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Geoscientific Model Development
Discussions

Simulating human impacts on global water resources using VIC-5

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Abstract. Questions related to historical and future water resources and water scarcity have been addressed by several macro-scale hydrological models over the last few decades. However, further advancements are needed to improve the integration of anthropogenic impacts and environmental flow requirements into hydrological models. The newly developed VIC-WUR model aims to increase the applicability of the VIC-5 model for water resource assessments, specifically by including human impacts and environmental flow requirements. To this end, VIC-WUR extends VIC-5 with modules for irrigation, domestic, industrial, energy and livestock water-use, environmental flow requirements for surface and groundwater systems, and dam operation. Model inputs of sectoral water demand were estimated independently and correlated well to reported national water withdrawals.

VIC-WUR results, based on the newly developed modules, corresponded with results from reported global water withdrawals and other hydrological models, although differences exist. The VICWUR irrigation withdrawals were high compared to the other models but closer to the reported values, decreasing the gap between simulated and reported withdrawals. Irrigation withdrawals were probably high due to the inclusion of groundwater withdrawals and paddy irrigation in the model. Domestic and industrial water withdrawals were slightly lower than the reported values. Domestic and industrial withdrawals were probably insufficient due to low water availability, as the potential water withdrawals are more in line with reported values. Livestock water withdrawals were within the range of reported values and other models.

The model additions comprehensively incorporate anthropogenic and environmental water use, which provides new opportunities for global water resource assessments. A preliminary assessment of environmental flow requirements shows competition between water resources allocated for human consumption and the environment, from ground and surface water sources. The improvements made here are a first step towards integrated water-food-energy nexus modelling.

1 Introduction

Questions related to historical and future water resources and scarcity have been addressed by several macro-scale hydrological models over the last few decades (Liang et al., 1994; Alcamo et al., 1997;

Hagemann and Gates, 2001; Takata et al., 2003; Krinner et al., 2005; Bondeau et al., 2007; Hanasaki et al., 2008; Van Beek and Bierkens, 2008; Best et al., 2011). Early efforts focussed on the simulation of natural water resources and the impacts of land cover and climate change on water availability (Oki et al., 1995; Nijssen et al., 2001a; Nijssen et al., 2001b). Recently, a larger focus has been on incorporating anthropogenic impacts, such as water withdrawals and dam operations, into water resource assessments (Alcamo et al., 2003; Haddeland et al., 2006b; Biemans et al., 2011; Wada et al., 2011b; Hanasaki et al., 2018).

Global water withdrawals increased eight-fold over the last century and are projected to increase further (Shiklomanov, 2000; Wada et al., 2011a). Although water withdrawals are only a small fraction of the total global runoff (Oki and Kanae, 2006), water scarcity can be severe due to the variability of water in both time and space (Postel et al., 1996). Already severe water scarcity is experienced by two-thirds of the global population for at least part of the year (Mekonnen and Hoekstra, 2016). To stabilize water availability for different sectors (e.g. irrigation, hydropower, and domestic uses) dams and reservoirs were built, which are able to strongly affect global river discharge (Nilsson et al., 2005; Grill et al., 2019). In addition, groundwater resources are being extensively exploited to meet increasing water demands (Rodell et al., 2009; Famiglietti, 2014).

However, further advancements are needed to improve the integration of anthropogenic impacts into hydrological models (Döll et al., 2016). Several models do not yet incorporate all aspects of anthropogenic water withdrawals such as domestic, manufacturing and energy (thermoelectric) water withdrawals from both ground and surface water. Although these sectors use less water than irrigation (Shiklomanov, 2000; Grobicki et al., 2005; Hejazi et al., 2014) they are locally important actors (Gleick et al., 2013), especially for the water-food-energy nexus (Bazilian et al., 2011). Sufficient water supply and availability are essential for meeting a range of local and global sustainable development goals related to water, food, energy and ecosystems (Bijl et al., 2018).

Environmental flow requirements (EFRs) are also often neglected in global water resource assessments (Pastor et al., 2014), even though they are “(...) necessary to sustain aquatic ecosystems which, in turn, support human cultures, economies, sustainable livelihoods, and well-being” (Brisbane Declaration, 2017). Various EFR methods are available for streamflow (Smakhtin et al., 2004; Richter et al., 2012; Pastor et al., 2014) and groundwater (Gleeson and Richter, 2018), although environmental limits for groundwater withdrawal have only recently been considered explicitly. Anthropogenic alterations already strongly affect freshwater ecosystems (Carpenter et al., 2011), with more than a quarter of all global rivers experiencing very high biodiversity threats (Vorosmarty et al., 2010). By neglecting EFRs, water availability for anthropogenic uses is likely over-estimated (Gerten et al., 2013).

One of widely-used macro-scale hydrological models is the Variable Infiltration Capacity (VIC) model. The model was originally developed as a land-surface model (Liang et al., 1994), but has been mostly used as a stand-alone hydrological model (Abdulla et al., 1996; Nijssen et al., 1997) using an offline routing module (Lohmann et al., 1996; Lohmann et al., 1998b, a). Where land-surface models focus on the vertical exchange of water and energy between the land surface and the atmosphere, hydrological models focus on the lateral movement and availability of water. By combining these two approaches, VIC simulations are strongly process-based and this, in turn, provides a good basis for climate-impact modelling. Recently version 5 of the VIC model (VIC-5) was released (Hamman et al., 2018), which focussed on improving the model infrastructure. These improvements are highly relevant when simulating anthropogenic impacts on global water resources.

VIC has been used extensively in studies ranging from: coupled regional climate model simulations (Zhu et al., 2009; Hamman et al., 2016), combined river discharge and water-temperature simulations (van Vliet et al., 2016), hydrological sensitivity to climate change (Hamlet and Lettenmaier, 1999; Nijssen et al., 2001a; Chegwidden et al., 2019), global streamflow simulations (Nijssen et al., 2001b), and real-time drought forecasting (Wood and Lettenmaier, 2006; Mo, 2008). Several studies used VIC to simulate the anthropogenic impacts of irrigation and dam operation on water resources (Haddeland et al., 2006a; Haddeland et al., 2006b; Zhou et al., 2015; Zhou et al., 2016) based on the model setup of Haddeland et al. (2006b). However, water withdrawals for other sectors and flow requirements for freshwater ecosystems were ignored in these studies.

Our study aims to increase the applicability of the VIC-5 model for water resource assessments, specifically by including human impacts and environmental flow requirements. Here the newly developed VIC-WUR model is presented (named after the developing team at Wageningen University and Research). The VIC-WUR model extends the existing VIC-5 model with several modules that simulate the anthropogenic impacts on water resources. These modules include: integrated routing, water use for various sectors (irrigation, domestic, industrial, energy and livestock), environmental flow requirements for both surface and groundwater systems, and dam operation.

The next section first describes the original VIC-5 hydrological model (Section 2.1), which calculates natural water resource availability. Subsequently the integration of the anthropogenic impact modules, which modify the water resource availability, are described (Section 2.2). Global anthropogenic water uses for each sector are also estimated (Section 2.3). To assess the performance of the newly developed modules, the VIC-WUR results were compared with reported global withdrawal data from Shiklomanov (2000) and Steinfeld et al. (2006) as well as various other state-of-the-art global hydrological models used in the Inter-Sectoral Impact Model Intercomparison Project (ISIMIP; Warszawski et al., 2014) and Water and Global Change project (WATCH; Harding et al., 2011) (Section 3.1). The results also contain a preliminary assessment of the water availability constrains imposed by EFRs.

2 Model development

2.1 VIC hydrological model

The basis of the VIC-WUR model is the Variable Infiltration Capacity model version 5 (VIC-5) (Liang et al., 1994; Hamman et al., 2018). VIC-5 is an open source macro-scale hydrological model that simulates the full water and energy balance on a (latitude – longitude) grid. Each grid cell accounts for sub-grid variability in land cover and topography, and allows for variable saturation across the grid cell. For each sub-grid the water and energy balance is computed individually (i.e. sub-grid do not exchange water or energy between one another). The methods used to calculate the water and energy balance are summarized in Appendix A, mainly based on the work of Liang et al. (1994). For the description of the global calibration and validation of the water balance one is referred to Nijssen et al. (2001b).

VIC version 5 (Hamman et al., 2018) upgrades did not change the model representation of physical processes, but improved the model infrastructure. Improvements include the use of NetCDF for

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input/output and the implementation of parallelization through Message Passing Interface (MPI). These changes increase computational speed and make VIC-5 better suited for (computationally expensive) global simulations. The most significant modification that enables new model applications is that VIC-5 also changed the processing order of the model. In previous versions all timesteps were processed for a single grid cell before continuing to the next cell (time-before-space). In VIC-5 all grid cells are processed before continuing to the next timestep (space-before-time). This development allows for interaction between grid cells every timestep, which is important for full integration of the anthropogenic impact modules, especially water withdrawals and dam operation.

For example, surface and subsurface runoff routing to produce river streamflow was typically done as a post-process operation (Lohmann et al., 1996; Hamman et al., 2017), due to the time-before-space processing order of previous versions. Therefore, water withdrawals could not be taken into account directly and studies using the model setup of Haddeland et al. (2006b) required multiple successive model runs. Since VIC-5 uses the space-before-time processing order, runoff routing could be simulated each timestep. The routing post-process was replaced by our newly developed routing module which simulates routing sequentially (upstream-to-downstream) to facilitate water withdrawals between cells.

2.2 Anthropogenic-impact modules

VIC-WUR extends the existing VIC-5 though the addition of several newly developed anthropogenic-impact modules (Figure 1). These modules include sector-specific water withdrawal and consumptions, environmental flow requirements for both surface and groundwater systems and dam operation for small (within-grid) and large dams.

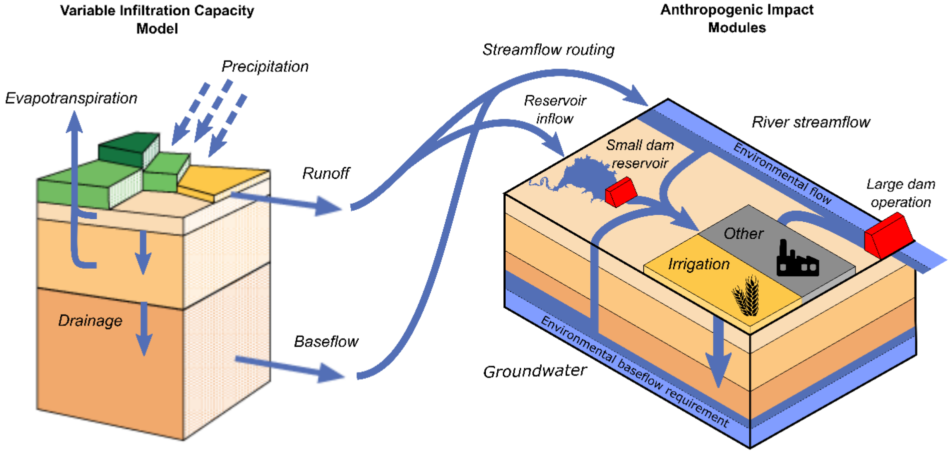

Figure 1: Schematic overview of the VIC-WUR model that includes the VIC-5 model and several anthropogenic impact modules. Water from river streamflow, groundwater and small (within-grid) reservoirs are available for withdrawal. Surface and groundwater withdrawals are constrained by environmental flow requirements. Withdrawn water is available for irrigation, domestic, industrial, energy and livestock use. Unconsumed irrigation water is returned to the soil column of the hydrological model. Unconsumed water for the other sectors is returned to the river streamflow. Small reservoirs fill using surface runoff from the cell they are located, while large dam reservoirs operate solely on rivers streamflow.

2.2.1 Water withdrawal and consumption

In VIC-WUR, sectoral water demands need to be specified for each grid cell (Section 2.3). To meet water demands, water can be withdrawn from river streamflow, small (within-grid) reservoirs and groundwater resources. Streamflow withdrawals are abstracted from the grid cell discharge (as generated by the routing module), reservoir withdrawals are abstracted from small dam reservoirs (located in the cell) and groundwater withdrawals are abstracted from the third layer soil moisture. The partitioning of water withdrawals between surface and ground water resources was based on the study of Döll et al. (2012), who estimated the groundwater withdrawal fraction for each sector in around 15.000 national and sub-national administrative units. Groundwater fractions were based mainly on information from the International Groundwater Resources Assessment Centre (IGRAC; un-igrac.org) database. Surface water withdrawals are partitioned between river streamflow and small reservoirs relative to water availability.

Water could also be withdrawn from the river streamflow of other ‘remote’ cells in delta areas. Since rivers cannot split in our routing module, the model is unable to simulate the redistribution of water resources in dendritic deltas. Therefore, streamflow at the river mouth is available for use in delta areas to simulate the actual water availability. Delta areas were delineated by the global delta map of Tessler et al. (2015).

When water demands cannot be met, water withdrawals are allocated to the domestic, energy, manufacturing, livestock and irrigation sector in that order. Withdrawn water is partly consumed, meaning the water evaporates and does not return to the hydrological model. Consumption rates were set at 0.15 and 0.10 for the domestic and industrial sectors respectively, based on the data of Shiklomanov (2000). The water consumption in the energy sector was based on Goldstein and Smith (2002) and varies per thermoelectric plant based on the fuel type and cooling system. For the livestock sector the assumption was made that all withdrawn water is consumed. Unconsumed water withdrawals for these sectors are returned as river streamflow. For the irrigation sector, consumption was determined by the calculated evapotranspiration. Unconsumed irrigation water remains in the soil column and eventually returns as subsurface runoff.

2.2.2 Environmental flow requirements

Water withdrawals can be constrained by environmental flow requirements (EFRs). These EFRs specify the timing and quantity of water needed to support terrestrial river ecosystems (Smakhtin et al., 2004; Pastor et al., 2019). Surface and groundwater withdrawals are constrained separately in VIC-WUR, based on the EFRs for streamflow and baseflow respectively. EFRs for streamflow specify the minimum river streamflow requirements while EFRs for baseflow specify the minimum subsurface runoff requirements (from groundwater to surface water). Since baseflow is a function groundwater availability in the hydrological model, baseflow requirements are used to constrain groundwater withdrawals.

Our study used the Variable Monthly Flow (VMF) method (Pastor et al., 2014) to calculate the EFRs for streamflows. VMF calculates the required streamflow as a fraction of the natural flow during high (30%), intermediate (45%) and low (60%) flow periods, as described in Appendix B. The VMF method performed favourably compared to other hydrological methods, such as the method proposed by Smakhtin et al. (2006) or the Q90-Q50 method, in 11 case studies where EFRs were calculated locally

(Pastor et al., 2014). The advantage of the VMF method is that the method accounts for the natural flow variability, which is essential to support freshwater ecosystems (Poff et al., 2010).

EFR methods for baseflow have been rather underdeveloped compared to EFR methods for streamflow. However, a presumptive standard of 90 % of the natural subsurface runoff through time was proposed by Gleeson and Richter (2018), as described in Appendix B. This standard should provide high levels of ecological protection, especially for groundwater dependent ecosystems. Note that part of the EFRs for baseflow are already captured in the EFRs for streamflow, especially during low-flow periods that are usually dominated by baseflows. However, the EFRs for baseflow specifically limit local groundwater withdrawals while EFRs for streamflow include the accumulated runoff from upstream areas. Also, the chemical composition of groundwater derived flows is inherently different, making them a non-substitutable water flow for environmental purposes (Gleeson and Richter, 2018).

2.2.3 Dam operation

Due to the lack of globally available information on local dam operations, several generic dam operation schemes were developed for macro-scale hydrological models to reproduce the effect of dams on natural streamflow (Haddeland et al., 2006a; Hanasaki et al., 2006; Zhao et al., 2016). In VIC-WUR a distinction is made between 'small' dam reservoirs (with an upstream area smaller than the cell area) and 'large' dam reservoirs, similar to Hanasaki et al. (2018), Wisser et al. (2010) and Döll et al. (2009). Small dam reservoirs act as buckets that fill using surface runoff of the grid-cell they are located in and reservoirs storage can be used for water withdrawals in the same cell. Large dam reservoirs are located in the main river and used the operation scheme of Hanasaki et al. (2006).

The scheme distinguishes between two dam types: (1) dams that do not account for water demands downstream (e.g. hydropower dams or flood protection dams) and (2) dams that do account for water demand downstream (e.g. irrigation dams). For dams that do not account for demands, dam release is aimed at reducing annual fluctuations in discharge. For dams that do account for demands, dam release is additionally adjusted to provide more water during periods of high demand. The operation scheme was validated by Hanasaki et al. (2006) for 28 reservoirs and was used in various studies (Hanasaki et al., 2008; Döll et al., 2009; Pokhrel et al., 2012; Voisin et al., 2013; Hanasaki et al., 2018). Here, the scheme was adjusted slightly to account for monthly varying EFRs and to reduce overflow releases. The full operation scheme is described in Appendix C.

The Global Reservoir and Dam (GRanD) database (Lehner et al., 2011) was used to specify location, capacity, function (purpose), and construction year of each dam. The capacity of multiple (small- and large) dams located in the same cell were combined.

2.3 Sectoral water demands

Water withdrawals are based on the irrigation, domestic, industry, energy and livestock water demand in each grid-cell. Water demands represent the potential water withdrawal, which is reduced when insufficient water is available. Irrigation demands were estimated based on the hydrological model while water demands for other sectors are provided to the model as an input. Domestic and industrial were estimated based on several socioeconomic predictors, while energy and livestock water demands were mostly data-driven (i.e. derived from power plant and livestock distribution data). Due to data limitations the energy sector was incomplete, and energy water demands were partly included in the industrial water demands (which combined the remaining energy and manufacturing water demands). For more details concerning sectoral water demand calculations the reader is referred to Appendix D.

2.3.1 Irrigation demands

Irrigation demands were set to increase soil moisture in the root zone so that water availability is not limiting crop evapotranspiration and growth. Preferably, irrigation was supplied to fill the soil to field capacity (Allen et al., 1998), which is the moisture content where water leaching is minimized. The exception is paddy rice irrigation (Brouwer et al., 1989), where irrigation was also supplied to keep the upper soil layer saturated. Water demands for paddy irrigation practices are relatively high compared to conventional irrigation practices due to increased evaporation and percolation. Therefore, the crop irrigation demands for these two irrigation practices were calculated and applied separately (i.e. in different sub-grids).

Total irrigation demands also included transportation and application losses. Note that transportation and application losses are not 'lost' but rather returned to the soil column without being used by the crop. The water loss fraction was based on Frenken and Gillet (2012), who estimated the irrigation efficiency for 22 United Nations sub-regions based on differences between calculated irrigation requirements and reported irrigation withdrawals. Potential total irrigation demands were validated independently and correlated well with reported withdrawals (adjusted $R^2 > 0.8$; Figure 2a).

2.3.2 Domestic and industrial demands

Domestic and industrial water withdrawals were estimated based on Gross Domestic Product (GDP) per capita and Gross Value Added (GVA) by industries respectively. These drivers do not fully capture the multitude of socioeconomic factors that influence water demands (Babel et al., 2007). However, the wide availability of data allows for extrapolation of water demands to data-scarce regions and future scenarios (using studies such as Chateau et al. (2014)).

Domestic water demands per capita (used for drinking, sanitation, hygiene and amenity uses) were estimated similar to Alcamo et al. (2003). Demands increased non-linearly with GDP per capita due to the acquisition of water using appliances as household become richer. A minimum water supply is needed for survival, and the saturation of water using appliances sets a maximum on domestic water demands. Industrial water demands (used for cooling, transportation and manufacturing) were estimated similar to Flörke et al. (2013) and Voß and Flörke (2010). Industrial demands increased linearly with GVA (as an indicator of industrial production). Since industrial water intensities (i.e. the water use per production unit) vary widely between different industries (Flörke and Alcamo, 2004 ; Vassolo and Döll, 2005; Voß and Flörke, 2010), the average water intensity was estimated for each country. Both domestic and industrial water demands were also influenced by technological developments that increase water-use efficiency over time, as in Flörke et al. (2013). Estimated domestic and industrial water demands were validated independently and correlated well to reported withdrawals (adjusted $R^2 > 0.8$; Figure 2b and Figure 2c).

Domestic water demands varied monthly based on air temperature variability as in Wada et al. (2011b). Using this approach, water demands were higher in summer than in winter, especially for counties with strong seasonal temperature differences. Domestic water demand per capita were downscaled using the HYDE3.2 gridded population maps (Goldewijk et al., 2017). Industrial water demands were kept constant throughout the year. Industrial demands were downscaled from national to grid cell values using the NASA Back Marble night-time light intensity map (Roman et al., 2018). National industrial water demands were allocated based on the relative light intensity per grid cell for each country.

2.3.3 Energy and livestock demands

Energy water demands (used for cooling of thermoelectric plants) were estimated using data from van Vliet et al. (2016). Water use intensity for generation (i.e. the water use per generation unit) was estimated based on the fuel and cooling system type (Goldstein and Smith, 2002), which was combined with the installed generation capacity. Note that the data only covered a selection of the total number of thermoelectric power plants worldwide. Around 27% of the total (non-renewable) global installed capacity between 1980 and 2011 was included in this dataset due to lack of information on cooling system types for the majority of thermoelectric plants. To avoid double counting, energy water demands were subtracted from the industrial water demands.

Livestock water demands (used for drinking and animal servicing) were estimated by combining the Gridded Livestock of the World (GLW3) map (Gilbert et al., 2018) with the livestock water requirement reported by Steinfeld et al. (2006). Eight varieties of livestock were considered: cattle, buffaloes, horses, sheep, goats, pigs, chicken and ducks. Drinking water demands varied monthly based on temperature as described by Steinfeld et al. (2006), whereby drinking water requirements were higher during higher temperatures.

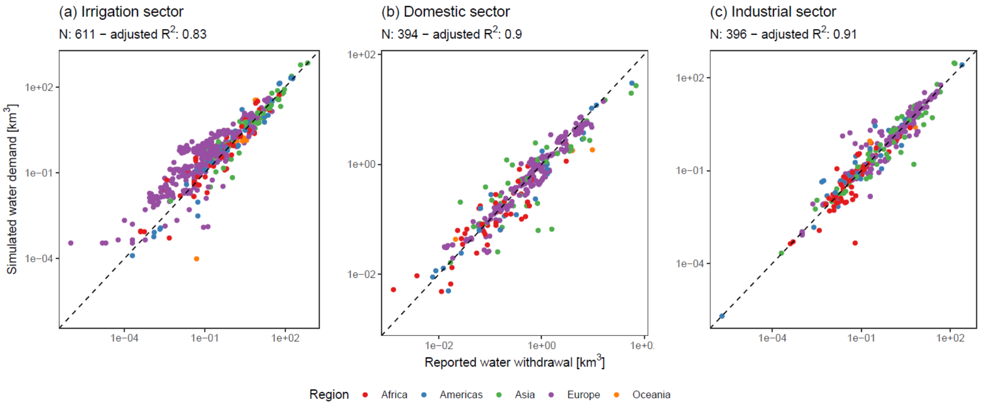

Figure 2: Comparison between reported and estimated national water withdrawal per year for the irrigation (a), domestic (b) and industrial (c) sector. Reported values are from the validation dataset. Note the log-log axis which is used to display the wide range of water withdrawals. The adjusted R squared is also based on the log-log values.

3 Model application

3.1 Setup

VIC-WUR results were generated between 1979 and 2016, excluding a spin-up period of one year (analysis period from 1980 to 2016). The model used a 6-hourly timestep and simulations were executed on a 0.5° by 0.5° grid (around 55 km at the equator) with three soil layers per grid cell. Soil and (natural) vegetation parameters were the same as in Nijssen et al. (2001c) (disaggregated to 0.5°), who used various sources to determine the soil (Cosby et al., 1984; Carter and Scholes, 1999) and vegetation parameters (Calder, 1993; Ducoudre et al., 1993; Sellers et al., 1994; Myneni et al., 1997).

Nijssen et al. (2001c) used the Advanced Very High Resolution Radiometer vegetation type database (Hansen et al., 2000) to spatially distinguish 13 land cover types. The land cover type 'cropland' in the original land-cover dataset was replaced by cropland extents from the MIRCA2000 cropland dataset (Portmann et al., 2010). MIRCA2000 distinguishes the monthly growing area(s) and season(s) of 26 irrigated and rain-fed crop types around the year 2000. Crop types were aggregated into three land cover types: rain-fed, irrigated and paddy rice cropland. The natural vegetation was proportionally rescaled to make up discrepancies between the natural vegetation and cropland extents.

Cropland coverage (the cropland area actually growing crops) varied monthly based on the crop growing areas of MIRCA2000. The remainder was treated as bare soil. Cropland vegetation parameters (e.g. Leaf

Area Index (LAI), displacement, vegetation roughness and albedo) vary based on the monthly crop growing seasons and the development-stage crop coefficients of the Food and Agricultural Organisation (Allen et al., 1998).

The latest WATCH forcing data Era Interim (aggregated to 6 hourly), developed by the EU Water and Global Change (WATCH; Harding et al., 2011) project, was used as climate forcing (WFDEI; Weedon et al., 2014). The dataset provides gridded historical climatic variables of minimum and maximum air temperature, precipitation (as the sum of snowfall and rainfall, GPCC bias-corrected), relative humidity, pressure and incoming shortwave and longwave radiation.

3.2 Results

The VIC-WUR model results were compared to several of the Inter-Sectoral Impact Model Intercomparison Project (Warszawski et al., 2014) simulation round 2a global hydrological impact models: H08 (Hanasaki et al., 2008), LPJmL (Sitch et al., 2003), VIC (Liang et al., 1994), PCR-GLOBWB (Wada et al., 2014) and WaterGAP (Muller Schmied et al., 2016). The ISIMIP2a outputs are comparable to the results of our study since the same meteorological and land cover inputs were used. The VIC and LPJmL models only provided data on the actual and potential irrigation withdrawal and consumption. H08 additionally provided data for the domestic sector, and PCR-GLOBWB additionally provided data for the domestic and livestock sector. To increase the number of models to compare to, the WaterGAP (Alcamo et al., 2003) output for the domestic and industrial (manufacturing plus energy) sector from the Water and Global Change project (Harding et al., 2011) was included as well. Note that the WaterGAP simulations were based on a different WATCH forcing dataset (WFD) (Weedon et al., 2011). Since our study used a present-day land-cover map, the actual and potential irrigation withdrawals were compared based on the so-called ‘pressoc’ (present-day land-cover) simulations. Domestic, industrial and livestock sectors were compared based on the ‘varsoc’ (variable human influences) simulations, since our study used varying socioeconomic predictors to estimate the water demand in these sectors. Results were compared between the years 1980 to 2005. The reported global water withdrawal of Shiklomanov (2000) and Steinfeld et al. (2006) are included as a reference.

3.2.1 Irrigation sector

Compared to other models the VIC-WUR potential and actual water withdrawals (without EFRs) were at the high end (Figure 3). Annual potential and actual irrigation withdrawals for VIC-WUR were around 3060 km^3 and 1870 km^3 respectively, while the ensemble mean potential and actual withdrawals were only 2200 km^3 and 1400 km^3 respectively. Especially in the African and Asian regions the irrigation withdrawals were high compared to the model ensemble. Irrigation withdrawals were probably high due to the inclusion of groundwater withdrawals and paddy irrigation in the model. All models (VIC-WUR included) indicated a lower actual irrigation withdrawal than reported. Due to the increased irrigation withdrawal, the deficit for VIC-WUR (around 710 km^3) was lower than the ensemble mean deficit (around 1170 km^3). This difference is often assumed to be met by non-renewable and/or unspecified withdrawals (Wada et al., 2010; Hanasaki et al., 2018).

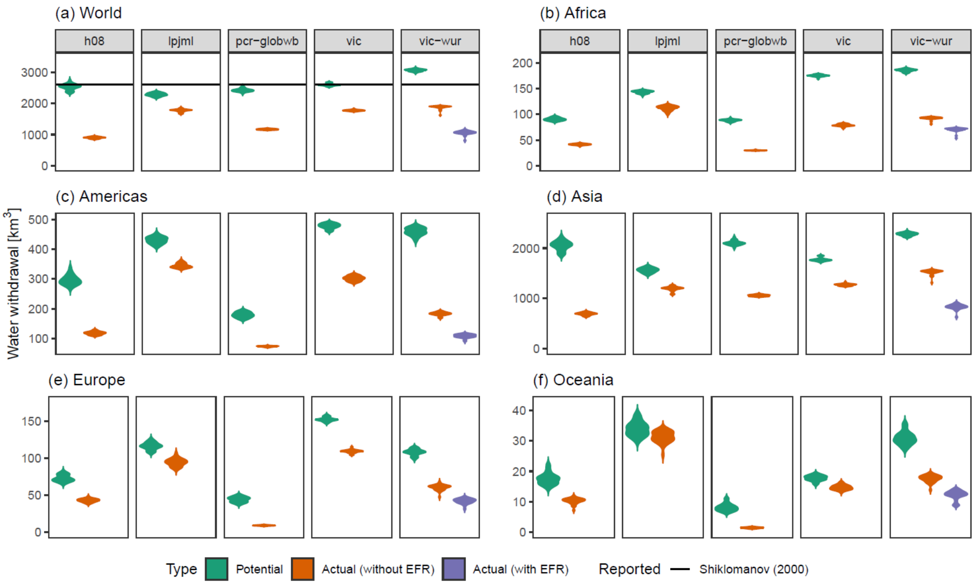

Figure 3: Irrigation withdrawals for the world (a) and each region (b-f) for five hydrological models between 1980 and 2005. Colours differentiate between the potential (green), actual excluding EFRs (orange) and actual including EFRs (purple) withdrawals. The spread indicates the inter-annual variation of the simulated irrigation withdrawals. Data was obtained from the ISIMIP project, except for the VIC-WUR model. The black line indicates the reported total irrigation withdrawal as estimated by Shiklomanov (2000). Note that the y-axis varies for each graph, making Asia is by far the largest contributor to irrigation withdrawals.

Limitations imposed by the environmental flow requirements reduced the actual (irrigation) water withdrawals by about 43% (Figure 4a). In total, 71% of the reduction could be attributed to limitations

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imposed on groundwater withdrawals. Therefore, the impact of the environmental flow requirements was largest in groundwater dependent regions (Figure 4b). However, surface water withdrawals increased by 11 % when limiting groundwater withdrawals on top of limiting surface water withdrawals, due to subsurface runoff increases.

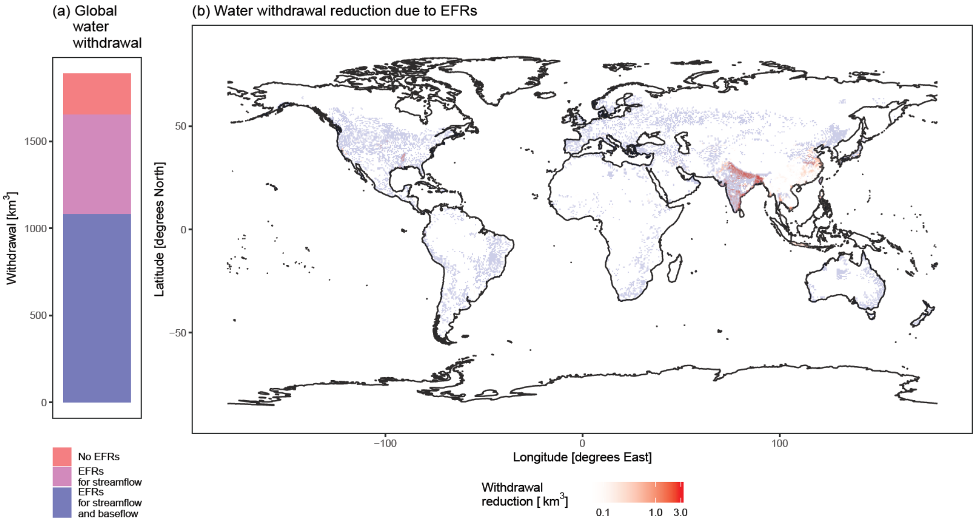

Figure 4: Average annual water withdrawals reductions when adhering to EFRs as (a) global gross total and (b) spatially distributed. Global gross totals (a) are separated into withdrawals without EFRs (red), withdrawals with EFRs for streamflow (purple) and withdrawals with EFRs for both streamflow and baseflow (blue). Note the log axis for the spatially distributed withdrawal reductions (b) to better display the spatial distribution of the reductions. Blue regions indicate areas where the withdrawal reduction is largely (> 75 %) caused by the EFRs for baseflow.

3.2.2 Domestic, industrial and livestock sector

In contrast to the irrigation withdrawals, the annual domestic (ranging from 195 km^3 to 275 km^3) and industrial (ranging from 461 km^3 to 637 km^3) withdrawals of VIC-WUR were slightly lower than that of reported values and other models (Figure 5a; Figure 5b). Domestic and industrial withdrawals were probably low due to insufficient water availability, as the potential water withdrawals are more in line with reported values. Note that the rising trend of the VIC-WUR domestic water withdrawals (on average 2.5 km^3 year^{-1}), WaterGAP (on average 2.3 km^3 year^{-1}) and H08 (on average 2.4 km^3 year^{-1}) was more gradual than that of PCR-GLOBWB (on average 8.3 km^3 year^{-1}) and Shiklomanov (2000) (on average 8.1 km^3 year^{-1}) between 1980 and 2000. This slope difference resulted from the different

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methods used to calculate domestic water demand. The annual livestock withdrawals of VIC-WUR (ranging from 25 km^3 to 27 km^3) were within range of reported values and other models (Figure 5c).

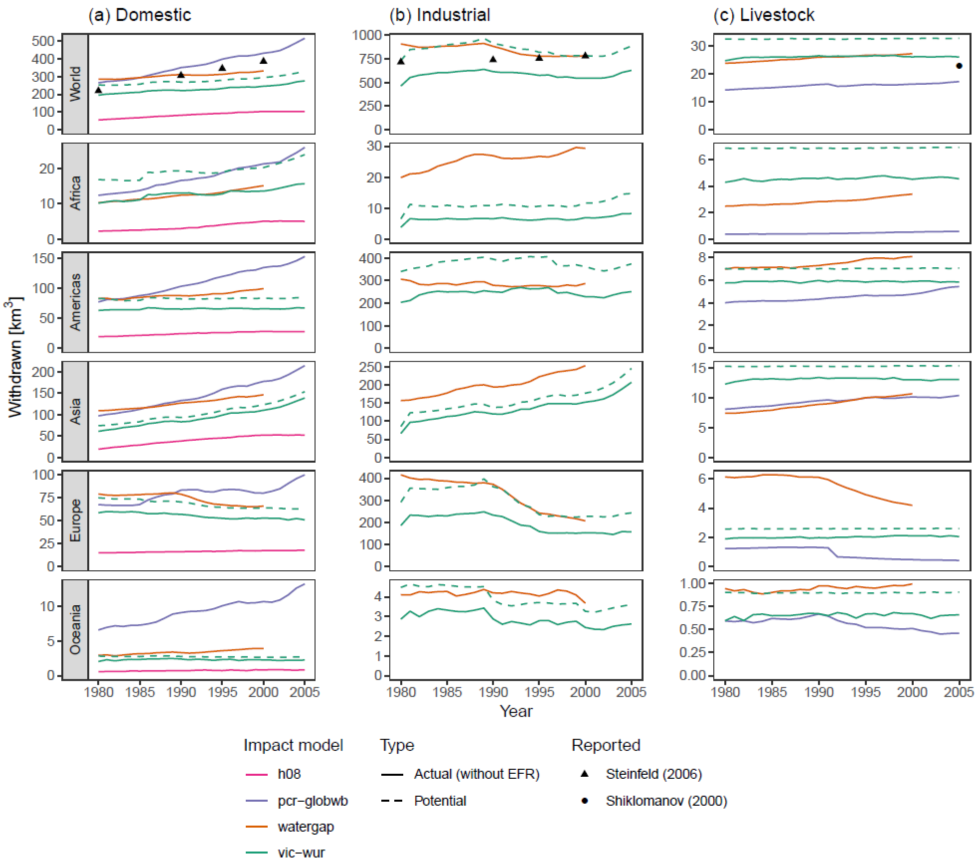

Figure 5: Domestic (a), industrial (b) and livestock (c) water withdrawals for the world and each region for five hydrological models between 1980 and 2005. Colours differentiate between models. Data was obtained from the ISIMIP project (for H08 and PCR-GLBOWB) and WATCH project (for WaterGAP), except for the VIC-WUR model. The black points indicates the reported total water withdrawals for each sector as estimated by Shiklomanov (2000) (for domestic and industrial sectors) and (Steinfeld et al., 2006) (for livestock sector). Note that the y-axis varies for each graph, making Asia, Europe and Asia the largest contributors to the domestic, industrial and livestock water withdrawals respectively.

4 Discussion

Our paper presents the newly developed VIC-WUR model that aims to provide new opportunities for global water resource assessments by integrating several anthropogenic-impact modules. The results of the VIC-WUR model are in line with reported water withdrawal values of Shiklomanov (2000) and

(Steinfeld et al., 2006), as well as the results of other hydrological models available via the ISIMIP and WATCH projects. However, there are some important differences.

Potential irrigation withdrawal differences between models reflect the differences in the representation of hydrological processes as well as the method used to calculate irrigation demands. Especially the differences between VIC and VIC-WUR are interesting since they both employ the same hydrological model. The increase in potential irrigation withdrawal between VIC and VIC-WUR can be attributed to the inclusion of paddy irrigation by VIC-WUR. VIC irrigates crops only when they experience water stress, while in VIC-WUR paddy irrigation is also used to saturate the top soil layer. The VIC-WUR potential withdrawals are 56 % higher than the VIC potential withdrawals for cells where rice is the major crop (> 50 % of cropland). Potential irrigation withdrawals for convention irrigation is actually higher for VIC than VIC-WUR, since the field capacity in VIC-WUR is tuned lower than that the field capacity in VIC (see Appendix D). The lower field capacity results in reduced percolation for conventional irrigation. The VIC potential withdrawals are 33 % higher than the VIC-WUR potential withdrawals for cells where no rice is present. This difference is reflected in the spatial distribution of water demands. Potential irrigation withdrawals of VIC-WUR are higher than those of VIC, except for the Americas and Europe where paddy irrigation is relatively limited.

Actual irrigation withdrawals of VIC-WUR are high compared to the other models. This difference can be explained, in part, since some models (LPJmL, VIC and H08) did not (yet) include groundwater withdrawals in their simulations. The high irrigation withdrawals of VIC-WUR decrease the gap between the reported and simulated irrigation withdrawals often assumed to be met by (non-renewable) groundwater withdrawals, in particular fossil groundwater stores (Wada et al., 2010). Often the regions where the simulated withdrawals are lower than the actual withdrawals are regions where unsustainable groundwater exploitation is reported by several studies (Gleeson et al., 2012; Rodell et al., 2018). To our knowledge no previous study has estimated the amount of global non-renewable groundwater withdrawals without using one of the models mentioned above. Therefore, the accuracy of actual irrigation withdrawal results cannot be verified.

Differences in domestic, industrial and livestock actual water withdrawals between models are difficult to explain since most studies use different methods to calculate and downscale sectoral water demands. Therefore, water availability is not the only factor affecting the actual water withdrawals. Inputs of potential domestic and industrial water withdrawals are close to the values reported by Shiklomanov (2000). However, the actual water withdrawals are lower, indicating limited water availability. The lack of water availability could be due to a number of factors: (1) The spatial distribution of water demands, (2) the division between groundwater and surface water withdrawals, and/or (3) simulations of water availability are insufficient in certain regions. Improvements would require more data to improve groundwater and surface water demands and/or regional verification of water availability.

Environmental Flow Requirements (EFRs) for both baseflow and streamflow are used to assess the water requirements for terrestrial river ecosystems. When adhering to EFRs the global water withdrawals are reduced substantially, especially due to groundwater withdrawal limitations. This limitation indicates competition between water allocated for anthropogenic uses and environmental purposes. In addition, groundwater withdrawal reductions upstream lead to increased surface water availability downstream. This interaction results in a trade-off between upstream groundwater withdrawals and downstream surface water withdrawals. Note that VIC-WUR does not include non-renewable groundwater withdrawals, while these withdrawals would affect baseflow to a lesser degree.

It can be discussed to what extent the EFRs for baseflow are too constricting, because it is based on the relatively stringent EFR for streamflow of Richter et al. (2012) (10 % of the natural streamflow). However, in the absence of any other standards, this baseflow standard remains the best available. However, the model setup allows for the evaluation of other standards as well. Note that, even when accounting for EFRs for baseflow on a grid scale, withdrawals can still have local and long-term impacts that are not captured by the model. The timing, location and depth of groundwater withdrawals are also important due to their interactions with the local geohydrology, as discussed by Gleeson and Richter (2018).

The newly developed model will be used for the assessment of trade-offs and synergies between the sectors in the water-food-energy nexus. However, there are some challenges when applying the methods as described in our paper to future water-food-energy nexus assessments. Firstly, an holistic approach is needed to assess trade-offs and synergies in the water-food-energy nexus. This approach should account for competition between resources in an integrated way and should also be captured in consistent scenarios. These scenarios should, for example, define future developments of manufacturing subsectors, hydropower and thermoelectric developments and water efficiencies. Secondly, the energy sector should be expanded upon. Currently only a limited number of thermoelectric power plants are included in the analysis while the rest is incorporated in the industrial water demands. By explicitly accounting for the energy water demand one is able to assess the impact of water scarcity on energy generation and manufacturing separately. Lastly, developments are needed to translate water scarcity into production losses. Currently, only the lack of water is assessed, while the interest is in determining the impact on food and energy production. Therefore, new modules have to be developed that estimate energy generation and food production based on the water availability of the VIC-WUR model.

5 Conclusion

The VIC-WUR model introduced in this paper aimed to provide new opportunities for global water resource assessments. Accordingly, several newly developed anthropogenic impact modules were integrated into the VIC-5 macro-scale hydrological model. The additions presented here comprehensively include anthropogenic and environmental water requirements and expand upon the previous efforts of Haddeland et al. (2006b).

The performance of the modules is in line with reported global water withdrawals and results of other hydrological models. While these additions are sufficient for global water resource assessments, further development is required in order to holistically assess trade-offs and synergies in the water-food-energy nexus. A preliminary assessment of environmental flow requirements already shows competition between water resources allocated for human consumption and the environment, from both ground and surface water sources.

6 Code availability

All code for the VIC-WUR model is freely available at github.com/wur-wsg/VIC (tag VIC-WUR.2.0.0; DOI 10.5281/zenodo.3399450) under the GNU General Public License, version 2 (GPL-2.0). VIC-WUR documentation can be found at vicwur.readthedocs.io. The original VIC model is freely available at github.com/UW-Hydro/VIC (tag VIC.5.0.1; DOI 10.5281/zenodo.267178) under the GNU General Public License, version 2 (GPL-2.0). VIC documentation can be found at vic.readthedocs.io. Documentation and scripts concerning inputs, configurations and analysis used in this study is freely available at github.com/bramdr/VIC-WUR_support (tag VIC-WUR.2.0.0; DOI 10.5281/zenodo.3401411) under the GNU General Public License, version 3 (GPL-3.0).

7 Appendix

7.1 Appendix A: VIC water and energy balance

In VIC each sub-grid computes the water and energy balance individually (i.e. sub-grid do not exchange water or energy between one another). For the water balance, incoming precipitation is partitioned between evapotranspiration, surface and subsurface runoff, and soil water storage. Potential evapotranspiration is based on the Penman-Monteith equation without the canopy resistance (Shuttleworth, 1993). The actual evapotranspiration is calculated by two methods, based on whether the land cover is vegetated or not (bare soil). Evapotranspiration of vegetation is constrained by stomatal, architectural and aerodynamic resistances and is partitioned between canopy evaporation and transpiration based on the intercepted water content of the canopy (Deardorff, 1978; Ducoudre et al., 1993). Bare soil evaporation is constrained by the saturated area of the upper soil layer. The saturated area is variable within the grid since (as the model name implies) the infiltration capacity of the soil is assumed heterogeneous (Franchini and Pacciani, 1991). Saturated areas evaporate at the potential evaporation rate while in unsaturated areas evaporation is limited. Surface runoff is produced by precipitation over saturated areas. Precipitation over unsaturated areas infiltrates into the upper soil layer and drains through the soil layers based on the gravitational hydraulic conductivity equations of Brooks and Corey (1964). In the first and second layer water is available for transpiration, while the third layer is assumed to be below the root zone. From the third layer baseflow is generated based on the non-linear Arno conceptualization (Franchini and Pacciani, 1991). Baseflow increases linearly with soil moisture content when the moisture content is low. At higher soil moisture contents the relation is non-linear, representing subsurface storm-flows.

For the energy balance, incoming net radiation is partitioned between sensible, latent, and ground heat fluxes and energy storage in the air below the canopy. The energy storage below the canopy is omitted if it is considered negligible (e.g. the canopy surface is open or close to the ground). The latent heat flux is determined by the evapotranspiration as calculated in the water balance. The sensible heat flux is calculated based on the difference between the air and surface temperature and the ground heat flux is calculated based on the difference between the soil and surface temperature. Since the incoming net radiation is also a function of the surface temperature (specifically the outgoing longwave radiation), the surface temperature is solved iteratively. Subsurface ground heat fluxes are calculated assuming an exponential temperature profile between the surface and the bottom of the soil column, where the bottom temperature is assumed constant. Later model developments included options for finite difference solutions of the ground temperature profile (Cherkauer and Lettenmaier, 1999), spatial distribution of soil temperatures (Cherkauer and Lettenmaier, 2003), a quasi-2-layer snow-pack snow model (Andreadis et al., 2009), and blowing snow sublimation (Bowling et al., 2004).

7.2 Appendix B: EFRs for streamflow and baselow

VIC-WUR used the Variable Monthly Flow (VMF) method (Pastor et al., 2014) to limit surface water withdrawals. The VMF method (Pastor et al., 2014) calculates the EFRs for streamflow as a fraction of the natural flow during high (Eq. A.1), intermediate (Eq. A.2) and low (Eq. A.3) flow periods. The presumptive standard Gleeson and Richter (2018) is used to limit groundwater withdrawals. This standard calculates the EFRs for baseflow as 90 % of the natural subsurface runoff through time (Eq. A.4). Here, daily instead of monthly EFRs were used to better capture the monthly flow variability.

$$EFR_{s,d} = 0.6 \cdot NF_{s,d} \qquad \text{Eq. (A.1)}$$

$$where\ NF_{s,d} \le 0.4 \cdot NF_{s,y}$$

$$EFR_{s,d} = 0.45 \cdot NF_{s,d} \qquad \text{Eq. (A.2)}$$

$$where\ 0.4 \cdot MF_{s,y} < NF_{s,d} \leq 0.8 \cdot NF_{s,y}$$

$$EFR_{s,d} = 0.3 \cdot NF_{s,d} \qquad \text{Eq. (A.3)}$$

$$where\ NF_{s,d} > 0.8 \cdot NF_{s,y}$$

$$EFR_{b,d} = 0.9 \cdot NF_{b,d} \qquad \text{Eq. (A.4)}$$

Where $EFR_{s,d}$ is the daily EFRs for streamflow [$m^3\ s^{-1}$], $EFR_{b,d}$ the daily EFRs for baseflow [$m^3\ s^{-1}$], $NF_{s,d}$ is the average natural daily streamflow [$m^3\ s^{-1}$], and $NF_{s,y}$ is the average natural yearly streamflow [$m^3\ s^{-1}$], and $NF_{b,d}$ is the average natural daily baseflow [$m^3\ s^{-1}$].

EFRs for streamflow and baseflow were based on VIC-WUR naturalized simulations between 1980 and 2010. Average natural daily flows were calculated as the multi-year daily average flow over the simulation period, followed by a 30-day moving average smoother.

7.3 Appendix C: Dam operation scheme

VIC-WUR used a dam operation scheme based on Hanasaki et al. (2006). Target release (i.e. the estimated optimal release) was calculated at the start of the operational year. The operational year starts at the month where the inflow drops below the average annual inflow, and thus the storage should be at its desired maximum. The scheme distinguished between two dam types: (1) dams that did not account for water demands downstream (e.g. hydropower dams or flood control) and (2) dams that did account for water demands downstream (e.g. irrigation dams). The original scheme of Hanasaki et al. (2006) also accounts for EFRs, which were fixed at half the annual mean inflow. Other studies lowered the requirements to a tenth of the mean annual inflow, increasing irrigation availability and preventing excessive releases (Biemans et al., 2011; Voisin et al., 2013). In our study the original dam operation scheme was adapted slightly to account for monthly varying EFRs.

For dams that did not account for demands, the initial release was set at the mean annual inflow corrected by the variable EFRs (Eq. A.1). For dams that did account for demands, the initial release was increased during periods of higher water demand. If demands were relatively high compared to the annual inflow, the release was corrected by the demand relative to the mean demand (Eq. A.2). If demands were relatively low compared to the annual inflow, release was corrected based on the actual water demand (Eq. A.3).

$$R'_m = EFR_{s,m} + (I_y - EFR_{s,y}) \qquad \text{Eq. (A.1)}$$

$$\quad where\ D_y = 0$$

$$R'_m = EFR_{s,m} + \left(I_y - EFR_{s,y}\right) * \frac{D_m}{D_y} \qquad \text{Eq. (A.2)}$$

$$\quad where\ D_y > 0\ and\ D_y > (I_y - EFR_{s,y})$$

$$R'_m = EFR_{s,m} + (I_y - EFR_{s,y}) - D_y + D_m \qquad \text{Eq. (A.3)}$$

$$\quad where\ D_y > 0\ and\ D_y \leq (I_y - EFR_{s,y})$$

Where R'_m is the initial monthly target release [$m^3\ s^{-1}$], $EFR_{s,m}$ is the average monthly EFR for streamflow demand [$m^3\ s^{-1}$], I_y is the average yearly inflow [$m^3\ s^{-1}$], $EFR_{s,y}$ is the average yearly EFR for streamflow [$m^3\ s^{-1}$], D_m is the average monthly water demand [$m^3\ s^{-1}$] and D_y is the average yearly water demand [$m^3\ s^{-1}$].

As in Hanasaki et al. (2006), the initial target release was adjusted based on storage and capacity. Target release was adjusted to compensate differences between the current storage and the desired maximum storage (Eq. A.4). Target release was additionally adjusted if the storage capacity is relatively low compared to the annual inflow, and unable to store large portions of the inflow for later release (Eq. A.5).

$$R_m = k \cdot R'_m \qquad \text{Eq. (A.4)}$$

$$\quad where\ c \geq 0.5$$

$$R_m = \left(\frac{c}{0.5}\right)^2 \cdot k \cdot R'_m + \left\{1 - \left(\frac{c}{0.5}\right)^2\right\} \cdot I_m \qquad \text{Eq. (A.5)}$$

$$\quad where\ 0 \leq c \leq 0.5$$

Where I_y is the average monthly inflow [m^3 s^{-1}], c the capacity parameter [-] calculated as the storage capacity divided by the mean annual inflow and k the storage parameter [-] calculated as current storage divided by the desired maximum storage. The desired maximum storage was set at 85% of the storage capacity as recommended by Hanasaki et al. (2006).

Water inflow, demand and EFRs were estimated based on the average of the past five years. Water demands were based on the water demands of downstream cells. Only a fraction of water demands were taken into account, based on the fraction of upstream area the dam controlled. For example: if a dam controlled 70% of the upstream area of a downstream cell, than 70% of its demands were taken into account. Fractions smaller than 25% were ignored.

The original dam operation scheme of Hanasaki et al. (2006) was shown to produce excessively high discharge events due to overflow releases (Masaki et al., 2018). These overflow releases occurred due to a mismatch between the expected and actual inflow. In our study, dam release was increased during high-storage events to prevent overflow and accompanying high discharge events. If dam storage was above the desired maximum storage, target dam release was increased to negate the difference (Eq. A.6). If dam storage was below the desired minimum storage, release is decreased (Eq. A.7). Dam release was adjusted exponentially based on the relative storage difference: small storage differences were only corrected slightly, but if the dam was close to overflowing or emptying, the difference was corrected strongly.

$$R_a = R_m + \frac{(S - C\alpha)}{\gamma} \cdot \left(\frac{\frac{S}{C} - \alpha}{1 - \alpha}\right)^b \qquad \text{Eq. (A.6)}$$

$$where\ S > C\alpha$$

$$R_a = R_m + \frac{(S - C(1-\alpha))}{\gamma} \cdot \left(\frac{(1-\alpha) - \frac{S}{C}}{1 - \alpha}\right)^b \qquad \text{Eq. (A.7)}$$

$$where\ S < C(1 - \alpha)$$

Where R_a is the actual dam release [m^3 s^{-1}], S the dam storage capacity [m^3], α the fraction of the capacity that is the desired maximum [-], β the exponent determining the correction increase [-] and γ the parameter determining the period when the release is corrected [s^{-1}]. In testing the exponent and period were tuned to 0.6 and 5 days respectively.

7.4 Appendix D: Water demand

7.4.1 Fitting and validation data

Data on irrigation, domestic and industrial water withdrawals were based on the AQUASTAT database (FAO, 2016), EUROSTAT database (EC, 2019) and United Nations World Water Development Report (Connor, 2015). Data on GDP per capita and GVA was abstracted from the Maddison Project Database 2018 (Bolt et al., 2018), Penn World Table 9.0 (Feenstra et al., 2015) and World Bank Development Indicators (World bank, 2010).

Available data for domestic an industrial withdrawals were divided into a dataset used for parameter fitting and a dataset used for validation. Domestic water demands were estimated for each United Nations sub-region, and thus data was divided per sub-region to ensure a good global coverage of data. In the same manner industrial water demand were divided per country. In case there is only a single data point, the data was added to both the fitting and validation data. To assess uncertainty introduced by dividing the dataset, a sensitivity analysis was performed. The dataset was divided into partial datasets 100 times at random. Each partial dataset was used to generate and map domestic and industrial water demands. Domestic water demands based on the partial datasets had a standard deviation of around 9% compared to water demands based on the total dataset (Figure A.1a). Industrial water demands based on the partial datasets had a standard deviation of around 3% compared to water demands based on the total dataset (Figure A.1b).

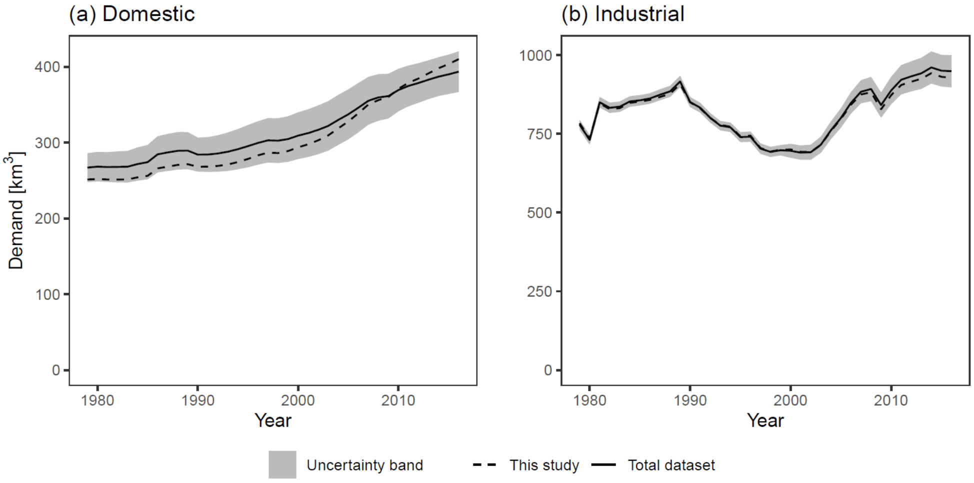

Figure A1: Sensitivity analysis of dividing the available data into a fitting and validation dataset for the domestic (a) and industrial (b) water demand estimations. Uncertainty bands are based on the standard deviation of 100 divisions. The solid line is the water demand estimation based on the total data. The striped line is the water demand estimation used in this study.

7.4.2 Irrigation sector

Conventional irrigation demands were calculated when soil moisture contents drop below the critical threshold where evapotranspiration will be limited. Demands were set to fill the soil up to field capacity (Eq. A.12). Paddy irrigation demands were set to always keep the soil moisture content of the upper soil layer saturated (Eq. A.13), similar to Hanasaki et al. (2008) and Wada et al. (2014). For paddy irrigation, the saturated hydraulic conductivity of the upper soil layer was reduced by its cubed root to simulate puddling practices, as recommended by the CROPWAT model (Smith, 1996). Total irrigation demands were adjusted by the irrigation efficiency (Eq. A.14). Paddy irrigation used an irrigation efficiency of 1 since the water losses were already incorporated in the water demand calculation.

$$ID'_{conventional} = FC - (W_1 + W_2) \qquad \text{Eq. (A.12)}$$

$$\text{where } W_1 + W_2 < W_{cr,1} + W_{cr,2}$$

$$ID'_{paddy} = W_{max,1} - W_1 \qquad \text{Eq. (A.13)}$$

$$\text{where } W_1 < W_{max,1}$$

$$ID = ID' * IE \qquad \text{Eq. (A.14)}$$

Where $ID'_{conventional}$ is the conventional crop irrigation demand [mm], ID'_{paddy} is the paddy crop irrigation demand [mm], ID is the total irrigation demand [mm], W_1 and W_2 are the soil moisture contents of the first and second soil layer respectively [mm], W_{cr} is the critical soil moisture content [mm], FC is the field capacity [mm], W_{max} the maximum soil moisture content [mm], and IE is the irrigation efficiency [mm mm^{-1}]. The field capacity was tuned to tuned to ($Wcr_1 + Wcr_2$) / 0.8. Note that our study used a lower field capacity compared to the (Haddeland et al., 2006b) model setup, since this provided a better fit.

7.4.3 Domestic sector

Domestic water demands were represented by using a sigmoid curve for the calculation of structural domestic water demands (Eq.A.15) and a efficiency rate for the calculation of water-use efficiency increases (Eq. A.16). These equations differ slightly from Alcamo et al. (2003) since our study used the base 10 logarithms of GDP and water withdrawals per capita as they provided a better fit.

$$DSW_y = DSW_{min} + (DSW_{max} - DSW_{min}) * \frac{1}{1+e^{-f(GDP_y - o)}} \qquad \text{Eq. (A.15)}$$

$$DW_y = 10^{DSW_y} \cdot TE^{y-y_{base}} \qquad \text{Eq. (A.16)}$$

Where DSW is the yearly structural domestic withdrawal [log10 m^3 cap^{-1}], DW the yearly domestic withdrawal [m^3 cap^{-1}], DW_{min} the minimum structural domestic withdrawal [log10 m^3 cap^{-1}], DW_{max} the maximum structural domestic withdrawal (without technological improvement) [log10 m^3 cap^{-1}], GDP the yearly gross domestic product [log10 USD$_{equivalent}$ cap^{-1}], f [-] and o [log10 USD$_{equivalent}$] the parameters that determine the range and steepness of the sigmoid curve, y the year index, TE the technological efficiency rate [-], and y_{base} the base year (taken to be 1980).

DW_{min} was set at 7.5 l cap^{-1} d^{-1} based on the World Health Organisation standard (Reed and Reed, 2013), DW_{max} was estimated at around 450 l cap^{-1} y^{-1} based on a global curve fit, and TE was set at 0.995, 0.99, and 0.98 for developing, transition and developed countries respectively (United Nations development status classification) based on Flörke et al. (2013). Curve parameters f and o were estimated for the 23 United Nations sub-regions based on the GDP per capita and domestic water withdrawal data. In case insufficient data was available to calculate parameters values, regional (4 sub-regions) or global (5 sub-regions) parameter estimates were used.

7.4.4 Industrial sector

Industrial water demands were represented by using a linear formula for the calculation of structural industrial water demands (Eq. A.17) and a efficiency rate for the calculation of water-use efficiency increases (Eq. A.18).

$$ISW_y = ISW_{int} \cdot GVA_y \qquad \text{Eq. (A.17)}$$

$$IW_y = \mathrm{ISW_y} \cdot TE^{y-y_{base}} \qquad \text{Eq. (A.18)}$$

Where *ISW* is the yearly structural industrial withdrawal [m^3], IW_{int} the country specific industrial water intensity [m $USD_{equivalent}^{-1}$], *IW* the yearly industrial withdrawal [m^3], *GVA* the yearly gross value added by industry [$USD_{equivalent}$], *y* the year index, y_{base} the base year (taken to be the year when the industrial water intensity is determined), and *TE* the technological efficiency rate [-].

TE was set at 0.976 and 1 for OECD and non-OECD countries respectively before the year 1980, 0.976 between the years 1980 and 2000 and 0.99 after the year 2000 based on Flörke et al. (2013). Industrial water intensities were estimated for the 246 United Nations countries based on the GVA and industrial water withdrawal data. In case insufficient data was available to calculate the industrial water intensities, either sub-regional (76 countries), regional (17 countries) or global (6 countries) intensities estimates were used.

7.4.5 Energy sector

For each thermoelectric power plant the water intensity was combined with their generation to calculate the water demands (Eq. A.19). Since there was no observed data about the actual annual generation, each plant was assumed to be running at its installed generation capacity throughout the year, similar to van Vliet et al. (2016).

$$EW_y = EW_{int} \cdot G_y \qquad \text{Eq. (A.19)}$$

Where *EW* is the yearly energy withdrawal [m^3], EW_{int} the energy water intensity [m^3 MWh^{-1}], *G* the yearly generation for each plant [MWh], and *y* the year index.

The energy water demands were subtracted from the industrial water demands at the location of each power plant. In cases where the grid cell industrial water demand was less than the energy water demand, national industrial water demands were lowered. In cases where even the national industrial water demands were less than the national energy water demand (5 countries), the energy water demands were lowered instead. This could be the case in countries where power plants do not operate at their installed capacity, as globally around 45% of the installed capacity is actually generated (based on data of van Vliet et al. (2016)). Energy demands were lowered until 10% of the national industrial water demand remains, to ensure some spatial coverage of industrial and energy water demands.

7.4.6 Livestock sector

Livestock water demands were estimated by combining the livestock population with the water requirements for each livestock variety (Eq. A.20).

$$LW_y = LW_{int} \cdot L \qquad \text{Eq. (A.20)}$$

Where LW is the yearly livestock withdrawal [m^3], LW_{int} the livestock water intensity [m^3 livestock^{-1}], L the livestock number for each variety [livestock].

8 Author contribution

Bram Droppers and Wietse H.P. Franssen developed and tested the model additions introduced in VIC-WUR. Bram Droppers generated and analysed the results. Michelle T.H. van Vliet, Bart Nijssen and Fulco Ludwig provided overall oversight and guidance. Bram Droppers prepared the manuscript with contributions from all co-authors.

9 Competing interests

The authors declare that they have no conflict of interest.

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