The role of storage capacity in coping with intra- and inter-annual water variability in large river basins

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The role of storage capacity in coping with intra- and inter-annual water variability in large river basins

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Abstract

Societies and economies are challenged by variable water supplies. Water storage infrastructure, on a range of scales, can help to mitigate hydrological variability. This study uses a water balance model to investigate how storage capacity can improve water security in the world's 403 most important river basins, by substituting water from wet months to dry months. We construct a new water balance model for 676 'basin-country units' (BCUs), which simulates runoff, water use (from surface and groundwater), evaporation and trans-boundary discharges. When hydrological variability and net withdrawals are taken into account, along with existing storage capacity, we find risks of water shortages in the Indian subcontinent, Northern China, Spain, the West of the US, Australia and several basins in Africa. Dividing basins into BCUs enabled assessment of upstream dependency in transboundary rivers. Including Environmental Water Requirements into the model, we find that in many basins in India, Northern China, South Africa, the US West Coast, the East of Brazil, Spain and in the Murray basin in Australia human water demand leads to over-abstraction of water resources important to the ecosystem. Then, a Sequent Peak Analysis is conducted to estimate how much storage would be needed to satisfy human water demand whilst not jeopardizing environmental flows. The results are consistent with the water balance model in that basins in India, Northern China, Western Australia, Spain, the US West Coast and several basins in Africa would need more storage to mitigate water supply variability and to meet water demand.

1. Introduction

As recent research has shown, hydrologic variability can be as, or more, important than average water availability as a threat to water security (Grey *et al* 2013, Hall *et al* 2014). More extreme weather conditions (Dai *et al* 2004) are likely to increase the risks of floods and droughts (Hirabayashi and Kanae 2009, Jongman *et al* 2012) with negative impacts on people and on the environment. According to the World Bank, drought is the largest cause of death due to natural catastrophes (Dilley 2005).

Water infrastructure, on a range of scales, plays a major role in coping with water supply variability and enhancing water security. Infrastructure is needed to store, access, move and regulate water (Grey and Sadoff 2007) and can consist of small-scale dams, weirs, irrigation systems, rainwater harvesting cisterns, large multi-purpose dams or inter-basin transfer schemes. Investments in water infrastructure as well as in institutions are needed to achieve water security which forms the basis for economic growth, poverty reduction and sustainable development (Grey and Sadoff 2007). Inadequate or deteriorating infrastructure, on the other hand, increases vulnerability to water scarcity (De Fraiture *et al* 2010, Garrick and Hall 2014, Hall *et al* 2014) especially under a changing climate. Nonetheless, storage infrastructure must be interpreted in its broadest sense, from local farm reservoirs, through to groundwater recharge and larger reservoirs, and is by no means a panacea. Inappropriate infrastructure investment can have a devastating effect on communities and ecosystems. Our aim herein, therefore, is to propose methods for appraising

sustainable amounts of storage provision in aggregate. We emphasize the importance of local appraisal, impact assessment and consultation in the actual implementation of storage schemes.

There are numerous papers defining and measuring water scarcity with a variety of different methods. We adopt the definition of water scarcity by Loon van and Lanen van (2013) who define it as a human effect on the hydrological system when water is over-exploited because water demand is higher than water availability. A review of different water scarcity definitions can be found in Pedro-Monzonís *et al* (2015). Comprehensive reviews of methods measuring water scarcity can be found, for instance, in Oki and Kanae (2006), Rijsberman (2006) and in Brown and Matlock (2011). Recent studies of water scarcity on a global scale include Alcamo *et al* (2003), Vorosmarty (2000), Wada *et al* (2011) and Hoekstra *et al* (2012) who use the ratio of water withdrawal, use or demand to availability as a metric of water scarcity. Most of these studies focus on the average ratio, which neglects the very significant effects of variability on water security (Hall *et al* 2014). By using monthly data rather than annual totals, Wada and Hoeckstra measured water scarcity resulting from intra-annual water variability. Yet, they use a 10 year average for each month and thereby cannot take into account inter-annual variability. Furthermore, they do not include storage. As Biemans *et al* (2011) emphasizes, water supply and water stress can be evaluated only when human changes to the hydrological cycle such as dams and reservoirs are also taken into account. Brown and Lall (2006) constructed a storage index which highlights countries in need of more storage infrastructure in order to buffer rainfall variability. However, the analysis on a country scale cannot account for hydro-climatic differences within a country which is especially important in large countries with different rainfall pattern across the regions such as China or the US. Moreover, their analysis included 'virtual water' in the water balance. Whilst that is an interesting idea, we believe that it detracted from the insights gained by studying physical water use from surface and groundwater sources.

Our study represents an improvement over previous work as it is conducted on a basin-country unit (BCUs) scale which consists of large global river basins (Global Runoff Data Centre 2007) intersected with country borders. One country may contain one or more BCUs. River basins that intersect more than one country are sub-divided into BCUs. In total we analyse 676 BCUs which cover the world's 403 major river basins. This scale was chosen as many water management decisions are taken on a basin level as well as at a country level. However, country borders are included in order to incorporate possible conflicts arising over water allocation within transboundary river basins.

Using a global water balance model, this paper examines the geography of water scarcity. Looking at past and present, we seek to establish how existing storage capacity helps to buffer intra- and inter-annual water variability and thereby mitigates resulting water scarcity. First, overall water scarcity is identified in each BCU considering existing storage and monthly total water use. Variability in surface water availability is analysed by considering monthly runoff totals from reconstructions for years 1979 to 2012, averaged over the BCU. Evaporation losses and potential groundwater withdrawals are included in the water balance. Water scarcity, or shortage, is defined as a situation when the aggregated current storage in a BCU is less than 20% of capacity. This definition takes account of dead storage in reservoirs and reflects the conditions under which water restrictions are typically applied. We go on to refine the analysis and examine dependency on storage in each BCU as well as upstream dependency in transboundary river basins. Combining upstream dependency with overall water scarcity, areas of potential conflicts over water allocation are detected. Third, impacts on the ecosystem are included in the analysis. Globally, the area of irrigated land is growing and the overall water extraction for human use is increasing. This poses a threat to the environment as these human interventions alter both variability and volume of river flows needed to maintain freshwater ecosystems such as fish and riparian vegetation (Grafton *et al* 2011). Aquatic ecosystems are adapted to hydrological variability, including droughts. However, when flow variability departs excessively from natural patterns, there is potential for major ecosystem disturbance. In the definition of environmental water requirements (EWR) it is therefore important to consider natural flow variability and the way in which this may be modified by human intervention (Pastor *et al* 2013). In this study we calculate monthly EWRs using the variable monthly flow (VMF) method (Pastor *et al* 2013) and compare them with actual water available after human demand is satisfied. Then, we conduct a sequent peak analysis (SPA) (Lele 1987, Adeloye and Montaseri 1998, McMahon *et al* 2007a) as an alternative methodology to measure water scarcity. In a SPA, hypothetical storage capacity required to meet water demand is estimated. Comparing required storage with existing capacity, the need for further infrastructure investments to cope with water variability is identified on a global scale.

The objective of this study is to inform decision makers about where water supply variability is causing water shortages in large river basins, given current storage infrastructure, and where it causes harm to the environment. The results show where policies and infrastructure investments are needed to sustain and improve global water security.

2. Methodology

2.1. Water balance model

The water balance model is based on the assumption that each BCU with sub-basins and tributaries can be approximated as one single reservoir with surface runoff and the possibility to withdraw groundwater for water supply. Whilst this is obviously an approximation, Young and Puentes ([1969](#)) found that in a multi-reservoir analysis, unregulated runoff, water demand, and storage can be aggregated over the study site. It is an assumption that has subsequently been adopted elsewhere (e.g., Coe [2000](#)). Average monthly surface runoff per BCU was derived from simulations using the global hydrological model MacPDM (Arnell [1999](#)), run at a 1° resolution. MacPDM is a macro-scale water balance model simulating streamflow from meteorological input data and catchment characteristics. MacPDM performs well in reproducing observed runoff, compared with other global hydrological models as shown in the Water Model Intercomparison Project (Haddeland *et al* [2011](#)). In the present study, MacPDM was driven with daily ERA-Interim reanalysis meteorological data (ERA Interim 2014) for the period 1979–2012. Groundwater withdrawal capacity, total water demand and evaporation losses were taken from the IMPACT model (Rosegrant *et al* [2002](#)) provided by the International Food Research Institute (IFPRI). Storage capacity was taken from the GRanD database (Lehner *et al* [2011](#)) and modified by IFPRI who excluded natural lakes as the storage lakes provide cannot usually be controlled to enhance water availability for human purposes. Storage thus refers to any kind of surface reservoir whose water can be managed and used for human activities in the industrial, domestic and agricultural sectors. Groundwater withdrawal capacity refers to the capacity to pump water. That is not necessarily the same as actual groundwater withdrawals, but in most regions of the world, groundwater abstractions are not consistently monitored, so withdrawal capacity is the best approximation of the contribution to the water balance from groundwater. The groundwater withdrawal that has been assumed may not be sustainable. Owing to a lack of data, the 2010 figures for groundwater withdrawal capacity were distributed equally over the 12 months of the year without accounting for possible variation within a year. Total water use includes consumptive water use from the industrial, domestic and agricultural sectors. Monthly values were used to describe intra-annual variation of water demand due to irrigation or hydropower demand.

Including country borders in our study on large global river basins enabled us to analyse the effect of transboundary flows on water security. This is especially relevant in BCUs such as the Nile in Egypt where most of the water supply comes from transboundary river flow (Conway [2005](#), Zeitoun and Mirumachi [2008](#)). Comprehensive information on transboundary runoff flows is rarely available on a global scale. On a country scale, the AQUASTAT database (FAO [2014](#)) contains data on water inflows and outflows. However, transboundary flows depend very much on how borders are drawn and trying to rescale national AQUASTAT data would not make sense in this case. Therefore, transboundary flows were calculated specifically for the country-basins used in this study. 305 out of the 676 country basins considered in this study are transboundary basins. Owing to a lack of global observed transboundary streamflow data, transboundary runoff was computed by routing from upstream to downstream BCUs in a basin, using flow accumulation data taken from the global hydrography HydroSHEDS (Lehner *et al* [2008](#)).

As a reservoir regulation rule we assumed that water is released to meet human water demand before recharging the reservoir. When the reservoir is filled completely, the residual runoff is spilled and flows out to one or several downstream BCUs or the sea. We assume that groundwater is used directly, so in the case that groundwater withdrawal capacity is higher than demand, demand is met entirely from groundwater. In BCU j at time t, the water balance model is therefore as follows:

$$
\begin{aligned}
&s_{j,t} = s_{j,t-1} + q_{j,t} + b_{j,t} - e_{j,t} - (u_{j,t} - g_j) \\
&b_{jk,t} = 0
\end{aligned}
\quad
\begin{cases}
0 \leqslant s_{j,t-1} + q_{j,t} + b_{j,t} - e_{j,t} - (u_{j,t} - g_j) \leqslant c_j \\
g_{j,t} \leqslant u_{j,t}
\end{cases}
\tag{1}
$$

$$
\begin{aligned}
&s_{j,t} = s_{j,t-1} + q_{j,t} + b_{j,t} - e_{j,t} \\
&b_{jk,t} = 0
\end{aligned}
\quad
\begin{cases}
0 \leqslant s_{j,t-1} + q_{j,t} + b_{j,t} - e_{j,t} - (u_{j,t} - g_j) \leqslant c_j \\
g_{j,t} > u_{j,t}
\end{cases}
\tag{2}
$$

$$
\begin{aligned}
&s_{j,t} = c_j \\
&b_{jk,t} = s_{j,t-1} + q_{j,t} + b_{j,t} - e_{j,t} - (u_{j,t} - g_j) - c_j
\end{aligned}
\quad
\begin{cases}
s_{j,t-1} + q_{j,t} + b_{j,t} - e_{j,t} - (u_{j,t} - g_j) > c_j \\
g_{j,t} \leqslant u_{j,t}
\end{cases}
\tag{3}
$$

$$
\begin{aligned}
&s_{j,t} = c_j \\
&b_{jk,t} = s_{j,t-1} + q_{j,t} + b_{j,t} - e_{j,t} - c_j
\end{aligned}
\quad
\begin{cases}
s_{j,t-1} + q_{j,t} + b_{j,t} - e_{j,t} > c_j \\
g_{j,t} > u_{j,t}
\end{cases}
\tag{4}
$$

with $s_{j,t}$ as storage in basin j in month t, $q_{j,t}$ as surface runoff, g_j as groundwater withdrawal (which is taken as the monthly average and is not time-varying), $u_{j,t}$ as total water use, $e_{j,t}$ as evaporation losses. c_j is storage capacity and $s_{j,0} = c_j$. The transboundary outflow from BCU j to BCU k downstream basins is written as

$$b_{jk,t} = \sum_{i=1}^{k} b_{ji,t} \qquad (5)$$

and the inflow to BCU j, $b_{j,t}$ is computed as the sum from n upstream basins:

$$b_{j,t} = \sum_{i=1}^{n} b_{ij,t}. \qquad (6)$$

Owing to a lack of data, institutional arrangements such as water treaties or specific reservoir management rules could not be included. The reservoir operation rule in this study maximizes storage and does not consider the possibility of multi-purpose use of reservoirs which would include drawing down reservoir levels in anticipation of floods.

Water shortage in BCU j is defined as the state in which storage is filled to 20% or less, i.e. $s_{j,t} = \leqslant 0.2c_j$, a situation which reflects the dead storage in reservoirs and which is typical of the conditions under which water restrictions are often applied. The number of months in which the storage water level is below the 20% threshold during the simulation period is counted and forms, as a percentage of all 408 months, the index of water scarcity.

2.2. Environmental water requirements
'EWR' are defined as 'quantity, quality and timing of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems' (The Brisbane Declaration 2007, p 1). Including EWRs in our water balance analysis ensures that ecosystem services such as biodiversity and recreation are maintained. There are several methods of EWR calculations (Tennant 1976, Tessmann 1980, Smakhtin *et al* 2004). This study uses the VMF method introduced by Pastor *et al* (2013) which is based on monthly naturalized flow data and thus can account for intra-annual variability. In contrast to Tessman's method, VMF allows some water withdrawal in low-flow seasons which helps the industry and the irrigation sector which need water especially in dry periods. On the other hand, it allocates at least 30% of the mean monthly flow (MMF) to the environment throughout the year. The VMF is calculated as follows. During the low-flow season (MMF < 40% of mean annual flow (MAF)) 60% of MMF is allocated to the environment; during the intermediate-flow season (MMF 40–80% of MAF) 45% of MMF and during high-flow season (MMF > 80% of MAF) only 30% are reserved for the environment. In extremely dry conditions (MMF < 1 m³ s⁻¹) there is no environmental flow allocation. EWRs $\text{ewr}_{j,t}$ are calculated using

naturalized flows. For each BCU, the EWRs are calculated as described above using average total surface runoff in the ith month $\overline{q_{j,i}}$ with $i = 1, 2,\ldots,$ 12. Then, the EWRs are compared to monthly outflows in each basin $b_{jk,t}$. The number of months in which $b_{jk,t} < \text{ewr}_{j,t}$ is counted and forms, as a percentage of all 408 months, the index of EWR violation.

2.3. Sequent peak analysis
The SPA is a method to estimate hypothetical storage capacity required to meet total water demand. Similar to the Rippl graphical mass curve (Rippl 1883) and the extended deficit analysis (Pegram 2000), SPA calculates the minimum required storage volume to either meet a specific target draft or to meet demand for a failure-free operation. Using the monthly runoff data described above, we calculate hypothetical storage capacity needed to satisfy human and environmental water needs using the following equation:

$$k_{j,t} = \min\left[0, k_{j,t-1} + u_{j,t} + \text{ewr}_{j,t} - q_{j,t} - b_{j,t} - g_j\right] \qquad (7)$$

with k_t as storage in t ($k_0 = 0$). Then, the required storage capacity is $c_j = \max(k_{j,t})$.

We use the simple SPA method in which net evaporation losses are ignored. Further discussion of the SPA can be found in Lele (1987), Adeloye and Montaseri (1998) and McMahon *et al* (2007a, 2007b).

3. Results

3.1. Water balance model
Applying equation (1) to the 676 BCUs considered in this study, global water scarcity was assessed using simulated surface runoff for the period 1979–2012. Figure 1 shows the results of equation (1) applied to all BCUs. The shortage scale ranges from 0 (no scarcity) to 1 (very water scarce) representing the percentages of months in which a BCU is classified as water scarce. Deserts, ice fields with no runoff and land areas with no large river basins are shaded in grey. Hotspots of water scarcity are the Indian subcontinent, Northern China, Spain, the West of the US, Australia and several basins in Africa. In India, the Ganges, the Indus, the Godavari, Krishna or the Penner River are extremely water scarce. In China, the Yellow River stands out as extremely water scarce. In Spain, the Guadalquivir, Guadiana and Tejo basins have difficulties to cope with water shortages. In the West of US, the San Joaquin River, Sacramento River and the Salinas basin show up, whereas in the South the Bravo, Colorado and Brazos rivers are identified as water scarce. In South America, most of the BCUs along the West Coast face water limitations: all basins along the coast of Ecuador, Peru and Northern and Central Chile. In Southern Africa, the Limpopo in South Africa, the Zambezi in
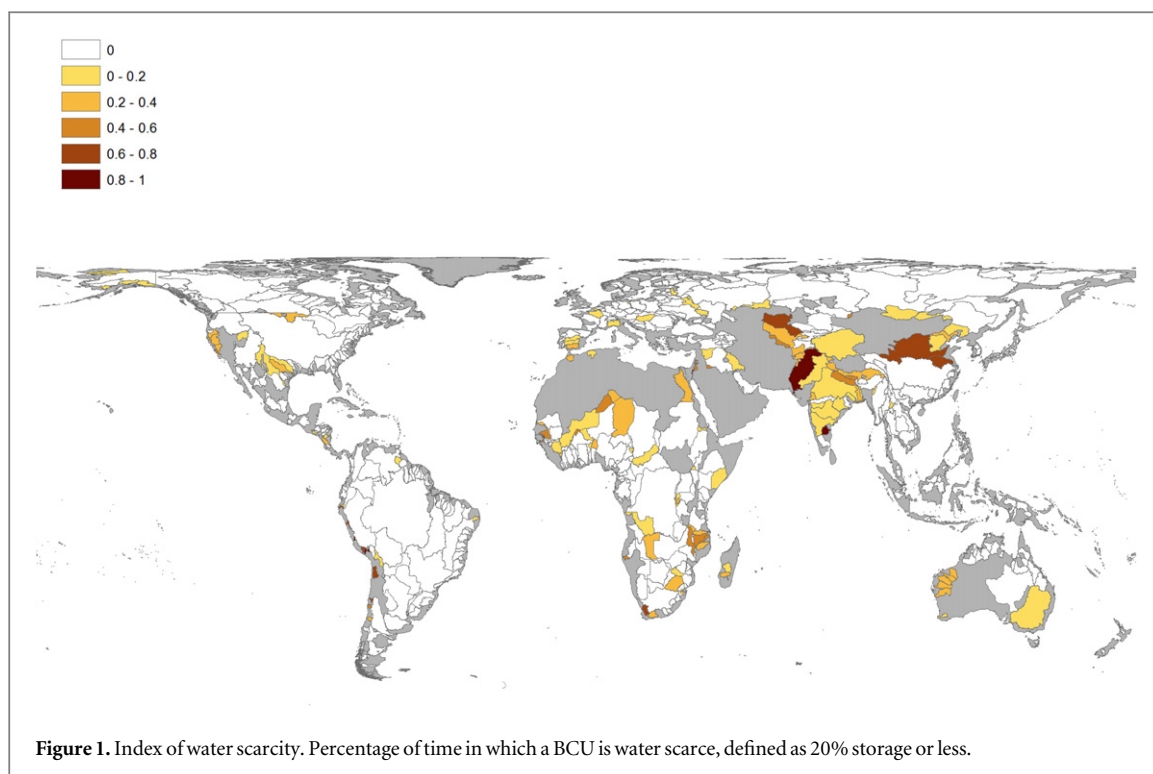
Figure 1. Index of water scarcity. Percentage of time in which a BCU is water scarce, defined as 20% storage or less.
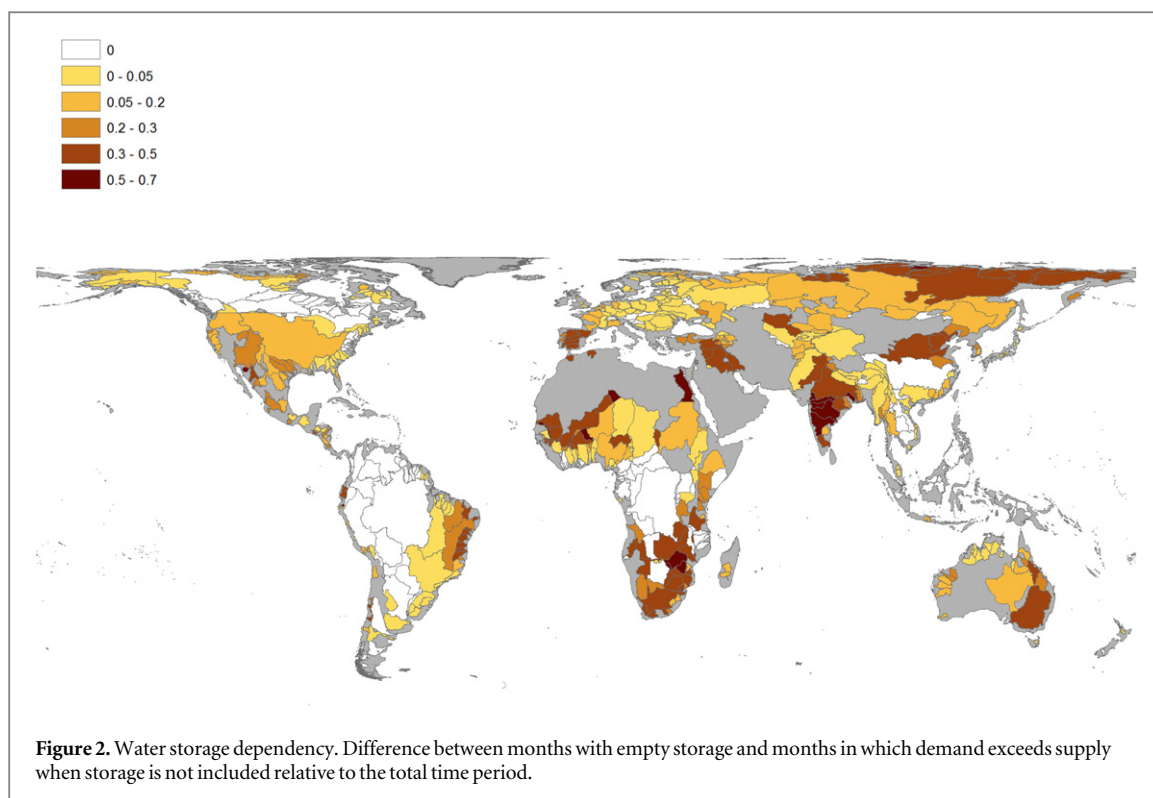
Table 1. Population at risk of water restrictions. Population at risk is the overall BCU population multiplied with the index of water scarcity. Shown here are global BCUs with population at risk higher than 1 million.

Drainage	Country	Population (millions)	Index of water scarcity	Population at risk (millions)
INDUS	Pakistan	162.97	0.93	151.56
HUANG HE (YELLOW RIVER)	China	172.94	0.70	121.06
HUAI HE	China	94.90	0.63	59.79
GANGES	Bangladesh	62.62	0.34	21.29
GANGES	India	483.5	0.05	24.18
GANGES	Nepal	29.66	0.44	13.05
PENNER RIVER	India	11.46	0.95	10.89
ARAL DRAINAGE	Uzbekistan	28.35	0.35	9.92
GODAVARI	India	71.08	0.13	9.24
ZAMBEZI	Malawi	12.40	0.47	5.83
DAMODAR RIVER	India	30.25	0.16	4.84
MAHI RIVER	India	13.31	0.32	4.26
LIAO HE	China	29.46	0.13	3.83
INDUS	Afghanistan	10.6	0.35	3.71
LUAN HE	China	12.26	0.23	2.82
LIMPOPO	South Africa	12.90	0.21	2.71
DALINGHE	China	4.3	0.40	1.72
YONGDING HE	China	84	0.02	1.68
NILE	Rwanda	7.65	0.20	1.53
NILE	Burundi	4.87	0.31	1.51
DEAD SEA	Jordan	2.5	0.58	1.45
ARAL DRAINAGE	Afghanistan	5.6	0.25	1.40
SEBOU	Morocco	6.33	0.21	1.33
CONGO	Burundi	3.85	0.33	1.27
SACRAMENTO RIVER	United States	3.23	0.35	1.13
KRISHNA	India	107	0.01	1.07

Angola, Malawi and Tanzania show water shortages. The Niger basin has difficulties to supply its population with water in nearly all basin countries: Algeria, Chad, Guinea, Mali and Benin. In Australia, the basins in the West were identified as water scarce due to

insufficient storage capacity whereas water scarcity in the Murray basins can be explained by high evaporation losses.

Estimating risk involves a combination of probability and consequences. In table 1 we show the

Figure 2. Water storage dependency. Difference between months with empty storage and months in which demand exceeds supply when storage is not included relative to the total time period.
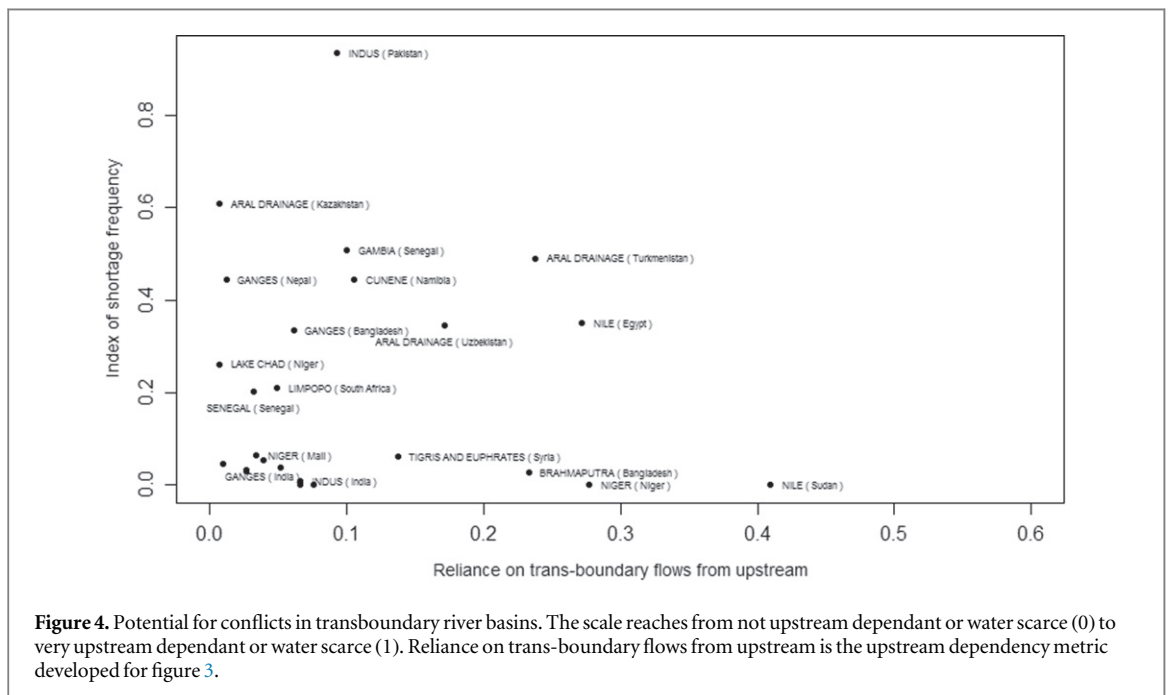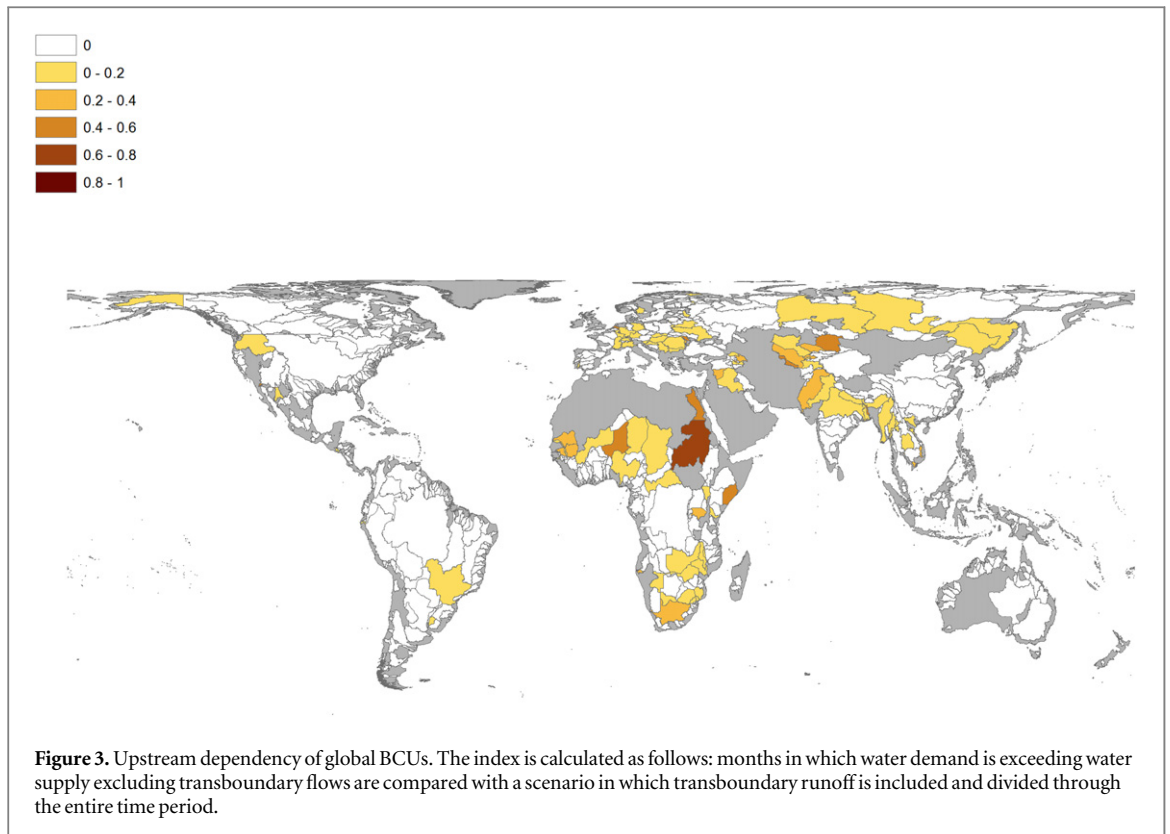
population at risk as an overall indicator of exposure multiplied by our shortage index. Population data for 2010 is taken from CIESIN (2015).

In figure 2 we quantify how dependent a BCU is on storage by plotting the difference between the shortage indices with and without storage i.e. with $c_j = 0$ in the latter case.

The results show that most basins except the basins in tropical regions show water storage dependency. Especially India, Northern China, Southern Africa, the entire US, the East of Brazil and the Murray basin in Australia are dependent on artificial storage capacity to provide reliable flow over the year. These results coincide with findings of Biemans *et al* (2011) who estimated the contributions of reservoirs to water supply for irrigation for the period 1981 to 2000. They found that the West Coast of the US as well as basins in China, India and central Asia rely heavily on reservoir storage.

Figure 3 shows upstream dependency independent of storage and reservoir regulation. Therefore, the frequency with which water demand exceeds water supply when transboundary flows are excluded is compared with the situation when transboundary runoff is included in the calculations. Egypt and Sudan in the Nile basin, Syria and Iraq in the Tigris and Euphrates basin, Kazakhstan, Uzbekistan and Turkmenistan in the Aral Drainage, Niger in the Niger basin, South Africa in the Orange basin and Pakistan in the Indus basin are the most upstream dependant BCUs before storage and other human interventions are taken into account.

Our upstream dependence metric quantified the *difference* between the shortage frequency with and without transboundary flows. The significance of upstream dependence, however, depends on how severe the water scarcity is in the BCU itself. Figure 4 shows important upstream dependent BCUs experiencing severe water scarcity despite the possibility to store water. Through the combination of these two metrics, potential trans-boundary conflicts over water in BCUs are shown, since the greatest potential for conflicts exists when upstream dependency is combined with water shortage within the BCU. The Nile in Sudan clearly stands out as highly dependent upon flows from upstream. However, it is in situations where a combination of upstream dependency and overall water scarcity occur, such as in the Indus in Pakistan or the Aral Drainage in Kazakhstan and Turkmenistan that trans-boundary conflicts over water resources might arise. These results, which are based on hydrological analysis, coincide with findings of other studies on conflict and cooperation in international river basins. A recent study by Bernauer (2014) analysed conflict and cooperation in global international river basins using International Rivers Conflict and Cooperation event data which include socio-economic factors such as GDP, population size, existence of democracy or upstream/downstream power. The following river basins from our study coincide with Bernauer's (2014) results: Indus and the Tigris and Euphrates as basins at risk and Indus, Niger, Nile and Senegal which were defined as cooperative international basins. A similar study by Yoffe *et al* (2003) highlighted 29 basins at risk from which 24

Figure 3. Upstream dependency of global BCUs. The index is calculated as follows: months in which water demand is exceeding water supply excluding transboundary flows are compared with a scenario in which transboundary runoff is included and divided through the entire time period.

Figure 4. Potential for conflicts in transboundary river basins. The scale reaches from not upstream dependant or water scarce (0) to very upstream dependant or water scarce (1). Reliance on trans-boundary flows from upstream is the upstream dependency metric developed for figure 3.
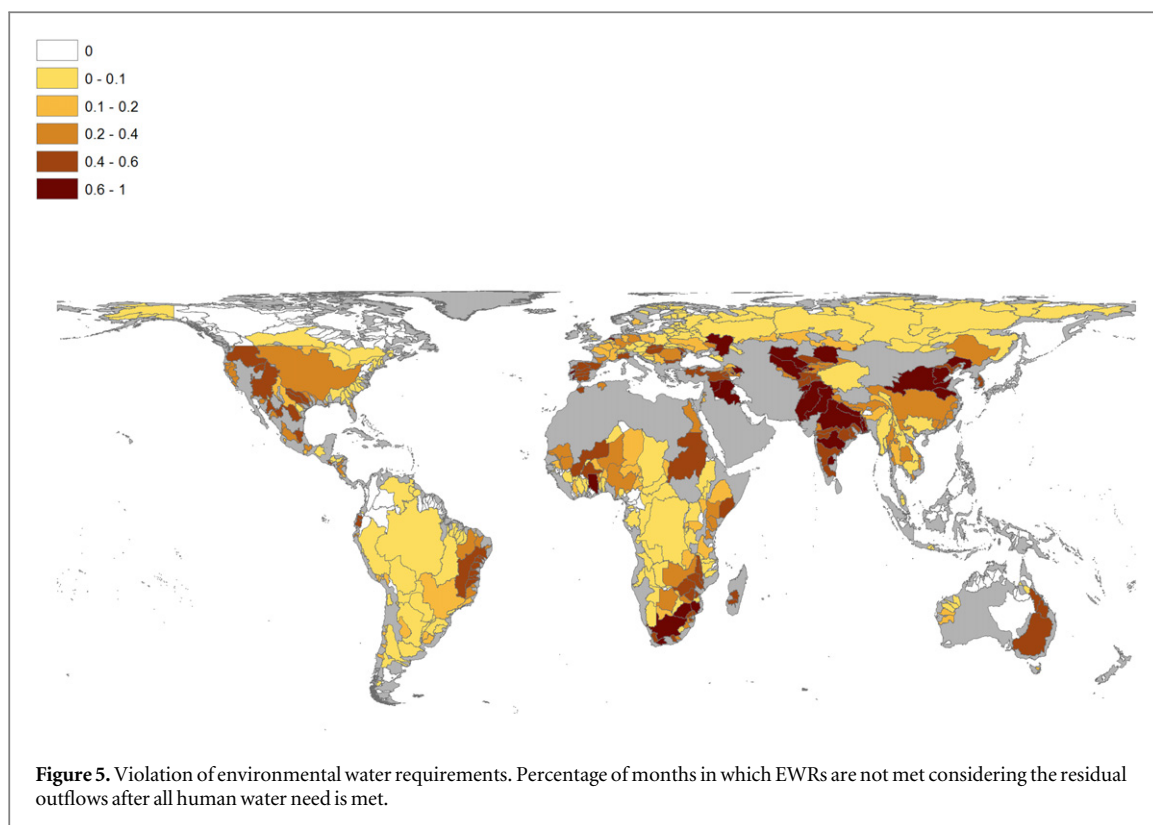
coincide with our river data set. 17 out of 24 basins coincided with our upstream dependent basins and 14 out of 24 are basins which resulted to be upstream dependent as well as water insecure in our study. Among those are the Aral Drainage, Ganges, Indus, Lake Chad, Limpopo, Nile, Senegal and the Tigris and Euphrates drainage. However, one has to add that the water balance model used in the present analysis does not include specific dam operation rules or political aspects such as water treaties. Results have to be interpreted accordingly.

3.2. Environmental water requirements
EWRs are calculated using Pastor *et al*'s (2013) VMF method for each month and each BCU using naturalized surface runoff. Then, the residual outflow of each BCU is compared with EWRs and the percentage of months in which the EWRs cannot be met is

Figure 5. Violation of environmental water requirements. Percentage of months in which EWRs are not met considering the residual outflows after all human water need is met.

derived. Figure 5 shows hotspots of EWR violation where water requirements for the ecosystem are chronically unmet: India, Northern China, South Africa, the US, especially the West Coast, the East of Brazil, Spain and the Murray basin in Australia. This coincides with the findings of numerous papers about the trade-off between water for agriculture and water for the environment in basins around the globe such as the Indian river basins (Smakhtin 2006), in the Yellow river basin in China (Sun *et al* 2008, Cui *et al* 2009) and the Murray Basin (e.g., Quiggin 2001, Goss 2003, Qureshi *et al* 2007, Garrick *et al* 2009, Grafton *et al* 2011). Specifically in water-scarce BCUs there is a high prevalence of EWR violations which is shown in our study through a significant positive correlation between our index of water restrictions and the frequency of EWR violations (Kendall's $\tau = 0.27$ with $p < 0.001$). Even more striking is the correlation between EWR violations and storage dependency ($\tau = 0.4$ with $p < 0.001$) which indicates that BCUs whose water security depends on storage are more likely to not meet EWRs needed for the ecosystem. This emphasizes that dam operation rules and flow dam releases play a major role in providing the necessary outflow variability to support the downstream ecosystem.

Table 2 lists all BCUs with a water restriction index higher than 0.5 and other indices developed in this paper.

Figure 6 shows the comparison of existing storage $s_{j,0}$ with storage capacity required to meet current water demand, both for human and environmental need, derived using a SPA. We size the storage such

that the EWR is not violated. Storage deficit is calculated in the following way:

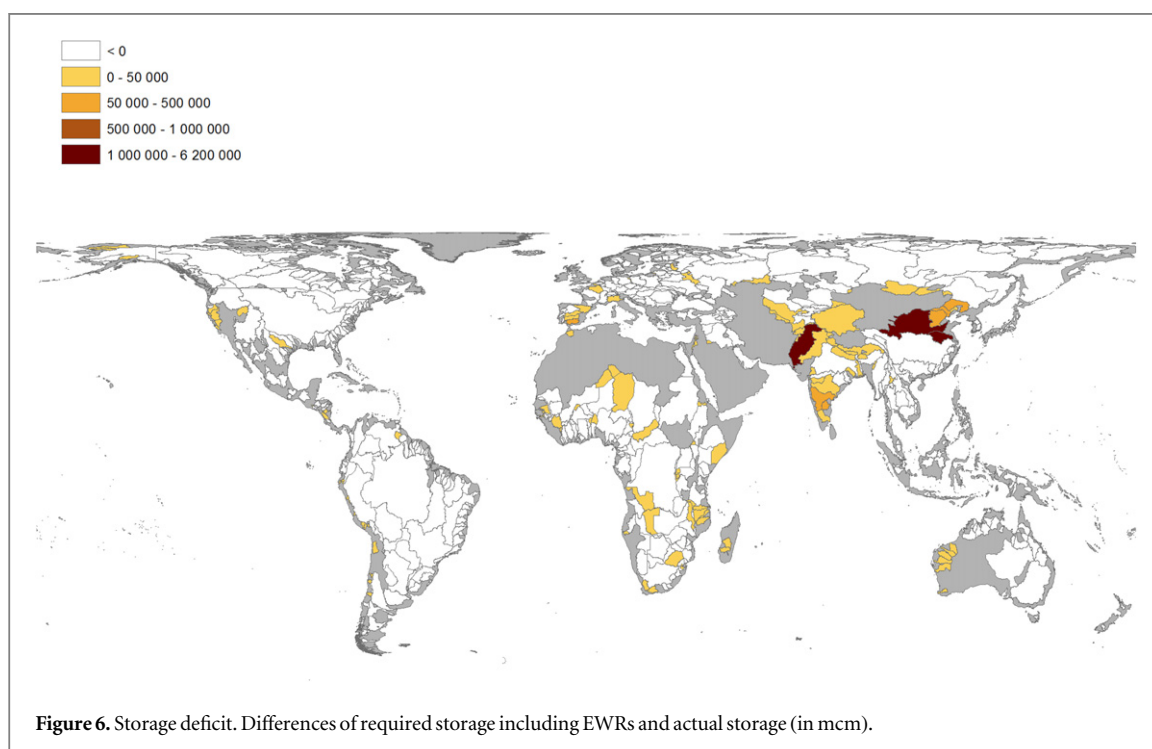$$\text{Storage \quad deficit}_j = \max\left(k_{j,t}\right) - s_{j,0}. \qquad (8)$$

According to our analysis, basins which require more storage are the hotspots in India, Northern China, Western Australia, Spain, the US West Coast and several basins in Africa. Comparing the results of the SPA and the water balance method one can see that both methodologies lead to similar results. As an overall conclusion one can say that so far, most basins are still water secure.

4. Discussion

21.2% of all BCUs show water shortages in at least one month. Although our method is very different and takes into account variability and storage, the overall percentage is not far off the analysis by Alcamo *et al* (2003) that 24% of all global river basins show 'severe water stress' using the ratio of water withdrawal to availability greater than 0.4 as indicator for water stress. It also coincides with Hoekstra *et al*'s (2012) analysis of global monthly blue water scarcity. Hoeckstra identified the basins in the Indian subcontinent as well as in Northern China, along the US West Coast, in South Africa, Spain and Australia as most water scarce. Differences are visible in the Murray and Eyre Lake basin in Australia where Hoeckstra measured higher water scarcity than our study. This is due to the fact that our study accounts for the possibility to store water. Figure 2 showed that both basins are dependent on water storage and therefore can mitigate the

Table 2. Water scarce BCUs.

Drainage	Country	Index of water scarcity	Frequency of EWR violation	Water storage dependency index	Upstream dependency index
CANETE	Peru	1.00	0.00	0.01	0.00
DEAD SEA	Israel	0.95	0.41	0.11	0.00
PENNER RIVER	India	0.95	0.80	0.07	0.00
INDUS	Pakistan	0.93	1.00	0.02	0.09
MAJES	Peru	0.90	0.15	0.06	0.00
DEAD SEA	Egypt	0.90	0.01	0.00	0.00
CHIRA	Peru	0.79	0.24	0.12	0.06
TARIM	Pakistan	0.77	0.21	0.09	0.00
LOA	Chile	0.73	0.02	0.10	0.00
OCONA	Peru	0.72	0.12	0.10	0.00
HUASCO	Chile	0.72	0.05	0.21	0.00
LOA	Bolivia	0.72	0.00	0.11	0.00
HUANG HE (YELLOW RIVER)	China	0.70	0.69	0.46	0.00
HUAI HE	China	0.63	0.83	0.25	0.00
ARAL DRAINAGE	Kazakhstan	0.61	0.96	0.31	0.01
GEBA	Senegal	0.60	0.05	0.00	0.00
SANTA	Peru	0.60	0.06	0.11	0.00
DORING	South Africa	0.60	0.58	0.36	0.00
DEAD SEA	Jordan	0.58	0.18	0.00	0.00
GANGES	China	0.51	0.39	0.00	0.00
GAMBIA	Senegal	0.51	0.20	0.05	0.10
VOLTA	Mali	0.50	0.06	0.26	0.00

Figure 6. Storage deficit. Differences of required storage including EWRs and actual storage (in mcm).

impacts of water scarcity. The same applies to the Mississippi, Colorado and St. Lawrence basins in the US as well as to the Orange basin in South Africa. In the study by Brown and Lall (2006) several indices were developed indicating the volume of required storage to meet annual water demand based on the seasonal rainfall cycle as well as annual water total demand. The three countries with highest water deficits identified by Brown and Lall (2006) are India, Pakistan and China which coincides with our findings.

According to our study most people at risk of water restrictions live in China (191.5 million people), Pakistan (151.8 million) and India (55.1 million), followed by Bangladesh (22.1 million), Nepal (13.1 million) and Uzbekistan (9.7 million). Desert regions with no major rivers are excluded from the analysis.

This paper highlighted global river basins with water shortages for human use as well as the environment due to inter- and intra-annual water variability. Using a monthly time scale and BCUs rather than countries this study

represents an improvement over previous work. Furthermore, possible reasons for the current status of water scarcity, such as dependence on available storage capacity or on transboundary flows from upstream were identified.

The global nature of the analysis means that there are several significant limitations. Storage capacity includes human made storage and ignores natural storage such as lakes and wetlands. This may lead to overestimation of the water restriction index in BCUs with large natural lakes such as Lake Victoria in the Nile basin or Lake Chad in the Chad basin. Groundwater withdrawal data represent the technological ability to pump up water and does not show how much of it is actually used, nor the quality of water abstracted. The 2010 annual withdrawal capacity was equally distributed over the months and therefore does not account for variations within a year. Furthermore, the storage numbers used in this work do not differentiate between main purposes of reservoirs such as agriculture, flood protection or hydropower. Although the GRanD database (Lehner *et al* 2011) offers information on main use, most reservoirs are multi-purpose reservoirs and excluding storage from reservoirs with hydropower as main purpose did not lead to improved results.

5. Conclusion

Consideration of water scarcity without proper analysis of hydrological variability, artificial storage and groundwater withdrawals can give an inaccurate picture of where the global hotspots of water scarcity are located. In this paper we have implemented a water balance model at the scale of BCUs in order to generate a metric of water scarcity. This paper showed in which BCUs inter- and intra-annual water variability leads to water shortages and highlights where this situation could be improved through larger storage capacity. However, in basins with a very large storage deficit (shown in figure 6) greatly exceeding average annual runoff, it may not be possible to mitigate the problem even with additional storage, which could lead to negative environmental impacts and higher evaporative losses as pointed out in Brown and Lall (2006). Our results highlight BCUs in which additional storage capacity can help mitigating the impacts of water variability. However, this paper should not be seen as a call for more large dams. Storage can be increased through a plethora of options such as smaller dams, aquifer recharge or rainwater harvesting and our paper does not recommend one of these options specifically. Our water balance model helps to explore trade-offs associated with investments in storage, whilst acknowledge the limitations of over-reliance on storage. Furthermore, the model can show trade-offs between water for human use and water for the environment. While storage is able to buffer water supply variability, the alteration of natural flows decreases the volume of water flowing downstream and changes the frequency and timing of high and low flows which are important for downstream river ecosystems (Richter and

Thomas 2007). It has been shown that dams can be operated to preserve EWRs (e.g., Harman and Stewardson 2005, Richter and Thomas 2007, Olden and Naiman 2010) but often flow requirements are still violated as our study has shown. Additionally, more reliable water supply due to increased storage can lead to increased water demand through a rebound effect which then might have further negative impact on EWRs.

The analysis is based upon synthetic runoff data derived from a reanalysis dataset. Use of reanalysis data and simulated runoff has known limitations (Haddeland *et al* 2011, Kopp and Lean 2011), but was necessary in order to provide global coverage. Additionally, detailed information on reservoir regulation rules and better groundwater use estimates, e.g. obtained from satellite data, could further improve the analysis. The same procedure could be applied to future precipitation and runoff series obtained from climate models, in order to explore the potential effects of climate change on water scarcity and the sensitivity of infrastructure investments to a changing climate. This will help to promote storage schemes that are robust to future uncertainties and adaptable in the face of a changing climate, as well as providing a platform for exploration of other policy responses, including demand reduction and groundwater resource management.

Appendix

$b_{j,t}$	Transboundary flow in basin j in month t
c_j	Storage capacity
$e_{j,t}$	Evaporation losses
$\text{ewr}_{j,t}$	Environmental water requirements
$k_{j,t}$	Storage in SPA
$q_{j,t}$	Surface runoff
$s_{j,t}$	Storage
$u_{j,t}$	Total water use

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