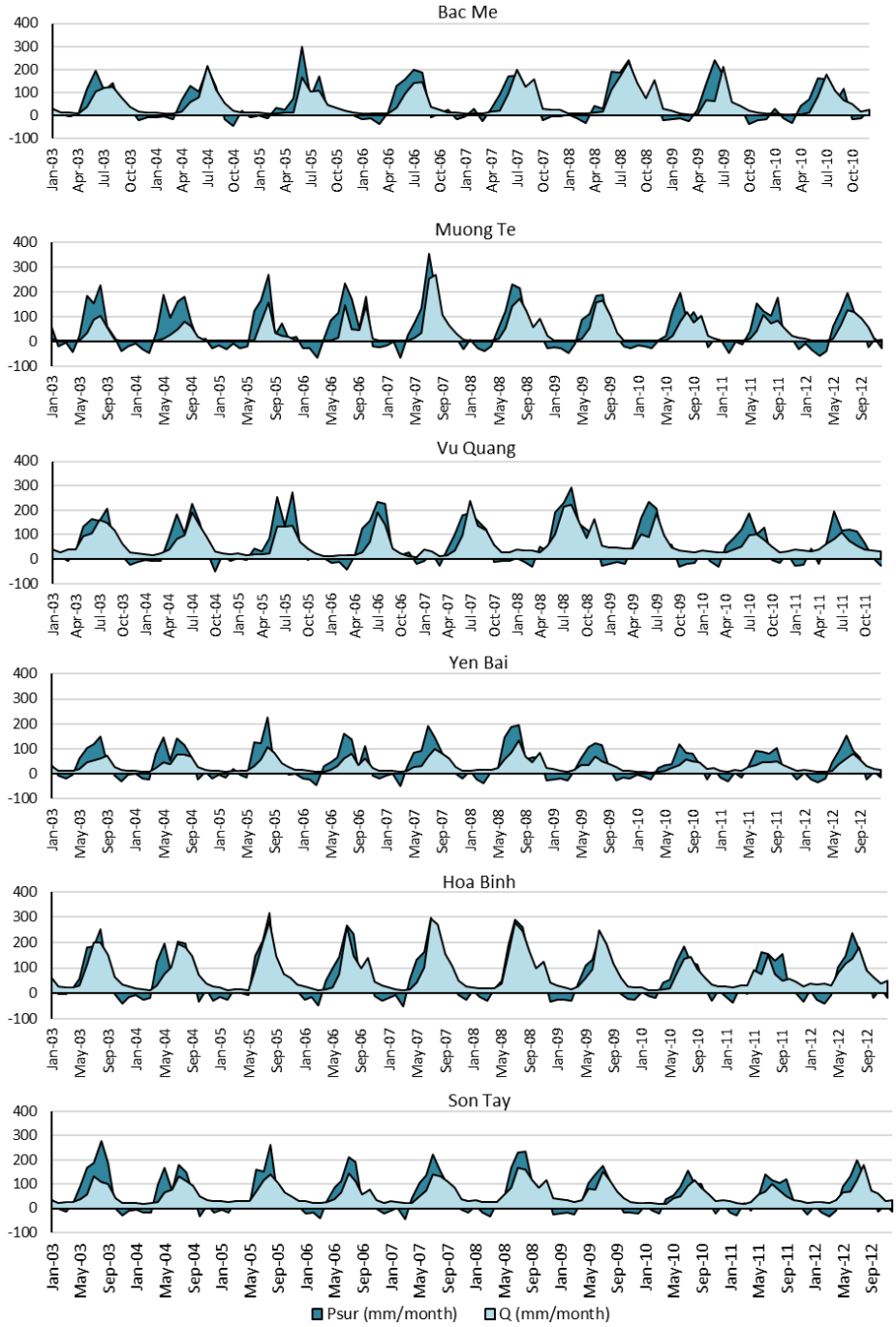

Figure 3.10. Graphs of upstream rainfall surplus (P_{sur}) from remote sensing and measured streamflow (Q) for each of the available discharge stations.

Table 3.7. Comparison of streamflow (Q) and rainfall surplus (P_{sur}) for each catchment, with slopes and R^2 for linear relationships between monthly Q and upstream P_{sur}. Also given are the values of Q/P_{sur} during the rainy season. All values represent the 2003-2012 period.

Station	Q/P_{sur} (%)	Q / P_{sur} slope (-)	R^2	Monsoonal Q/P_{sur}				
				May	*Jun*	*Jul*	*Aug*	*Sep*
Bac Me	98.4	0.60	0.66	0.37	0.55	1.07	1.78	1.68
Mường Tè	80.9	0.49	0.61	0.15	0.28	0.61	0.80	1.03
Vũ Quang	101.0	0.50	0.67	0.71	0.54	0.91	0.86	1.57
Yên Bái	87.2	0.33	0.65	0.52	0.43	0.45	0.61	1.39
Hòa Bình	125.9	0.65	0.72	0.59	0.71	0.94	0.98	2.21
Sơn Tây	107.2	0.43	0.71	0.74	0.56	0.71	0.79	1.43

runoff. The highest Q/P_{sur} value of 0.65 is found for Hòa Bình catchment, whereas only 33% of P_{sur} contributes to surface runoff upstream of Yên Bái. A reason for this difference is likely the catchment topography, with a lower average slope in the upstream catchment of the latter station. Also, average annual P_{sur} is substantially higher in Hòa Bình with 730 mm as opposed to 460 mm for Yên Bái, increasing the frequency of occurrence of saturated conditions in the soil profile.

Although multi-annual Q and P_{sur} are congruent at the subcatchment scale, it is not obvious that a correlation on the monthly scale should be expected. Especially in dry months when the catchment storage is relatively empty, a low and stable Q is observed (likely driven by baseflow) that is not significantly affected by variability in monthly P_{sur}. During wet months, however, the progression of the Q/P_{sur} ratio is representative of the changing response of the catchment to rainfall. Table 3.7 lists average Q/P_{sur} for each of the months in the rainy season. A similar pattern is observed for all catchments, in which the ratio increases as the rainy season progresses and exceeds 1 at the end of monsoon in September. This consistent increase of Q/P_{sur} suggests that in the Red River Basin saturation excess processes are dominant in runoff generation, rather than Hortonian runoff occurring during high-intensity precipitation events (Easton et al., 2012; Liu et al., 2008). Monthly Q/P_{sur} values substantially higher than 1 could occur due to groundwater flow between catchments, or human actions; e.g. when large volumes of water are released from the reservoirs. These releases occur in particular during the monsoonal months, when flood buffering capacity is required and a maximum water level is maintained (Ngo, 2006). Although these management actions are expected to affect Q/P_{sur}, the natural processes of streamflow generation still appear clearly in the figures in Table 3.7.

As correlation between Q and P_{sur} is logically weak for specific months, it is not yet feasible to predict Q for every month solely from remote sensing. This could change when ET_{act} GSDPs come available on a daily basis, which will enable a detailed

investigation of the relation between cumulative P_{sur} from the start of the hydrological year and the Q/P_{sur} term (Easton et al., 2012). However, a clear relation is observed between monthly P_{sur} and ΔS, as the storage capacity of the Red River Basin is not fully satisfied for the major part of the year. Therefore, it is possible to express volumetric ΔS as a function of remotely sensed P_{sur}. Figure 3.11 provides a plot of monthly ΔS versus P_{sur}, upstream of Sơn Tây. A clockwise hysteresis pattern can be observed. Linear models were derived that enable the prediction of ΔS without the need for ground observations. From December until the start of the rainy season in April, the slope of the models is near to 1 with a relatively stable intercept in the order of 23-29 mm, which can be viewed as the contribution of groundwater to streamflow. The slope of the model decreases as storage fills up and the contribution of P_{sur} to Q increases. As P_{sur} values decrease in September and October due to declining rainfall, the low intercept is representative of the rain water from previous months that is now taken out of storage to contribute to streamflow. Errors in the derived models for monsoonal months are partly caused by human interventions in Red River water management, and this approach is expected to work even better in more "natural" river basins.

3.4 Discussion

With the increasing availability of global actual evapotranspiration data in the public domain, in addition to rainfall and land use / land cover, it is now possible to quantify the main components of the water balance for river basins in a distributed

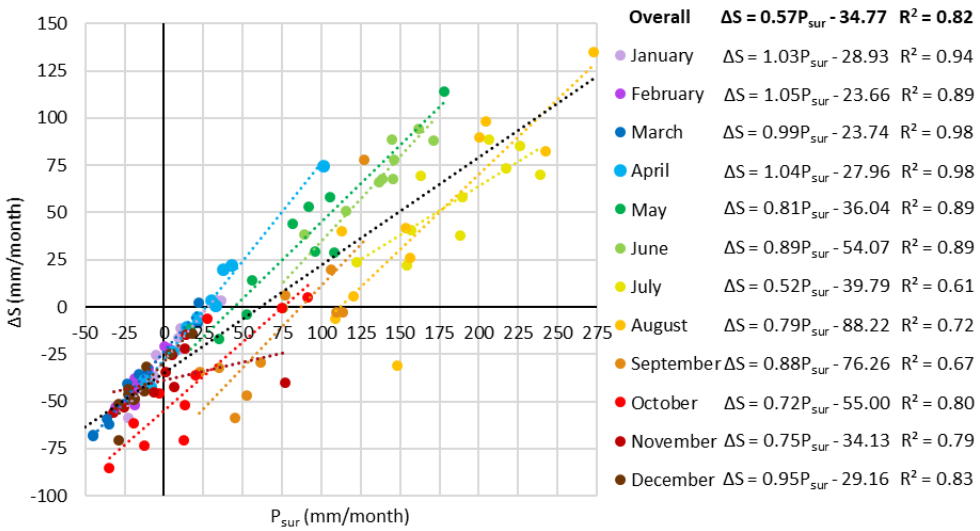

Figure 3.11. Monthly changes in storage upstream of Sơn Tây station (ΔS) plotted against the rainfall surplus (P_{sur}). Dashed lines indicate the lines of best fit for each month (colored) and the entire year (black). For each month, the derived linear model is given on the right with its respective coefficient of determination (R^2).

manner. This chapter shows that rainfall surplus can be successfully computed from global satellite-derived data products for monthly, annual and multi-annual time scales. The total annual water yield of 102.6 km3 computed for the entire Red River Basin is an estimation of long-term river outflow, which is especially valuable because of the lack of streamflow gauges in the Red River Delta (Duc et al., 2011). In non-saturated conditions, spatially distributed monthly P_{sur} is strongly related to changes in storage, and monthly ΔS can thus be quantitatively determined from satellite data. These findings demonstrate that assessments of rainfall surplus from satellite-derived P and ET_{act} can allow for sound water accounting in ungauged river basins that was previously impossible due to missing ground data.

It was found that the SSEBop ET_{act} product succeeds in closing the water balance of the Red River Basin with respect to TRMM rainfall and longer-term streamflow records, while the other products seem to have a tendency to overestimate ET_{act}. The range of average annual ET_{act} values according to different products is found to be rather large (268 mm/yr) and illustrates the need for a thorough comparison. For areas with frequent cloud cover, a part of this range is likely attributed to the various ways in which the ET_{act} algorithms deal with cloud-covered skies and data gaps. The observed difference between the individual models is somewhat inconsistent with the very low errors in satellite-derived ET_{act} that were found in the review by Karimi and Bastiaanssen (Karimi and Bastiaanssen, 2015), which illustrates the current disparity between region-specific ET_{act} estimate with opportunities for parameter-tuning and extractions from global datasets. It was found that for the Red River Basin spatial patterns of MOD16, SSEBop and ALEXI are similar, and this finding has been used to compute the areal ET_{act} patterns from these three ET_{act} products with equal weight. The fundamental differences between a relatively simple, largely LST-based model (SSEBop), an algorithm with more advanced physics incorporating temporal LST variability and a separation between evaporation and transpiration (ALEXI), and a method strongly reliant on LAI (MOD16), support the assumption that the selected models complement each other in terms of performance over a heterogeneous terrain. The consistency between satellite-derived P_{sur} and measured Q in terms of both inter- and intra-annual variability, as well as their agreement for individual subcatchments, put confidence in the constructed ensemble ET_{act} maps.

Previous studies in other basins have yielded differing outcomes regarding the relative performance of the respective ET_{act} algorithms. Therefore, the appropriate choice of models for basin-scale normalization is expected to vary from basin to basin. In future studies, depending on the properties of the river basin at hand, different types of ensemble products may be suitable. It is advised that future research focuses on reviewing the strengths and weaknesses of the ET_{act} GSDPs with respect to different LULC types and climate zones, with the aim to achieve a reliable satellite-derived ET_{act} estimation on the global scale. When doing so, the uncertainties associated with each of the components of the water balance, including streamflow records, should receive sufficient attention. It should be noted that the ET_{act} products applied in this research are in differing stages of

development and substantial progress is to be expected in the next few years. For example, future versions of the ALEXI product will implement microwave-based LST (Holmes et al., 2015) to provide estimates of ET_{act} over all-sky conditions which is particularly important over the Red River basin during persistently cloudy periods. This use of microwave LST will help constrain estimates of ET_{act} during such periods, which currently rely on gap-filling techniques with high uncertainty and are likely responsible for some of the overestimation of ET_{act} seen in this study.

Analyses of global remote sensing products provide a valuable first outlook on the main hydrological processes within a river basin, especially after verification against the longer term total river outflow to ensure mass balance and consistency. Hydrological models are capable of providing complementary information, for example on non-linear sub-soil flow processes that determine runoff, infiltration, storage change, percolation and recharge. These processes govern the partitioning of rainfall surplus into groundwater and surface water. Models also facilitate analyses on a daily time scale, for which only a few ET_{act} GSDPs are currently available. It is already common practice to use satellite-derived information, in particular P and LULC, as inputs to hydrological models. However, results of remote sensing-based quantifications of monthly ET_{act}, P_{sur} and ΔS, as well as multi-annual Q can also be used to train and constrain hydrological models and water management decision tools. Examples are already available in which remotely-sensed ET_{act} is used to constrain hydrological models, or for calibration purposes (Carroll et al., 2015; Cheema et al., 2014; Immerzeel and Droogers, 2008; Livneh and Lettenmaier, 2012; Muthuwatta et al., 2009; Vervoort et al., 2014; Winsemius et al., 2008). P minus ET_{act} appears to be highly correlated with the root zone storage capacity (Wang-Erlandsson et al., 2016). By using satellite-derived information as a reality check, model performance can be improved. This is in particular relevant in areas with abundant water withdrawals, which require a lot of assumptions to simulate but are implicitly included in remotely-sensed ET_{act} (van Eekelen et al., 2015).

Currently, much attention goes out to the development of global hydrological models (GHMs). Several reviews of the current state of art were recently published (Bierkens, 2015; Döll et al., 2015; Sood and Smakhtin, 2014). There are even on-going attempts to create the first operational, hyper-resolution GHM (Bierkens et al., 2015). Integration with remote sensing is identified as one of the promising trends in GHMs to reduce uncertainties (Sood and Smakhtin, 2014). The latest generation of GHMs is capable of spatially explicit assessments of the consumed fraction of applied irrigation water, thus no longer requiring an estimate of efficiencies as input (Jägermeyr et al., 2015; Wada et al., 2014). However, these models still quantify water withdrawals for irrigation by supplying water until optimal growing conditions are achieved, an approach that is likely to lead to an overestimation of withdrawals (Peña-Arancibia et al., 2016). Alternatively, non-physically based statistical methods are used to quantify water withdrawals for different water using sectors (Vandecasteele et al., 2014; Yamada et al., 2012). With ET_{act} maps now readily available on the global scale, it is a logical next step to start

incorporating these products in GHMs, either as model constraints or in the calibration procedure. This could lead to a more realistic representation of withdrawals (Droogers et al., 2010; Peña-Arancibia et al., 2016; Santos et al., 2008), and therefore of non-consumed water and reuse.

3.5 Conclusions

This chapter demonstrates how an integration of readily available global satellite-derived data products can shed light on river basin hydrology. With the availability of rainfall (P), land use / land cover (LULC) and the newly available actual evapotranspiration (ET_{act}) data on the global scale, such analyses can now be performed for all river basins as pre-analyses to numerical hydrology studies. The consistency between different P and ET_{act} products and downstream river discharge should first be evaluated by applying the law of mass conservation on the multi-annual scale. Even for a challenging basin in terms of atmospheric conditions such as the Red River Basin, satisfactory and meaningful conclusions were drawn. Average annual water yield of the basin is 102.6 km^3, of which 29% is generated in China. Forests were found to be the main water producer, while also irrigated cropland is not a net water consumer on the annual scale. In addition, it proved possible to model monthly storage changes solely based on satellite-derived P and ET_{act}. The ratio of streamflow (Q) over rainfall surplus (P_{sur}) increases steadily during the rainy season, signifying the importance of saturation excess processes in runoff generation. This is a first step into determining the partitioning between fast surface runoff and slow groundwater runoff.

Although our comparison for the Red River shows that the range between values of individual evapotranspiration products is still substantial, it is concluded that there is a high potential for applying monthly remotely sensed ET_{act}, P_{sur}, storage changes and multi-annual Q to constrain or calibrate hydrological models. This facilitates quantification of hydrological processes that take place on the daily or weekly time scale, or processes that cannot be assessed by remote sensing alone. These include indirect water reuse of return flows, which are transported downstream through both surface water and groundwater pathways. Further studies are required to examine the performance of the ET_{act} products for different geographical regions, climate zones and land use types, in order to ultimately facilitate the coupling between these products and (global) hydrological models. In the meantime, it is concluded that the proposed methodology based on spatial correlations among individual ET_{act} products and absolute calibration of longer-term P - Q has a satisfactory performance under the conditions encountered in the Red River Basin.

4

A NOVEL METHOD TO QUANTIFY CONSUMED FRACTIONS AND NON-CONSUMPTIVE USE OF IRRIGATION WATER: APPLICATION TO THE INDUS BASIN IRRIGATION SYSTEM OF PAKISTAN

In this chapter, we demonstrate a novel method for spatial quantification of the Consumed Fraction (CF) of withdrawn irrigation water based on satellite remote sensing and the Budyko Hypothesis. This method was applied to evaluate consumption of irrigation water (ET_{blue}), total water supply, and non-consumptive use across the Indus Basin Irrigation System (IBIS) of Pakistan. An average ET_{blue} of 707 mm/yr from irrigated cropland was found for 2004 - 2012, with values per Canal Command Area (CCA) varying from 421 mm/yr to 1,011 mm/yr. Although canal supply (662 mm/yr on average) in most CCAs was largely sufficient to sustain ET_{blue}, a similar volume of additional pumping (690 mm/yr) was required to comply with hydro-climatological principles prescribed by Budyko theory. CF values between 0.38 and 0.66 were computed at CCA level, with an average value of 0.52. Co-occurrence of relatively low CF values, high additional water supply, and long-term canal diversions similar to ETblue, implies that the IBIS is characterized by extensive reuse of non-consumed flows within CCAs. In addition, the notably higher CF of 0.71 - 0.93 of the full IBIS indicates that return flow reuse between CCAs cannot be neglected. These conclusions imply that the IBIS network of irrigators is adapted to extensively recover and reuse drainage flows on different spatial scales. Water saving and efficiency enhancement measures should therefore be implemented with great caution. By relying on globally available satellite products and limited additional data, this novel method to determine Consumed Fractions and non-consumed flows can support policy makers worldwide to make irrigation systems more efficient without detriment to downstream users.

Chapter based on Simons, G.W.H., Bastiaanssen, W.G.M., Cheema, M.J.M., Ahmad, B., 2020. A novel method to quantify consumed fractions and non-consumptive use of irrigation water: application to the Indus Basin Irrigation System of Pakistan. Agricultural Water Management, 236; doi:10.1016/j.agwat.2020.106174.

4.1 Introduction

Pressure on water resources is expected to increase in many of the world's river basins due to population growth and the associated increase in demand for food, fiber and biofuels. Changing precipitation, evapotranspiration and carbon fluxes are projected to further exacerbate water shortages. Recent policy reports and development programs supported by global institutions, as well as scientific and popular articles, promote irrigation efficiency improvements as a solution to water scarcity (e.g. World Bank, 2016; Siyal et al., 2016; Sultana et al., 2016; USAID, 2016). This perspective contradicts, however, with the growing body of work conveying the notion that aiming for more efficient water use in agriculture will not solve the water crisis (FAO, 2017; Grafton et al., 2018; Lankford, 2012; Perry, 2011).

The latter studies address the paradoxical effect of intended water savings having adverse effects, by in fact boosting water consumption (Scott et al. 2014). This efficiency paradox occurs when farmers find new use for the "freed up" water, by expanding irrigated areas, introducing new crops with higher water requirements, or switching from deficit to full irrigation (Berbel et al., 2015; Gómez and Pérez-Blanco, 2014; Sanchis-Ibor et al., 2017). By now, the occurrence of this phenomenon, its preconditions, and implications, have been well-described in a large number of case studies (e.g. Pfeiffer and Lin 2014; Contor and Taylor, 2013; Lecina et al., 2010; Rodriguez Díaz et al., 2012; Ward and Pulido-Velazquez, 2008). When no policy mechanisms are in place that incentivize farmers to reduce withdrawals or restrict either irrigated area or consumptive water use, there is a high risk of efficiency-enhancing measures leading to reduced non-consumed flows (i.e., return flows).

For effective planning of irrigation technology improvements and policies, it is therefore essential to understand the dependencies between water users (anthropogenic as well as natural) across a river basin. Reuse of non-consumed flows within and between sectors is facilitated by both natural pathways and human interventions, and results in a complex interplay between surface water and groundwater flows (Grogan et al., 2017). Intensity and complexity of reuse networks typically increase with scale (Simons et al., 2015; Wu et al., 2019b). Environmental flow requirements of downstream ecosystems are often neglected, while their vulnerability to changes in agricultural non-consumed flows is potentially very high (Carrillo-Guerrero et al., 2013; Pastor et al., 2014).

As the conclusion of a literature review on impacts of drip irrigation introduction, Van der Kooij et al. (2013) called for an increased awareness of the scale-dependency of efficiencies and unintended re-allocations of water flows. To achieve this objective and to account for spatial tradeoffs in policies and regulations, quantitative data on consumed and non-consumed portions of withdrawals are required. Quantifying consumed fractions on different scales would support assessments of the likely scope for water saving by irrigation modernization or policy alterations (Berbel and Mateos, 2014). In addition, it would support implementation of evapotranspiration caps in water rights systems, a key policy

instrument to ensure water availability to downstream users (e.g. Dagnino and Ward, 2012; Bastiaanssen et al., 2008).

Data availability is currently a major limiting factor in the uptake of existing water reuse frameworks and indicators (Simons et al., 2015). Wiener et al. (2016) demonstrated how water reuse can be well-characterized for a watershed where extensive records of withdrawals, consumptive use and non-consumed flows are available. This is, however, not the case for most river basins. Governmental line agencies are struggling with the quantitative assessment of consumed fractions. Estimates of consumed fractions are therefore commonly limited to static literature values assumed at country level based on prevailing irrigation types, despite spatially varying biophysical factors having significant effects (Jägermeyr et al., 2015). Plot-level efficiency measured in an experimental setting remains the main source of quantitative information (Bos et al., 2005; Bos and Nugteren, 1990). However, simply extrapolating these values to larger spatial scales can lead to misunderstanding and mismanagement (Merks, 2018; Molden and Sakthivadivel, 1999).

By definition, an assessment of consumed fractions in an irrigation context requires estimates of (i) the volume of water that is withdrawn for irrigation, and (ii) the fraction of this water that evaporates. To quantify the latter, over the past years the scientific community has turned to satellite remote sensing. Global satellite-derived data products can provide spatiotemporal insight in key hydrological parameters such as precipitation, actual evapotranspiration, soil moisture changes, runoff and storage change (Bastiaanssen and Harshadeep, 2005; Poortinga et al., 2017; Gijs Simons et al., 2016). Local estimates of consumed irrigation water can for example be obtained by analyzing evapotranspiration of nearby sites with similar land use, but known to be solely rainfed (van Eekelen et al., 2015). As satellites cannot measure water withdrawals, coupling remote sensing with simulation models has been explored for evaluating irrigation dynamics (Droogers et al., 2010; Peña-Arancibia et al., 2016; Santos et al., 2008). Promising results were achieved, but site-specific calibration remains necessary, prohibiting an easily scalable monitoring approach. In addition, some global-scale hydrological models compute consumed fractions by partitioning irrigation water into consumed and non-consumed flows (e.g. Jägermeyr et al. 2015). This enables scenario studies on the global scale, but applicability for basin-level monitoring purposes remains limited.

Application of the Budyko Hypothesis (Budyko, 1974) is an approach that has not yet been pursued by the scientific community for quantifying consumptive use of irrigation water. The Budyko curve prescribes the theoretical partitioning of precipitation into streamflow and evapotranspiration based on water and energy climatologies. It has frequently been applied successfully for purposes of developing, constraining and validating water balance models (e.g. Zhang et al. 2008; Gentine et al. 2012; Chen et al. 2013; Poortinga et al. 2017). Although initially developed for natural river basins in dynamic equilibrium and with precipitation as the sole source of water supply, derivatives of the original Budyko approach have

recently been successfully applied to evaluate the water balance of systems with anthropogenic supply or storage of water (e.g. Du et al. 2016; Greve et al. 2016; Gunkel and Lange 2017; Tang et al. 2017).

In this chapter, we present a novel method for quantifying consumed fractions of irrigation systems based on Budyko theory and satellite-derived data products of evapotranspiration and precipitation. The approach is demonstrated by describing its application to the Indus Basin Irrigation System, which is the largest continuous irrigation system in the world. Consumptive use, irrigation water supply and non-consumed flows are presented, and findings are discussed in relation to water reuse and water saving potential.

4.2 Materials and methods

4.2.1 *Study area*

This study focuses on the Pakistani part of the Indus Basin Irrigation System (IBIS), excluding the Canal Command Areas (CCAs) upstream of Jinnah Barrage (Figure 4.1). IBIS receives its water mainly from snow melt and glacial waters in the upstream high-mountain areas of the Himalayas, Karakoram and Hindu Kush (Immerzeel et al., 2010), as well as from extraordinary rainfall falling on the windward slopes of the Himalayan mountains. The major part of IBIS surface area has an arid climate and rainfall in catchment areas is a secondary source of water. The monsoonal regime causes rainfall during the dry *rabi* season, in the months November to April, to be only 30% of that in the rainy *kharif* season, from May to October (Habib, 2004). Surface water flow is concentrated in the Indus River and its tributaries Jhelum, Chenab, Ravi, Sutlej and Kabul. Water is buffered and distributed by a system comprising 3 major reservoirs, 18 barrages and headworks, 2 major siphons, and 12 inter-river link canals, serving a gross irrigable command area of over 16 million hectares in total (Qureshi, 2011). After extensive consumptive use for irrigation and, to a far lesser extent, municipal and industrial purposes, remaining streamflow downstream of the IBIS supports the rich diversity of vegetation and wildlife of the Indus Delta, where the Indus River eventually drains into the Arabian Sea. Annual environmental flow requirements are in place to combat inundation, sea water intrusion and coastal erosion (Kalhoro et al., 2016). Drainage flows, largely of poor quality, are also transported out of the system to evaporation ponds, or directly to the sea through the Left Bank Outflow Drainage (LBOD) canal (Basharat and Rizvi, 2016).

Cropping intensities in the IBIS have increased over the past decades and crop water requirements are, at the system scale, not fulfilled by the sum of surface water withdrawals and rainfall (Ullah et al., 2001). This discrepancy between water supply and demand is especially experienced by tail-end farmers, who typically have 32% less water available than head-end farmers (Qureshi et al., 2010). Inadequacy and unreliability of surface water supply has driven farmers to augment water shortages by pumping groundwater resources. Reported amounts

vary from 52 to 61 km³/yr, approaching the volume of annually replenished groundwater of 55 - 63 km³/yr (Laghari et al., 2012; PBS, 2014; Watto and Mugera, 2016). Falling groundwater tables are observed in areas with fresh groundwater, most notably in the northeastern part of the province of Punjab (Mekonnen et al., 2015). Particularly Eastern Punjab is a hotspot of groundwater depletion, with water table decline possibly exacerbated by transboundary impacts from extensive groundwater pumping across the Indian border (Cheema et al., 2014; Iqbal et al., 2017; Watto and Mugera, 2016). The situation is different in Sindh Province, where groundwater quality is generally marginal to hazardous and groundwater abstractions only constitute 4 - 8% of total water use (Qureshi et al., 2008; van Steenbergen et al., 2015; Young et al., 2019). Structural waterlogging is a serious problem here, with over half of all CCA surface area increasingly affected by shallow water tables due to high surface water supplies and a low level of groundwater pumping, as well as poorly functioning drainage facilities and salinization (Basharat and Rizvi, 2016; van Steenbergen et al., 2015).

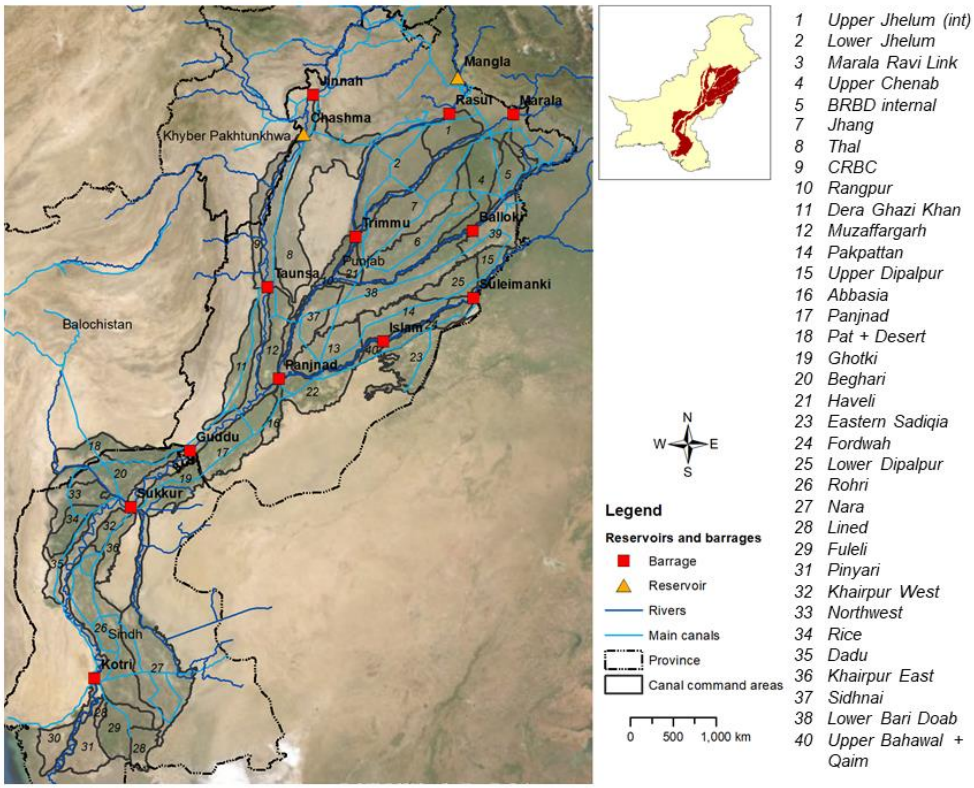

Figure 4.1. Indus Basin Irrigation System in Pakistan and its canal command areas.

4.2.2 *Analytical framework and calculation steps*

The conceptual framework proposed by Simons et al. (2015) is followed in this study, thus defining the Consumed Fraction (*CF*) as the ratio between consumptive use of irrigation water and total water withdrawal. Following the common definitions of *green* and *blue* water (Falkenmark and Rockström, 2006), the component of actual evapotranspiration (ET_{act}) from surface or groundwater resource is denoted as ET_{blue}, and rain-dependent ET_{act} is termed ET_{green}:

$$ET_{act} = ET_{green} + ET_{blue} \tag{4.1}$$

Note that ET_{blue} is also referred to as incremental evapotranspiration (Hoogeveen et al., 2015), or secondary evaporation (Van Dijk et al., 2018). ET_{green} is referred to as net precipitation in classical formulations of irrigation water requirements (Jensen and Allen, 2016). The equation for computing *CF* then becomes:

$$CF = \frac{ET_{blue}}{Q_w} \tag{4.2}$$

where Q_w comprises withdrawals from surface water and/or groundwater for irrigation. In the context of an IBIS CCA, it is relevant to distinguish two types of inflow:

$$Q_w = Q_{div} + Q_{add} \tag{4.3}$$

where Q_{div} represents the volume of surface water diverted at the main canal head. Q_{add} comprises additional sources of water, such as local non-consumed flows that are pumped up, fossil groundwater abstraction, or drainage water from upstream CCAs entering through surface or sub-surface pathways other than the main canal.

The non-consumed portion of applied irrigation water is then calculated as the difference between total blue water supply and consumptive use of irrigation water:

$$Q_{nc} = Q_w - ET_{blue} \tag{4.4}$$

The proposed procedure for partitioning ET_{act} into ET_{green} and ET_{blue} is based on the Budyko Hypothesis (BH), which describes an empirical relation between ET_{act}, reference evapotranspiration (ET_0) and precipitation (P) for areas in dynamic equilibrium and with negligible storage changes (Sposito, 2017). The original Budyko equation has been reformulated several times in order to account for systematic differences between watersheds. This study applies the commonly used Budyko reformulation derived by Fu (1981):

$$\frac{ET_{green}}{P} = 1 + \frac{ET_0}{P} - \left(1 + \left(\frac{ET_0}{P}\right)^{\omega}\right)^{\frac{1}{\omega}} \tag{4.5}$$

where ω is a free parameter that describes the shape of the Budyko curve. ω can be viewed as an integrated catchment characteristic, determined by catchment-specific properties such as climate, land cover and soil hydraulics (Condon and Maxwell, 2017). Higher ω values indicate a higher ET_{green} under the same ET_0 / P ratio (the aridity index), and are thus related to a greater capacity of a basin to retain water for evapotranspiration. Gunkel and Lange (2017) reviewed previous studies to find a range of ω values between approximately 1 and 5, where values below 2 are generally observed for larger, drier basins such as the Lower Indus.

In many river basins the original BH assumptions are nowadays violated by extensive human influence on the water balance, e.g. by irrigation or interbasin transfers. However, Du et al. (2016) demonstrated that accounting for alternative water sources such as canal water supply and storage changes, in addition to precipitation, can allow for application of the BH in arid, irrigated regions on longer time scales. For the irrigated IBIS, on a multi-annual time scale under the assumption of zero storage changes, this means that:

$$\frac{ET_{act}}{P_{adj}} = 1 + \frac{ET_{ref}}{P_{adj}} - \left(1 + \left(\frac{ET_{ref}}{P_{adj}}\right)^{\omega}\right)^{\frac{1}{\omega}} \tag{4.6}$$

where:

$$P_{adj} = P + Q_w \tag{4.7}$$

Based on spatially distributed P, ET_0 and ω data (see Section 4.2.3), eq. (4.5) can be solved for ET_{green}. By subtracting computed ET_{green} from satellite-derived ET_{act}, ET_{blue} can be calculated as the portion of consumptive water use that cannot be accounted for by rainfall according to the BH (Figure 4.2, left panel). Under the assumption that eq. (4.4) is valid at the pixel scale (Viola et al., 2017), this step yields spatial data of both rainfall- and irrigation-dependent ET. Subsequently, in order to estimate the supply side of CF, eq. (4.6) is applied to find the value of P_{adj} for which ET_{act} / P_{adj} equals the theoretical value of this ratio prescribed by Budyko theory, as illustrated in the right panel of Figure 4.2. In this case, 1 - ET_{act} / P_{adj} equals the runoff fraction R_f. Subtracting P from the comprehensive supply term P_{adj}, then, yields the estimate of Q_w required for quantifying CF (eq. 4.2). If records of Q_{div} are available, Q_{add} can be computed by applying eq. (4.3) to explore reuse of water and (unsustainable) groundwater pumping. An overview of the full approach is presented in Figure 4.3.

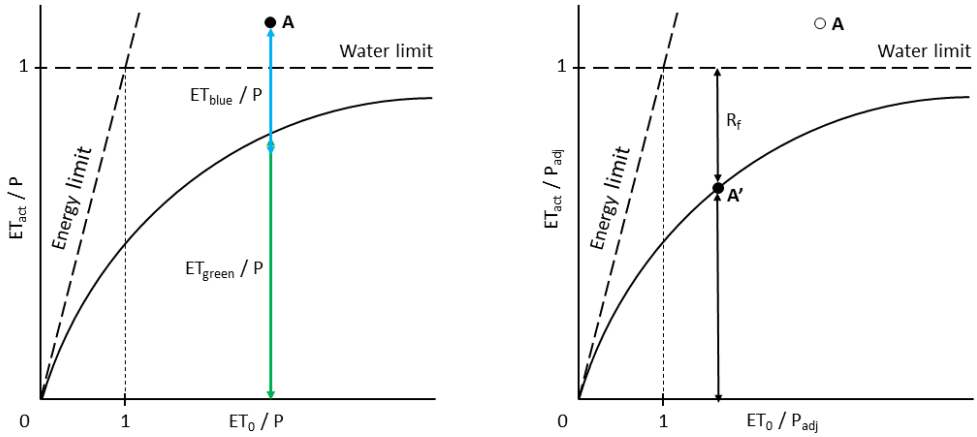

Figure 4.2. Location of an irrigated basin (A) in the Budyko framework when considering rainfall (*P*) as the sole term on the supply side (left), and its new location A' on the Budyko curve when considering all sources of water (P_{adj}, right). ET_{act} and ET_0 refer to actual and reference evapotranspiration, respectively.

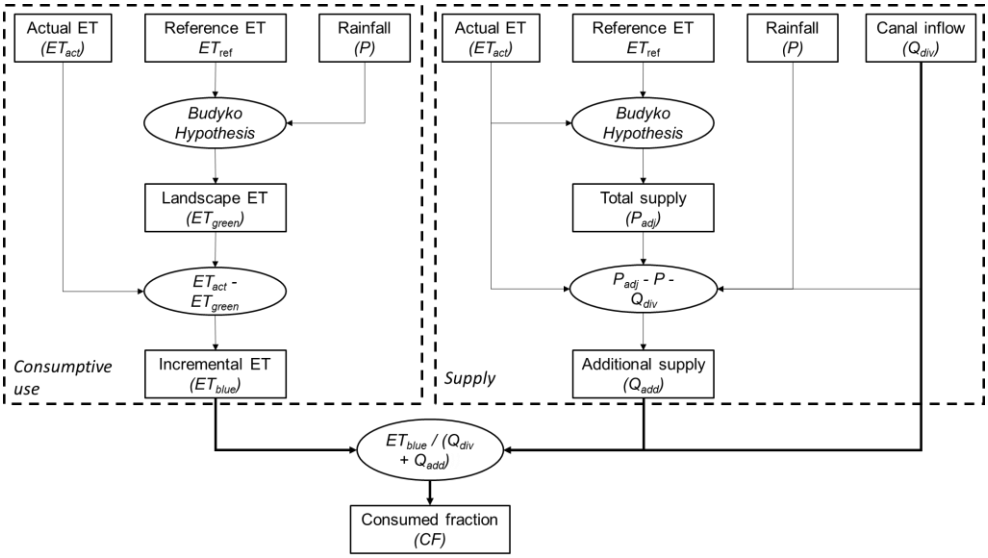

Figure 4.3. Analytical framework and calculation steps.

4.2.3 *Datasets*

This study uses ET_{act} data for 2004 - 2012 from the Operational Simplified Surface Energy Balance (SSEBop) v4 model, which is one of several global-scale satellite-derived ET_{act} products available in the public domain (Senay, 2018; Senay et al., 2013). SSEBop is a surface energy balance model that calculates the latent heat flux from land surface temperature measured by the satellite-based MODIS sensor. It is based on pixel-specific pre-defined temperature differences between cold (wet) and hot (dry) conditions, where air temperature from climate models is used as an indicator for the coldest land surface temperature. The performance of SSEBop relative to other global ET_{act} products and field measurements has been evaluated in multiple studies, and has been generally found favorable (e.g. Simons et al 2016; FAO 2019). Another reason for using SSEBop in this study is the availability of a corresponding ET_0 product in the public domain, which ensures consistency between ET_0 and ET_{act} as required for BH application.

Although SSEBop performance in terms of spatial and temporal dynamics has previously been found satisfactory, systematic biases can occur depending on the region of interest and the algorithm should be calibrated based on auxiliary data (Senay, 2018). This relates to the use of a "maximum ET scaling factor" (K) in the SSEBop algorithm, which depends on the aerodynamic roughness, the degree of advection and prevailing weather conditions, among others. Based on independent estimates of ET_{act}, e.g. from field experiments or the conservation of water mass at the river basin scale, a potential bias correction of the global SSEBop product in a river basin of interest is recommended.

In this study, we take the approach of inventorying previous efforts to quantify ET_{act} in the IBIS, and correcting long-term SSEBop ET_{act} for these values. Several previous studies have been performed in the Indus Basin, applying locally calibrated models to assess water consumption of irrigated crops. Next to ET_{act} and ET_0, data on rainfall and the Budyko ω parameter are required for application of the BH. Monthly rainfall data at ~5km resolution were obtained from the quasi-global satellite-derived Climate Hazards Group InfraRed Precipitation with Stations (CHIRPS) v2.0 dataset (Funk et al., 2015). For Pakistan in 2004 - 2012, data from approximately 35 rainfall stations are incorporated in the CHIRPS algorithm to enhance satellite rainfall estimates. Data on ω were acquired from the study by Xu et al. (2013), who produced spatially discrete data on ω on the global scale using a Neural Network model fed by ET_{act}, ET_0, P and streamflow data for 256 river basins. Finally, monthly data on canal diversions and reservoir releases, required for partitioning calculated withdrawals into Q_{div} and Q_{add}, were made available by the Water And Power Development Agency of Pakistan (WAPDA) for the years 2004 - 2012.

Table 4.1 presents the identified studies quantifying annual ET_{act} for at least a part of the IBIS. Based on the values presented in these studies and SSEBop values for the corresponding years and areas, a correction factor of 0.78 was applied to the

original global SSEBop data to correct for overestimation. This linear bias correction is justified due to the linear relation of K to ET_{act} in the SSEBop formulation.

Next to ET_{act} and ET_0, data on rainfall and the Budyko ω parameter are required for application of the BH. Monthly rainfall data at ~5km resolution were obtained from the quasi-global satellite-derived Climate Hazards Group InfraRed Precipitation with Stations (CHIRPS) v2.0 dataset (Funk et al., 2015). For Pakistan in 2004 - 2012, data from approximately 35 rainfall stations are incorporated in the CHIRPS algorithm to enhance satellite rainfall estimates. Data on ω were acquired from the study by Xu et al. (2013), who produced spatially discrete data on ω on the global scale using a Neural Network model fed by ET_{act}, ET_0, P and streamflow data for 256 river basins. Finally, monthly data on canal diversions and reservoir releases, required for partitioning calculated withdrawals into Q_{div} and Q_{add}, were made available by the Water And Power Development Agency of Pakistan (WAPDA) for the years 2004 - 2012.

Table 4.1. Overview of different actual evapotranspiration (ET_{act}) studies and SSEBop values for corresponding areas and periods. The *SSEBop_cor* column presents ET_{act} values after correction with a factor of 0.78.

Area	Period	Literature ET_{act} mm/yr	Source	*SSEBop* mm/yr	*SSEBop_cor* mm/yr
Lower Chenab	2005 - 2012	793	Usman et al., 2015	1,145	893
	2005 - 2011	853	Awan and Ismaeel, 2014	1,150	897
Hakra	2008 - 2014	963	Liaqat et al., 2016	1,112	868
All CCAs	2009 - 2010	854 – 1,208*	Liaqat et al., 2015	656 – 1,257	512 - 980
Entire IBIS - irrigated (incl India)	2007	974	Bastiaanssen et al., 2012	1,198	934
Pakistani IBIS	1993 - 1994	970	Bastiaanssen et al., 2002, 2003	-	-
	2001 - 2002	850	Ahmad et al., 2009	-	-
	2004 - 2012	-	-	1,187	926

**This study only provides annual ET_{act} averages at the CCA level. Listed values are minimum and maximum.*

4.3 Results and discussion

4.3.1 *Evapotranspiration of irrigation water*

Figure 4.4 shows the position of the 40 IBIS CCAs in Budyko space, based on area-averaged values of mean annual ET_{act}, ET_0 and P over the period May 2004 - April 2012 (eight full hydrological years). Each of the CCAs has a unique theoretical Budyko curve depending on ω. For reference, Figure 4.4 presents the curves corresponding with minimum and maximum ω at the CCA level, as well as one for the average ω value for entire IBIS. All CCAs are located well above the Budyko curves. It should be noted that the y-axis is plotted on a logarithmic scale (base 2) to account for the relatively large distances to the theoretical curves. The arid climate in the IBIS is demonstrated by the high aridity indices plotted on the x-axis, with CCAs in Punjab generally having lower ET_{act}/P and ET_0/P values than those in Sindh[4]. This is caused by the northeast-southwest rainfall gradient in IBIS. Overall, given Budyko theory, Figure 4.4 matches expectations with regards to an irrigated system, as for none of the CCAs the rate of water consumption can be explained by natural water supply through rainfall. The theoretical lines in Figure 4 can be used to infer the ET_{act} value associated with P, i.e. ET_{green} in eq. (4.5).

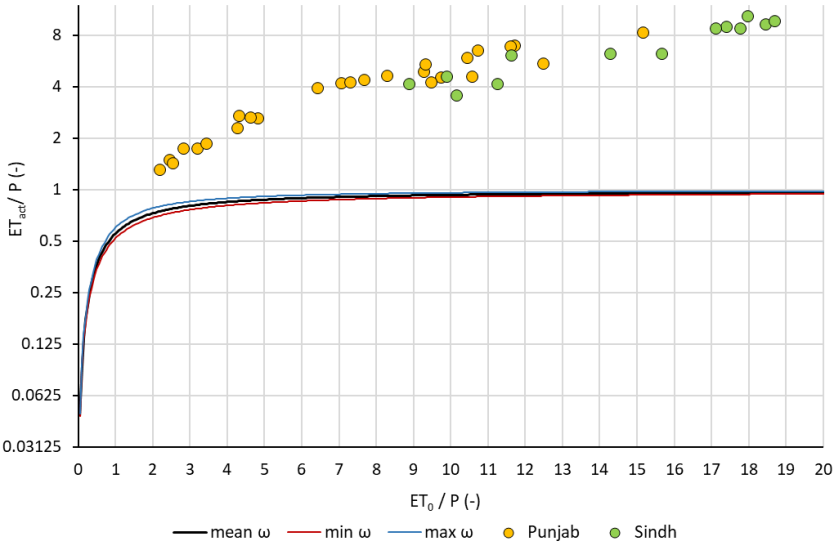

Figure 4.4. Ratios between actual evapotranspiration (ET_{act}) and precipitation (P), and between reference evapotranspiration (ET_0) and P, of IBIS canal command areas in 2004 – 2012. Depicted Budyko curves are based on ω values of 1.88, 1.76, and 2.05; the command area-level mean, minimum, and maximum values, respectively.

[4] *"Punjab" CCAs include CRBC, partly located in KPK province. Northwest and Pat+Desert CCAs, (partly) located in Balochistan, are grouped under "Sindh" CCAs.*

According to the theoretical concept illustrated in Figure 4.2, ET_{act} can now be partitioned into ET_{blue} and ET_{green} for each CCA, based on the distance to the CCA-specific theoretical Budyko curves. Figure 4.5 shows the resulting maps of annual ET_{blue} and ET_{green}, averaged for 2004 - 2012. Whereas annual ET_{green} follows a relatively smooth spatial pattern corresponding with the rainfall gradient, ET_{blue} is much more heterogeneous and depends on e.g. crop type, canal operations, groundwater pumping behavior, soil salinity and groundwater quality. High values for ET_{blue} are particularly observed in the central part of IBIS and in southern Sindh, particularly in areas close to the main river. Locally, values of over 1,200 mm of annual ET_{blue} occur in Rohri, Lined, and Khaipur West CCAs. Low ET_{blue} values approaching zero are found at the edges of many CCAs where irrigation is absent, and further from the main canal inlets. It is striking that a major part of Thal CCA surface area has negligible ET_{blue}, which corresponds with the large extent of rainfed agriculture in this CCA reported by the land use / land cover map of Cheema and Bastiaanssen (2010).

Figure 4.6 presents CCA-level averages for annual ET_{blue}, ET_{green}, and shows that ET_{green} generally follows the variability of CCA-averaged rainfall amounts precipitation, as is to be expected. Several CCAs in Punjab depend on rainfall for a substantial portion of their water consumption, with ET_{green} in four CCAs (Marala Ravi Link, Thal, Upper Jhelum, BRBD internal) accounting for over half of total ET_{act}. This is very different in Sindh Province, where annual ET_{green} for all CCAs is at 25% of total ET_{act}, or less. Here, arid conditions require supply of high volumes of irrigation water to satisfy crop water requirements. In Punjab Province, annual

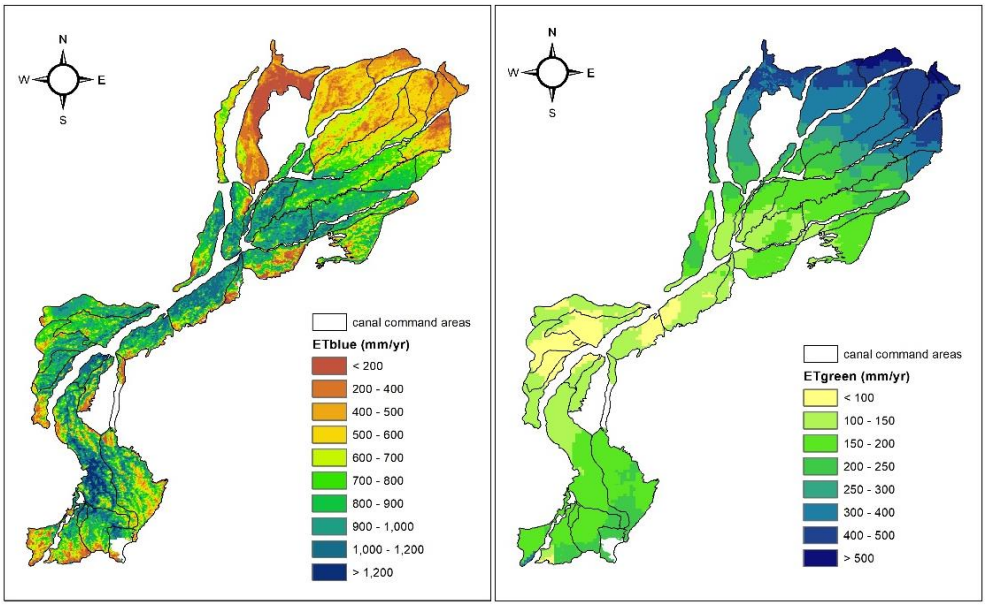

Figure 4.5. Annual blue water evapotranspiration (ET_{blue}, left) and green water evapotranspiration (ET_{green}, right) across the IBIS, averaged for 2004 – 2012.

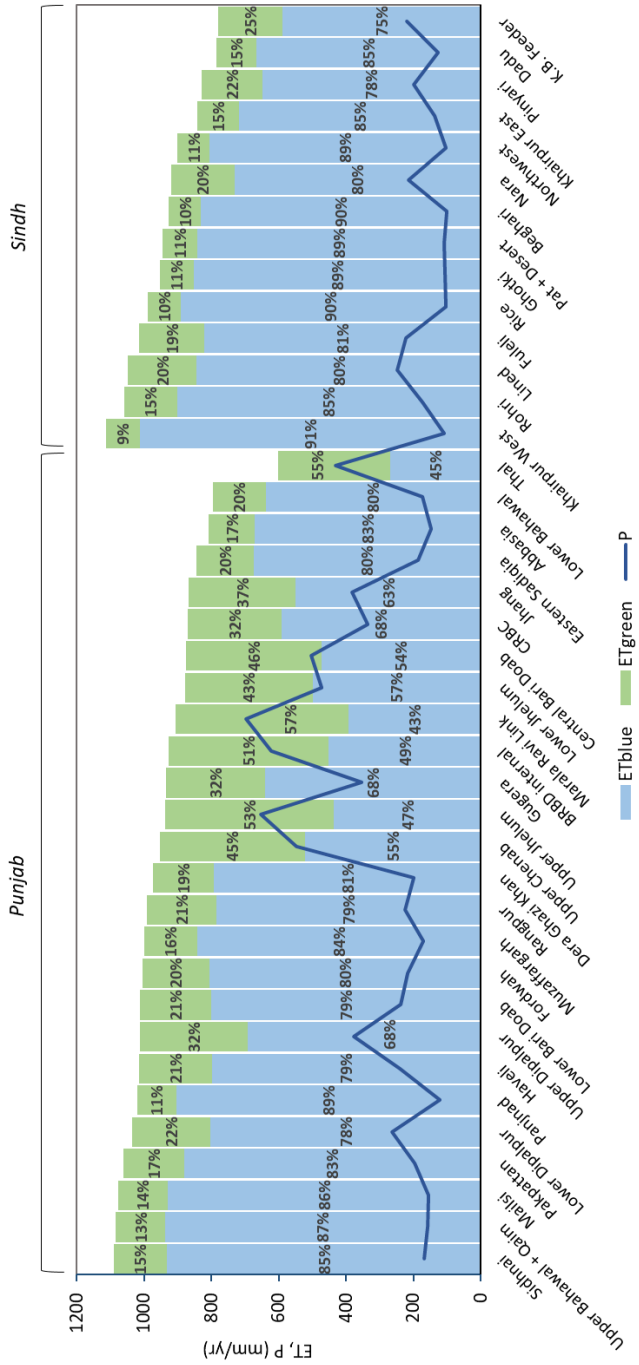

Figure 4.6. Annual average blue water evapotranspiration (ET_{blue}), green water evapotranspiration (ET_{green}), and precipitation (P) for each canal command area. Percentages represent ET_{blue} and ET_{green} amounts with respect to total ET.

ET_{blue} values vary between 268 mm/yr (Thal) and 937 mm/yr (Upper Bahawal + Qaim). In Sindh, minimum and maximum annual ET_{blue} is 588 mm/yr (K.B. Feeder) and 1,011 mm/yr (Khaipur West), respectively.

In Table 4.2, ET_{blue} and ET_{green} results are aggregated for provinces, as well as for the agro-climatic zones distinguished by Ullah et al (2001). Relatively low ET_{act} values in the mixed cropping zone can be explained by cultivation of (fruit) crops that are less water-demanding and by high seepage due to presence of sandy soils (Liaqat et al., 2015; Ullah et al., 2001). The table shows how, despite similar overall ET_{act} values at the provincial level, the relative attribution of this consumed water to rainfall and additional irrigation water differs substantially between the provinces. The ratio of ET_{green} over P presented in the far-right column of Table 4.2 can be viewed as the percentage of effective rainfall, which on the annual scale for the entire IBIS amounts to 85%. It should be noted that presented values do not include "unofficial" irrigation outside CCA boundaries, and that a thorough review of CCA boundaries is beyond the scope of the current research.

4.3.2 Canal diversions and additional water supply

The distance of the CCAs to the theoretical Budyko curves in Figure 4.4 is indicative of water sources other than P. Figure 4.7 presents the CCAs in Budyko space once again, now with measured Q_{div} added to the supply side of both ratios. By adding Q_{div} as a supply of water, both the ET ratio (vertical axis) and aridity index (horizontal axis) decrease, reflecting a situation with wetter land surface climatology. In consequence, all CCA points have moved substantially towards the Budyko lines. It should be noted that values are annual averages for the 2004 - 2012

Table 4.2. Precipitation (P), actual evapotranspiration (ET_{act}), blue water evapotranspiration (ET_{blue}), and green water evapotranspiration (ET_{green}) for the agro-climatic zones and provinces of the IBIS.

	Agro-climatic zone	Area (km²)	P	ET_{act}	ET_{blue}			ET_{green}			
			mm	*mm*	*mm*	*BCM*	*% of ET_{act}*	*mm*	*BCM*	*% of ET_{act}*	*% of P*
Punjab	Mixed cropping	10,494	435	602	268	2.8	45%	334	3.5	55%	77%
	Rice wheat	12,527	541	929	505	6.3	54%	423	5.3	46%	78%
	Cotton wheat	55,840	189	986	814	45.4	83%	172	9.6	17%	91%
	Sugarcane wheat	26,524	425	899	555	14.7	62%	344	9.1	38%	81%
	Total	**105,385**	**321**	**915**	**649**	**69.3**	**71%**	**266**	**27.5**	**29%**	**83%**
Sindh	Cotton wheat	29,472	174	984	828	24.4	84%	156	4.6	16%	90%
	Rice wheat	30,419	154	915	778	23.7	85%	137	4.2	15%	89%
	Total	**59,891**	**168**	**950**	**801**	**48.0**	**84%**	**149**	**8.8**	**16%**	**89%**
IBIS total		**165,276**	**263**	**927**	**707**	**117.3**	**76%**	**220**	**36.3**	**24%**	**85%**

period, and only CCAs are shown for which at least one full hydrological year of Q_{div} data is available during this period (years included per CCA are listed in Table 4.3).

Most points are still above the theoretical Budyko lines, suggesting that the sum of precipitation and canal water diversions is unable to explain all water supplied to the crops. Strikingly, as opposed to Figure 4.4, Sindh CCAs are now generally closer to the Budyko curve than those in Punjab. This can be explained by relatively high surface irrigation allocations in Sindh. As described by van Steenbergen et al (2015), excessive canal supplies in several of Sindh CCAs are known to lead to extensive water logging. A famous example is Rice CCA (no. 34), which approaches the theoretical Budyko value in Figure 4.7. K.B. Feeder (no. 30) is located below the Budyko curves, as a substantial part of diverted water is transported for domestic use to the megacity of Karachi, adjacent to the CCA (Phul et al., 2010). This CCA is therefore excluded from further analyses. Most Punjab CCAs are still far from the theoretical Budyko curves, indicating that a relatively large portion of their water supply comes from sources other than main canal headwaters.

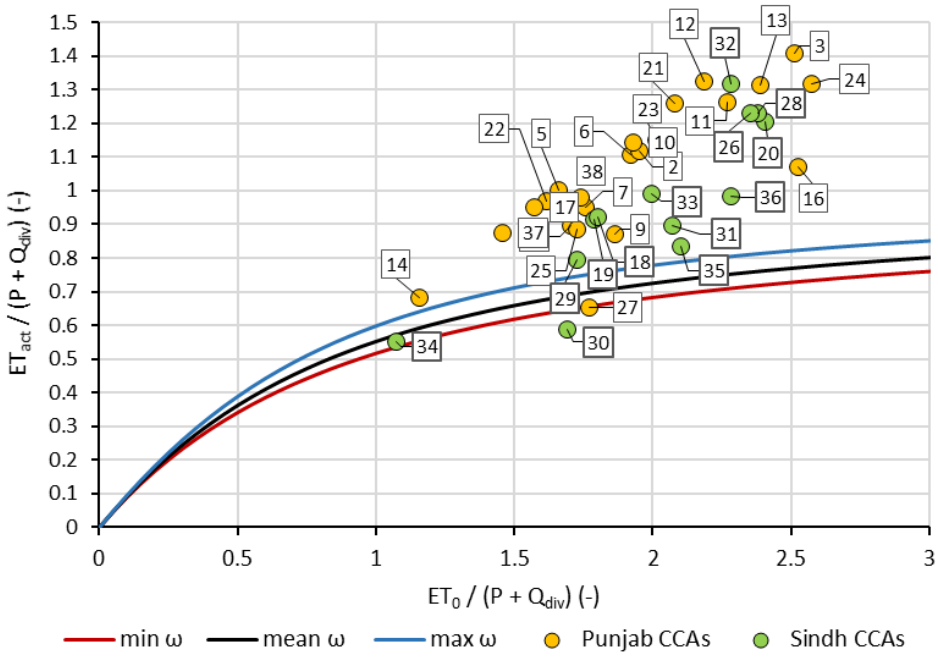

Figure 4.7. Ratios between actual evapotranspiration (ET_{act}) and the sum of precipitation (P) and canal water supply (Q_{div}), and between reference evapotranspiration (ET_0) and ($P + Q_{div}$), of canal command areas in the IBIS in 2004 – 2012. Budyko curves are based on ω values of 1.876, 1.76, and 2.05; the CCA-level mean, minimum, and maximum values, respectively. CCA numbering is as listed in Figure 1.

Table 4.3. Annual blue water fluxes for the IBIS canal command areas. Not presented due to insufficient availability of canal diversion data are Gugera, Mailsi, and Lower Bahawal. K.B. Feeder is also not shown, as a substantial portion of canal water is used for Karachi urban water supply (see main text).

ID	CCA	Area (km^2)	ET_{blue} (mm) (hm^3)	Q_w (mm) (hm^3)	Q_{div} (mm) (hm^3)	Q_{add} (mm) (hm^3)	Q_{nc} (mm) (hm^3)	Period
1	Upper Jhelum (int)	2830	432 (1,222)	1,144 (3238)	401 (1,134)	743 (2,104)	712 (2,016)	2004 - 2007
2	Lower Jhelum	7489	494 (3,697)	1,118 (8375)	445 (3,332)	673 (5,043)	625 (4,679)	2004 - 2007, 2010 - 2012
3	Marala Ravi Link	855	421 (360)	923 (789)	270 (231)	653 (558)	502 (429)	2004 - 2007
4	Upper Chenab	4,334	531 (2,300)	1,205 (5,222)	465 (2,015)	740 (3,207)	674 (2,923)	2004 - 2007
5	BRBD internal	2,197	438 (963)	998 (2,193)	288 (634)	710 (1,559)	560 (1,230)	2004 - 2007, 2010 - 2012
7	Jhang	9,113	536 (4,882)	1,049 (9,561)	259 (2,356)	791 (7,205)	513 (4,680)	2006 - 2007
8	Thal	10,494	267 (2,797)	489 (5133)	489 (5131)	0 (1)	223 (2,336)	2004 - 2007, 2010 - 2012
9	CRBC	2,745	574 (1575)	1,265 (3,473)	418 (1,149)	847 (2,325)	691 (1,898)	2007
10	Rangpur	1,606	764 (1227)	1,512 (2429)	444 (714)	1,068 (1,715)	748 (1,202)	2006 - 2007, 2010 - 2012
11	Dera Ghazi Khan	4,188	788 (3299)	1,573 (6,588)	877 (3,674)	696 (2,914)	785 (3,289)	2004 - 2007, 2010 - 2012
12	Muzaffar garh	3,662	835 (3057)	1,676 (6,137)	847 (3,100)	829 (3,037)	841 (3,080)	2004 - 2007, 2010 - 2012
14	Pakpattan	4,278	857 (3667)	1,640 (7,016)	725 (3,103)	915 (3,913)	783 (3,348)	2006 - 2007, 2010 - 2012
15	Upper Dipalpur	1,438	685 (985)	1,288 (1,851)	497 (714)	791 (1,137)	603 (867)	2006 - 2007
16	Abbasia	1,199	659 (789)	997 (1,195)	583 (699)	414 (497)	339 (406)	2004 - 2007
17	Panjnad	6,017	910 (5,474)	1,686 (10,147)	653 (3,929)	1,033 (6,218)	777 (4,673)	2004 - 2007, 2010 - 2012
18	Pat + Desert	4,410	841 (3,711)	1,537 (6,780)	915 (4,033)	623 (2,747)	696 (3,069)	2004 - 2012
19	Ghotki	3,819	852 (3,253)	1,511 (5,772)	933 (3,565)	578 (2,207)	660 (2,519)	2004 - 2012

Table 4.3. (Continued).

ID	CCA	Area (km^2)	ET_{blue} (mm) (hm^3)	Q_w (mm) (hm^3)	Q_{div} (mm) (hm^3)	Q_{add} (mm) (hm^3)	Q_{nc} (mm) (hm^3)	Period
20	Beghari	4,627	831 (3,845)	1,480 (6,848)	671 (3,107)	809 (3,742)	649 (3,003)	2004 - 2012
21	Haveli	816	802 (654)	1,680 (1,370)	646 (527)	1,034 (843)	879 (717)	2004 - 2007
23	Eastern Sadiqia	5,130	669 (3,434)	1,114 (5,717)	768 (3,938)	347 (1,779)	445 (2,283)	2006 - 2007, 2010 - 2012
24	Fordwah	2,136	787 (1,681)	1,416 (3,025)	554 (1,184)	862 (1,841)	630 (1,345)	2004 - 2012
25	Lower Dipalpur	2,890	776 (2,242)	1,603 (4,632)	525 (1,516)	1,078 (3,116)	827 (2,391)	2006 - 2007, 2010 - 2012
26	Rohri	11,446	902 (10,321)	1,671 (19,123)	682 (7,811)	988 (11,312)	769 (8,801)	2004 - 2012
27	Nara	10,996	731 (8,041)	1,378 (15152)	802 (8,817)	576 (6,335)	647 (7,111)	2004 - 2012
28	Lined	2,402	843 (2,025)	1,568 (3,767)	585 (1,406)	983 (2,361)	725 (1,742)	2004 - 2012
29	Fuleli	4,294	820 (3,521)	1,409 (6,052)	1,052 (4,517)	357 (1,535)	589 (2,531)	2004 - 2012
31	Pinyari	3,576	649 (2,320)	1,074 (3,841)	722 (2,581)	352 (1,260)	425 (1,521)	2004 - 2012
32	Khairpur West	1,336	1011 (1,351)	2,046 (2,733)	742 (992)	1,304 (1,742)	1035 (1,382)	2004 - 2012
33	Northwest	3,907	804 (3,143)	1,525 (5,958)	808 (3,159)	717 (2,800)	721 (2,816)	2004 - 2012
34	Rice	2,261	891 (2,014)	1,779 (4,022)	1,687 (3,813)	92 (209)	888 (2,008)	2004 - 2012
35	Dadu	2,211	667 (1,474)	1,100 (2,432)	820 (1,813)	280 (620)	433 (958)	2004 - 2012
36	Khairpur East	1,876	717 (1,345)	1,131 (2,123)	715 (1,342)	416 (780)	415 (778)	2004 - 2012
37	Sidhnai	3,508	927 (3,253)	1,854 (6,505)	647 (2,271)	1207 (4,234)	927 (3,252)	2004 - 2007, 2010 - 2012
38	Lower Bari Doab	7,935	795 (6310)	1,577 (1,2515)	664 (5,271)	913 (7,244)	782 (6,205)	2004 - 2007, 2010 - 2012
40	Upper Bahawal + Qaim	555	937 (520)	1,899 (1,054)	1,418 (787)	481 (267)	962 (534)	2004 - 2007, 2010 - 2012
Area-weighted average (mm)			**707**	**1,352**	**662**	**690**	**645**	Varying

What follows from Figure 4.7 is that ET_{act} in most CCAs is attributable to sources of water in addition to rainfall and canal diversions, as most CCA points plot well above the curve. Total Q_w can now be computed from the distance between the actual data points in Budyko space and the theoretical Budyko curve as prescribed by the CCA-specific ω values, by solving eq. (4.6) for the comprehensive supply term P_{adj} and subsequently applying eq. (4.7). A full overview of all blue water fluxes per CCA, including additional supply Q_{add} as the difference between Q_w and Q_{div}, is provided in Table 4.3. It is clear that relatively large volumes of Q_{add} are calculated for almost all CCAs. At the same time, a substantial amount of non-consumed water (Q_{nc}) is computed as, apparently, Q_w has to exceed ET_{blue} substantially to maintain the hydrological processes imbedded in the Budyko Hypothesis. On average, this annual water balance looks as follows for the IBIS CCAs:

$$Q_{div} + Q_{add} = ET_{blue} + Q_{nc} \tag{4.8}$$

$$662 + 690 = 707 + 645$$

with all values in mm per year.

As described in Section 4.2.2, Q_{add} can be a combination of different sources of water, both depending on hydrological processes *within* the respective CCA and *between* CCAs. It is interesting to explore the Q_{add} term further, as it provides insight into the nature of reuse of non-consumed flows in the IBIS and potentially also includes unsustainable groundwater pumping. Figure 4.8 presents Q_{add} relative to other water supply components for all CCAs. Dependency on Q_{add} differs highly among the areas, with values ranging between 5% (Rice) and 61% (Khaipur West). At the provincial level, these values amount to 47% and 38% of total water supply including precipitation for Punjab and Sindh respectively. This difference could be explained for example by coarser soils with more percolation losses, the degree to which canal water allocation meet crop water requirements, and groundwater quality issues.

Evaluating multi-annual Q_{div} against ET_{blue} provides insight in the long-term blue water balance and the source of Q_{add}. In CCAs where ET_{blue} exceeds Q_{div}, Q_{add} must structurally depend on non-consumed flows from upstream CCAs, rainfall recharge outside of CCA (or total IBIS) boundaries, or unsustainable groundwater use. On the other hand, positive values for Q_{div} - ET_{blue} indicate a net positive contribution of blue water in the corresponding CCA to the aquifer system. Table 4.3 shows that, on average, Q_{div} (662 mm) is largely able to sustain ET_{blue} (707 mm, or 107% of Q_{div}). However, Figure 4.9 demonstrates that $Q_{div} - ET_{blue}$ varies greatly per CCA and, in fact, per province. Clearly, Jhang, Panjnad, Lower Bari Doab, and Rohri are examples of CCAs requiring substantial volumes of water on the long-term in addition to Q_{div} to explain irrigation consumptive use. An example of the opposite phenomenal is Rice canal, which due to excessive canal supply has a blue water surplus of 1.8 BCM. Looking at the provincial level, substantial differences exist between Punjab and

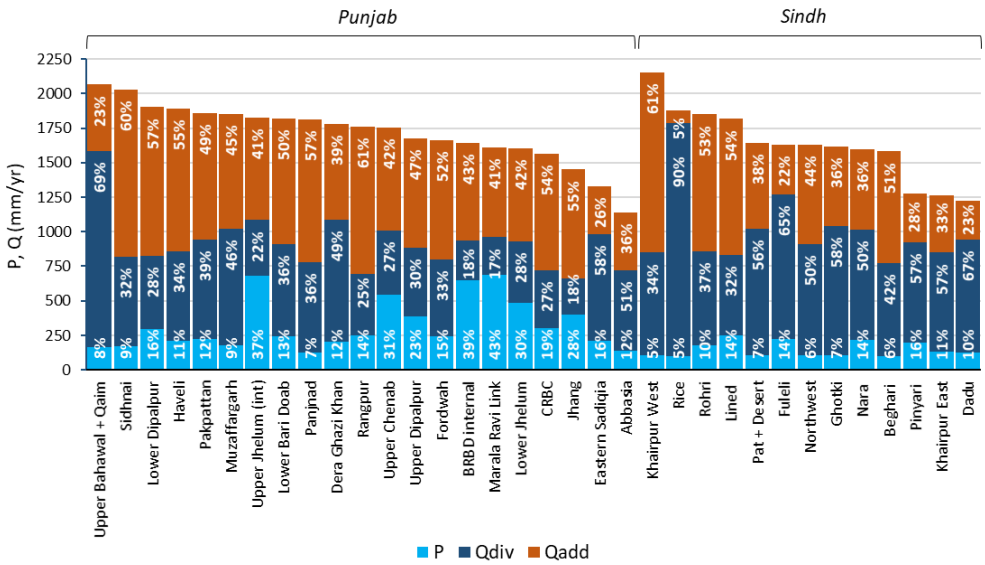

Figure 4.8. Different sources of water for each canal command area: precipitation (*P*), canal water (Q_{div}), and additional supply (Q_{add}). Percentages indicate the extent to which water use in each command area depends on sources other than rainfall or water diverted to the main canal.

Sindh. Annual ET_{blue} in Punjab is approximately 7 BCM (15%) higher than Q_{div}, whereas for Sindh a minor positive Q_{div} - ET_{blue} value is calculated. The above analysis shows that consumptive use in Punjab CCAs is more dependent on return flows and aquifer recharge generated outside CCA boundaries, and / or fossil groundwater pumping. The latter has received elaborate attention in recent scientific literature and model assessments. Although local falling water tables due to unsustainable groundwater use are a well-known point of concern, especially in Punjab, they cannot be regarded as dominant in explaining Q_{add} volumes. Since long-term Q_{add} is substantially higher than Q_{div} - ET_{blue} in all CCAs, the main source of Q_{add} must lie within the CCA and must be replenished within the annual time frame. This finding is supported by previous analyses of GRACE water storage data, in which groundwater depletion over the Upper Indus Plain in 2003 - 2010 was estimated at 1.48 BCM/yr or 13.5 mm/yr (Iqbal et al., 2016). This corresponds to only 4% of annual Q_{add} computed for the relevant CCAs. The groundwater balance presented by Young et al. (2019), based on a comprehensive literature review, similarly suggests that the recharge and discharge components of the overall aquifer system are largely in balance.

Overall, the Budyko-based analysis paints a picture of a system where discrepancies between crop water demands and canal water supply during the irrigation season lead to pumping of a mixture of Q_{div} and Q_{add}. Based on the magnitude of Q_{add} volumes in both Punjab and Sindh compared to other blue water fluxes, it can be safely stated that this additional supply term mainly consists of

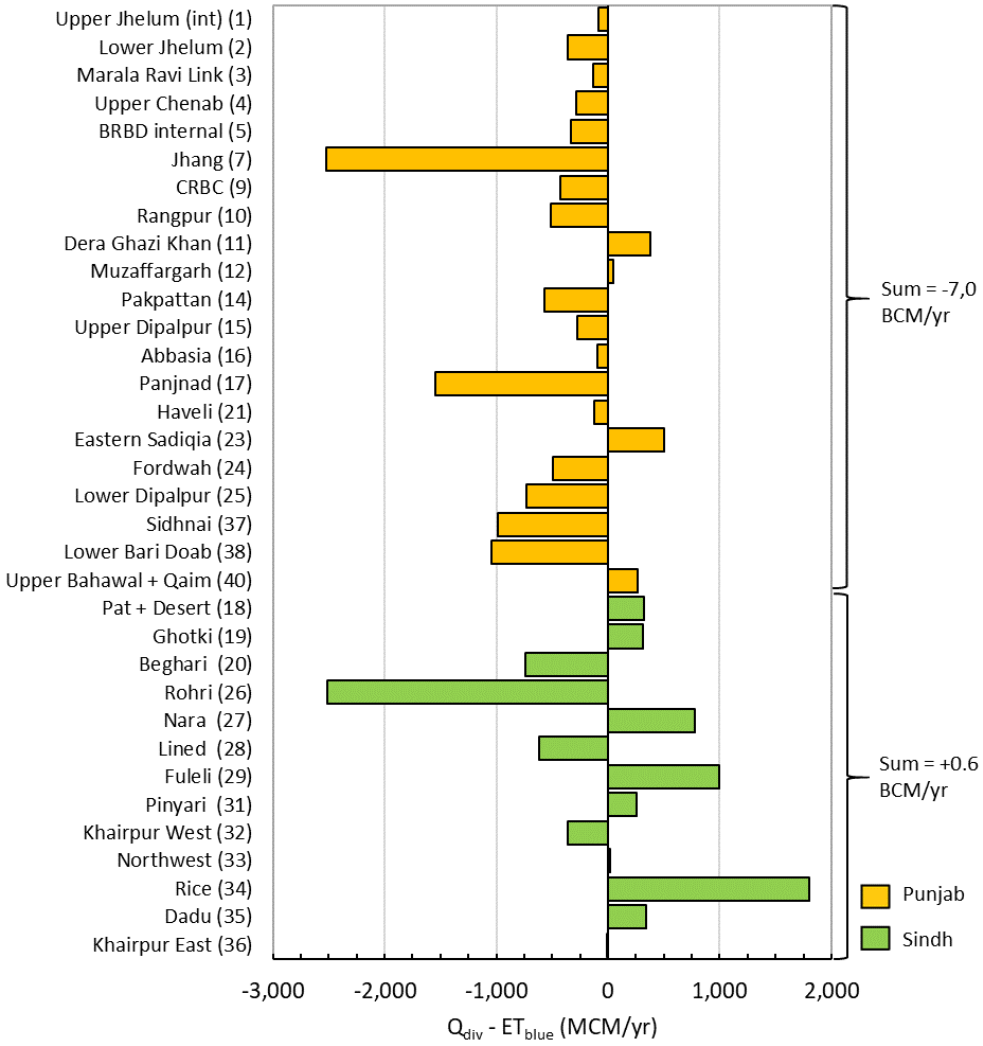

Figure 4.9. Difference between canal water supply (Q_{div}) and blue water evapotranspiration (ET_{blue}) for each of the canal command areas. Not presented due to insufficient availability of Q_{div} data are Gugera, Mailsi, and Lower Bahawal. K.B. Feeder is also not shown, as a substantial portion of Q_{div} is used for Karachi urban water supply (see main text).

local (within-CCA) non-consumed flows (Q_{nc}). In this regard, it is interesting to note the similar magnitude of Q_{add} and Q_{nc} presented in Table 4.3. Irrigation in IBIS CCAs is characterized by the pumping of considerable volumes of non-consumed flows generated within the same CCA, which for a major part drain back into the system and are withdrawn again in a next cycle.

4.3.3 Consumed fractions and implications for agricultural water management

Thanks to the availability of ET_{blue} and Q_w data, eq. (4.1) can now be applied to calculate consumed fractions of water withdrawals at the CCA level. Figure 4.10 shows the resulting map of *CF* across IBIS. Although *CF* values differ between CCAs, *CF* values in Sindh are generally found to be higher than in Punjab. Overall, *CF* ranges between 0.38 (Upper Jhelum) and 0.66 (Abbasia) at the level of the IBIS main CCAs, with an average size of 4,036 km^2. The average *CF* at CCA level for entire IBIS, weighted according to total Q_w, is 0.52.

To provide reference for the BH-based results, Table 4.4 gives an overview of IBIS irrigation efficiency values found in scientific literature. Though efficiency definitions are not consistent among these studies, they typically incorporate "losses" of diverted water in the processes of conveyance through canals and application to the field crop. The Budyko-based analysis generally yields higher values than irrigation efficiencies previously assumed for Pakistan, which vary between 0.3 and 0.49. This suggests that irrigation in the IBIS is more "efficient" than previously reported, mostly based on local-scale measurements. In comparison to literature efficiency estimates separating beneficial and non-beneficial consumption, it should be noted that ET_{blue} does not discriminate between crop transpiration and soil evaporation, which logically yields somewhat higher *CF* values.

Evaluating *CF* values on different spatial scales leads to insight in the system-scale reuse of non-consumed flows. In this study, it is assumed that the CCA level is the minimum scale on which Budyko theory assumptions are valid. *CF* of the entire IBIS can be estimated by dividing Budyko-derived ET_{blue} by the total water supply to the system. As long-term net groundwater recharge is virtually zero, a conservative estimate of *CF* can be computed based on total releases of the main reservoirs at the IBIS head, which in 2004 - 2012 amounted to 163.6 km^3/yr or 990 mm/yr on average (PBS, 2014). *CF* of entire IBIS can then be estimated by the ratio between ET_{blue} (Table 4.3) and reservoir releases, i.e. 707 / 990 = 0.71. However, as not all of the released water is used for irrigation purposes, a different estimate can be calculated based on total official surface water withdrawals of 125 BCM/yr (Young et al. 2019), or 756 mm/yr which leads to a total system *CF* of 0.93. Although the real supply volume arguably lies somewhere in between, both estimates are well above the CCA average of 0.52 and signify a relatively efficient system despite substantial water "losses" on smaller scales. This indicates that non-consumed flows to unconfined aquifers, drainage canals, and baseflow contribution to rivers cause water reuse processes to extend beyond CCA borders. In reality, informal irrigation outside official CCA boundaries leads to higher ET_{blue} and thus an even greater return flow reuse and system *CF*. When increasing the scope of the analysis to the full transboundary Indus Basin, *CF* may be further enhanced by lateral groundwater flows between India and Pakistan (Khan et al. 2017).

This study has successfully quantified total water supply and consumed fractions in the IBIS command areas, demonstrating the production of considerable volumes

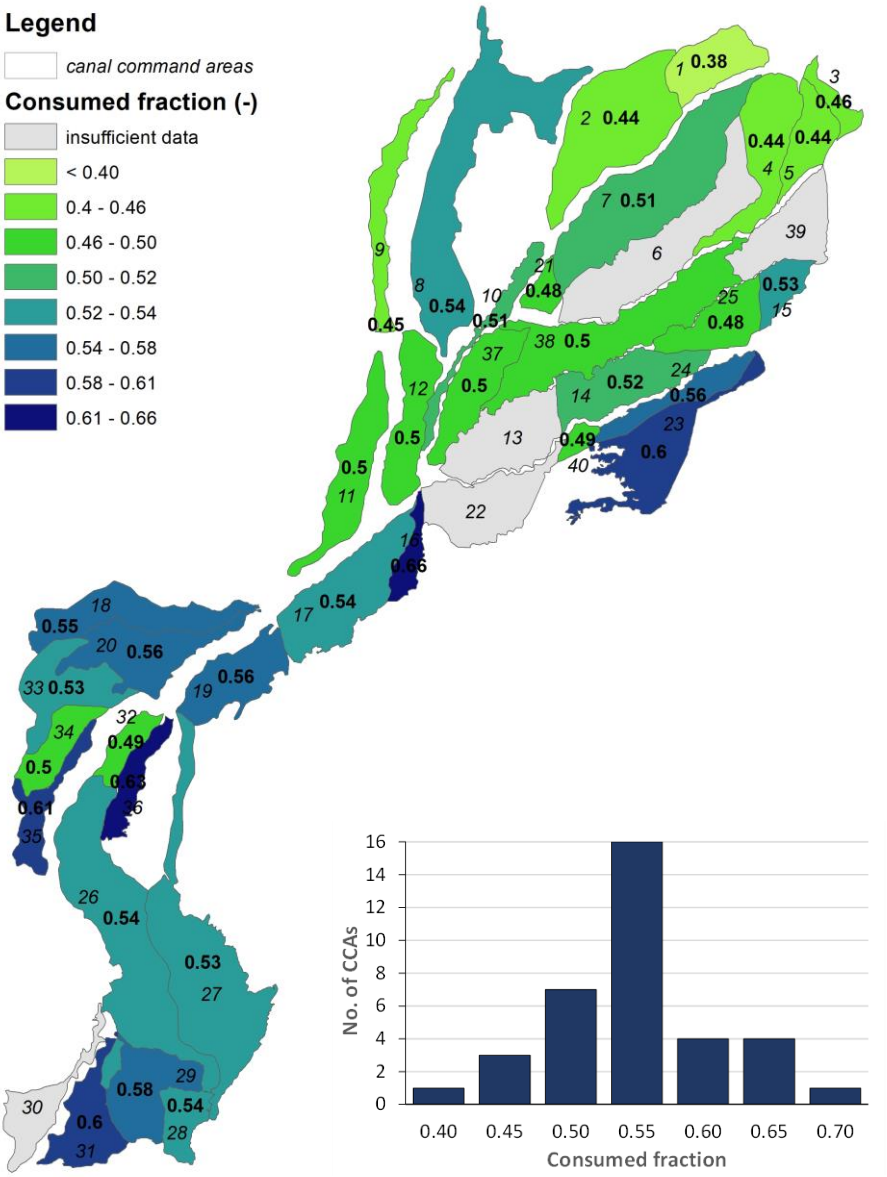

Figure 4.10. Map and histogram of consumed fractions per canal command area.

of non-consumed flows. As discussed above, this water is not only extensively reused within the CCAs, for example to mitigate differences in head vs. tail canal supplies, but also leave CCA boundaries for pumping downstream. This notion of a dense and complex network of water (re)use is supported by various studies. According to Van Steenbergen and Gohar (2005), an estimated 79% of pumped

Table 4.4. Selected IBIS irrigation efficiency values from various literature sources. It should be noted that definitions vary and, therefore, not all values can be directly intercompared.

Data source	Area	Value	Definition as used in source
Khan et al. (2006)	Rechna Doab	0.32	Surface water irrigation efficiency
Hussain et al. (2011)	IBIS	0.35	Canal conveyance efficiency * watercourse conveyance efficiency * field channel efficiency * field application efficiency
Basharat and Tariq (2013)	Lower Bari Doab Canal	0.49	Conveyance efficiency * watercourse efficiency * field application efficiency
Yu et al. (2015)	Punjab and Sindh	0.35	Canal efficiency * watercourse efficiency * field efficiency
Qureshi et al. (2010)	Pakistan	0.3	Overall irrigation efficiency
Shakir et al. (2010)	IBIS	0.4	Irrigation efficiency "from canal head to the field level"
Jägermeyr et al. (2015)	IBIS	0.24	Beneficial irrigation efficiency (transpiration / withdrawals)
Rohwer et al. (2007)	Pakistan	0.32	Actual project efficiency

groundwater in IBIS originates from canal seepage, percolation from the river, and non-consumed flows. Karimi et al. (2013) report a basin-scale "classical efficiency" of 84% for the full Indus Basin, incorporating transboundary lateral flows. Grogan et al. (2017) showed that the Indus flow regime will significantly shift when consumed fractions are altered, due to extensive reuse of non-consumed flows. It is evident that further increases of system-scale *CF* will impact flow volumes and patterns downstream of Kotri Barrage and, therefore, hydrological and sedimentation regimes in the Indus Delta (Salik et al., 2016).

The results of this study exemplify the need to account for the system scale when considering efficiency improvement measures in the IBIS. In practice, increases in evapotranspiration in the IBIS are often achieved by a reduction in groundwater recharge, exacerbating the decline of the groundwater table and reducing water availability to downstream users (Ahmad et al., 2007). By providing spatially disaggregated *CF* values, the proposed approach facilitates a more effective and tailored development of water conservation measures in the different CCAs. It is found that in many CCAs, field-scale efficiency improvements may impact on an existing equilibrium of non-consumed flows and reuse of these flows by others as part of their Q_{add}. However, in areas where observations of rapidly falling groundwater tables coincide with a relatively low *CF*, such as on the Upper Indus Plain (Figure 4.10), appropriate measures could result in a greater sustainability of the system. Similarly, occurrence of low *CF* values in areas with hazardous groundwater quality (particularly found in Sindh), may justify interventions to minimize recharge of saline groundwater bodies.

4.4 Conclusions and recommendations

A new method for spatially quantifying consumptive use of irrigation water based on the Budyko Hypothesis was successfully demonstrated for the IBIS in Pakistan. The innovation is twofold, as the approach (i) distinguishes green and blue water consumption using reference evapotranspiration and precipitation data, and (ii) computes total water supply to support Consumed Fraction estimates, which are essential for understanding system-scale water use and potential for water savings. It was found that out of the average annual ET_{act} of 927 mm/yr, 707 mm/yr (76%) depends on irrigation water. ET_{blue} values vary greatly among CCAs with a range of 421 to 1,011 mm/yr, as a consequence of differing canal headwater volumes, crop types, climate conditions, and groundwater quality, among others. By evaluating Budyko-based total blue water supply against long-term main canal diversions, it was concluded that most command areas rely substantially on water not diverted at the head of the primary canal, with additional supply Q_{add} (690 mm/yr) on average even slightly exceeding Q_{div} (662 mm/yr).

The average consumed fraction of the IBIS canal command areas was computed at 0.52, with CCA values ranging between 0.38 and 0.66. From the relatively low *CF* values, high additional water supplies, and long-term canal supplies largely sufficient to sustain ET_{blue}, the conclusion can be drawn that the IBIS is characterized by extensive reuse of non-consumed flows within CCAs. At the same time, a notably higher *CF* at the system scale indicates that reuse of non-consumed water facilitated by lateral connectivity between CCAs cannot be disregarded. These conclusions imply that, although the IBIS is generally not regarded as an efficient irrigation system, it is in fact tailored to recover and reuse drainage flows on different spatial scales. Water saving measures should therefore be implemented with caution. It is recommended to supplement the results of this study with ancillary information on groundwater quality and groundwater table time series, to identify locations where *CF* increases may be beneficial on the system scale. It should be noted that the accuracy of the CCA map used in this study is continuously under revision by government institutions, allowing for more refined *CF* assessments in the future, e.g. by accounting for irrigated area dynamics and city boundaries.

By providing quantitative estimates of previously unexposed parameters ET_{blue}, *CF* and Q_{nc} per CCA, the proposed approach contributes significantly to the understanding of water consumption and reuse in the IBIS. Results of the consecutive steps of the Budyko-based approach (climatology and ET partitioning, consumptive use, and assessments of water supply components) were shown to be in agreement with the existing knowledge base on the IBIS. A big advantage of the method over alternative approaches is that estimates of ET_{blue}, Q_{add} and Q_{nc} were produced without the need for complex hydrological models, data on soil parameters, or assumptions on curve numbers. In addition, although diversion data were used for partitioning total withdrawals into canal water and additional supply, they are not required for the basic ET_{blue} and *CF* analyses. As the use of

global satellite-derived data products allows the method to be replicated worldwide, the proposed method holds great potential for more accurate evaluation of consumptive use and dependencies among water users in river basins, facilitating targeted and more effective water allocation policies and water conservation measures.

This study fits in a recent body of work exploring the potential of the Budyko Hypothesis, in various reformulations, to function under differing conditions in terms of spatial and temporal scales, storage changes, and degree of anthropogenic impact on the natural water balance. CCAs, typically with areas of several thousands of km^2, were assumed appropriate units for BH-based analysis. Analyses were based on multi-annual input datasets to allow for assumption of zero storage change, and seasonal-scale results were deemed incongruous with BH preconditions and were therefore not presented. It is recommended for future studies to further explore opportunities and limitations of Budyko-based analyses in an irrigation context, with regards to appropriate spatiotemporal dimensions and, potentially, more complex BH formulations to account for non-steady states or incorporate physical catchment parameters in a more explicit way. By using pixel-based satellite data products on evapotranspiration and precipitation, the proposed method is highly flexible in terms of scale and can easily be applied to other basins and Budyko formulations.

5

VIRTUAL TRACERS TO DETECT SOURCES OF WATER AND TRACK WATER REUSE ACROSS THE SEGURA RIVER BASIN, SPAIN

Water managers around the world face the increasingly challenging task to evaluate impacts of technological measures and policy mechanisms from the local to the river basin scale. A toolset providing quantitative, actionable information on dependencies and trade-offs between upstream and downstream water users is currently lacking. Yet, any intervention needs to be assessed in terms of consequences for downstream water users. This chapter evaluates the potential of a tracer-like approach, implemented in the water allocation software WEAP, to quantitatively track non-consumed flows and their downstream reuse in the river basin context. The WEAP-VirtualTracer (WEAP-VT) approach was successfully applied to one of Europe's driest river basins, the Segura River Basin in Spain. For each water demand site, the different original sources of water supply, dependency on upstream return flows, and downstream reuse of its non-consumed flow were assessed. Based on these results, agricultural, urban, and environmental water users were evaluated in terms of their suitability for water saving measures and their vulnerability to reduction of upstream return flows. A scenario analysis simulating improvement of local efficiency improvements shows that specific irrigation schemes and ecosystems become deprived of water. Hence, efficiency improvement in water-scarce basins should be considered with caution. The demonstrated ability to quantify key water reuse indicators for individual water users and at different aggregation levels makes WEAP-VT a valuable tool to support water resources management decisions.

Chapter based on G.W.H. Simons, P. Droogers, S. Contreras, J. Sieber, W.G.M. Bastiaanssen, 2020. Virtual Tracers to detect sources of water and track water reuse across a river basin. Water, 12(8), 2315; doi:10.3390/w12082315

5.1 Introduction

Competition for water resources is intensifying in many river basins around the world. Climate change is projected to significantly impact spatial patterns and temporal dynamics of water availability (Flörke et al., 2018; Wijngaard et al., 2018). Ambitions to save water are especially common in (semi-)arid basins with valuable economic benefits and ecosystem services associated with the use of water. A frequently applied response involves the introduction of modern technology or management practices which aim at enhancing the *efficiency* of a water use; i.e. increasing the ratio of consumptive use over withdrawal. However, downstream reuse of non-consumed water, either planned or unplanned, potentially limits the basin-scale beneficial effects of such local-scale interventions (Williams and Grafton, 2019; Zhang et al., 2019). The relevance of accounting for dependencies and trade-offs through water reuse increases with the degree of water resources development and complexity of the network of water users in a basin.

Numerous case studies provide empirical evidence that the degree to which water can actually be saved at the basin level is often minimal (FAO, 2017). Although reduction of water withdrawals can provide benefits in various ways (Gleick et al., 2011), local efficiency increases in practice often do not actually release water to alternative users. If no legal or physical restrictions are in place, water users are inclined to find further productive use for the abstracted water, such as by growing more water-intensive crops or expanding their irrigated acreage (Koech and Langat, 2018). More permanent reductions of non-consumed water due to upstream water savings can affect downstream agriculture, urban areas, and/or ecosystems (Berbel et al., 2015; Contor and Taylor, 2013; Dumont et al., 2013). Sound water management and effective implementation of new technology therefore rely on reliable knowledge of upstream-downstream interactions.

Various conceptual and analytical frameworks have been developed with the purpose of characterizing a system of water users based on return flows and their reuse (Simons et al., 2015). The practical application of these methods is, however, greatly limited by a lack of spatiotemporal data on fundamental variables such as water withdrawal, consumption, and non-consumptive use. Wiener *et al.* (Wiener et al., 2016) demonstrated how water reuse can be well-characterized for a watershed where extensive records of withdrawals, consumptive use and non-consumed flows are available. In the context of irrigated rice, Chinh (2012) and Hafeez et al. (2007) disposed of pumping records and flow data in drains and channels to characterize return flow reuse and its scale effects. Although such studies produced valuable information on local reuse situations, the need for extensive field data provides limited basis for upscaling to most of the world's river basins, where this type of information is needed the most.

The scarcity of field data can be partly resolved by satellite-derived information (Simons et al., 2016). Innovative approaches have recently been devised to assess key variables such as water consumption, consumed fractions of irrigation water, and applied irrigation water (Simons et al., 2020; van Eekelen et al., 2015; Vogels

et al., 2020). These advances are encouraging as they limit the degree to which complex, physically-based modelling is required to separate surface water from soil and groundwater fluxes, avoiding the need for spatial data to describe the soil profile and vegetation properties.

The nature of satellite imagery, however, is unsuitable for an explicit assessment of lateral flows between water users. At best, some bulk flows and reuse factors can be computed. Complementary simulation modelling can allow for assessing the direct interactions between water users and quantifying water reuse indicators, under current conditions as well as different water management scenarios. Dynamic models also allow for investigating the temporal dimension of reuse and impacts of alterations in timing of water availability. A range of modeling approaches has been developed and applied to quantify non-consumed water, especially in the context of irrigation systems (e.g. Chien and Fang, 2012; Hu et al., 2017; Mohan and Vijayalakshmi, 2009). The number of studies explicitly looking into actual reuse of these flows by downstream water users is, however, much more limited. Wu et al. (2019b, 2019a) used a modified SWAT model to evaluate return flow reuse processes on different scales for a paddy rice irrigation system, but this type of examples is not very common. Hence, no modeling approach for comprehensively assessing basin-level water (re)use between different types of users is currently available.

A key challenge to such a model-based assessment is the explicit dimensioning of flows between water users, as well as mixing processes occurring when return flows are injected into sources and streams. In field studies, to tackle similar issues, artificial and environmental tracers are widely used for identifying water origins and flow paths (Leibundgut et al., 2009). In an irrigation context, for example, Vallet-Coulomb et al. (2017) made use of this principle to partition groundwater recharge into rainfall infiltration and irrigation return flows. Beard et al. (2019) recently were among the first to implement a tracer-like approach in a modeling environment to assess the contribution of treated wastewater to surface water used for irrigation. However, by working with a single tracer substance, their approach does not allow for evaluating downstream reuse of flows from specific sources and assessing dependencies between individual water users. To our knowledge, so far no studies have applied a tracer approach in water resources modeling with the purpose of assessing indicators of reuse at the level of individual water users.

The objective of this chapter is therefore to explore the potential of applying a "VirtualTracer" approach in a water allocation model for tracking water sources and reuse. The Segura River Basin in South-Eastern Spain was selected as the pilot area because of its exposure to water shortages and the regional significance of irrigated agriculture.

5.2 Materials and methods

5.2.1 *Study area*

The Segura River Basin covers an area of 18,930 km2 in the semi-arid southeastern corner of the Iberian Peninsula (Figure 5.1). Average precipitation in the region ranges from 1,000 mm/year in the headwater sections to below 300 mm/year in the driest lowlands, while reference evapotranspiration averages 1,500 mm/year. The river network is comprised by 1,553 km of permanent and intermittent streams: a primary channel (Segura River) and various right-side (Taibilla, Moratalla, Argos, Quipar, Mulas and Guadalentín) and left-side tributaries (Mundo-Camarillas system). At the headwaters, the Segura and Mundo rivers contribute, on average, 68% of the total surface water resources available in the region. The left-side tributaries have an intermittent flow regime and provide discharge only after very intense rainfall events (CHS, 2015).

The basin is home to a population of 2.1 million. The majority of these inhabitants live in the cities of Murcia and Cartagena as well as urban area adjacent to the Mar Menor, a coastal saltwater lagoon of high ecological importance and significance to the tourism industry. Upstream mountainous areas are largely covered by forest and shrubland (Figure 5.1). Downstream, in particular the Campo de Cartagena region is known for its intensive agricultural activity, despite a structural water deficit of more than 400 mm per year (Castejón-Porcel et al., 2018). Overall, 43% of the basin is covered by agricultural land, of which 1/3 is under irrigation. A wide range of fruit, vegetables and flowers is grown across the alluvial plains of the Segura Basin. The water demand of irrigated agriculture amounts to 85% of total water demand in the basin (Martınez-Paz et al., 2018). Particular sites of ecological importance are the 120 wetlands in the basin, of which 70 are subject to special protection in the framework of the European Habitats directive (Aldaya et al., 2019).

The Segura River Basin is one of the most water-stressed regions in the Mediterranean basin. Management of the scarce water resources is the responsibility of the Confederación Hidrográfica del Segura (CHS), the basin water authority. To satisfy water demand in the absence of sufficient resources within the basin, the Tajo-Segura and Negratín (Guadalquivir) interbasin aqueducts were constructed to provide additional water. The contribution of these interbasin transfers accounts for approximately 20% of the average annual water demand (Sanchis Ibor et al., 2011). Expansion of crop cultivation, population growth and prolonged drought events, along with groundwater pumping restrictions imposed by CHS, has led to the installation of desalination plants to further alleviate the gap between supply and demand. This development has been successful in reducing vulnerability to drought episodes (Morote et al., 2019). Still, significant overexploitation of aquifers occurs to satisfy crop water requirements, with approximately 50% of the annually abstracted volume considered as non-renewable (Uche et al., 2014).

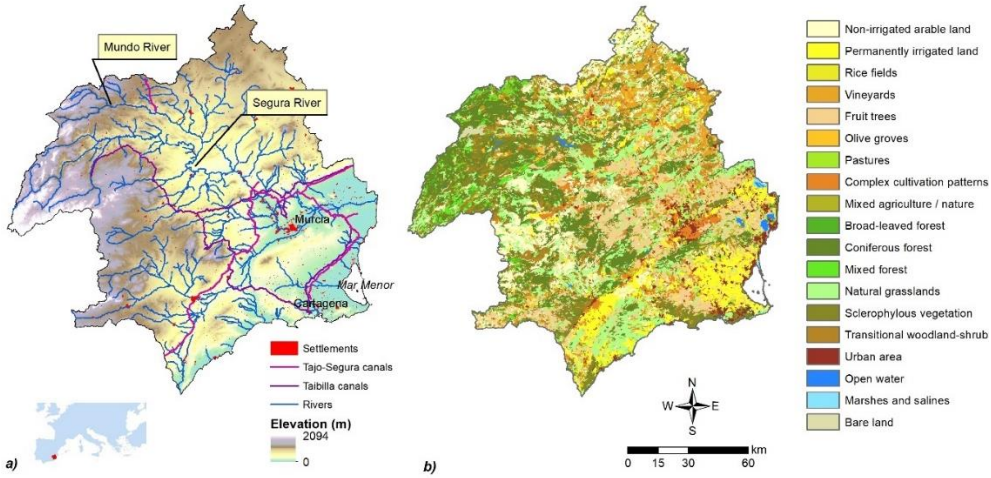

Figure 5.1. Maps of the Segura River Basin: (*a*) main rivers, infrastructure and elevation, (*b*) land use / land cover.

5.2.2 *Concepts and analytical framework*

This chapter follows the water reuse framework of Simons et al. (2015). Reuse is defined as the downstream re-application of non-consumed water from an upstream water use, where the latter can comprise any deliberate application of water to a specified purpose (Perry, 2011). Non-consumed water finds it way downstream through both surface water and groundwater, therefore requiring both to be part of the reuse framework. Water can be reused for e.g. agricultural and landscape irrigation, industrial processes, domestic use, or aquaculture, and does necessarily involve a treatment process. A special form of water reuse is the dependency of natural systems on previously abstracted water for delivering valuable ecosystem services, e.g. by inundation of wetlands. A glossary of all water balance terms used in this study is included in Appendix A.

A crucial starting point for analyzing water reuse is the fraction of withdrawn water that is removed from the system because it evaporated, was transpired by plants, incorporated into products or crops, or consumed by people or livestock. This Consumed Fraction is defined as follows:

$$CF = \frac{Q_c}{Q_w} \tag{5.1}$$

where Q_c is consumed water, and Q_w is the total volume of water withdrawn from various sources (both surface and groundwater). Dynamics of on-site water recycling (referred to as "direct water reuse" in some studies) are reflected in the *CF* value, with multiple cycles leading to a higher *CF*. Thus, when *CF* is computed for

larger spatial domains, it is indicative of the extent to which water reuse occurs within that domain.

The non-consumed water (return flow) enters a network of natural and/or artificial hydrological flow paths and may ultimately be recovered for reuse at a downstream location. In order to perform a thorough assessment of water reuse across a system, this is the flow of water that needs to be tracked in space and time.

Simons et al. (2015) reviewed several indicators developed to characterize water reuse, and discussed these in terms of their value to decision makers, appropriate scales of application, and data requirements. The indicators Degree of Return flow Reuse (*DRR*) and Reuse Dependency (*RD*), as initially proposed by Chinh (2012) in the context of a paddy rice system, were identified as potentially holding the most direct information on water reuse, but with limited practical application so far due to the high demand for input data. These are two key indicators examined in this chapter at the level of a water user.

The original *DRR* definition needs to be modified to be applicable for a more generic context, beyond irrigation only, and to focus on managed or "blue" water fluxes (Falkenmark and Rockström, 2006). For a water user x with n users located downstream, *DRR* can be computed as follows:

$$DRR_x = \frac{\sum_{i=1}^{n}(\chi_i * Q_{w,i})}{Q_{nc,x}} \tag{5.2}$$

where $\chi_{,i}$ is the mixing ratio between non-consumed flow from water user x and total flow through the source medium (e.g. a stream, aquifer, or reservoir) of downstream user i, $Q_{w,i}$ is the volume withdrawn by a downstream user i, and $Q_{nc,x}$ is volume of non-consumed water from x. In case of multiple reuses and inflow of additional water at locations downstream of the point of discharge of x, a sequence of unique χ_i values should be provided for each downstream reuse.

RD is complementary to *DRR*, in the sense that it uses similar input variables to express the dependency of supply to a water user on upstream return flows. For a water user x with n users located upstream, it is defined as follows:

$$RD_x = \frac{\sum_{i=1}^{n}(\chi_i * Q_{w,x})}{Q_{w,x}} \tag{5.3}$$

where $Q_{w,x}$ is the volume of water withdrawn by x.

In addition to indicators based on quantitative assessment of actual flows, relating water demand to supply holds added value for water reuse analyses. To this end, it is important to distinguish between gross demand, i.e. the total demand of a water user including on-site recycling and return flows, and net demand, which corrects for these terms. Local water shortages occur when supply is inadequate to meet gross demand, causing part of the demand to be *unmet*. Expressing this in a relative

indicator yields the coverage (C), or the percentage of gross demand that is met by the supply to a water user:

$$C_x = \frac{Q_{w_x}}{D_{gross_x}} \tag{5.4}$$

where D_{gross_x} is its gross water demand. Quantifying C helps to identify users that are deprived of water. It is particularly useful for evaluating different management scenarios, as for example a water user with low C is likely to respond differently to changes in supply than a user with full coverage.

The CF, RD, DRR, and C indicators can be computed for individual water users, as well as aggregated to the sectoral or basin levels. Together, they provide a toolset for a comprehensive evaluation of water reuse processes across a river basin.

5.2.3 *Modeling approach*

WEAP

The Water Evaluation And Planning (WEAP) system was used as the basic water resources model (Yates et al., 2005). WEAP is a commonly used tool in strategic water resource planning and scenario assessment in many regions around the world (e.g. (Gedefaw et al., 2019; Miraji et al., 2019; Salomón-Sirolesi and Farinós-Dasí, 2019). WEAP uses the basic principle of water balance accounting: total inflows equal total outflows, save for any change in storage (in reservoirs, aquifers and soil). It represents a particular water system, with its main supply and demand nodes and the links between them, both numerically and graphically. Catchment attributes such as river and groundwater systems, demand sites, wastewater treatment plants, catchment and administrative political boundaries are projected in a spatial environment. The concept-based representation of WEAP means that different scenarios can be quickly set up and compared. The system is scalable and exists of various modules that can be enabled and disabled.

WEAP users specify allocation rules by assigning priorities and supply preferences for each node; these preferences are mutable, both in space and time. WEAP then employs a priority-based optimization algorithm to allocate water in times of shortage. The challenge is to distribute the supply remaining after satisfaction of catchment demand. Water delivery to various demand elements is optimized, according to their ranked priority and accounting for in-stream flow requirements. This is accomplished using an iterative, linear programming algorithm. The demands of the same priority are referred to as “equity groups”. WEAP allocates equal percentages of water to the members of the same equity group when the system is supply-limited.

VirtualTracer approach

To evaluate water reuse processes in a river basin, a VirtualTracer (VT) approach was developed making use of the water quality modelling functionality in WEAP. Unique tracers are added as conservative water quality constituents to the non-consumed flow of each agricultural and urban demand site, in a concentration of 1 g/L. Concentrations of each of these tracers are tracked across a basin and reported for the inflows of each downstream demand site. Similarly, the VT approach tracks original sources of water across the basin by introducing a unique tracer for each source. A standardized workflow in Microsoft Excel was developed using VBA-API scripting, to export the large amounts of WEAP results on flows and concentrations and report key water reuse indicators in an automated manner.

The WEAP modelling software was adjusted in several ways to implement the VT approach. For each demand site, outflow concentrations of all simulated tracers were set to equal inflow concentrations of the previous timestep. In order to track tracers as they traverse reservoirs, two new reservoir water quality methods were introduced: "Same as Inflow", and "Simple Mixing". Same as Inflow sets the reservoir outflow concentrations of all water quality constituents equal to the inflow concentration, whereas the Simple Mixing method tracks the concentration of water in storage, using a weighted average to calculate a new concentration for each tracer each timestep, taking into account upstream inflows, evaporation and releases downstream. Thorough mixing of water in storage is assumed, so that the outflow concentration equals this average concentration. The Simple Mixing approach is required for reservoirs in cases where water is released from storage even though there is no inflow of water that timestep. Such a Simple Mixing approach is essential for tracer studies as the current one; for water quality modeling more advanced reservoir mixing processes might be needed.

The VT approach was developed to evaluate reuse between demand sites in WEAP. It is, however, flexible in terms of scale, as the user determines the nature of the entity represented by a single demand site. For models where a demand site represents a system of various individual users (e.g. an irrigation scheme consisting of multiple fields), reuse within the demand site is not explicitly considered by the VT approach. This internal / direct reuse should be accounted for in the WEAP loss rate and reuse rate parameters.

The Segura River Basin model

The schematization of the WEAP-VT model of the Segura River Basin was based on the 2010-2015 basin model developed for CHS using the SIMGES water resources model and the AquaTool interface (Andreu Alvarez et al., 2007; Andreu et al., 1996; CHS, 2013). For building the topology of the WEAP model, a lumped-aggregation strategy was adopted in order to reduce complexity of the original model while retaining the major water demand sites and infrastructure. The resulting schematization includes 16 river sections, 6 diversion canals, 32 reservoirs, 15 aquifers, 9 sources of external supply (desalination plants and interbasin transfer),

37 demand sites, 127 distribution canals, 29 drainage canals, and 9 environmental flow requirements (**Figure 5.2**). For the reservoirs, volume-elevation curves, monthly net evaporation values, and operational parameters are specified in the WEAP model. As part of the VT approach, 32 water quality constituents were introduced; 15 and 13 tracers for the agricultural and urban demand sites respectively, and 4 tracers representing the original sources of water in the basin: runoff generated in the catchment, aquifers, interbasin aqueducts, and desalination plants (Aldaya et al., 2019). A monthly timestep was used, corresponding with the application of the model for strategic evaluations.

Three different categories of water demand sites are considered in the model: agricultural (termed as *Unidad de Demanda Agraria*, or UDA, by CHS), urban (UDU), and environmental (UDE). The irrigation districts identified by CHS were aggregated into 15 nodes, coded as SUDAs. SUDA15 constitutes irrigated lands located in the Jucar Basin. For each irrigation demand site, cropped areas were provided for the 17 main crops in the catchment along with crop-specific water requirements varying per month. Thus, irrigation demands are calculated by the model based on a total of 255 separate units (17 crops times 15 irrigation systems). Water demands of the UDUs are calculated based on the number of equivalent inhabitants (permanent + seasonal population) at the municipal level and monthly dynamics in demand. Yearly environmental water demands of the wetlands and salt marshes in the basin are incorporated based on the difference between actual evapotranspiration and effective rainfall, as presented in the CHS River Basin

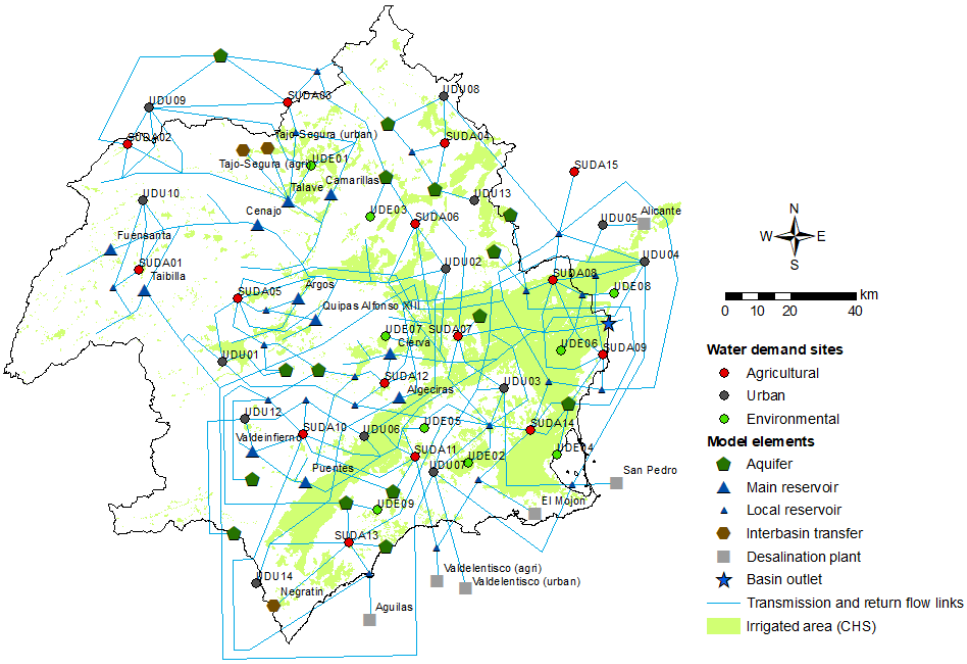

Figure 5.2. Schematization of the WEAP Segura model.

Management Plan (RBMP) (CHS, 2013).

Table 5.3 provides annual water demands of all water demand sites included in the WEAP-VT model.

The model was configured for the period 1999 - 2011. All results presented in this chapter are valid for the ten-year period of 2002 - 2011, with the first three years used for initialization purposes. Input data on catchment runoff, interbasin and desalination supplies, crop water requirements, UDU and UDE demands, and consumed fractions at the demand site level were obtained from the RBMP (CHS, 2013). The RBMP data on water demand concern managed water and are already corrected for rainfall and evapotranspiration of water in the soil profile. The WEAP-VT model therefore produces results on the use and reuse of these blue water flows.

5.3 Results and discussion

5.3.1 *Basin-scale analysis*

Table 5.1 presents the basin-scale blue water cycle of the Segura River Basin obtained from the WEAP model, and lists results from previous studies for reference. Hunink et al. (2019) applied data assimilation techniques using a combination of observations and the Water Accounting Plus framework to obtain overall water balance numbers. Contreras and Hunink (2015) used observations from CHS and other reported sources and used those data in the United Nations System of Environmental-Economic Accounting for Water (UN-SEEAW). Aldaya et al. (2019) performed a similar analysis, based on the blue water natural regime for the period 1980/81 - 2011/12 combined with more recent information on the supply of external water resources. As can be seen from Table 5.1, the WEAP-VT results are largely consistent with these previous studies. Deviations shown in the Table can to a large extent be explained by differences in period and methodological setup, where the dynamic modelling approach of WEAP-VT can be expected to provide a more accurate representation of basin-level consequences of interactions between water users. Based on Table 5.1, the current model is considered suitable to be used in a demonstration of the VirtualTracer approach to analyze sources of water and reuse between various users. Obviously, the model could benefit from further calibration and validation with more local data (if they come available) for application as an operational management tool.

As shown by Table 5.1, annual renewable water resources generated within the basin amounts to 58% of the net blue water supply. There is a considerable net groundwater depletion of 124 hm^3/yr to complement water resources obtained from the catchment, reservoirs, interbasin transfer aqueducts, and desalination plants. Clearly, most of the available water is consumed by irrigated agriculture. The overall consumed fraction is quite high at 0.86. This shows how the basin is effectively closed, with hardly any opportunities for additional development of water resources and limited scope for further expansion of reuse.

Table 5.1. Blue water cycle of the Segura River Basin (*ΔS* = storage change, *CF* = Consumed Fraction).

	Average annual flows (hm³/yr)			
	This study	Hunink et al. (2019)	Contreras and Hunink (2015)	Aldaya et al. (2019)
Period:	*2002 - 2011*	*1981 - 2000*	*2000 - 2010*	*various*
Segura catchment	763	-	-	854
Interbasin transfer (Tajo + Guadalquivir)	337	283	408	322
Desalination	106	158	158	193
Reservoir ΔS	-12	-30	-	0
Groundwater ΔS	124	185	243	231
Total inflows	**1,319**			
Irrigation	907	948	835	-*
Environment	43	44	-	39
Urban	136	128	96	96
Reservoir ET	53	-	-	75
Outflow	180	121	123	123
Total outflows	**1,319**			
Basin-level CF	**0.86**			

**1,366 hm³ of water use is reported, of which 124 hm³ returns to the system. These figures are, however, based on demand data rather than actual consumption.*

Consumption and supply of water are related to demands in Table 5.2. Gross water demands of both the irrigation and urban sectors, including direct reuse and return flows, are not fully satisfied by supply. On average, annual shortages amount to 241 hm³/yr (17%) and 19 hm³/yr (9%) respectively. It should be noted that the flows in Table 5.2 are summed for the individual water demand sites, causing double accounting to occur as a consequence of water reuse. CF values in Table 5.2 should therefore be interpreted as averages per sector at the user level, as opposed to the basin-level value in Table 5.1. With this in mind, values of 0.76 and 0.73 are relatively high, which is indicative of extensive local recycling (direct reuse) occurring within the demand sites defined in WEAP-VT.

Table 5.3 provides a disaggregation of gross demand, net demand, and the main relevant flows for the individual demand sites.

Table 5.2. Annual average supply and demand of different types of water users aggregated for the entire Segura Basin using the WEAP analysis (*CF* = Consumed Fraction).

	Irrigation	Urban	Environment	Total
Gross demand (hm^3/yr)	1,436	207	44	1,687
Net demand (hm^3/yr)	1,105	149	44	1,298
Supply (hm^3/yr)	1,195	188	43	1,425
Consumption (hm^3/yr)	907	136	43	1,086
Return flows (hm^3/yr)	288	52	0	340
Unmet demand (hm^3/yr)	241	19	1	261
CF (sum of individual demand sites) (-)	0.76	0.73		

5.3.2 *Analysis of original water sources and return flow reuse*

Original sources

Water users in the Segura River Basin obtain their resources from four distinct sources of water: runoff generated in the catchment, aquifers, interbasin transfers, and desalination plants. Timing of water delivered from the upstream mountainous catchments is managed through reservoir operations. Recharge of and extraction from the aquifers occur at the level of individual water users. On the other hand, the interbasin transfers and desalination plants are centrally managed pieces of infrastructure. To account for the physical, political and financial aspects associated with each of these water sources, it is relevant to understand the dependency of the various water users on each of these four distinct sources. Water extracted from a surface or groundwater source at a downstream point in the basin, may in fact have had a different original source due to being withdrawn and discharged upstream.

Figure 5.3 shows the WEAP-VT modelling results. The breakdown of original sources of water is presented for each water demand site. For some sites, such as the SUDA01 irrigation system which is located upstream, 100% of the water is withdrawn from the river and originates as renewable surface water resources from the upstream catchment. This is, however, not the case for most of the other agricultural demand sites, which have more complex networks of supply and reuse and rely on three or four different original water sources. Maintaining adequate water supply to these demand sites is especially challenging when these fractions are relatively equal, such as for SUDA10, SUDA11, SUDA12, and SUDA14. In general, the complexity of sources increases from the basin edges towards the river mouth.

From a scientific point of view, these kinds of results are essential to better understand water flows in complex and over-exploited basins. From a management perspective, these results are key to making more balanced and informed decisions on water allocation and abstractions. The monthly pattern of supply from each of the water sources to agricultural, urban and environmental water demand sites is

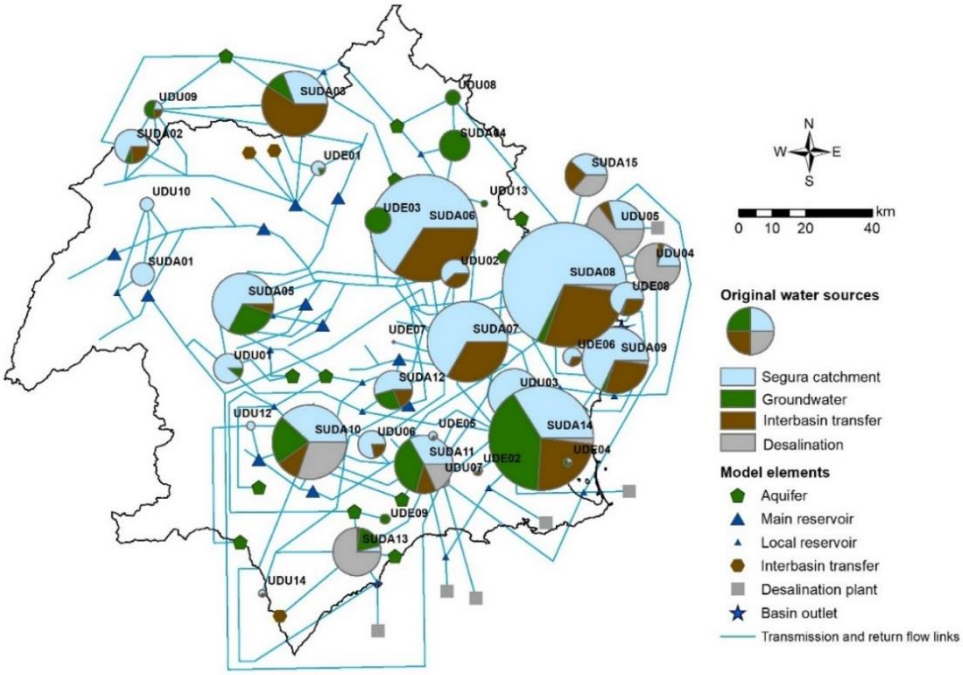

Figure 5.3. Original sources of water per demand site. Sizes of the pie charts are in proportion to total annual supply to each demand site. SUDA, UDU, and UDE refer to agricultural, urban, and environmental demand sites respectively.

presented in Figure 5.4. Clearly, the peak supply of internally renewable water resources of the Segura catchment water occurs in July and August, which reflects the large storage capacity of local reservoirs to hold the water for several months. A striking observation is that water supplied to the environment varies in terms of original sources. Over 20% of the water supply to wetlands and salt marshes originates from the interbasin transfers during 9 out of 12 months, reaching a maximum of 24% in March and July. This can be fully attributed to return flows from agricultural and urban sites, as no water from the interbasin aqueducts is allocated directly to the environment. This is important information, as the Tajo-Segura aqueduct is primarily intended, and viewed, as a source of water for irrigation with limited impact on environmental flows (Pérez-Blanco et al., 2020). The WEAP-VT results show that changes in transferred volumes to the Segura Basin will not only affect agriculture, but also supply to urban (with 15% of annual inflow supplied originally by interbasin transfers) and environmental water demand sites. This is particularly relevant in the context of climate change, which is expected to reduce supply from the Upper Tajo by at least 70% (Pellicer-Martínez and Martínez-Paz, 2018). In a similar fashion, Figure 5.4 shows that desalinated water, with urban water supply as its primary purpose, also constitutes a minor source of supply to agricultural and environmental demand sites.

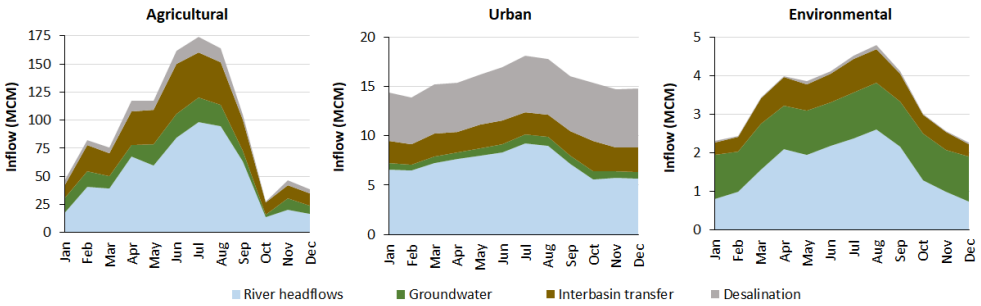

Figure 5.4. Original sources of water for different types of water use in the Segura River Basin. Percentages indicate the overall contribution of the respective source to annual average supply.

Reuse of demand site return flows

As mentioned earlier, downstream implications must be assessed when planning policy mechanisms or technological interventions that may affect local withdrawals or the partitioning between consumptive and non-consumptive use. This requires knowledge of the downstream reuse of return flows, particularly in the context of irrigation systems, which typically have substantial return flows and are often targeted by water saving measures. In the Segura River Basin, there is on-going debate on modernization of traditional irrigation canals (*azarbes*), and how this will affect riverine ecosystems and downstream water users (Trapote Jaume et al., 2015). Figure 5.5 shows as examples for three irrigation systems (SUDA05, SUDA06, SUDA07) the volume of non-consumed flow and which downstream system benefits from this return flow. For example, via the complex network of canals and local reservoirs, nine irrigation systems, urban supply and environmental sites depend on a total of 22.7 hm^3/yr of return flow generated by SUDA05. By contrast, 38.1 hm^3/yr (73%) of the reused return flow of the large irrigation system SUDA07 is used by a single demand site, SUDA08.

Whether the supply to downstream users is significantly affected by changes in non-consumed flow at a demand site, depends on the *DRR* indicator: the ratio of actual return flow reuse (such as presented in Figure 5.5) to the total volume of non-consumed water that is released back into the system. Typically, such upstream-downstream interactions through water reuse make up a balanced system that has evolved over many years, and trade-offs may occur once the upstream situation is changed. This is particularly the case for water users with both high *DRR* values and high return flow volumes.

Table 5.3 lists annual average *DRR* values, as well as other key indicators and flow volumes, for each demand site in the Segura Basin. Strikingly, several demand sites have *DRR* values above 1, which indicates that their non-consumed flow is withdrawn more than once downstream. This is also the case for SUDA05, SUDA06, and SUDA07, presented in Figure **5.5**. In other words, the return flow of these irrigation systems is already "overcommitted" to downstream reuse, and

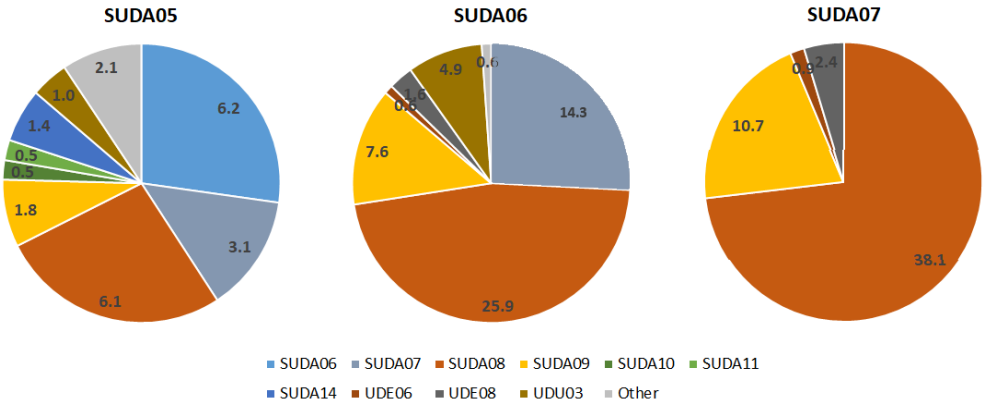

Figure 5.5. Destinations of return flow for three agricultural demand sites with high downstream reuse volumes (hm³/yr). All downstream sites at which less than 0.5 hm³/yr of return flow is reused are grouped as "Other".

increasing local consumptive use would lead to further intensification of the system and potential downstream water shortages.

Table 5.3. Demand, supply, return flow and water reuse indicators calculated at the demand site level. D_{gross} = gross demand (hm³/yr), D_{net} = net demand (hm³/yr), Q_w = total withdrawal (hm³/yr), Q_{nc} = non-consumed flow (hm³/yr), CF = Consumed Fraction (-), DRR = Degree of Return flow Reuse (-), RD = Reuse Dependency (-), C = Coverage (%).

	Demand site	D_{gross}	D_{net}	Q_w	Q_{nc}	CF	DRR	RD	C
Agricultural	SUDA01	15.0	6.5	8.9	5	0.44	1.36	0.02	59%
	SUDA02	19.3	11.6	18.7	7.5	0.60	1.25	0.42	97%
	SUDA03	85.4	66.8	68.5	14.9	0.78	0.87	0.03	80%
	SUDA04	43.7	36.2	15.2	2.6	0.83	0.00	0.13	35%
	SUDA05	84.3	58.1	61.5	19.1	0.69	1.19	0.03	73%
	SUDA06	185.6	140.7	185.6	44.9	0.76	1.23	0.06	100%
	SUDA07	105.1	58.8	105.1	46.3	0.56	1.12	0.21	100%
	SUDA08	250.3	169.7	250.3	80.6	0.68	0.24	0.43	100%
	SUDA09	71.3	61.9	71.3	9.4	0.87	0.89	0.60	100%
	SUDA10	94.1	79.4	89.3	13.9	0.84	1.13	0.04	95%
	SUDA11	101.0	83.3	54.8	9.6	0.82	0.88	0.08	54%
	SUDA12	42.6	32.4	23.8	5.7	0.76	1.19	0.06	56%
	SUDA13	83.2	75.0	36.8	3.7	0.90	0.00	0.05	44%
	SUDA14	223.1	200.7	176.4	17.7	0.90	0.07	0.03	79%

Table 5.3. (Continued).

	Demand site	D_{gross}	D_{net}	Q_w	Q_{nc}	*CF*	*DRR*	*RD*	*C*
	SUDA15	32.2	24.0	28.7	7.3	0.74	0.00	0.54	89%
	Total	**1436.2**	**1105.2**	**1194.9**	**288.2**	**0.76**	**0.76**	**0.19**	**83%**
Urban	UDU01	16.1	10.0	13.8	5.2	0.62	0.65	0.00	86%
	UDU02	12.8	11.5	12.8	2.2	0.83	0.62	0.04	100%
	UDU03	44.5	38.3	44.5	6.2	0.86	0.91	0.17	100%
	UDU04	39.0	25.3	34.3	12.2	0.64	1.35	0.08	88%
	UDU05	61.6	43.1	52.6	15.8	0.70	0.24	0.01	85%
	UDU06	14.2	7.1	12.2	6.8	0.45	0.92	0.03	86%
	UDU07	3.6	3.1	3.1	0.4	0.86	0.96	0.00	89%
	UDU08	3.5	2.5	3.5	0.9	0.73	2.01	0.00	100%
	UDU09	5.3	3.7	5.2	1.7	0.68	0.95	0.42	98%
	UDU10	2.9	2.0	2.9	0.9	0.70	0.98	0.00	100%
	UDU12	1.8	0.9	1.1	0.6	0.50	1.64	0.00	61%
	UDU13	0.7	0.5	0.7	0.2	0.70	0.89	0.00	100%
	UDU14	0.8	0.6	0.8	0.2	0.70	0.28	0.03	100%
	Total	**206.8**	**148.6**	**187.5**	**53.3**	**0.73**	**0.80**	**0.07**	**91%**
Environmental	UDE01	4.3	4.3	3.3	-	-	-	0.31	77%
	UDE02	1.3	1.3	1.3	-	-	-	1.00	100%
	UDE03	10.7	10.7	10.7	-	-	-	0.00	100%
	UDE04	1.2	1.2	1.2	-	-	-	0.99	100%
	UDE05	1.2	1.2	1.2	-	-	-	0.66	100%
	UDE06	5.5	5.5	5.5	-	-	-	0.64	100%
	UDE07	0.1	0.1	0.1	-	-	-	0.20	100%
	UDE08	17.9	17.9	17.9	-	-	-	0.56	100%
	UDE09	1.5	1.5	1.5	-	-	-	0.00	100%
	Total	**43.7**	**43.7**	**42.7**	**-**	**-**	**-**	**0.41**	**98%**

Reuse Dependency

The relevance of upstream return flow reductions to supply to an individual water user, depends on the fraction of its supply that was previously discharged by upstream users as return flow (Reuse Dependency, *RD*). Tabulated *RD* values per demand site are listed in Table 5.3. Figure 5.6 visualizes *RD*, disaggregated for the fraction to which each individual upstream site contributes to the water supply.

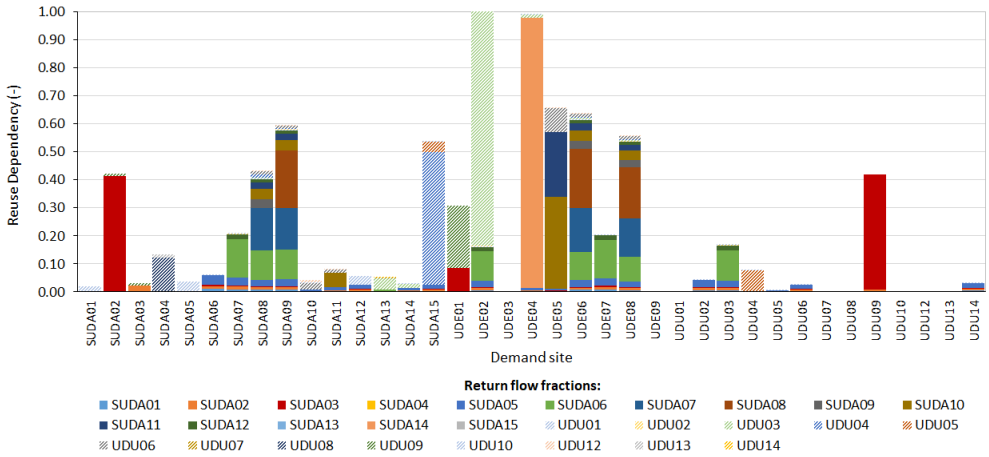

Figure 5.6. Reuse Dependency of each water demand site, disaggregated per upstream user contributing to the supply.

There is clearly a large variability of overall *RD* values in the basin. While in particular several urban demand sites only withdraw water that was not previously used upstream, there are two downstream environmental demands (UDE02 andUDE04) which rely fully on water that was already withdrawn at least once. With regards to irrigation schemes, SUDA08 and SUDA09 rely for 43 and 60% of their water resources on non-consumed water, and in particular the water drainage and percolation processes of SUDA06 and SUDA07. Over half (54%) of the water supplied to the irrigated area in the Jucar Basin, SUDA15, is comprised of return flows. It is interesting to note the number of different colors in each bar in Figure 5.6, i.e. the number of upstream users whose return flow is a source of supply. Demand sites with a relatively high overall *RD*, combined with a low amount of return flow sources, can be considered the sites most vulnerable to upstream changes in consumption. Examples include SUDA02, SUDA15, UDE04, UDE05, and UDU09.

Environmental water demands are typically the most vulnerable in river basins, as water supply is less well-monitored and sites of ecological relevance are commonly located downstream. Figure 5.7 presents the *RD* aggregated for each of the three water use sectors evaluated in the WEAP-VT model. Clearly, also in the Segura Basin, ecological demand sites are the most vulnerable to changes in upstream return flows. In the current situation, 41% of all water supply to the basin's wetlands and salt marshes depends on return flows from upstream water users. The major portion (85%) of return flows reused at environmental demand sites originate from irrigated agriculture.

The Natural Park of El Hondo (2,495 ha, UDE08) is one of the ecologically most valuable areas in the Segura Basin, and its most important environmental water user in terms of annual volume. The park is included in the Ramsar convention list

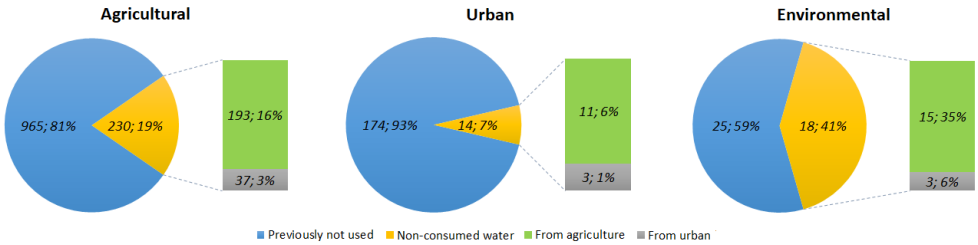

Figure 5.7. Sector-specific dependency on non-consumed water and water not previously withdrawn (blue). Values are in hm³/yr, percentages are relative to total water supply.

of protected wetlands and as a Special Protection Area under the European Union Directive on the Conservation of Wild Birds. As mentioned in Table 5.3, average RD of the El Hondo Natural Park is 0.56 on an annual basis. Figure 5.8 demonstrates how water supply to the El Hondo wetlands depends on upstream return flows through the year. Panel *a)* shows that, on average, *RD* drops to 27% during the wet January month, while it reaches a peak of over 70% in August. During the dry year 2006, as shown in panel *b)*, *RD* fluctuates around 0.80 for the months March to November. Particularly under these conditions, reduced return flows of irrigation systems such as SUDA07 and SUDA08 would negatively impact water availability to the El Hondo wetlands.

5.3.3 *Scenario analyses of unmet demand and coverage*

As shown in Table 5.2, despite the supply of external water resources and water reuse within the basin, there is still an overall unmet demand in the Segura River Basin. In total, 19 out of 37 demand sites experience an insufficient supply during an average year. Local efficiency increases are typical mitigation measures considered by water managers under these conditions. Two scenarios were simulated to explore the potential impacts of such measures. Both scenarios consider Consumed Fraction (*CF*) increases of all agricultural and urban demand

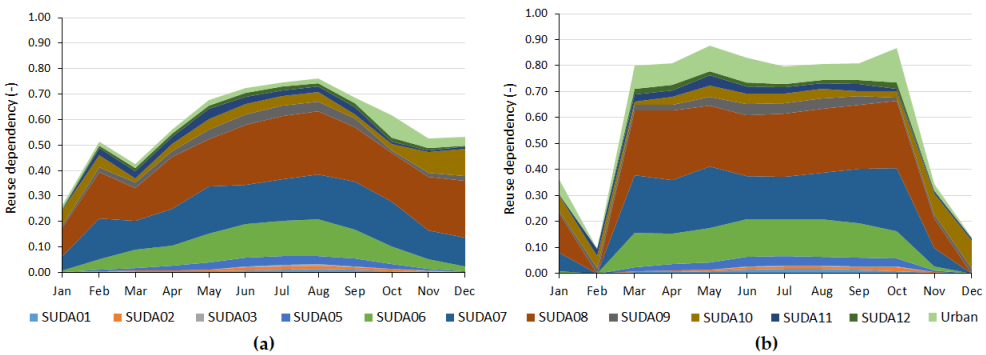

Figure 5.8. Monthly dependency of the El Hondo wetlands on upstream non-consumed water: (a) annual average for the period 2002 - 2011; (b) in 2006, a dry year.

sites to 0.9, which is the maximum local *CF* occurring in the baseline situation. In the first scenario (*CF_90*), it is assumed that no legal or geographical limitations are in place to stop water users from maximizing consumptive use of the volume of water withdrawn under baseline conditions (2002 – 2011). The second scenario (*CF_90_pol*) represents the situation where caps on consumptive use per demand site are enforced by the water authorities. There is no possibility to increase consumption in this scenario, thus effectively leading to reduced gross demands.

Table 5.4 summarizes the demand and supply situation for the three main water-using sectors and the overall basin, under both scenarios. All key water balance components at the demand site level are provided in Appendix F (Table F.1). In *CF_90*, with stable demands but increasing efficiencies, consumption logically increases while return flows decrease. The high reduction of return flows leads to a rise of unmet demand at the basin level of 28%, and thus a decrease in coverage (supply divided by gross demand). Less water is available to demand sites with a high Reuse Dependency, resulting in a reduction of total supply. Figure 5.9 shows the changes in unmet demands at each demand site. Unmet demands of all sites with existing water shortages are increased, while unmet demands also newly occur at 9 sites.

Under the *CF_90_pol* scenario, the reduction in gross demand is almost fully reflected in the decrease of total supply to the demand sites. Combined with higher efficiency rates of individual demand sites, this logically leads to a substantial decrease in return flows. As the assumed restrictions on consumptive use are enforced on the basin scale, this scenario does not lead to a mere reallocation of flows, but instead reduces overall unmet demand in the basin by 18%. Figure 5.9 visualizes the changes in unmet demand for each site. Despite lower gross water demand, the overall reduction in unmet demand does result in a minor increase of consumption at the basin level, as some of the water shortages under baseline

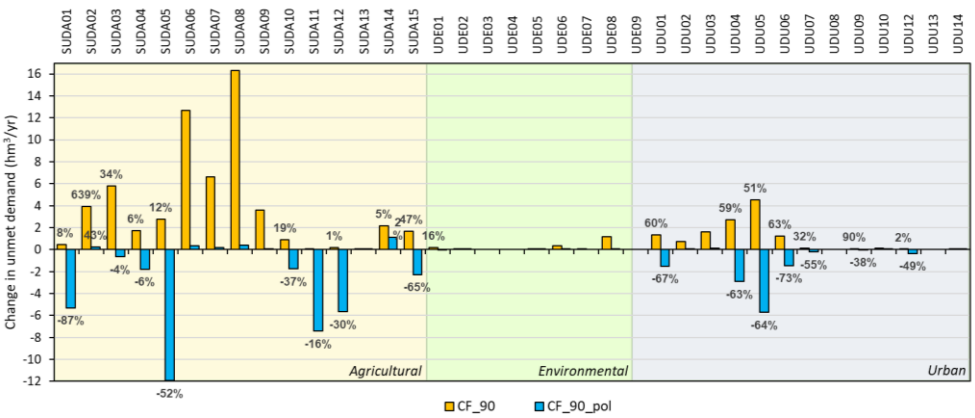

Figure 5.9. Changes of unmet demand in the two scenarios, with respect to baseline conditions. Percentages indicate the relative unmet demand change for all demand sites with nonzero unmet demand under reference conditions.

Table 5.4. Water demands, supply, and consumption simulated under the two scenarios. All values are in hm³/yr. Percentages show changes of basin-wide totals compared to the reference values provided in Table 5.2. SUDA, UDU, and UDE refer to the totals for all agricultural, urban, and environmental demand sites, respectively.

	CF_90					*CF_90_pol*				
	SUDA	**UDU**	**UDE**	**Total**		**SUDA**	**UDU**	**UDE**	**Total**	
Gross demand	1,436	207	44	1,687	*0%*	1,228	165	44	1,437	*-15%*
Net demand	1,105	149	44	1,298	*0%*	1,105	149	44	1,298	*0%*
Supply	1,136	175	41	1,352	*-5%*	1,021	158	43	1,222	*-14%*
Consumption	1,022	158	41	1,221	*+12%*	919	142	43	1,104	*+2%*
Return flows	114	18	0	132	*-61%*	102	16	0	118	*-65%*
Unmet demand	300	32	3	335	*+28%*	207	7	1	215	*-18%*
Coverage	79%	85%	93%	80%		83%	96%	98%	85%	
Basin-level *CF*				0.94	*+0.08*				0.87	*+0.01*

conditions are alleviated by higher volumes of water remaining in streams and reservoirs. The basin-level *CF* remains largely stable, thus maintaining a similar level of river outflow as under baseline conditions.

5.3.4 *General implications for water resources management*

Limited knowledge of use, reuse, and original sources of water has led to ineffective or even harmful water management decisions. Subsequent conflicts over water resources have been described extensively for larger scales such as river basins (e.g. Karimov et al., 2010; Molle et al., 2018). Also on smaller scales, between irrigation systems or even within one canal system, those conflicts are often based on uncertainty of source of water and reuse (e.g. Gonçalves et al., 2020; Ricart and Rico, 2019).

Implementing the WEAP-VT methodology, as presented in this chapter, contributes to water resources management by quantitatively tracking use and reuse of water per unique combination of source and destination. This allows the results to be interpreted at the full range of spatial scales, from the individual demand site to the entire basin. In addition, the use of a dynamic model incorporates the monthly variability of both water availability and demand. By allowing for evaluation of different scenarios related to consumed fractions and water demand, the methodology not only serves to characterize a system but can also support targeted interventions related to management, policy, and technology. In this manner, it is complementary to various water accounting methods that have been developed for the basin level and annual time scales, such as Water Accounting + (Karimi et al., 2013a).

The detailed information that is generated by the WEAP-VT methodology complies with the data requirements of informative water reuse indicators such as RD and *DRR*. The opportunity to quantify these indicators on different spatial scales, as well as for each month of the year, can support a range of water resources management applications which require information on upstream-downstream interactions. *DRR* assessments are typically relevant for irrigated agriculture located upstream, where users with a high value play an important role in the water reuse cascade and are in most cases not an appropriate target for water saving measures. Analyzing RD of high-value downstream water users, such as nature conservation areas, sheds light on their dependency on upstream return flows and can inform policies for ensuring adequate supply.

The scenario simulations for the Segura River Basin represent the extreme ends of possible impacts of efficiency-enhancing interventions in terms of trade-offs between users: (1) interventions that fail, from a basin perspective, in terms of water conservation and satisfying user needs (*CF_90),* and (2) interventions which succeed in motivating water users to fully adapt their withdrawals to their consumptive needs (*CF_90_pol*). Results of these two scenarios are of broader interest to inform policy measures in water-scarce basins. Two main benefits can be observed from the *CF_90_pol* scenario: (i) alleviation of part of the water shortages occurring in the basin, and (ii) a reduction in withdrawals of a comparable volume to the current extraction of non-renewable groundwater, indicating scope for reducing aquifer depletion when water allocation policies are properly implemented (and, specifically for the Segura case, supply of external resources remains constant). Another benefit that is often assumed, "freeing up" of water at the basin level by reducing consumptive use, does not occur due to extensive water reuse and an already high *CF* under baseline conditions. In fact, water consumption under *CF_90_pol* even slightly increases. This shows that potential impacts of local alterations of consumed and non-consumed flows are particularly complex in a system that is already under stress, i.e. experiencing substantial unmet demand. Even when incentives are in place for users to adapt their withdrawals to their consumptive needs, thus leaving more water in-stream, overall basin consumption may increase as downstream water shortages are consequently reduced. This is an important notion that needs to be explicitly considered in the on-going debate of scale dependency of efficiencies and water saving options. Also, the scenario results demonstrate that the use of a modelling tool is required to evaluate the complex interlinkages and nonlinear relationships occurring in a system with intensive water reuse and various water users experiencing stress.

In its current form, WEAP-VT was developed specifically to perform quantitative water reuse analyses across river basins. Depending on the local context, additional aspects of water withdrawals and return flows should be considered when making decisions on water allocation and local-scale efficiency improvements. In many river basins, including the Segura, water quality is an important issue to take into account. On the one hand, generating significant return flows in irrigation schemes

can be a deliberate choice to avoid salinization of soils (leaching). On the other hand, return flows typically have higher pollution loads than non-return flows. When more water is left in-stream due to enhanced efficiency and effective policy mechanisms, more water of higher quality remains available for downstream use. Combining WEAP-VT with modelling of water quality to evaluate such processes and trade-offs is a recommended subject for future research.

The presented WEAP-VT application for the Segura Basin made use of the data available from the Segura River Basin Management Plan, which contains recorded data for the key inputs required such as river streamflow at various points, water demands, and consumed fractions at the user level (CHS, 2013). To allow for applications in river basins that are less well-monitored, the WEAP-VT approach is also directly compatible with recently developed methods to assess consumed fractions for irrigation schemes (Simons et al., 2020). Furthermore, the integrated rainfall-runoff module in WEAP allows for simulation of upstream catchment processes in case inflow measurements are not available.

5.4 Conclusions

This chapter successfully demonstrates that the WEAP-VirtualTracer approach can be applied to quantitatively track non-consumed flows and their (un-)planned reuse across complex systems. Analyses can be performed under baseline conditions as well as different management scenarios, to reveal the impact of local interventions across a river basin in terms of (adequacy of) supply, consumption, and return flows. Outputs of a WEAP-VT analysis comply with the input data requirements of pre-developed water reuse indicators, which can be quantified on different spatial scales and for monthly to multi-annual periods.

For each water demand site in the Segura River Basin, the various original sources of water supply, dependency on upstream return flows, and downstream reuse of its own return flows were evaluated. Based on these results, agricultural, urban, and environmental water demand sites were characterized in terms of their suitability for water saving measures and their vulnerability to reduction of upstream return flows. As the Segura River Basin can be considered illustrative for (semi-)arid basins with high competition for water resources, it can be concluded that WEAP-VT holds great potential for supporting sensible water saving measures and targeted efficiency improvements in such basins worldwide.

6

CONCLUSIONS AND IMPLICATIONS

This chapter presents the main research findings and places them in a broader societal and scientific perspective. Section 6.1 provides the main conclusions in the context of the five guiding research questions. Section 6.2 discusses how the various methods developed and demonstrated in this dissertation can be connected in an integrated toolbox to inform water managers on indirect reuse. Finally, recommendations for future research directions are given in Section 6.3.

6.1 General conclusions

6.1.1 *What are the key hydrological processes associated with water reuse in a river basin among users of varying nature, and how should these be described in a sound accounting framework?*

Indirect water reuse is the result of a wide range of processes, driven by climatological, geographical, technological, environmental, and socio-economic factors. Worldwide case studies over the past decades have shown that inadequate and ambiguous concepts and terminology have, often unintentionally, been deployed to describe complex water reuse systems, resulting in adverse impacts of interventions by water managers. In general, it can be stated that the more a river basin approaches the point of closure due to progressive water resources development, the greater the challenge as well as the need to apply sound accounting principles to basin water flows. This information is essential to support the implementation of effective water-related technologies, practices, and policies.

This dissertation concludes that a quantitative, multi-scale approach, rooted in basic hydrological principles, should be the starting point for understanding a water reuse system. To this end, a framework is proposed based on the generic *hydrological fractions* concept, as described by Frederiksen and Allen (2011), Perry (2011), and originally Willardson et al. (1994), tailored towards the hydrological distinctions that are relevant in the context of a water reuse analysis. The fundamental hydrological fraction to examine at the user level is the non-consumed fraction, which expresses how much of locally withdrawn water returns to the system for potential reuse downstream. Whether this water is indeed recoverable by downstream users depends, among others, on its destination, profitability of recovery, and its quality after initial use, with all of these aspects potentially varying over time. Conceptually, it is therefore important to distinguish recoverable and non-recoverable portions of the non-consumed fraction, which requires zooming out beyond the user level and assessing spatial hydrological connectivity and temporal dynamics of water flows. Both surface and groundwater need to be

explicitly included in an accounting framework for a comprehensive picture. Furthermore, a distinction between *blue* and *green* water use and artificial vs. natural flow pathways is important to shed light on opportunities for water managers to intervene.

The hydrological fractions school of thought typically emphasizes the danger of reducing non-consumed flows, potentially leading to a reallocation of water from downstream to upstream users because of existing indirect reuse. Although the occurrence of large-scale informal reuse is increasingly evident (and one of the fundamental reasons for embarking on this research), a preconceived assumption of the vast majority of return flows being reused in any context is just as unhelpful to water managers as overlooking reuse altogether. Both types of oversimplification need to be avoided by methodically quantifying the relevant fractions on different spatial scales, from the individual user to the full system. The basic framework proposed here purposefully excludes notions of benefits or productivity of water consumption, which typically are subject to priorities and interests of different stakeholders.

6.1.2 *What are the knowledge and data gaps related to existing methods for evaluating non-consumed water and its downstream reuse?*

The importance of incorporating indirect water reuse in hydrological analyses is by now widely acknowledged. This is evidenced, among others, by the extensive scientific debate on local irrigation efficiencies vs. basin-scale water accounting. As a result, multiple indicators have been developed which address one or several of the relevant dimensions to water reuse outlined in the previous section.

A comprehensive review of the available indicators shows that they differ substantially in terms of how explicit water reuse processes are described. Some indicators make use of proxy variables with a supposed correlation with water reuse, such as the total amount of water withdrawn upstream from a location (*Water Reuse Index*), or the geographical situation of a user within a basin (*Downstreamness*). Generally, the more explicitly an indicator incorporates quantitative water flows and the more dimensions of reuse it addresses, the more actionable it is for policy makers. For a given practical application in water management, it is advised that one or multiple water reuse indicators are critically selected depending on the purpose at hand.

Although a substantial body of work on water reuse indicators from a conceptual perspective is available, actual application of these indicators in scientific articles, technical reports and policy documents has been limited. This is mainly due to major challenges in gathering the required data for quantifying the indicators, particularly those in requirement of quantitative flow estimates. Water reuse itself is extremely difficult to measure in the field. Supply and return flow data at the user level are often sensitive and difficult to obtain. As demonstrated in this dissertation, even for well-monitored basins (e.g. Arkansas River Basin, Murray-Darling River Basin), these data are at present insufficiently available. Therefore, alternative

ways for gathering data on (non-)consumed fractions and connectivity between water users need to be explored.

Reviewing data needs of the various reuse indicators, and acknowledging the significant challenges in obtaining ground data on the relevant processes, the main knowledge gaps identified are twofold:

(i) the lack of a method for quantifying consumed fractions and return flows on different spatial scales;
(ii) the lack of a method for explicitly quantifying spatial flows between water users, as well as mixing processes occurring when return flows merge with existing sources and streams.

Satellite remote sensing and simulation modelling have previously shown potential in producing related information. However, no previous research has been identified that develops and demonstrates a consistent methodology integrating these methods to evaluate water (re)use processes across river basins.

6.1.3 *What is the potential of satellite-derived data products to evaluate spatiotemporal dynamics of water availability and water use?*

To satisfy the aforementioned data requirements, an essential first step is to have reliable data on water consumption through actual evapotranspiration (ET_{act}). For addressing the spatiotemporal dynamics of reuse, these need to be available across spatial scales and for a long time series. The availability of global satellite-derived ET_{act} data in the public domain is rapidly increasing, and if found to be of sufficient quality, these data could provide a promising basis for analyses of water reuse.

Chapter 3 of this dissertation demonstrates how the integration of different ET_{act} datasets supports an ensemble product which is consistent with remotely sensed rainfall (P) and measured streamflow (Q) data for the Red River Basin in China and Vietnam. The individual ET_{act} datasets result from fundamentally different algorithms, relying to various extents on inputs such as land surface temperature and leaf area index, and each has a different approach to solving cloud cover issues. Constructing an ensemble product makes use of the complementary qualities of the algorithms over heterogeneous terrains.

It was found that, in conjunction with remotely sensed rainfall (P) and land use / land cover datasets, satellite-derived ET_{act} supports application of sound water accounting on the yearly and multi-annual scale. In addition, it proved possible to model monthly storage changes solely based on satellite derived P and ET_{act}. Even for a challenging basin in terms of atmospheric conditions such as the Red River Basin, meaningful conclusions were drawn on the hydrological system, without applying sophisticated simulation models. A main conclusion is therefore that monthly satellite-derived ET_{act} products provide a promising basis for stand-alone hydrological analyses, as well as for feeding, constraining, and calibrating hydrological models. Integration with more advanced simulation algorithms facilitates quantification of hydrological processes that take place on the daily or

weekly time scale, or processes that cannot be assessed by remote sensing alone, such as withdrawals, reuse, and the partitioning of rainfall surplus into groundwater and surface water.

6.1.4 *Can the consumptive and non-consumptive portions of water use be quantified based on satellite remote sensing data?*

In order to support assessment of water reuse, total water consumption needs to be expressed against the volume of water withdrawn (the Consumed Fraction, *CF*). Here, two major challenges come into play, as water withdrawal data are generally very difficult to obtain, and separation of water consumption into artificially withdrawn water ("blue" water) and ET_{act} dependent on local precipitation ("green" water) is challenging. Similar challenges have prompted scholars to explore the use of the Budyko Hypothesis (BH) in regions with anthropogenic impacts on the water balance, which so far has yielded encouraging results.

This research demonstrates a new method for spatially quantifying consumptive use of irrigation water based on the BH for the Indus Basin Irrigation System (IBIS) in Pakistan. Green and blue water consumption (ET_{green} and ET_{blue}, respectively) were successfully separated using reference evapotranspiration and precipitation data, and total water supply was calculated at the canal command area level.

It was found that ET_{act} in IBIS on average is 927 mm/yr, out of which 76% depends on irrigation water. The average *CF* of IBIS canal command areas was calculated at 0.52, ranging between a minimum of 0.38 and a maximum 0.66. By comparing BH-derived total blue water supply with long-term main canal diversions, it was concluded that most command areas rely substantially on water not diverted at the primary canal head. The relatively low *CF* values and the fact that long-term canal supplies largely suffice to sustain ET_{blue}, indicate that the IBIS is characterized by extensive reuse of non-consumed flows *within* CCAs (local pumping). At the same time, a notably higher *CF* at the IBIS level (0.71 – 0.93) shows that reuse of non-consumed water *between* CCAs cannot be neglected, by capturing drainage water from upstream CCAs entering from pathways other than the primary canal. Although the IBIS is generally not considered an efficient irrigation system, it is thus in fact well-adapted to reuse return flows on different spatial scales.

The demonstrated methodology supports quantification of both sides of the *CF* equation, thereby significantly advancing the understanding of system-scale water use and potential for water savings. In addition, it can be a starting point for evaluating reuse and dependencies between water users in river basins, facilitating targeted and more effective water allocation policies and water conservation measures. The use of global satellite-derived data products allows the procedure to be replicated in irrigated basins worldwide.

6.1.5 *How can spatial interactions and trade-offs between water users be tracked and visualized to support more effective water resources management?*

Full reliance on satellite-derived data limits the extent to which interactions and trade-offs between water users can be understood. A water resources model, including user-level water demands, is required to simulate these lateral processes and evaluate how they respond to different (management) scenarios. As demonstrated in this dissertation, integrating a VirtualTracer module into a water resources model can address these processes and produce the spatiotemporal outputs for quantification of water reuse on different scales.

The proposed WEAP-VirtualTracer (WEAP-VT) approach, based on a modified water quality model, was successfully applied to track non-consumed flows and their (un-)planned reuse across the Segura River Basin. At the water user level, the various original sources of water supply (surface water, groundwater, interbasin transfers, desalination plants), dependency on upstream return flows, and downstream reuse of its non-consumed flow were quantitatively evaluated. Agricultural, urban, and environmental water demand sites were characterized in terms of the potential for water saving measures and their vulnerability to reduction of upstream return flows (i.e. upstream classical efficiency increases). Due to the incorporation of water demand calculations, the basin-wide impacts of local interventions were evaluated in terms of (adequacy of) supply and water shortages.

Although calibration for individual basins is preferred, the basic input data for a WEAP-VT are derived from remote sensing and GIS data from the public domain. Consumed Fraction estimates, as derived from remote sensing by applying the Budyko Theory-based approach presented in Chapter 4, are an important input into the model. For the irrigation sector, assumptions of literature-based values of efficiencies that do not consider local conditions is therewith no longer required. The outputs of a WEAP-VT analysis are compatible with the input data needs of pre-developed water reuse indicators (see Section 6.1.1), which can subsequently be quantified on different spatial scales and for monthly to multi-annual periods. Therefore, the WEAP-VT approach holds significant potential for supporting sensible water saving measures and targeted local efficiency improvements in (semi-)arid basins around the globe. Section 6.2.4 elaborates on how the implementation of WEAP-VT in practice, as a final step in a technical analysis of water reuse, integrates the results from the other methods proposed in this dissertation and provides quantitative information on key water reuse indicators

6.2 A water reuse toolbox to support water managers

This dissertation essentially presents four primary building blocks that need to be part of a water reuse toolbox: (i) routines for extracting hydrological information from satellite-derived data products, (ii) an algorithm for estimating (non-)consumed fractions at different scales, (iii) a water resources model with a VirtualTracer approach for tracking water use and reuse, and (iv) meaningful

indicators to inform the development, implementation, monitoring, and evaluation of water management policies. The general structure of a toolbox for water reuse assessments, including the points of interaction with water managers, is schematized in Figure 6.1. This section discusses the rationale for connecting the individual building blocks and describes some of the practical applications of an integrated toolbox for water reuse assessment.

6.2.1 *Hydrological information from Global Satellite-derived Data Products*

The increasing availability of satellite-derived hydrological information in the public domain offers ample opportunity for obtaining insight in water resources conditions in poorly gauged regions. Parameters such as precipitation, actual

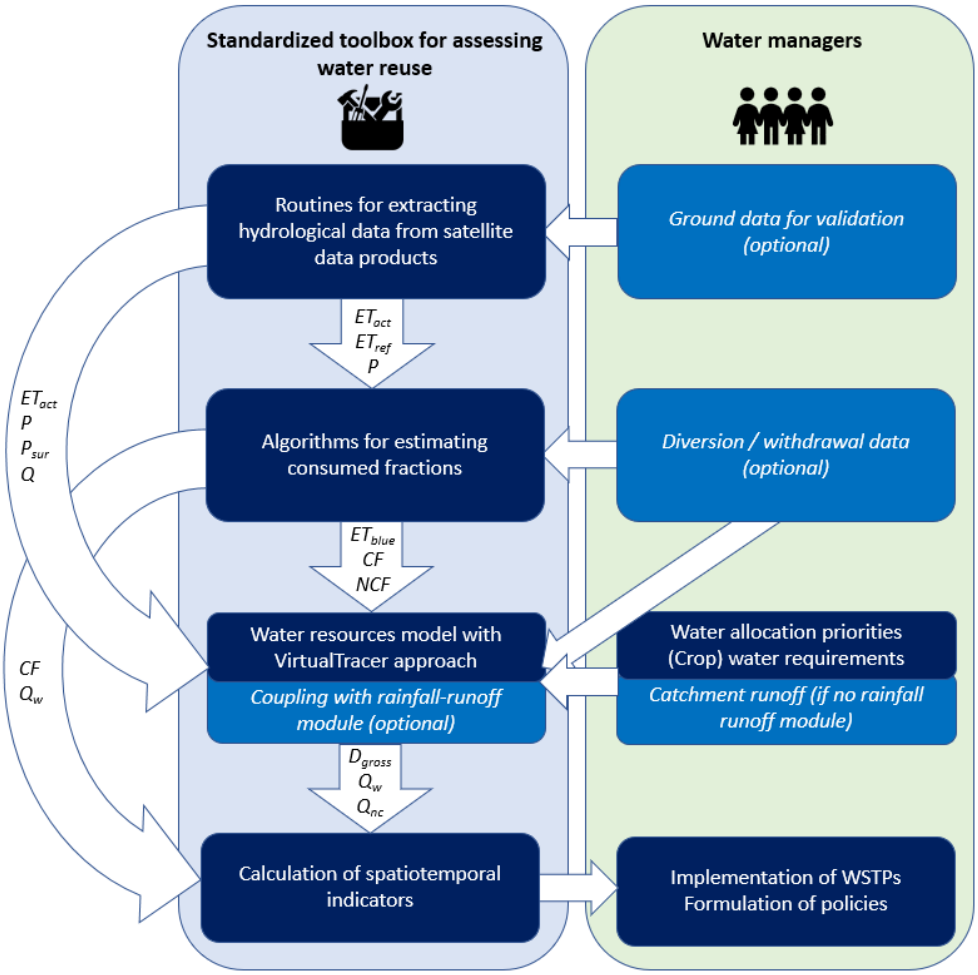

Figure 6.1. Schematic overview of the linkages between the components of a water reuse toolbox, and points of interaction with policy makers.

evapotranspiration, and reference evapotranspiration are instrumental to assessments of water reuse and are now available from various publicly accessible databases encompassing the entire globe. Recent developments linking these data archives to cloud computing facilities, such as through Google Earth Engine, further enhance the potential for incorporation in practical decision support tools.

As demonstrated in Chapter 3, assessments of rainfall surplus, storage dynamics, and long-term runoff can be performed by integrating monthly data of sufficient time period. The combination with satellite-derived land use maps supports Water Accounting studies, which communicate water resources related information and the services generated from consumptive use in a geographical domain to different stakeholders. The performance of available GSDPs is expected to differ regionally, depending on factors such as climate and land use. Therefore, auxiliary ground observations provided by water management authorities are helpful to identify the most suitable GSDP for a specific area or apply bias-correction or data integration procedures.

Next to the stand-alone analyses described above, the use of satellite-derived data products constitutes an essential foundation for subsequent operations executed in the water reuse toolbox. As shown in Chapter 4, ET_{act}, ET_{ref} and P are direct inputs into algorithms for determining CF. In addition, particularly in data-scarce areas, satellite-derived data products are crucial to feeding, calibrating, and constraining simulation models. These are the two ways in which GSDPs form the fundament of a water reuse assessment (Figure 6.1).

6.2.2 *Assessment of (non-)consumed fractions*

Monthly spatial data on ET_{act}, ET_{ref} and P from GSDPs are the main inputs into the algorithm for Budyko-based computation of the consumed and non-consumed fractions of water use, primarily targeted at irrigation. This component of the toolbox produces spatiotemporal estimates of total water supply, consumption of irrigation water, consumed fraction, and non-consumed flows. In their own right, these variables are of great value to decision makers, since alternative ways to assess them (other than by extensive field measurement campaigns) hardly exist.

As shown in Chapter 4, evaluating these parameters on different scales already provides implicit estimates of the degree of water reuse occurring in a basin. If additional data on particular components of water supply (e.g. canal water diversions, groundwater pumping) are inserted by water managers, the total water supply estimated by the Budyko algorithm can be partitioned into different sources. Calculating consumed fractions and non-consumed flows allows water managers to perform a preliminary identification of water users of which the scope for enhancing local efficiency may still be significant. It is important for CF to be defined relative to total blue water supply, as this is the supply component that is directly manageable and potentially affected by any new technologies or regulations put in place. However, as lateral interaction between users is not explicitly described,

such decisions should be taken carefully and evaluated in the light of the hydrological location within a basin.

The output of the Budyko-based analysis is used in the subsequent modelling component of the toolbox. *CF* is an important input that can be introduced into the model separately for each water demand site. Especially for basins with very limited field data available, estimates of supply from different sources and consumed blue water (ET_{blue}) can be used to constrain and/or validate the water resources model.

6.2.3 *Water resources model with a VirtualTracer module*

A water resources model is required for modelling lateral flows and relating demand to supply at the water user level. A model such as WEAP, demonstrated in Chapter 5, calculates demands, inflows, and outflows for each water demand site. The VirtualTracer module, developed for WEAP as part of this research, allows for tracking interactions and dependencies between water users. In this way, water reuse indicators can be quantified at different spatial levels. In addition, monthly simulation timesteps allow for examination of the results per month, season, year, and for long-term multi-annual periods.

The WEAP-VT demonstration in Chapter 5 relies on data on flows and supplies provided by the river basin authority. In many basins these data may not be available. The water resources model can then be expanded with a rainfall-runoff module for calculation of inflows to water sources, thus accounting for green water fluxes. Next to user-level *CF* values, data from GSDPs on precipitation, ET_{act} and long-term runoff data produced from the preceding steps can be used to feed, calibrate, or constrain the WEAP-VT model. The ideal picture is to ultimately have an integrated model of the green and blue water cycle, constrained and parameterized by satellite-derived information, including withdrawals from different sources, and incorporating connectivity through both surface water and groundwater. Follow-up research steps (see Section 6.3) and technological progress are expected to contribute to realizing this future vision.

This building block of the toolbox has the strongest interaction with water managers. The model allows for simulation of different water management and allocation scenarios, by varying local demands, efficiencies, and water distribution priorities. Through its graphical user interface, it provides a platform for visualizing and interpreting results and parameterizing scenarios. Although not further explored in this dissertation, this is also the step where water quality requirements, and deterioration after use, can be introduced.

6.2.4 *Water reuse indicators*

An elaborate review of previously developed indicators for expressing water reuse is provided in Chapter 2 of this dissertation. The WEAP-VT module allows for expanding this list with information on water demand, as expressed by the

Coverage (*C*) indicator (% of demand satisfied by supply). Water managers can have different objectives when deploying the water reuse toolbox, or individual components of it. Depending on these objectives, a selection of suitable indicators needs to be made.

As explained in Chapter 2, water reuse indicators can be organized in three groups, each of which are suitable for a certain type of application in water management. The first class (A) of indicators attribute a single value to a system of multiple water users, offering a basic indication of the scope for enhancing system-level efficiency and achieving water savings. Class B of reuse indicators directly or indirectly describe a water user's dependency on the supply of non-consumed water from upstream users, thereby characterizing a user's vulnerability to changes in upstream water resources conditions. Finally, Class C of reuse indicators define a water user based on the downstream reuse of its own non-consumed water, and thereby highlight its importance within the network of water use.

Table 6.1 highlights a number of typical applications of selected indicators quantified by the water reuse toolbox. To illustrate, a general narrative of a comprehensive application of indicators quantified by the water reuse toolbox could look as described below.

Table 6.1. Overview of example applications of key indicators quantified with the WEAP-VT methodology.

Key Indicator	Class	Application
Basin-scale Consumed Fraction (-)	A	• Assess the overall potential of „freeing up" water resources or planning additional water resources development
Reuse Dependency (-)	B	• Vulnerability assessments of individual water users, including ecosystems • Setting up water rights systems based on minimum return flows • Support upstream-downstream financing schemes, such as Payment for Ecosystem Services schemes and water funds
Coverage (%, supply / demand)	B	• Determine water shortages experienced across a basin • Evaluate impacts of local water saving and efficiency-enhancing measures on water stress across a basin
Degree of Return flow Reuse (-)	C	• Identify suitable locations for efficiency improvements (e.g. irrigation modernization) with beneficial impact on the basin scale • Support implementation of caps on consumptive use • Setting up water right systems based on minimum return flows (with a certain quality) • Better understanding of value of water in water market contexts

The water authority (WA) of a semi-arid river basin has established that an important urban area is experiencing regular, considerable shortages of supply. They order for an assessment of basin-scale *CF* to evaluate the scope for saving water, with the aim to release this additional water to the city currently under stress. Satellite-derived data and Budyko-based analyses show that basin-level *CF* is in fact already quite high, to the extent that although some water saving is still possible, most surface water and groundwater resources are already committed through direct use and indirect reuse. Still, due to the high priority attributed to the urban water use, the WA decides to implement a set of Water Saving Technologies and Practices (WSTPs) at irrigated sites where detrimental effects to downstream users are expected to be minimal. The WA identifies water users to be targeted from indicators quantified by the water reuse toolbox, based on low values of user-level *CF* (i.e. low *classical efficiency*), as well as low *DRR* to account for downstream implications. Then, different scenarios involving local WSTP implementation (e.g. introduction of high-tech irrigation systems) are simulated by modifying user-level *CF* and other model inputs where relevant, e.g. those related to adjusted land management. The results of these model runs are assessed in terms of their positive impacts on Coverage (*C*) for the high-priority city originally established as underserved, as well as any decrease in *C* for other, downstream users. Here, special attention is paid to downstream protected ecosystems previously identified as having high *RD* values under baseline conditions, to ensure that they remain unaffected. Additional policy mechanisms to complement the identified WSTPs, such as caps on consumptive use or minimum return flow releases, are evaluated with WEAP-VT to determine effective flow volumes. Finally, once the WSTPs and accompanying regulations are implemented in reality, their enforcement, monitoring and evaluation is supported by satellite-derived measurements of ET_{act} and Budyko-based estimates of ET_{blue} and blue water supply.

Obviously, the above description is an oversimplification when compared to real-life decision-making procedures, as it disregards interests of different stakeholders, excludes participatory processes, does not account for data inaccuracies, etc. Still, it does provide a basic idea of how the information generated by the water reuse toolbox can support different phases of policy development, monitoring and evaluation. Especially when further scientific and technological advances are achieved (see Section 6.3), implementation of the proposed tools can safeguard fundamental hydrological principles in water management. A simplified schematization is given by Figure 6.2.

6.3 Recommendations for future research

As demonstrated in this dissertation, evaluating dynamics and patterns of indirect water reuse across river basins involves a sequence of steps involving collection, processing, integration, and interpretation of geospatial data using various methods, as well as expressing and visualizing the resulting information in an actionable manner. The setup presented in Figure 6.1 visualizes the basic

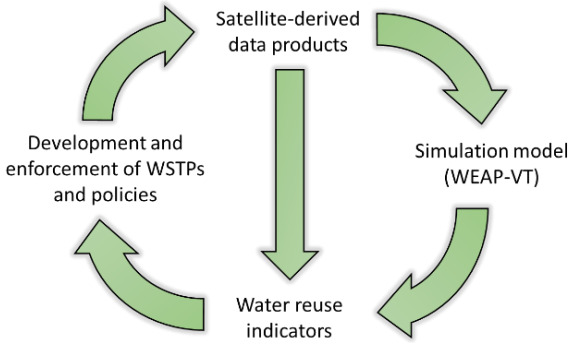

Figure 6.2. Incorporation of water reuse assessments in WSTP and policy development and enforcement.

procedures required for evaluating water reuse based on satellite remote sensing and simulation modelling. Due to the innovative nature of the individual building blocks of the proposed approach, as well as their integration, there is considerable scope for improvement as further scientific progress is made. Below, some directions are given with regards to the key points in the approach where additional scientific research could provide the greatest leaps forward.

Availability of accurate ET_{act} data with at least monthly time intervals is an essential foundation for the assessment of consumed fractions and reuse dynamics. Current global-scale ET_{act} products, such as utilized in this dissertation, typically have a spatial resolution of 500m – 1km. Improved spatial detail will help to attribute figures of consumptive use to individual water users, and enhance the flexibility of water consumption data to be applied across spatial scales. As sensor technology, data processing algorithms and computing power advance, progress is being made in determining ET_{act} at higher resolutions (e.g. Allies et al., 2020; Singh et al., 2020). Although these studies have been mostly restricted to the regional level, initiatives such as OpenET[5] aim to achieve upscaling to the global scale. The challenges involved in covering an extreme variability of terrains and climate conditions with a satisfactory accuracy level requires a method that is robust and (automatically) adaptable. Following the findings in Chapter 3, it is recommended to pursue an ensemble approach involving complementary ET_{act} models to address these challenges of upscaling. Future research should seek to obtain a thorough understanding of the strengths and weaknesses of individual algorithms in different geographical regions, climate zones and land use types. This is a prerequisite to determine the most appropriate way of integrating them in an ensemble product. Great collaborative efforts will be needed from the scientific community to achieve the required level of data integration and knowledge sharing.

[5] (*https://openetdata.org/faq.pdf*, retrieved 01-Jan-2021)

It is evident from this dissertation that *CF* is a crucial parameter to quantify in the process of evaluating water reuse, both at the level of the individual user and the overall hydrological system. Several encouraging lines of research towards *CF* estimation have been identified. Chapter 4 elaborately explores a Budyko theory-based approach, yielding promising results. As the boundaries of Budyko Hypothesis applications are pushed further beyond its originally intended scope, both in this research and by many other recent studies, it is crucial to gain a better understanding of its applicability under different conditions. Future research should therefore focus on evaluating the validity of the primary assumptions of Budyko theory across a range of spatial scales. A main research question to be answered is how the minimally appropriate surface area for application of Budyko theory (basin, sub-basin, canal command area, or even pixel level?) can be determined under different circumstances related to geography, hydrology, climate and human impact on the water cycle. Similar fundamental research is required regarding the limits of BH application for estimating ET_{blue} and blue water supply on different time scales, varying from monthly and seasonal to annual and multi-annual.

The importance of *CF* also warrants the exploration of other, independent methods for its estimation. As mentioned in Section 5.1, several experimental methods have been developed for determining ET_{blue} (numerator of the *CF* ratio) from satellite remote sensing in irrigated contexts, based on ET_{act} of nearby rainfed pixels or object-based classification. Although these studies provide encouraging results, they still require assumptions of efficiencies for conversion to applied irrigation amounts and therefore do not allow for calculating *CF* values (Foster et al., 2020). Constraining hydrological models with remotely sensed ET_{act}, or independently determined ET_{blue}, can enhance the understanding of hydrological connectivity between water users, leading to a more realistic representation of withdrawals, non-consumed water and reuse. Such a model would need to realistically represent green and blue water cycles, be capable of including different types of water users, withdrawals from different sources, and incorporate spatial connectivity (including groundwater) with a sufficient level of detail.

An interesting future study could investigate the feasibility of expressing *CF* as a function of various environmental factors, based on a set of calibrated hydrological models constrained by weekly or monthly ET_{act} for different study areas. The main objective of such a study would be to find out whether it is feasible to estimate *CF* from a limited number of input parameters, such as e.g. slope, NDVI, and soil water holding capacity, in an approach not unlike the pedotransfer functions used in soil hydrology (e.g. Zhang and Schaap, 2019). A well-calibrated model would allow for simulation of varying irrigation schedules within feasible boundary values, all leading to the same weekly ET_{act} derived from satellite-derived data products. This could yield a distribution of different *CF* values and an opportunity to determine the extent to which irrigation behavior determines *CF.*

Finally, it should be noted that this research focuses on the development and testing of methods and tools for characterizing individual water users and the hydrologically connected network of water users in which they are situated. Recent research suggests that this connectivity may even transcend river basin borders, through processes of regional groundwater flow and atmospheric moisture transfer (e.g. de Kok et al., 2018; Gleeson et al., 2020). In any case, the drivers and consequences of water reuse do transcend scientific disciplines and can be agronomical, economic, chemical, social, and political in nature. When aiming to sustainably alter the hydrological reality, quantitative hydrology should therefore be complemented with multidisciplinary research.

APPENDICES

Appendix A Glossary

Term	Definition
Anthropogenic flows	Flows that are regulated by man-made hydraulic infrastructure such as drains, sewerage, aqueducts, etc.
Blue water	All freshwater stored in lakes, streams groundwater, glaciers and snow
Consumed water	Water that is no longer available because it evaporated, was transpired by plants, incorporated into products or crops, or consumed by people or livestock
Gross demand	The water requirement of a water user after return flows and on-site recycling are taken into account
Gross inflow	The total amount of water that flows into the domain, this includes precipitation plus any inflow from surface or groundwater sources and desalinized water
Natural flows	Flows that are defined by natural processes
Net demand	The water requirement of a water user before non-consumed flows and on-site recycling are taken into account
Non-consumed water	Water that is not consumed in the process of water withdrawal
Non-recoverable water	Non-consumed water that cannot be reused at a downstream location for various reasons
Recoverable water	Non-consumed water that can be captured and reused at a downstream location
Reserved flow	Surface water that has been reserved to meet committed flows, navigational flow, and environmental flow
Return flow	See: *Non-consumed water*
Unmet demand	The amount of a user's gross water demand that is not met by supply.
Water recycling	Reuse of water on-site for the same purpose
Water reuse	Downstream re-application of non-consumed water for further use with or without prior treatment. Water reuse includes the dependency of natural systems on return flows, e.g. for inundation of wetlands

Water use	Any deliberate application of water to a specified purpose. Part of the water will evaporate, another part will return to the catchment where it was withdrawn, and yet another part may return to another catchment or the sea

Appendix B Rainfall data

Table B.1. Metadata of rainfall stations

	Station name	Latitude	Longitude	Country	Elevation (m)	Source
1	Ba Vi	21.10	105.43	Vietnam	20	NCHMF
2	Bac Can	22.15	105.83	Vietnam	176	GSOD
3	Bac Ha	22.53	104.28	Vietnam	107	NCHMF
4	Bac Me	22.73	105.37	Vietnam	380	NCHMF
5	Bac Quang	22.50	104.87	Vietnam		NCHMF
6	Bac Yen	21.25	104.42	Vietnam	65	NCHMF
7	Ban Cung	20.75	105.05	Vietnam		NCHMF
8	Bao Lac	22.95	105.67	Vietnam	283	NCHMF
9	Binh Lu	22.37	103.61	Vietnam	636	NCHMF
10	Bounneua	21.63	101.88	Lao PDR	923	GSOD
11	Cao Bang	22.67	106.25	Vietnam	260	GSOD
12	Chiem Hoa	22.15	105.27	Vietnam	56	NCHMF
13	Cho Ra	22.45	105.72	Vietnam	210	NCHMF
14	Chuxiong	25.02	101.52	China	1773	GSOD
15	Co Noi	21.13	104.15	Vietnam	704	NCHMF
16	Cuc Phuong	20.23	105.72	Vietnam		NCHMF
17	Dali	25.70	100.18	China	1992	GSOD
18	Dien Bien	21.35	103.00	Vietnam		NCHMF
19	Dinh Hoa	21.90	105.63	Vietnam		NCHMF
20	Guangnan	24.07	105.07	China	1251	GSOD
21	Ha Dong	20.97	105.77	Vietnam	8	NCHMF
22	Ha Giang	22.82	104.97	Vietnam	113	NCHMF
23	Ha Noi	21.02	105.80	Vietnam	6	NCHMF
24	Ham Yen	22.07	105.03	Vietnam	54	NCHMF
25	Hoa Binh	20.82	105.33	Vietnam	23	NCHMF
26	Hoang Su Phi	22.75	104.68	Vietnam		NCHMF
27	Jiangcheng	22.62	101.82	China	1121	GSOD
28	Lai Chau	22.05	103.15	Vietnam	244	NCHMF
29	Lang Son	21.83	106.77	Vietnam	258	GSOD
30	Lao Cai	22.50	103.97	Vietnam	112	NCHMF
31	Lincang	23.95	100.22	China	1503	GSOD
32	Longzhou	22.37	106.75	China	129	GSOD
33	Luy Cen	22.58	104.40	Vietnam	133	NCHMF

Table B.1. (Continued).

	Station name	Latitude	Longitude	Country	Elevation (m)	Source
34	Mai Chau	20.65	105.05	Vietnam	160	NCHMF
35	Mengzi	23.38	103.38	China	1302	GSOD
36	Moc Chau	20.83	104.68	Vietnam	958	NCHMF
37	Mu Cang Chai	21.85	104.08	Vietnam	975	NCHMF
38	Muong Cha	21.97	102.87	Vietnam	487	NCHMF
39	Muong Nhe	22.18	102.45	Vietnam	500	NCHMF
40	Muong Te	22.37	102.83	Vietnam	310	NCHMF
41	Nam Dinh	20.43	106.15	Vietnam	3	NCHMF
42	Nam Giang	22.26	103.17	Vietnam		NCHMF
43	Nam Muc	21.88	103.30	Vietnam	494	NCHMF
44	Napo	23.30	105.95	China	794	GSOD
45	Nguyen Binh	21.84	104.65	Vietnam		NCHMF
46	Nho Quan	20.32	105.75	Vietnam	12	NCHMF
47	Ninh Binh	20.25	105.98	Vietnam	2	NCHMF
48	Phu Ho	21.45	105.23	Vietnam		NCHMF
49	Phu Lien	20.80	106.63	Vietnam	119	GSOD
50	Phu Ly	20.52	105.92	Vietnam	3	NCHMF
51	Phu Yen	21.27	104.63	Vietnam	182	NCHMF
52	Quynh Nhai	21.85	103.57	Vietnam	802	NCHMF
53	Sam Neua	20.42	104.07	Lao PDR	1000	GSOD
54	Sapa	22.35	103.82	Vietnam	1570	NCHMF
55	Simao	22.77	100.98	China	1303	GSOD
56	Sin Ho	22.37	103.23	Vietnam	1529	NCHMF
57	Son La	21.33	103.90	Vietnam	676	NCHMF
58	Son Tay	21.13	105.50	Vietnam	15	NCHMF
59	Tam Dao	20.90	104.45	Vietnam		NCHMF
60	Tam Duong	22.42	103.48	Vietnam	900	NCHMF
61	Thai Nguyen	21.60	105.83	Vietnam	32	GSOD
62	Than Uyen	21.95	103.88	Vietnam		NCHMF
63	Thanh Hoa	19.80	105.78	Vietnam	7	GSOD
64	Thanh Son	21.19	105.16	Vietnam	50	NCHMF
65	Thuan Chau	21.43	103.68	Vietnam	652	NCHMF
66	Tuan Giao	21.58	103.42	Vietnam	570	NCHMF
67	Tuyen Quang	21.82	105.22	Vietnam	81	NCHMF

Table B.1. (Continued).

	Station name	Latitude	Longitude	Country	Elevation (m)	Source
67	Tuyen Quang	21.82	105.22	Vietnam	81	NCHMF
68	Van Chan	22.05	104.15	Vietnam	257	NCHMF
69	Viengsay	20.42	104.23	Lao PDR	913	GSOD
70	Viet Tri	21.27	105.42	Vietnam	17	NCHMF
71	Vinh Yen	22.27	104.88	Vietnam		NCHMF
72	Wujiaba	25.02	102.68	China	1892	GSOD
73	Yen Bai	21.70	104.87	Vietnam		NCHMF
74	Yen Chau	21.07	104.27	Vietnam	59	NCHMF
75	Yuanjiang	23.60	101.98	China	398	GSOD
76	Yuanmou	25.73	101.87	China	1120	GSOD

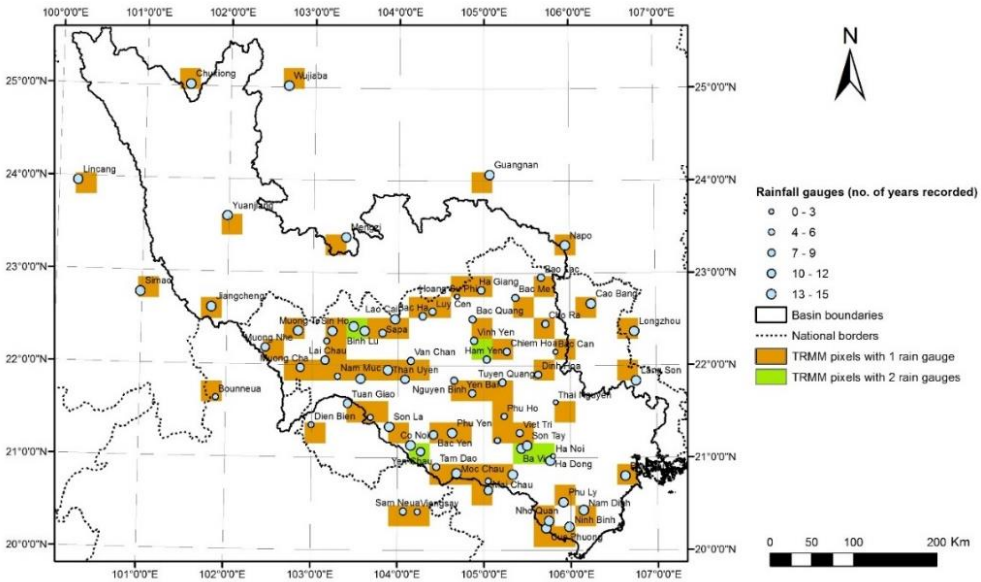

Figure B.1. TRMM pixels with one (red) or two (green) rainfall gauges

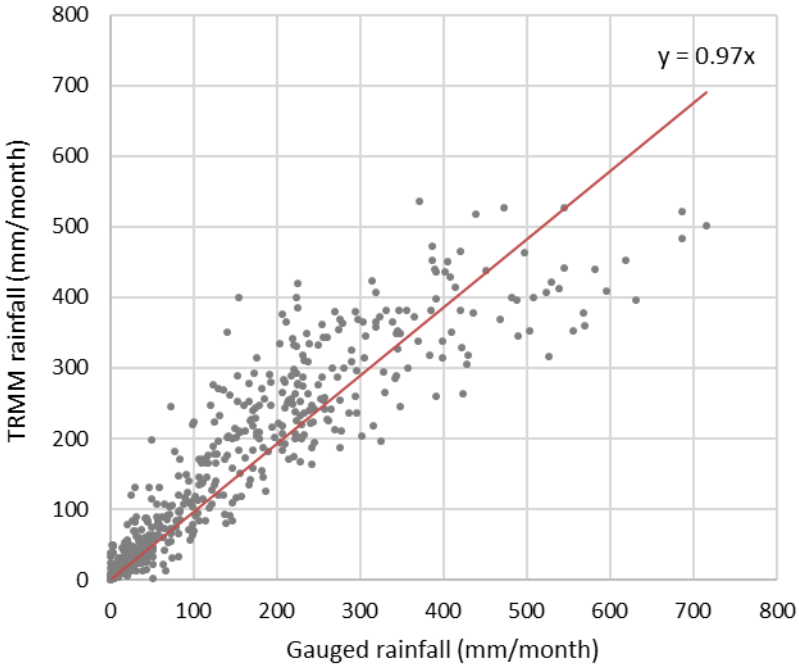

Figure B.2. Comparison of TRMM data with measured monthly rainfall averaged per pixel for gauges in pixels with multiple stations. The red line gives the linear regression best fit with 0 intercept.

Appendix C Maps of annual ET_{act} in the Red River Basin (2003-2012)

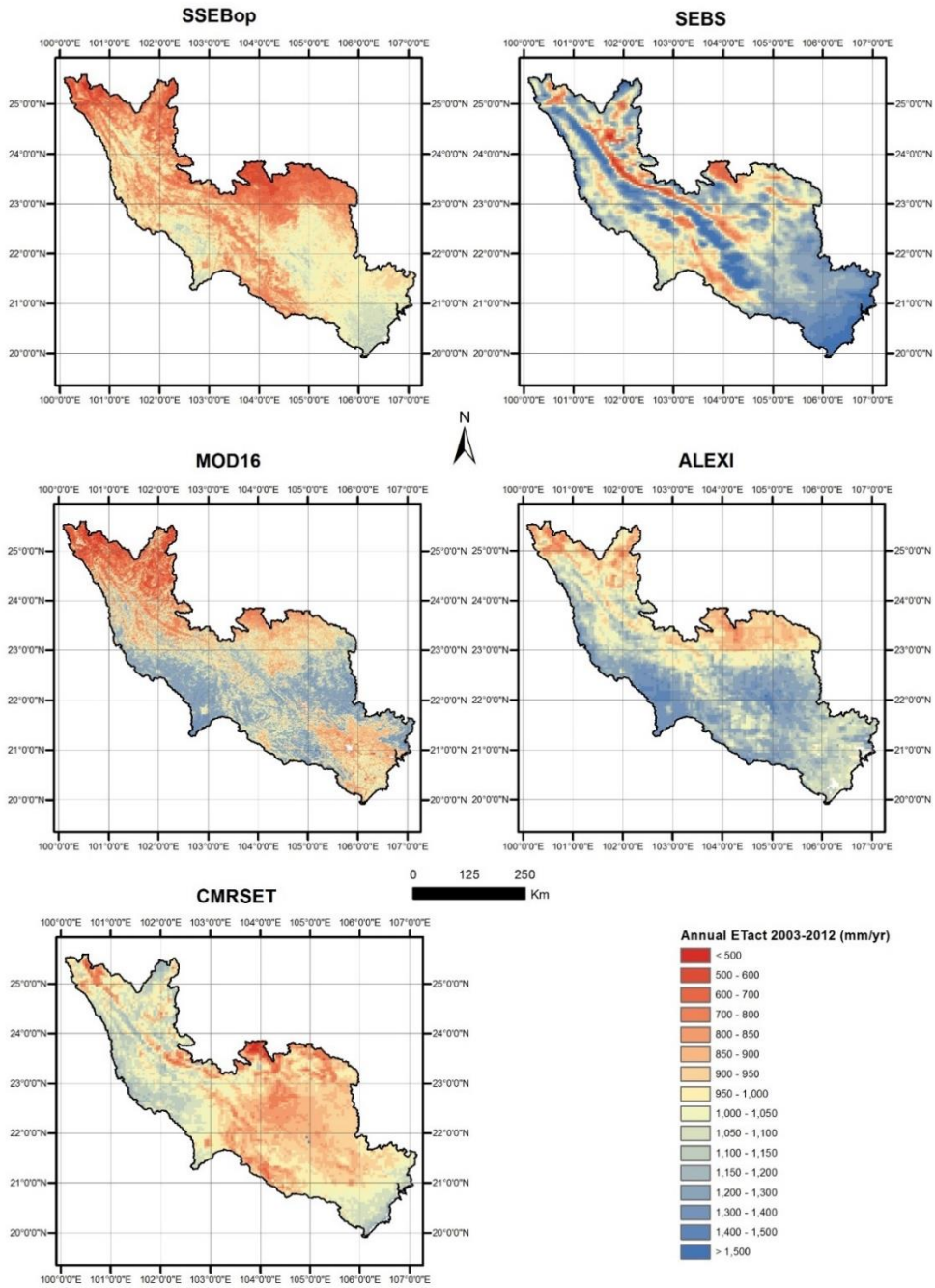

Figure C.1. Overview of annual ET_{act} in the Red River Basin according to five GSDPs.

Appendix D Coefficient of Variation of annual average ET_{act}

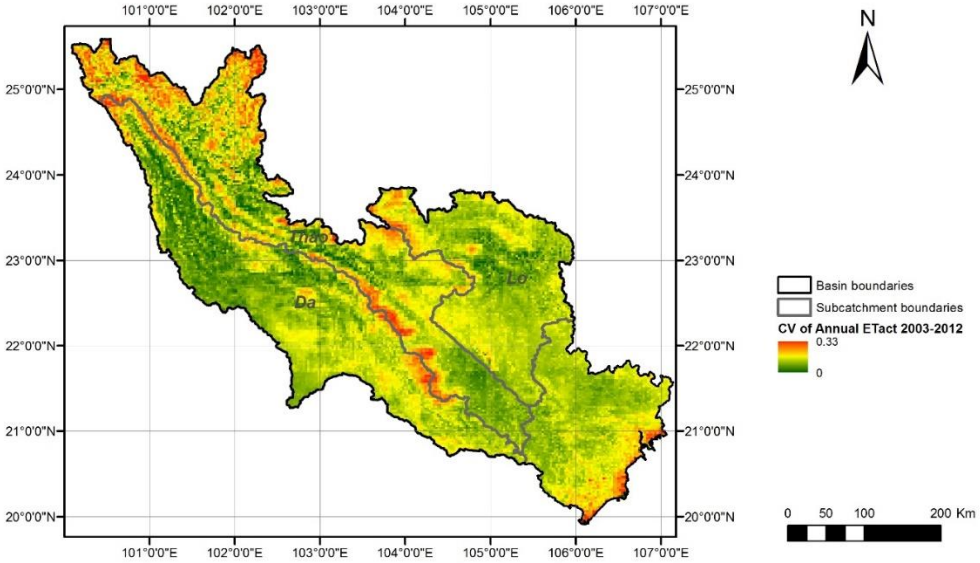

Figure D.1. Coefficient of variation (CV) of annual average ET_{act} based on five different products.

Appendix E Streamflow data

Table E.1. Metadata of streamflow stations in the Red River Basin.

	Station name	Latitude	Longitude	Country	Source
1	Muong Te	22.47	102.62	Vietnam	NCHMF
2	Hoa Binh	20.81	105.32	Vietnam	NCHMF
3	Lao Cai	22.50	103.95	Vietnam	NCHMF
4	Yen Bai	21.70	104.88	Vietnam	NCHMF
5	Bac Me	22.73	105.37	Vietnam	NCHMF
6	Vu Quang	21.57	105.25	Vietnam	NCHMF
7	Son Tay	21.15	105.50	Vietnam	NCHMF

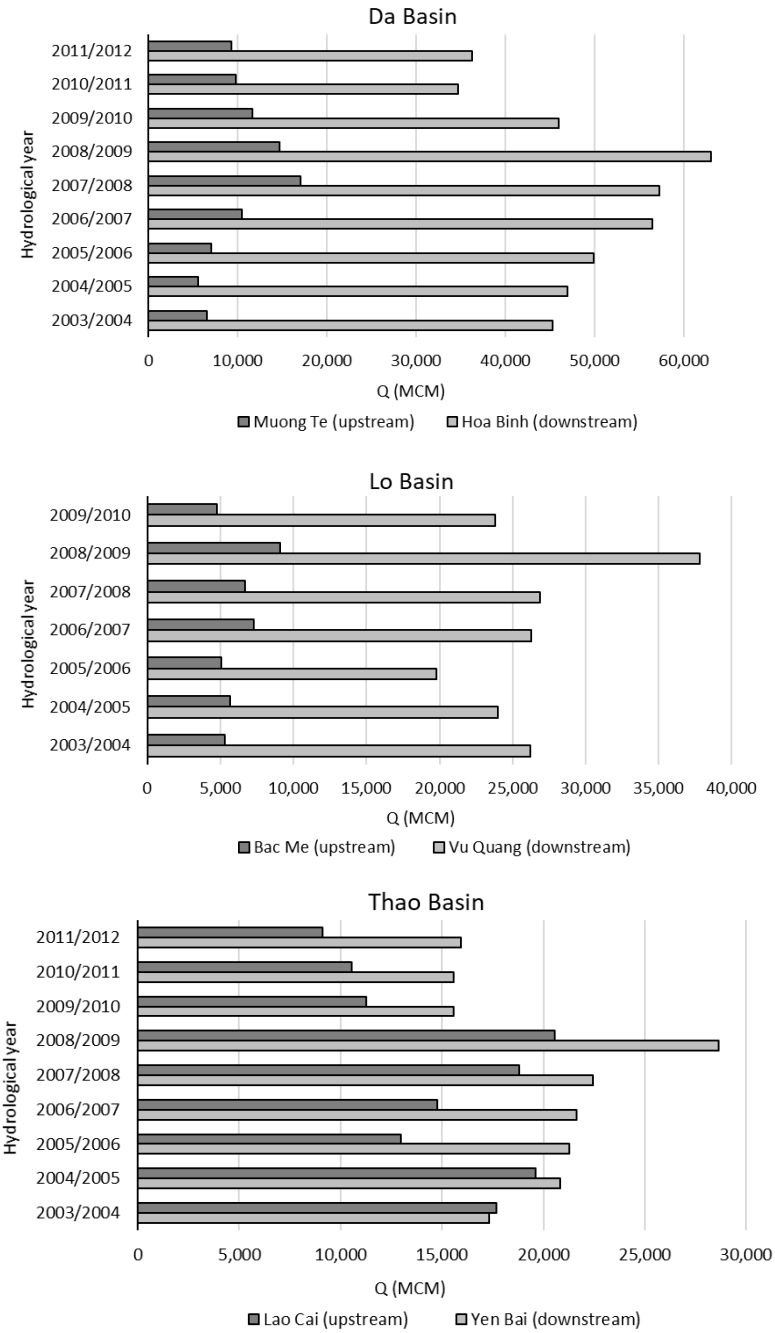

Figure E.1. Runoff in million cubic meters (hm³) generated per hydrological year in each of the subbasins, according to streamflow (*Q*) records.

Appendix F WEAP-VT scenario results per demand site

Table F.1. Demand, supply, return flow, and coverage at the demand site level under the *CF_90* and *CF_90_pol* scenarios, in which the Consumed Fraction of all agricultural and urban demand sites is set to 0.9. D_{gross} = gross demand (hm³/yr), D_{net} = net demand (hm³/yr), Q_w = total withdrawal (hm³/yr), Q_{nc} = non-consumed flow (hm³/yr),, C = Coverage (%).

Demand site	*CF_90*					*CF_90_pol*				
	D_{gross}	D_{net}	Q_w	Q_{nc}	C	D_{gross}	D_{net}	Q_w	Q_{nc}	C
SUDA01	15.0	6.5	8.4	0.8	56%	7.2	6.5	6.5	0.6	89%
SUDA02	19.3	11.6	14.8	1.5	77%	12.9	11.6	12.0	1.2	93%
SUDA03	85.4	66.8	62.7	6.3	73%	74.2	66.8	57.9	5.8	78%
SUDA04	43.7	36.2	13.5	1.3	31%	40.3	36.2	13.5	1.4	34%
SUDA05	84.3	58.1	58.7	5.9	70%	64.6	58.1	53.7	5.4	83%
SUDA06	185.6	140.7	172.9	17.3	93%	156.4	140.7	156.0	15.6	100%
SUDA07	105.1	58.8	98.5	9.8	94%	65.4	58.8	65.2	6.5	100%
SUDA08	250.3	169.7	233.9	23.4	93%	188.5	169.7	188.1	18.8	100%
SUDA09	71.3	61.9	67.7	6.8	95%	68.8	61.9	68.7	6.9	100%
SUDA10	94.1	79.4	88.4	8.8	94%	88.2	79.4	85.2	8.5	97%
SUDA11	101.0	83.3	54.7	5.5	54%	92.6	83.3	53.8	5.4	58%
SUDA12	42.6	32.4	23.6	2.4	55%	36.0	32.4	22.8	2.3	63%
SUDA13	83.2	75.0	36.8	3.7	44%	83.3	75.0	36.8	3.7	44%
SUDA14	223.1	200.7	174.2	17.4	78%	223.0	200.7	175.2	17.5	79%
SUDA15	32.2	24.0	27.0	2.7	84%	26.7	24.0	25.4	2.5	95%
Total	**1,436**	**1,105**	**1,136**	**113.6**	**79%**	**1,228**	**1,105**	**1,021**	**102.1**	**83%**
UDU01	16.1	10.0	12.4	1.2	77%	11.1	10.0	10.3	1.0	93%
UDU02	12.8	11.5	12.1	1.2	94%	12.8	11.5	12.8	1.3	100%
UDU03	44.5	38.3	42.9	4.3	96%	42.5	38.3	42.4	4.2	100%
UDU04	39.0	25.3	31.6	3.2	81%	28.1	25.3	26.4	2.6	94%
UDU05	61.6	43.1	48.1	4.8	78%	47.9	43.1	44.7	4.5	93%
UDU06	14.2	7.1	11.0	1.1	77%	7.9	7.1	7.4	0.7	93%
UDU07	3.6	3.1	3.0	0.3	85%	3.4	3.1	3.2	0.3	95%
UDU08	3.5	2.5	3.5	0.3	100%	2.8	2.5	2.8	0.3	100%
UDU09	5.3	3.7	5.2	0.5	97%	4.1	3.7	4.0	0.4	99%
UDU10	2.9	2.0	2.7	0.3	95%	2.3	2.0	2.2	0.2	100%
UDU12	1.8	0.9	1.1	0.1	61%	1.0	0.9	0.7	0.1	65%

Table F.1. (Continued).

Demand site	*CF_90*					*CF_90_pol*				
	D_{gross}	D_{net}	Q_w	Q_{nc}	C	D_{gross}	D_{net}	Q_w	Q_{nc}	C
UDU13	0.7	0.5	0.7	0.1	100%	0.5	0.5	0.5	0.1	100%
UDU14	0.8	0.6	0.8	0.1	95%	0.6	0.6	0.6	0.1	100%
Total	**206.8**	**148.6**	**175.0**	**17.5**	**85%**	**165.1**	**148.6**	**158.1**	**15.8**	**96%**
UDE01	4.3	4.3	3.2	-	73%	4.3	4.3	3.3	-	77%
UDE02	1.3	1.3	1.2	-	95%	1.3	1.3	1.3	-	100%
UDE03	10.7	10.7	10.7	-	100%	10.7	10.7	10.7	-	100%
UDE04	1.2	1.2	1.2	-	100%	1.2	1.2	1.2	-	100%
UDE05	1.2	1.2	1.1	-	93%	1.2	1.2	1.2	-	100%
UDE06	5.5	5.5	5.1	-	93%	5.5	5.5	5.5	-	100%
UDE07	0.1	0.1	0.1	-	95%	0.1	0.1	0.1	-	100%
UDE08	17.9	17.9	16.7	-	94%	17.9	17.9	17.8	-	100%
UDE09	1.5	1.5	1.5	-	100%	1.5	1.5	1.5	-	100%
Total	**43.7**	**43.7**	**40.9**	**-**	**93%**	**43.7**	**43.7**	**42.6**	**-**	**97%**

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Key Qualifications

Gijs Simons is a hydrologist and remote sensing expert with over ten years of experience in managing and executing international projects in the field of water resources management. Countries of work experience include Cambodia, Egypt, Ethiopia, Lao PDR, Madagascar, Mali, Morocco, Mozambique, Myanmar, Nepal, The Netherlands, Philippines, Spain, Sudan, Uganda, and Vietnam. Gijs is skilled in hydrological modelling, satellite image interpretation and GIS analysis using a variety of tools on the full range of spatial scales from a single agricultural field to an entire river basin. In addition, Gijs has developed and conducted multiple training courses and workshops on remote sensing, GIS and hydrological modelling.

Gijs is currently the managing director of the FutureWater office in Wageningen.

Educational Background

2013 – 2021	PhD: Tracking water reuse in river basins with multiple users, Delft University of Technology, Delft, The Netherlands
2007 – 2009	MSc Hydrology, Utrecht University, Utrecht, The Netherlands
2004 – 2007	BSc Earth Sciences with a specialization in Physical Geography, Utrecht University, Utrecht, The Netherlands

Professional Experience

2014 – present	Hydrologist and Managing Director, FutureWater, Wageningen, The Netherlands
2010 – 2014	Hydrologist and Remote Sensing Specialist, WaterWatch BV / eLEAF Competence Center, Wageningen, The Netherlands
2010	Hydrogeologist, IF Technology, Arnhem, The Netherlands

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First author:

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