UNIVERSIDADE DE LISBOA
INSTITUTO SUPERIOR TÉCNICO

LECTURE

How to assess water scarcity for water management purposes

Rodrigo de Almada Cardoso Proença de Oliveira

HABILITATION IN CIVIL ENGINEERING

April 2025

Table of Contents

1 Introduction and justification

A significant portion of the world's population faces inadequate access to water, negatively affecting their health, well-being, and economic prosperity. Climate change and growing demands for water, driven by improved living standards and changing lifestyles, further intensify these challenges. Additionally, ecosystems suffer greatly from insufficient water availability.

Water shortages manifest themselves in various forms and over different time scales. To effectively address this complex challenge, it is essential to clearly define and distinguish between the concepts of drought, water scarcity, water stress and aridity.

Aridity is defined as long term lack of available, life-sustainable moisture in terrestrial ecosystems (UNCCD, 2024).

A drought event is defined as a period when a climate variable, namely precipitation, deviates from a reference condition which may lead to a temporary water shortage. The key aspects highlighted in this definition is the temporary duration and the natural cause of the drought.

Water scarcity refers to a prolonged condition resulting from excessive water use relative to its availability. Unlike drought, it is a persistent, human-induced issue that arises due to overconsumption, limited natural water resources, or inadequate infrastructure and institutional frameworks that fail to meet existing consumptive and non-consumptive demands. Within this context, it is useful to differentiate between absolute (or physical) scarcity, where natural water resources are insufficient to satisfy water demands, and economic scarcity, which stems from inadequate infrastructure, insufficient institutional and legal structures, or a shortage of skilled human resources required for effective water resource management.

Water scarcity may reveal itself when rivers, reservoirs and aquifers contain insufficient water of adequate quality to satisfy existing water needs, but also when the water in the soil is unable to support crop growth. This distinction highlights the concepts of blue water—referring to surface water and groundwater resources—and green water, which denotes water stored in the soil that is available for plant uptake. It also emphasizes that water scarcity may arise from contamination and inadequate sanitation infrastructures.

Water stress is a more inclusive and broader concept that refers to the inability to meet human and ecological demand for water. It considers several physical aspects related to water resources, including water scarcity, but also water quality, environmental flows, and the accessibility of water (Schulte and Morrison, 2014). The primary distinction between water stress and water scarcity lies in the explicit consideration of ecosystem requirements in the definition of water stress. However, these two terms are frequently used interchangeably.

Many river basins worldwide are increasingly facing water scarcity driven by climate variability and human-induced pressures, particularly due to rising water demands across municipal, industrial, and agricultural sectors. By altering alter precipitation and streamflow patterns, climate change is anticipated to further amplify uncertainty in water availability, particularly in regions already experiencing water scarcity (Arnell et al., 2011; IPCC, 2022). Consequently, water security challenges have become a critical concern, with freshwater scarcity impacting not only the environment but also the social and economic sectors reliant on these essential resources. The water crisis has been ranked as one of the top five global risks that may affect the world's social, economic, and environmental welfare (Mekonnen and Hoekstra, 2016; WEF, 2020).

This lecture investigates methodologies for accurately assessing water scarcity, emphasizing the importance of accounting the role reservoirs play in mitigating water scarcity, especially in highly regulated river basins characterized by Mediterranean climates.

The lecture concludes by examining water management strategies aimed at addressing water scarcity as both a global and local challenge, with particular focus on the role of reservoirs. Reservoirs have long been central to water scarcity mitigation, especially in Mediterranean climates where agriculture heavily depends on irrigation. These regions frequently experience prolonged dry seasons, making reliable water storage crucial for sustaining crop production. While reservoirs offer substantial support in these contexts, it is also important to recognize their limitations, including environmental trade-offs and long-term sustainability concerns.

2 How to evaluate water scarcity

2.1 Drought indices and water scarcity indices

Aridity is usually quantified by the *Aridity Index* (AI), defined as the ratio between long term averages of precipitation and potential evapotranspiration. A study by UNCCD concluded that between 1961-1990 and 1991-2020 drylands, defined as those having an AI less than 0.65, have expanded 4.3 million km^2 and reached 40% of global land (excluding Antarctica) (Figure 1). People living in drylands doubled in number, reaching 2.3 billion (more than a quarter of global population). Climate models suggest that as many as 5 billion people could inhabit dryland by 2100 (UNCCD, 2024). This trend primarily resulted from rising air temperatures and increased variability in precipitation, leading to elevated evapotranspiration rates and reduced infiltration and soil moisture. Additionally, higher atmospheric CO_2 concentrations contributed by altering stomatal function in plants.

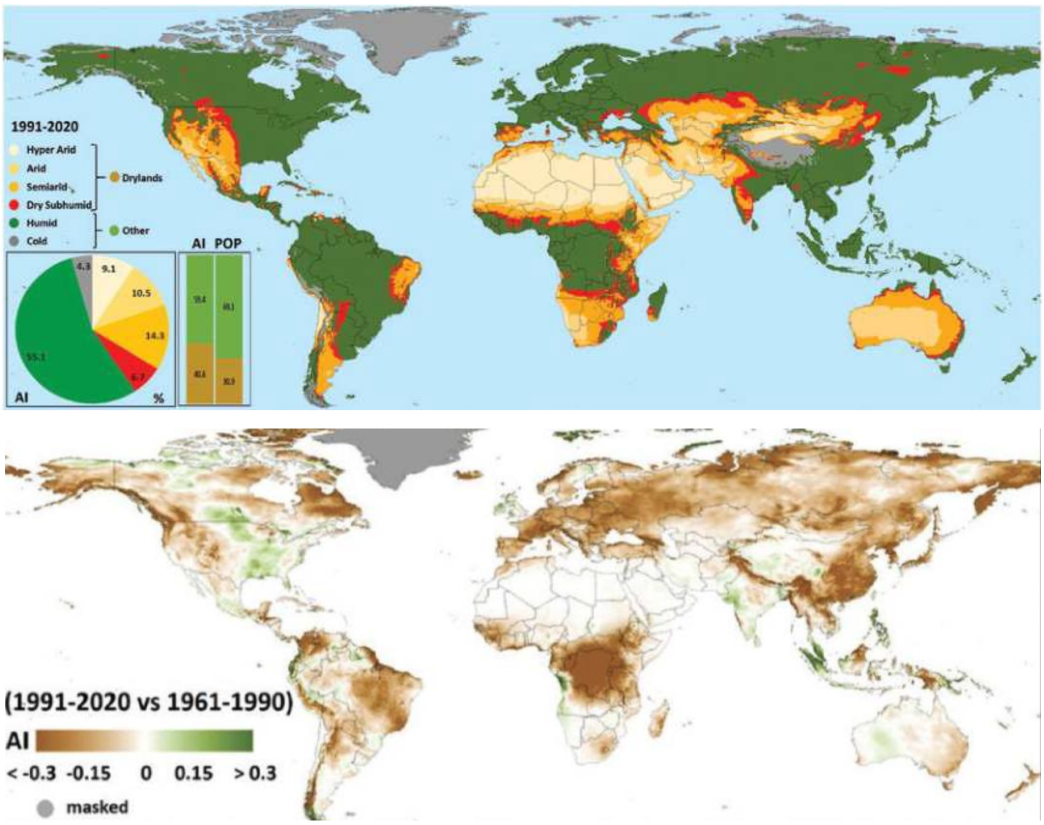

Figure 1 - Aridity Index (AI) for 1991-2020 (top) and difference of Aridity Index (AI) over 1961-1990 and 1991-2020 (bottom) (source: UNCCD, 2024)

Drought indexes use precipitation and other meteorological variables to evaluate the frequency, severity, and duration of drought events defined as periods when the climate variable, namely precipitation, deviates from a reference condition (Mishra and Singh, 2010). Among the drought indexes, the most widely used are the Palmer Drought Severity Index (PDSI) (Palmer, 1965), the Standardized Precipitation Index (SPI) (Mckee et al., 1993), and the Standardized Precipitation Evapotranspiration Index (SPEI) (Vicente-Serrano et al., 2010). Drought indexes based on runoff estimates have also been proposed by Shukla and Wood (2008), Wu et al. (2018) and Zou et al. (2018).

Drought affects 55 million people annually, with the low to middle income countries being more vulnerable. Figure 2 show the major drought events in 2022 – 2024.

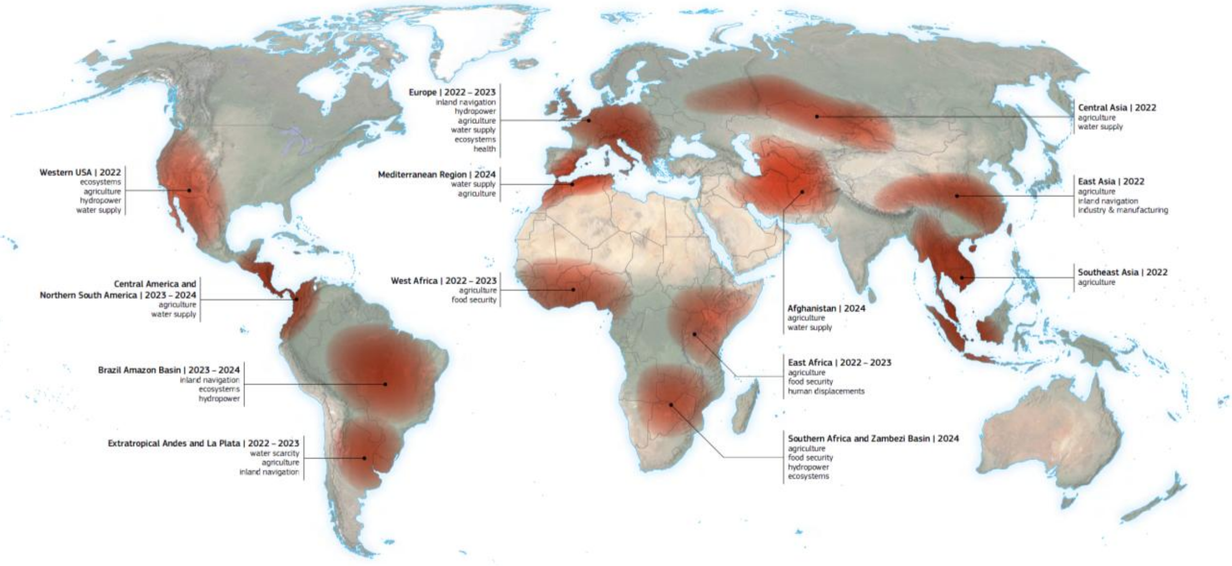

Figure 2 – Major drought events in 2022 – 2024 (source: UNCCD, 2024)

Finally, water scarcity indexes aim to assess the level of human-induced quantitative pressures on the river basin water resources, usually by comparing water demands or withdrawals with water availability. These indexes explicitly include estimates of water withdrawals or demands on their definition or include proxy variables, such as the population data. Most of the times, water availability is quantified by a long-term annual average of total runoff in natural conditions, to consider both surface and groundwater resources.

A complete review of several water scarcity indexes can be found in Rijsberman (2006), Pedro-Monzonís et al. (2015), Damkjaer and Taylor (2017), Liu et al. (2017) and Hussain et al. (2022).

2.2 Water Stress Index

The most well know water scarcity index is the Water Stress Index, proposed by Falkenmark et al. (1989), which measures the amount of water available in a watershed per capita. It is determined by the ratio between the long-term mean annual water availability (AWA^i) and the population (Pop^i) of watershed i (Equation 1).

$$WSI^i = \frac{AWA^i}{Pop^i} \tag{1}$$

AWA is defined as the long-term average of total annual flow (surface and groundwater flows), under natural conditions. In a hydrological year, the total annual flow is expected to be the streamflow annual volume, as the variation of groundwater storage is assumed to be small. As such AWA, may be determined from monitoring records of streamflow or from the results of a hydrological model.

The index aims at evaluating the capacity to supply basic human needs, such as drinking water, sanitary services, and food preparation. The authors proposed threshold values of 1 700 m³, 1 000 m³ and 500 m³ per capita per year, below which a country experiences water stress, scarcity and absolute scarcity, respectively (Rijsberman, 2006). The use of the expression "water stress" does not fully follow the above-mentioned distinction between water scarcity and water stress.

Figure 3 shows a global map of water stress measured by the Water Stress Index and Figure 4 presents the values computed for Europe by Eurostat. Both highlight the limitation of the index, with Portugal being a good example. The country is positioned high in the Eurostat table, above Netherlands, France and the United Kingdom, due to its superior average annual runoff, despite existing water management challenges due to seasonality and interannual variability of water availability.

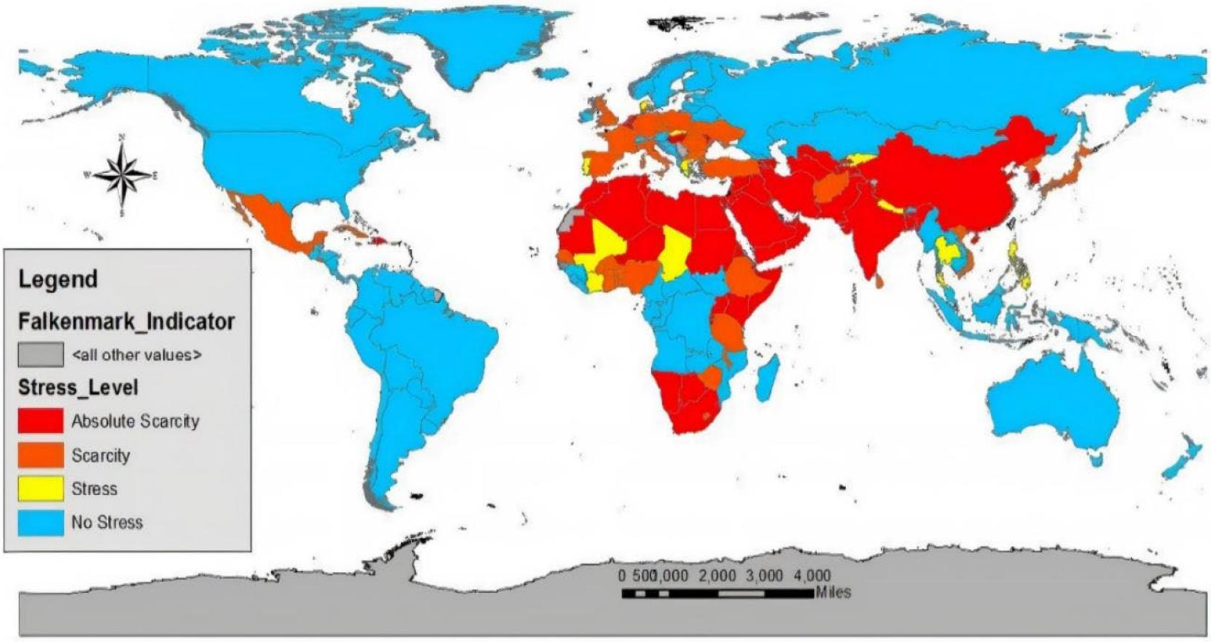

Figure 3 - Water Stress, measured by the Water Stress (or Falkenmark) index (source: University of Texas)

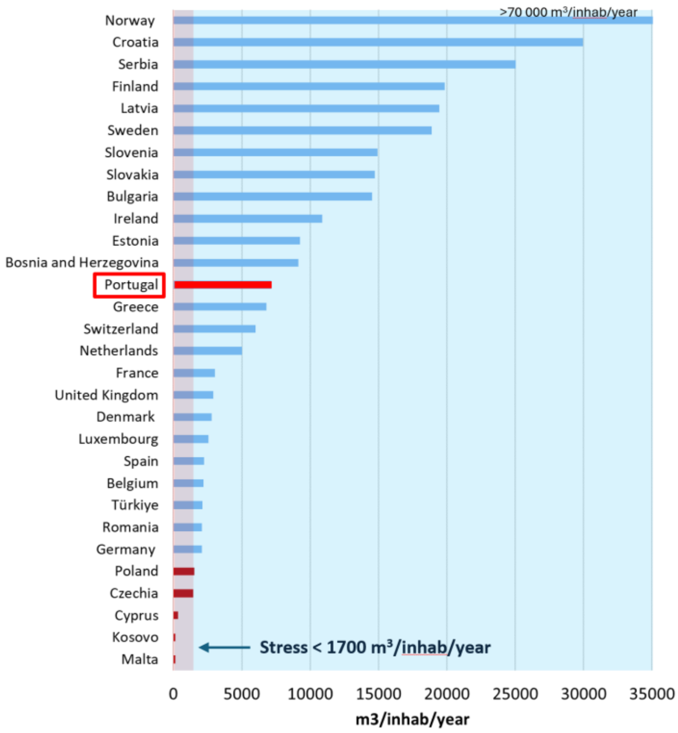

Figure 4 - Water Scarcity in Europe, measured by the Water Scarcity Index (source: Eurostat)

2.3 Criticality Ratio or the Water Exploitation Index

The Criticality Ratio (Alcamo et al., 2000) or the Water Exploitation Index *(WEI)* (EEA, 2010) is another index widely used that measures the level of water scarcity by the ratio of annual water withdrawals to annual water availability. The ratio is also known as the Withdrawal-to-Availability ratio (*WTA)* (Damkjaer and Taylor, 2017) or the Water Resources Vulnerability Index (Raskin et al., 1997; Rijsberman, 2006). It is used to measure the achievement of target 6.4 of the Sustainable Development Goals (SDG), which seeks to ensure sustainable withdrawals and supply of freshwater to address water scarcity. In that context it is known as indicator 6.4.2 or Water Stress Indicator (FAO, 2019), not to be confused with the Water Stress Index, from Falkenmark (1989).

The Water Exploitation Index is defined as the ratio of average annual water withdrawals for agriculture, industry and urban uses ($\bar{A}^i$) to annual water availability (Alcamo et al., 2000) (Eq.2).

$$WEI^i = \frac{\bar{A}^i}{AWA^i} \qquad (2)$$

Higher values of *WEI* indicate more water uses in a river basin and more shortages during low flow periods. The authors define severe levels of water stress when WEI > 0.4. A discussion surrounding threshold values persist since some experts believe that water resources can be used with WEI up to 0.6. Other experts say that the sustainability of water resources can only be achieved with WEI< 0.2 (Alcamo et al., 2000).

Figure 5 presents the global map of water stress obtained in the SDG indicator 6.4.2 portal powered by UN-WATER. The map shows annual average values by country, which hides sub-national, inter-annual and seasonal variability.

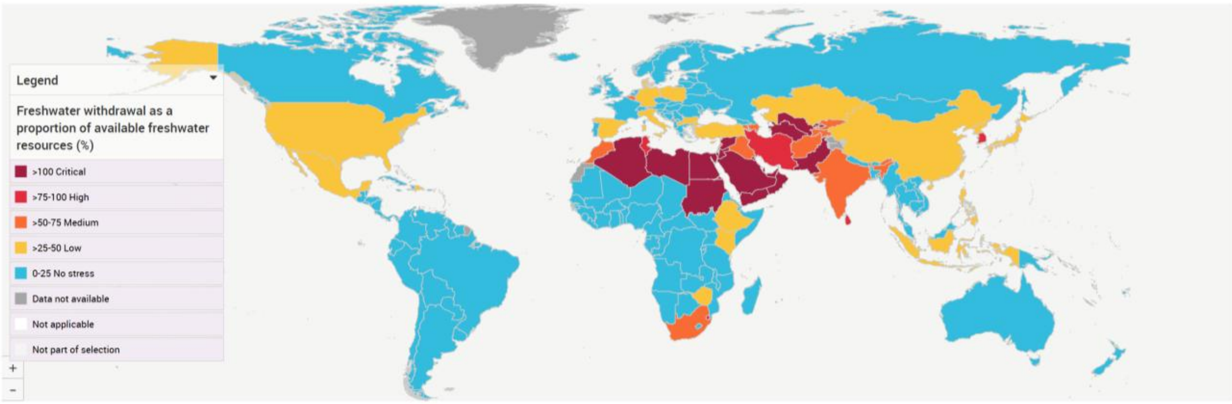

Figure 5 – Level of water stress, measured by the WEI index, aka indicator SDG 6.4.2 or Water Stress Indicator (source: UN-WATER)

Mekonnen and Hoekstra (2016) have computed global water scarcity on a monthly basis at the level of grid cells of 30 × 30 arc min (Figure 6). They have found that two thirds of the global population (4 billion people) experience severe water scarcity at least one month per year and that half a billion people face severe water scarcity all year round.

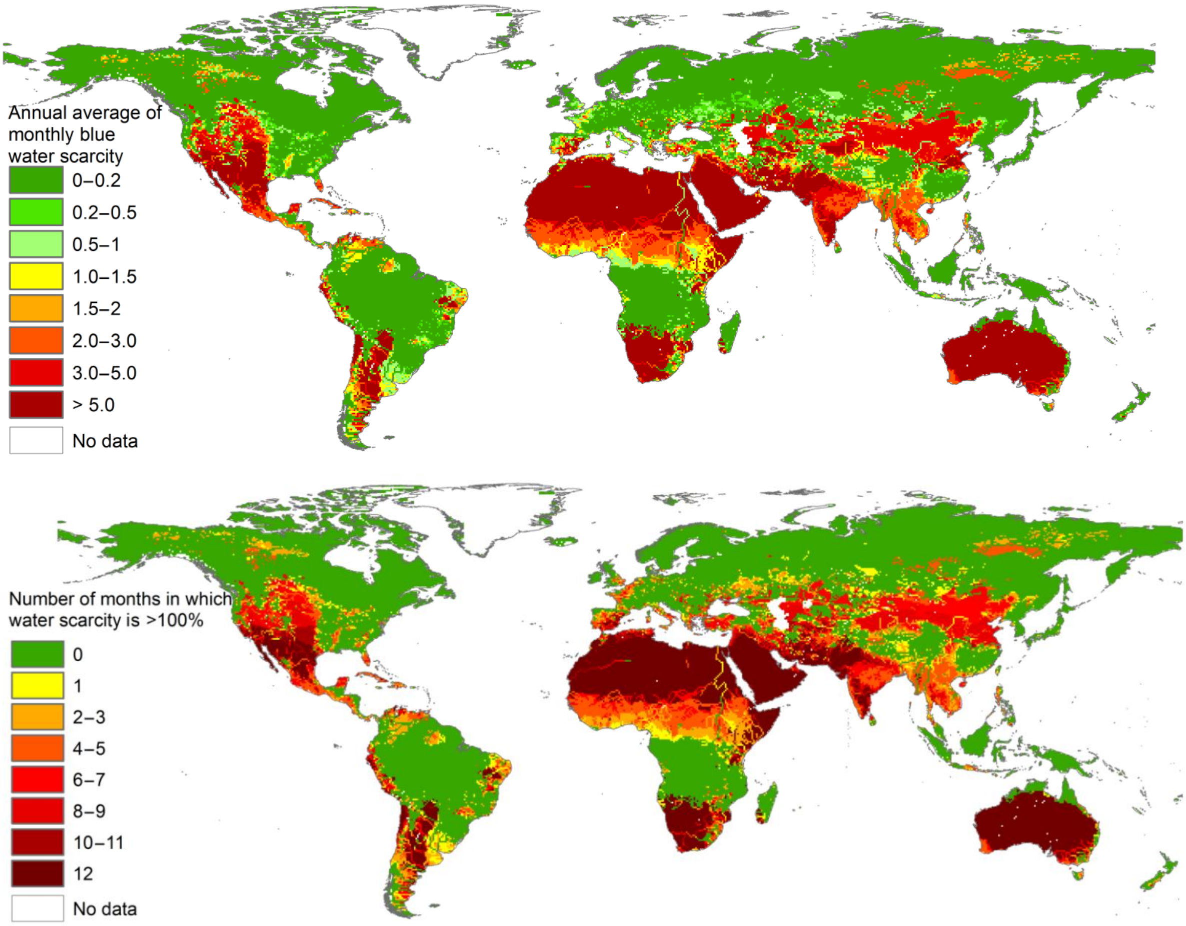

Figure 6 - Annual average monthly blue water scarcity (top) and number of months in which water scarcity is > 100% (bottom), both computed with a 30 × 30 arc min resolution for 1996–2005 period (source: Mekonnen and Hoeskstra, 2016)

When calculating both the Water Stress Index (WSI) and the Water Exploitation Index (WEI), the long-term mean annual streamflow is typically used as a proxy for average annual water availability. However, this approach presents challenges in cases where national boundaries differ significantly from river basin limits, or where substantial man-made water transfers occur. In those cases, there is a need to include in the analysis transfers to external basins as withdrawals from donor basin, and resources originating from external basins as water sources of the receiving basin.

Consider the river basin depicted in Figure 7, with sub-basin 1 located upstream of sub-basin 2. P_t^i and ET_t^i are the precipitation and the evapotranspiration occurring in the sub-basin i, respectively, which determine the total flow generated in the basins in a month t, under natural conditions. Q_t^1 is the streamflow from sub-basin 1 into sub-basin 2, and Q_t^2 is the outflow from sub-basin 2. The values of Q_t^1 and Q_t^2 are dependent on the natural flows generated in these sub-basins, and (T_t^i) represents possible water transfers to or from the sub-basins. A_t^i is the sum of water abstractions, in sub-basin *i* in month *t*, to urban, industrial and agriculture uses, from surface and groundwater sources. R_t^i is the total return from these uses. ΔS_t^i includes the changes of water stored in sub-basin *i*, both underground (soil moisture and groundwater) and over the surface, *e.g,* lakes and artificial reservoirs.

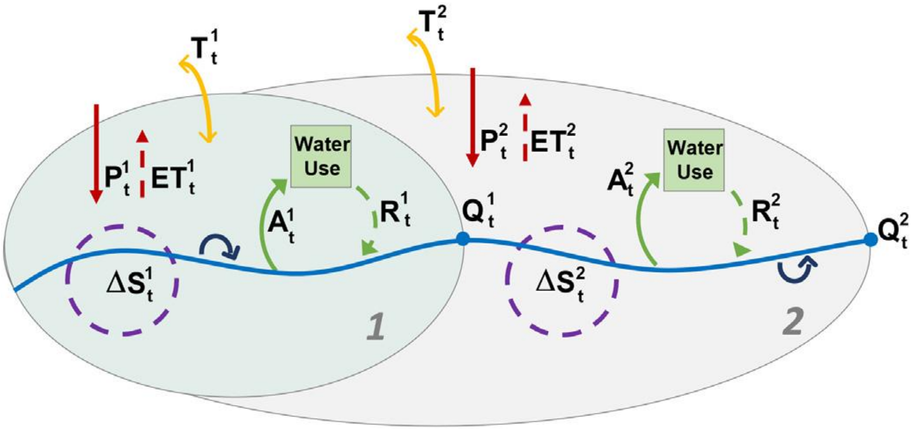

Figure 7 – The scheme of a highly stressed river basin system with fresh water and Groundwater (source: Sonderman and Oliveira, 2022a)

Equations 3 and 4 show how to calculate *AWA* for upstream and downstream river basins, respectively. If transfers from external river basins are negligible, AWA of the upstream river basin is equal to the average annual natural streamflow, $\bar{Q}_{Nat}$, which corresponds to the difference between average annual precipitation ($\bar{P}$) and average annual evapotranspiration ($\overline{ET}$), as the change of groundwater storage (soil moisture and aquifer storage) ($\overline{\Delta S}_{Nat}$) is assumed to be zero (Eq.3). *AWA* of the downstream river basin is the average annual natural streamflow at the outlet, deducted by withdrawals at the upstream river basin (FAO, 2019) (Eq.4).

$$AWA^1 = \bar{Q}_{Nat}^1 = \bar{P}^1 - \overline{ET}^1 - \overline{\Delta S}_{Nat}^1 = \bar{P}^1 - \overline{ET}^1 \tag{3}$$

$$AWA^2 = \bar{Q}_{Nat}^1 + \bar{Q}_{Nat}^2 - \bar{A}^1 = \bar{Q}_{Nat}^1 - \bar{A}^1 + \bar{P}^2 - \overline{ET}^2 \tag{4}$$

If needed, equations 3 and 4 may be modified to include in the estimates of *AWA* transfers from external river basins and in the case of transboundary river basins any transfers from upstream basins defined in existing treaties (FAO, 2019).

Equations 5 and 6 are used to determine RWR_t^i in upstream and downstream sub-basins, respectively basins 1 and 2 in Figure 1. In each month t, changes in the stored amount of water ΔS_t^i in surface systems (artificial reservoirs) and subsurface systems (aquifers) must be added to or deducted from the available water resources. A positive value of ΔS_t^i represents an increase of water storage in the basin, which is not available to sat isfy demands in month t. A negative value represents the use of water stored in previous months to satisfy demands in month t. Q_t^1 and Q_t^2 are the stream flows occurring in month t at the outlets of river basins 1 and 2, respectively, contingent on the water use and water storage at both basins.

$$RWR_t^1 = P_t^1 - ET_t^1 - \Delta S_t^1 + T_t^1 = Q_t^1 + (A_t^1 - R_t^1) \tag{5}$$

$$RWR_t^2 = P_t^1 - ET_t^1 - \Delta S_t^1 + T_t^1 - (A_t^1 - R_t^1) + P_t^2 - ET_t^2 - \Delta S_t^2 + T_t^2 = Q_t^2 + (A_t^2 - R_t^2) \tag{6}$$

The determination of RWR_t^i through Equations 5 and 6 may be achieved in two ways. If long and reliable streamflow records of Q_t^i exist, RWR_t^i can be estimated by adding $\left(A_t^i - R_t^i\right)$ to Q_t^i. Alternatively, estimates of P_t^i, ET_t^i, T_t^i and ΔS_t^i are needed. These estimates can be obtained from mathematical models that simulate the river basin hydrological processes, including groundwater flow and storage, surface water storage, allocation and use. These simulation capabilities are offered by some integrated models that encompass hydrological processes and water management processes, or by the joint use of a rainfall-runoff model and a water management and allocation model.

When applying equations 5 and 6 to evaluate available water resources at country level, the expressions internal and external water resources appear. The glossary of FAO defines renewable resources as the total resources that are offered by the average annual natural inflow and runoff that feed each hydro system (catchment area or aquifer). Internal renewable water resources (IRWR) comprise the average annual flow of rivers and groundwater generated from endogenous precipitation, whereas external renewable water resources (ERWR) are defined as the part of the country's renewable water resources which is not generated in the country. The ERWR include inflows from upstream countries (groundwater and surface water), and part of the water of border lakes or rivers. Non-renewable water resources are groundwater bodies (deep aquifers) that have a negligible rate of recharge on the human timescale and thus can be considered non-renewable.

2.4 Water Exploitation Index +

The Water Stress Index (WSI) and Water Exploitation Index (WEI) do not consider flow variability, seasonality, nor the regulating capacity of reservoirs. Consequently, they are inadequate for assessing water scarcity in regions with a Mediterranean climate, such as Southern Europe.

WEI+, an improved version of WEI, addresses these issues. It was proposed by the Working Group on Water Scarcity and Droughts of the Common Implementation Strategy for the Water Framework Directive to evaluate water stress conditions in Europe (EEA, 2019).

The WEI+ quantifies the renewable water resources and the water use for consumptive uses at a monthly time step to better capture water-scarce periods when abstractions surpass water resources availability (Bariamis et al., 2016; Zal et al., 2017). It also assumes regulated conditions, instead of natural conditions.

Instead of water abstractions, $WEI+$ considers net consumption for consumptive uses, determined by deduction return flows, R_t^i, from water abstractions, A_t^i. The use of net water consumption, instead of water abstraction (gross values), provides a more accurate and less pessimistic view of water scarcity, as net abstractions correspond to a percentage of total water abstraction, which can be as low as 10-20% for urban uses (Pérez-Blanco et al., 2020). The non-consumed fractions of water abstraction contribute to the satisfaction of downstream uses.

WEI+ also considers ecosystem needs by deducting environmental flow requirements, $Qenv_t^i$, from the renewable water resources, RWR_t^i, as this volume is not available for consumptive uses.

Equation 5 is used to determine the value of WEI+ at sub-basin *i*, in month *t*.

$$WEI+_t^i = \frac{A_t^i - R_t^i}{RWR_t^i - Qenv_t^i} \tag{8}$$

The WEI+ value varies between 0 and 1. A non-stressed period is defined when the WEI+ value is below 20%; a severe water stress period occurs when WEI+ value exceeds 40%; and a highly stressed period occurs when WEI+ value is above 60% (Faergemann, 2012).

Water abstractions for consumptive uses, A_t^i, includes water abstraction for cooling of power plants but not water used for hydropower generation (Faergemann, 2012). However, as water allocation for hydropower generation and other non-consumptive uses affects streamflow and water storage in reservoirs, hydropower generation is indirectly included in WEI+ computations.

The variation of stored volume in reservoirs considers evaporation losses as a volume that is not available to meet water demands. The variation of stored volume groundwater system results from the water balance of aquifer recharge, pumping and river-aquifer transfers, with the latter being positive or negative depending on the hydraulic conditions of the river and adjacent aquifers.

Decisions on the modelling approach may influence the WEI+ computations, especially in what concerns the simulation of groundwater flow and the river-aquifer interaction. An aquifer can be considered as the underground unit of the surface system sub-basin, with no water transfers to other surface or underground units, or as individual underground units, hydraulically connected to various rivers of multiples surface sub-basins. The first approach does not respect the unity of individual aquifer systems, but the model results provide $\Delta S_{Aquifer}$ for each sub-basin, which can be used in the calculation of WEI+. The second approach respects this individuality and can properly reproduce complex river-aquifer systems. However, it is not possible to obtain $\Delta S_{Aquifer}$ for each sub-basin. RWR values must be determined from Q_t^i. and $\left(A_t^i - R_t^i\right)$.

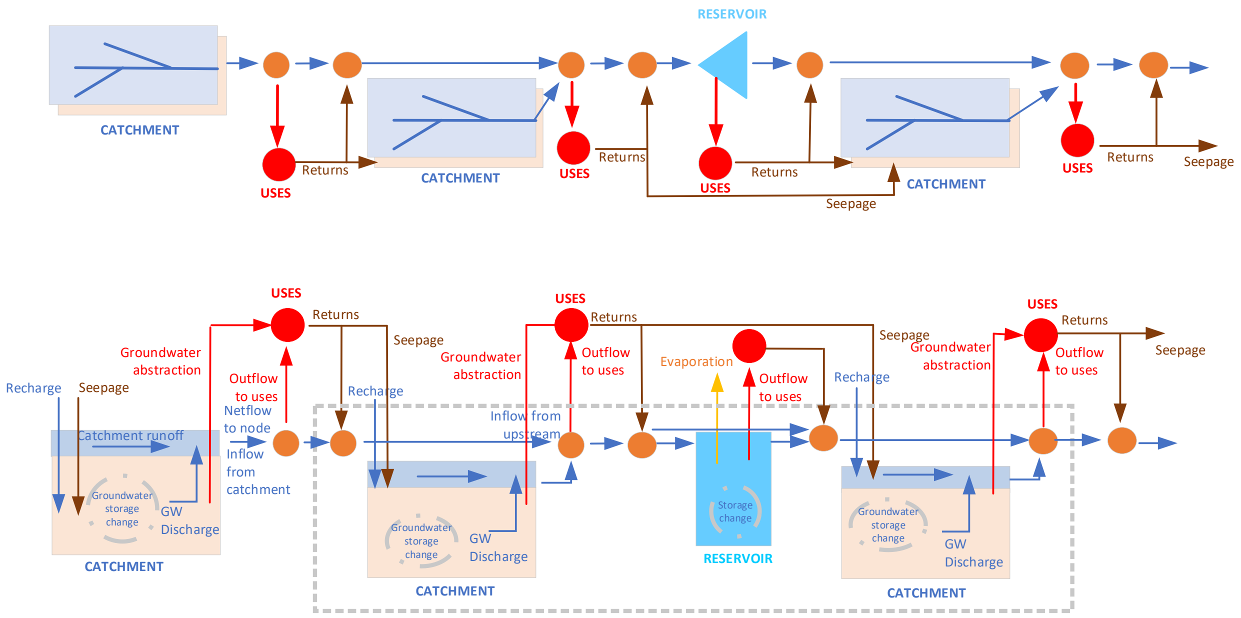

Figure 8 – Illustration of the water balance

Calculating WEI+ values is a complex task, and relatively few studies have provided detailed applications of this indicator. Initially proposed by the European Commission and the European Environment Agency (EEA), the WEI+ index is predominantly applied within Europe but is gradually gaining attention globally. The European Commission and the EEA have been working to use the WISE Water Accounts database and the LISFLOOD hydrological model in assessing WEI+ for various European river basins (Figure 9). Several researchers have applied WEI+ across major European river basins. Bariamis et al. (2016), de Roo et al. (2012), and Zal et al. (2017) have estimated WEI+ values for Europe's larger river basins. Bisselink et al. (2018) and Karabulut et al. (2016) specifically applied WEI+ assessments to the Danube River basin. Beyond Europe, Sun and Zhou (2020) employed WEI+ to evaluate water scarcity in China's Jinghe River basin. Additionally, alternative and simplified methods for estimating WEI+ were proposed by Casadei et al. (2020) for the Upper Tiber River basin and by De Roo et al. (2016) for the Sava River basin.

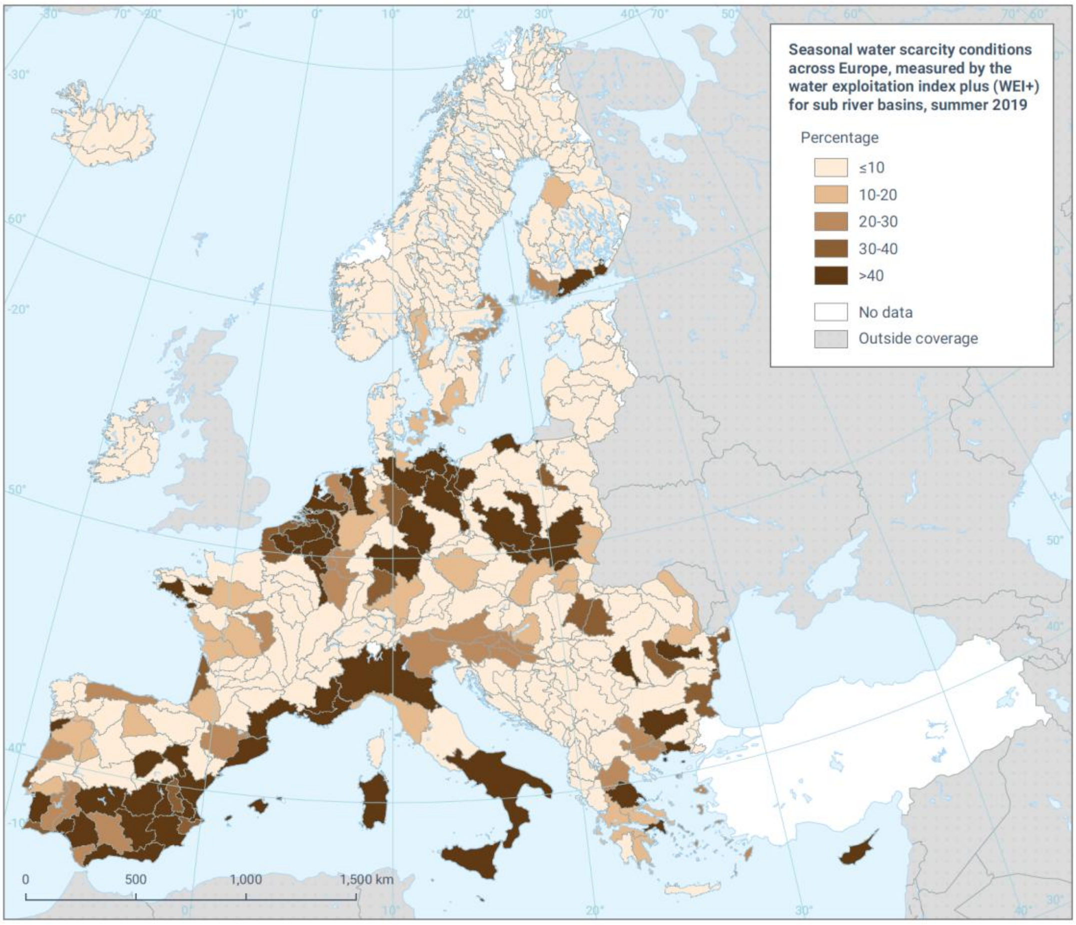

Figure 9 – WEI+ in the summer of 2019 (EEA, 2024)

Applications of WEI+ in river basins characterized by Mediterranean climates remain rare. Noted examples include studies by Contreras and Hunink (2015) on the Segura River basin in Spain and by Sondermann and Oliveira (2022) on the transboundary Tagus River basin.

The substantial data requirements of WEI+ computations require the adoption of mathematical models to supplement and enhance ground-based and satellite monitoring data. Hydrological models estimate natural water flows, including evapotranspiration, groundwater recharge, and streamflow. In parallel, water allocation and management models provide insights into reservoir storage volumes, water withdrawals, consumption, and return flows, offering a comprehensive perspective on water distribution and usage.

Decisions on the modelling approach may influence the WEI+ computations, especially in what concerns the simulation of groundwater flow and the river-aquifer interaction.

2.5 Economic and social water scarcity indexes

In addition to the above-mentioned indicators that focus on absolute or physical water scarcity, additional indexes were developed to capture the impacts of water capacity and the societal capacity to deal with them. The Social Water Scarcity Index (Ohlsson, 2000) and the Water Poverty Index (Sullivan, 2002) are examples of these indexes.

Ohlsson (2000) proposed the Social Water Scarcity Index (SWSI) by dividing the Water Stress Index (WSI)—which measures per capita water availability—by the Human Development Index (HDI), an indicator of a country's achievements in the fundamental dimensions of human development: longevity, education, and standard of living. The SWSI thus highlights societal capacity to effectively manage and respond to water scarcity challenges.

The Water Poverty Index (WPI) is a composite indicator calculated as a weighted sum of five components: resources, access, capacity, use, and environment. Collectively, these components evaluate water availability, accessibility, the capacity to effectively manage and use water, and the environmental condition of water bodies. Analysing WPI results enables differentiation between countries that, despite having limited water resources, possess robust institutional frameworks and technological capacities for efficient water management, and those with abundant water resources yet inadequate means or strategies to utilize them sustainably.

$$WPI = \frac{w_R \cdot Resources + w_A \cdot Access + w_C \cdot Capacity + w_U \cdot Use + w_E \cdot Environment}{w_R + w_A + w_C + w_U + w_E} \quad (9)$$

Figure 10 shows the Water Poverty Index computed by Lawrence et al (2002). There is a significant amount of work and detail in computing each of the components of these values and in weighting them, but it is interesting to note the insight that the Water Poverty Index provide. Compare two extreme examples of countries: Spain, with a WPI of 0.63, despite having a resource component of 0.08, and Congo DR, with a WPI of 0.47, having a resources component of 0.12. Oman and Qatar reach a Water Poverty Index of 0.57, despite having a resource component of 0.02.

The added complexity in collecting data and the difficulty in understanding some variables and results hinders the widespread application of these economic and social water scarcity indexes.

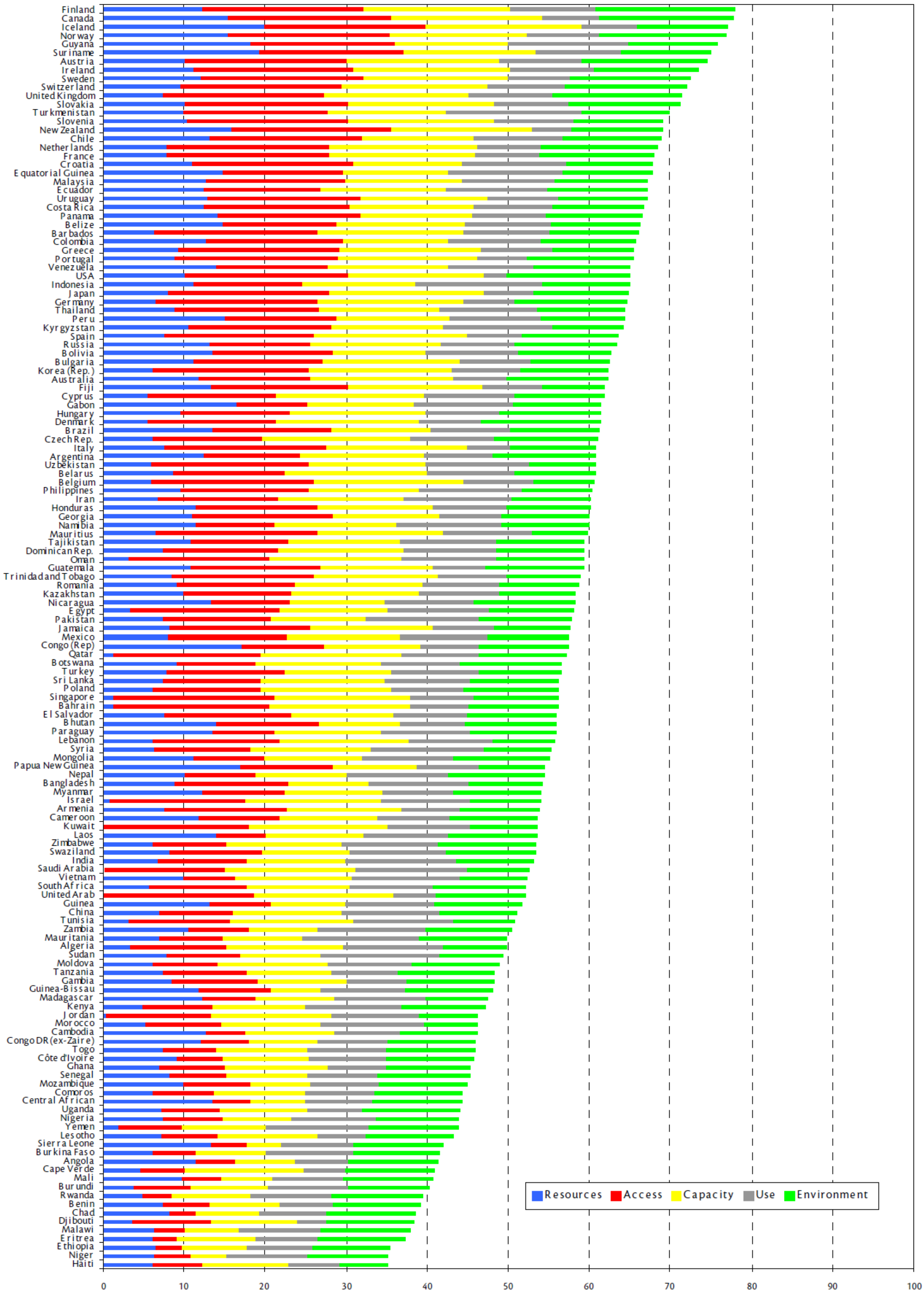

Figure 10 - The Water Poverty Index (source: Lawrence, Meigh and Sullivan 2002, The Water Poverty Index: An International Comparison, Keele Economic Research Papers).

2.6 Water accounting

Water accounting offers a streamlined and user-friendly framework for reporting water availability and allocation within a river basin. It offers a detailed view of how water is used in a country, river basin, water resource system or a specific water body and the amount that remains available for additional uses. The approach is based in a water balance framework that accounts for the water inputs and outputs within a basin, including flows associated with various water uses. Molden (1997), Molden and Sakthivadivel (2010) and Karimi et al. (2012) present the main principles of water accounting.

By consolidating and presenting water resource data in a structured and organized manner, it facilitates a comprehensive understanding of how water is distributed and utilized across various sectors and regions, which is valuable for identifying critical management challenges such as water scarcity, inefficiencies, and imbalances in allocation and to support informed decision-making, especially in basins already experiencing water stress (Momblanch et al., 2014). If the water accounting system also includes economic data, it is possible to assess water productivity and the water footprint of services and products (Craswell et al., 2007).

To effectively represent a river basin water balance, water accounting must ensure completeness, consistency, transparency, and relevance. A thorough evaluation of all components of the natural water cycle—including precipitation, evapotranspiration, surface water, and groundwater—is essential. Additionally, human-influenced flows such as reservoir storage, water withdrawals, and allocations must be included. This comprehensive assessment should achieve precise closure of the water balance and integrate both quantitative and qualitative analyses.

Water accounting methodologies should utilize consistent and standardized approaches with clearly documented data sources, enabling comparability across different regions and time periods. Molden and Sakthivadivel (2010) and Perry (2010) highlight the importance of using accurate and consistent terminology to facilitate cross-sectoral understanding of water flows and water balances, laying the groundwork for consensus-driven solutions in integrated river basin management.

Incorporating verification mechanisms strengthens the credibility and reliability of the data. Lastly, the outputs must be relevant and actionable, directly supporting decision-makers and aligning with governance frameworks. This approach ensures that the needs of diverse water users—such as agriculture, industry, ecosystems, and domestic sectors—are effectively addressed.

Multiple water accounting methodologies have been developed by national and international organizations, each emphasizing different components of water accounting. Delavar et al. (2022); Karimi et al. (2013); Momblanch et al. (2018) review some of the available water account methodologies. Much of the literature focuses on water use in agriculture, a natural emphasis given that it is the largest water-consuming sector and involves complex flow patterns which challenge some concepts such as efficiency and productivity (Perry, 2010). In contrast, Lyu et al. (2023) discuss the application of these methodologies in analysing industrial water use. Some frameworks also integrate economic data, allowing for assessments of water productivity and the water footprint of goods and services (Craswell et al., 2007). This diversity offers flexibility to accommodate different analytical objectives and data availability contexts but can compromise standardization and the sharing of concepts.

The first widespread water accounting framework is the *System of Environmental-Economic Accounting for Water (SEEA-Water)*, proposed by the United Nations as a subset of its environmental accounting systems in the *Handbook of National Accounting: Integrated Environmental and Economic Accounting* (UN, 2012). SEEA-Water provides a robust structure for integrating hydrological and economic data, enabling comprehensive assessments of water supply, demand, and usage by human activities. It facilitates cross-country and temporal comparability, making it a valuable tool for evaluating water resource allocation and linking water management with economic planning.

The framework addresses the stock and flows of water between the environment and the economy, emphasizing water abstraction, discharge, supply, use, and reuse in economic activities. By organizing hydrological and economic data into a set of core standard tables recommended for all countries, along with supplementary tables addressing social aspects and experimental data, SEEA-Water effectively connects water resource systems with economic activities. Its flexibility allows it to be applied across diverse contexts and at various spatial scales, such as river basins or administrative regions.

However, the SEEA-Water framework also has limitations. Notably, environmental flow requirements—essential for sustaining freshwater ecosystems—are not explicitly included in the standard SEEA-Water tables. This omission reduces the framework's suitability for addressing environmental sustainability priorities without incorporating additional analyses or supplementary tables. Furthermore, because economic accounts are typically compiled at the administrative region level, the framework is less effective when applied to non-administrative boundaries, such as river basins. This constraint may limit its utility in integrated water management planning (Esen and Hein, 2020b; Momblanch et al., 2018; Pedro-Monzonís et al., 2016b; Vicente et al., 2019b, 2016b)

Different from the globally standardized SEEA-Water framework, the *Australian Water Accounting Standard (AWAS)* framework was primarily developed to support detailed water management in Australia (BoM, 2010). Currently, there are some applications in other countries, for instance in Spain and South Africa (Godfrey and Chalmers, 2012; Momblanch et al., 2014). AWAS is a governance-oriented framework that focuses on water reporting to various stakeholders, particularly users, by applying principles of financial accounting to specific needs of water resource management, emphasizing accountability and transparency. The framework includes elements like water assets, liabilities, changes in storage and water rights, and it explicitly incorporates water rights, entitlements, and liabilities, reflecting the water governance system in Australia, where water trading and allocations are key features.

Finally, the *Water Accounting (WA)* framework, developed by the International Water Management Institute (IWMI), was designed to track green water use (evapotranspiration) and to minimize errors in evaluating return flows (Molden and Sakthivadivel, 2010). An enhanced version, known as *Water Accounting Plus (WA+),* offers a simplified and user-friendly framework (Karimi et al., 2013). The WA+ methodology has been widely applied to river basins and agricultural systems in water-scarce regions, at both local and regional scales (Karimi et al., 2013; Molden and Sakthivadivel, 2010).

The WA+ framework organizes its outputs into four sheets to summarise the water balance and management situation, each one with a specific purpose. These are the resource base, the withdrawals, the evapotranspiration and the productivity sheets.

The *Resource Base Sheet* provides hydrological information on water inflows and outflows, depletion processes and available water within a domain. Figure 11 presents the scheme of the resources base sheet, proposed by Karimi et al. (2013).

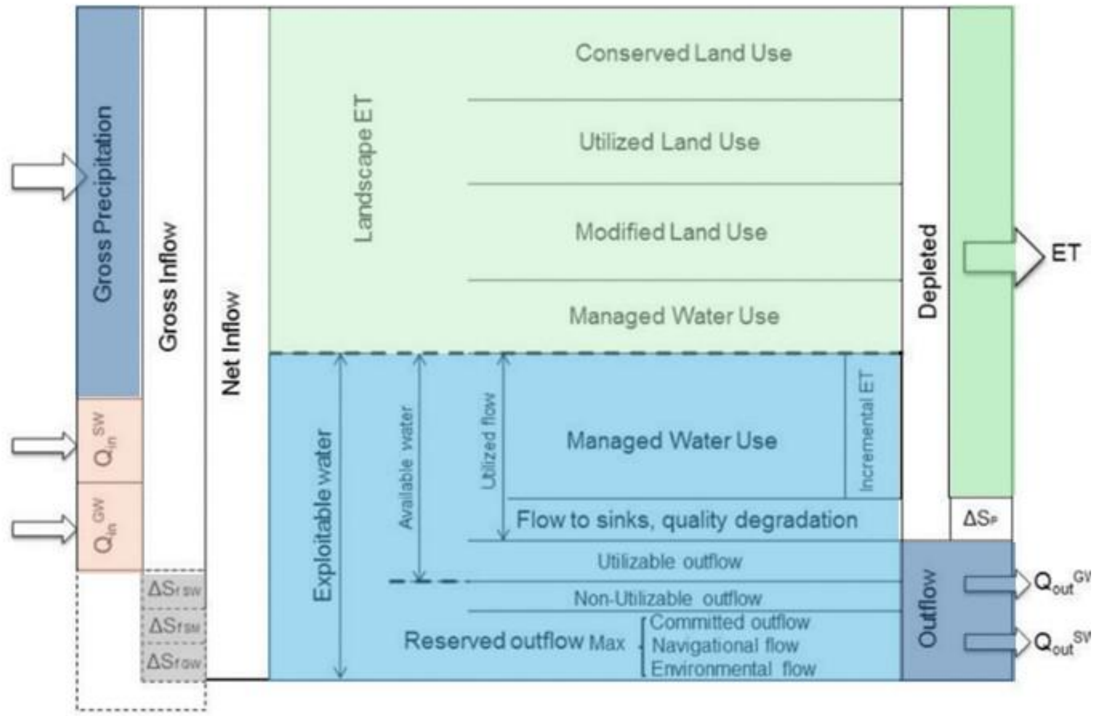

Figure 11 - Scheme of the resources sheet (Karimi et al., 2013)

Gross Inflow encompasses all incoming water to the system, primarily from precipitation which feeds surface and groundwater sources. *Net Inflow* accounts for changes in water storage, making it smaller than *Gross Inflow* when water is stored in reservoirs or aquifers, and larger when water is released from storage. *Net Inflow* is divided into two main components: *Green Water*, representing water lost to the atmosphere through landscape evapotranspiration and unavailable for further use, and *Blue Water*, which refers to the exploitable surface and ground water, whose returns after use remain accessible for downstream uses. *Water Diversions* describe the allocation of water to consumptive and non-consumptive uses, part of it returns to the system and may be further used. The part that does not return due to evaporation or to its inclusion in products, is designated *Water Depletion*. *Water Depletion* also includes contaminated water bodies and contaminated returned volumes to an extent that it is unfit for certain uses, as well as evaporation from water bodies or deep percolation to unreachable underground layers. The difference between *Net Inflow* and *Landscape Evapotranspiration* is the *Exploitable Water,* which corresponds to the volume accessible for current or future consumptive or non-consumptive uses. The water available for consumptive uses (*Available Water*) is determined by deducting from *Exploitable Water* the *Non-Utilisable Flow* (e.g., flow during flood events which is not retained in reservoirs) and the committed or *Reserved Flows* for downstream uses, hydropower, navigation, and environmental purposes. These uses must be viewed collectively as the water flowing in rivers meet different non-consumptive uses simultaneously or in sequence. *Utilisable Water r*efers to uncommitted water which is still available for additional use. Finally, *Water Outflow* encompasses both *Reserved Outflow* and *Utilisable Water*.

The *Withdrawal Sheet* describes the consumptive use of surface and groundwater sources to urban hubs, irrigation, and industries, without neglecting evaporation losses from reservoirs and return flows to surface and groundwater systems. It should be noticed that the above estimates of Withdrawal reductions correspond to average values and reliability of water demand is not evaluated in the analysis, which is an important topic in Southern Europe where often demands are not fully and always met.

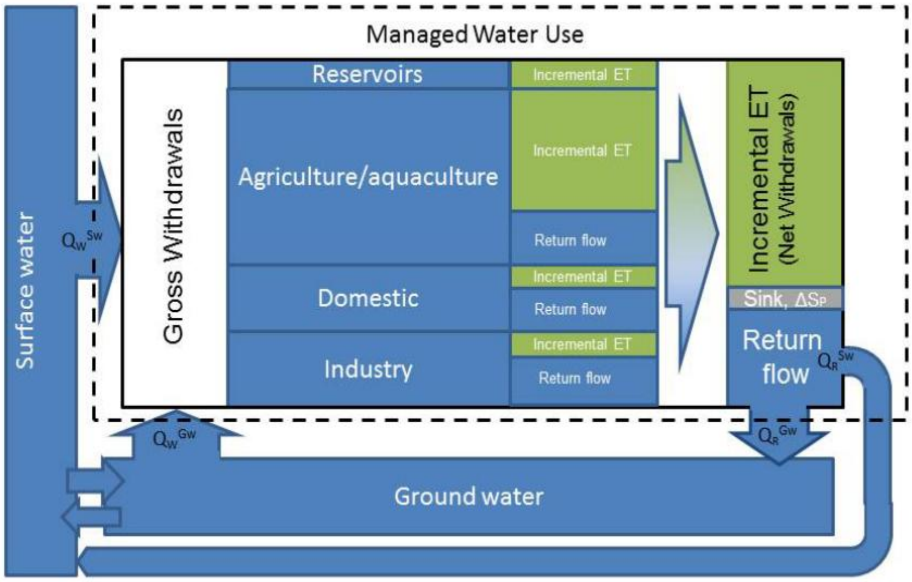

Figure 12 - Scheme of the withdrawals scheme (Karimi et al., 2013)

The *Evapotranspiration Sheet* and the *Productivity Sheet* focus on green water outflows through evaporation and transpiration and biomass production in crop lands, pastures, forests and other natural landscapes. Four land use classes are considered:

- *Conserved Land Use* - areas that remain untouched or minimally impacted by human activities
- *Utilised Land Use* - areas that provide various ecosystem services with minimal human interference
- *Modified Land Use* - land significantly altered by human activity for purposes such as food production and infrastructure development, including roads to connect populations
- *Managed Water Use* - land where water resources are actively regulated and controlled

The *Evapotranspiration Sheet* outlines the volumes related to *Interception*, *Evaporation*, and *Transpiration*, assessing whether these processes contribute to *Beneficial* or *Non-Beneficial* water uses. *Beneficial Uses* primarily include rain-fed agriculture and ecosystem support, though this classification can be subjective. It depends on factors such as local context, cultural values, economic priorities, and environmental goals.

The *Productivity Sheet* reports agricultural production per unit of water, focusing on *Biomass Services* and *Benefits*, while excluding *Water Productivity* in other sectors. Bio-physical land productivity (kg ha^{-1}) and water productivity (kg m^{-3}) for crop and pasture are evaluated for each land categories defined in WA+:

Under the **WA+ framework**, each sheet generates a set of indicators that offer detailed insights into water resource availability and allocation to different uses (Karimi et al., 2013). We hereby highlight the indicators associated with the sheets related to the overall water balance, namely the *Resource Base Sheet* and the *Withdrawal Sheet*.

The indicators for the *Resource Base Sheet* are E*xploitable Water, Storage Change, Available Water, Basin Closure* and *Reserved Outflow*. The *Exploitable Water Fraction* represents the share of net inflows that is not lost to landscape evapotranspiration during the year under analysis and therefore produces run-off and is available for use in the basin or region.

$$Exploitable\ Water_{\ Fraction} = \frac{Exploitable\ Water}{Net\ Inflow} \tag{10}$$

Storage Change Fraction represents the share of *Exploitable Water* during the year under analysis that is dependent on temporary storage in reservoirs and aquifer systems.

$$Storage\ Change_{\ Fraction} = \frac{Storage\ Change}{Exploitable\ Water} \tag{11}$$

Available Water Fraction is the share of *Exploitable Water* that is not reserved to meet water flow requirements for non-consumptive water uses.

$$Available\ Water_{\ Fraction} = \frac{Available\ Water}{Exploitable\ Water} \tag{12}$$

Basin Closure Fraction represents the share of available water that is utilised during the year under analysis by allocation to consumptive uses or depletion. Non-consumptive uses are not considered in this ratio, as they are not a part of *Available Water*. The basin or region is classified as closed basin when this Fraction is close to unity, indicating a severe level of water scarcity.

$$Basin\ closure_{\ Fraction} = \frac{Utilised\ Flow}{Available\ Water} \tag{13}$$

Finally, *Reserved Outflow Fraction* represents the share of the total outflow (surface and groundwater) that is committed to navigation and environmental flows and, therefore, cannot be withdrawn from the river network.

$$Reserved\ outflow_{\ Fraction} = \frac{Reserved\ Outflow}{Total\ Outflow} \tag{14}$$

The indicators of the *Withdrawal Sheet* are *Groundwater Withdrawal Fraction, Classical Irrigation Efficiency* and *Recoverable Fraction*. *Groundwater Withdrawal Fraction* represents the share of total withdrawals in a basin or region that are supplied from groundwater sources.

$$Groundwater\ Withdrawal_{\ Fraction} = \frac{Groundwater\ Withdrawals}{Total\ Withdrawals} \tag{15}$$

Classical Irrigation Efficiency represents the share of the total withdrawals for agriculture purposes that evaporates or transpires (*Incremental ET*) and that does not return to the system.

$$Classical\ Irrigation\ Efficiency = \frac{Incremental\ ET}{Total\ Withdrawals\ for\ Agriculture} \tag{16}$$

Finally, *Recoverable Fraction* represents the portion of all withdrawals from surface and groundwater sources that returns to the system.

$$Recoverable_{Fraction} = \frac{Return\ Flow}{Total\ Withdrawals} \tag{17}$$

The growing demand to increase water productivity and to achieve the best use of water by the various stakeholders requires a deep analysis of economic, environmental, and social benefits of both consumptive and non-consumptive water uses. By summarizing and standardizing data, water accounting serves as an important first step toward achieving this goal. However, the difficulties in obtaining consistent economic data across the different sectors and in estimating the cost and benefits of ecosystems services remain a challenge.

The Water accounting framework has notable limitations, primarily related to the use of average estimates of stocks and flows, neglecting crucial concepts such as water supply reliability, vulnerability, and resilience. These factors are particularly important in basins with a Mediterranean climate, where water availability is highly variable, and acute water scarcity is common. While water managers rely on reservoir and aquifer storage to meet demands, dry periods frequently result in demands being only partially met. Effective water management requires a more comprehensive understanding of variability and risk to ensure sustainable and reliable resource use.

Furthermore, the growing demand to increase water productivity and to achieve the best use of water by the various stakeholders requires a deep analysis of economic, environmental, and social benefits of both consumptive and non-consumptive water uses. By summarizing and standardizing data, water accounting serves as an important first step toward achieving this goal. However, the difficulties in obtaining consistent economic data across the different sectors and in estimating the cost and benefits of ecosystems services remain a challenge.

2.7 Water resource system performance indicators

The traditional concepts of risk and water supply reliability and vulnerability, which have been referenced in scientific literature since at least the early 1970s, may also be applied to assess water scarcity. Hashimoto et al (1982a, 1982b) review the concepts of reliability, resilience and vulnerability for water resource system performance evaluation. These concepts where later merged into a *Sustainability Index (SI)* and a *Water Allocation Index (WAI)* (Sandoval-Solis et al. 2011; Milano et al. 2013; Loucks and van Beek 2017).

Reliability (Rel^u) measures the probability to meet water demand. Assume that T_t^u is the target demand volume to be satisfied in month *t* at unit *u*. If X_t^u is supplied to meet the target T_t^u, the water deficit D_t^u can be computed by equation (18). When D_t^u is zero, the target is fully satisfied and a satisfactory state is achieved in month *t* at unit *u*; otherwise, a monthly failure occurs at unit *u*. To account for small numeric errors arising from the model, the target is considered satisfied when X_t^u exceeds a given percentage α of the monthly target T_t^u. This percentage is 95% for urban and industrial uses, and 90% for agriculture. Given a record of *n* years, reliability (Rel^u) is determined as the percentage of years where the deficit was null in all months (Equation 19).

$$D_t^u = \begin{cases} 0 & , \quad if\ X_t^u \geq \alpha T_t^u \\ \alpha T_t^u - X_t^u & , \quad if\ X_t^u < \alpha T_t^u \end{cases} \tag{18}$$

$$Rel^u = 1 - \frac{Number\ of\ years\ with\ at\ least\ one\ D_t^u > 0}{n} \tag{19}$$

Resilience (Resu) measures the probability of the system to recover from a failure, which is the same as the inverse of the failure average duration, in months (Equation 20).

$$Res^u = \frac{Number\ of\ occurrences\ D_t^u = 0 \mid D_{t-1}^u > 0}{Number\ of\ months\ D_{t-1}^u > 0} \tag{20}$$

Vulnerability (Vul^u) measures the average extent of the deficit volume when a failure occurs (Equation 21).

$$Vul^u = \frac{(\sum_{t=1}^{12n} D_t^u)/Number\ of\ years\ with\ at\ least\ one\ D_t^u > 0}{T^u} \tag{21}$$

The sustainability index *SI* is defined as a geometric average of *M* performance criteria, in this study reliability, resilience and vulnerability (Equation 22).

$$SI^u = (Rel^u * Res^u * (1 - Vul^u))^{1/3} \tag{22}$$

From the sustainability values (SI^u) determined for each demand unit *u*, a group value for each sub-basin can be determined as the weighted average of all sustainability values of the units included in sub-basin *i*. In Equation 23, the weights are determined from the annual supply target of unit *u* (T^u) and the annual target of sub-basin *i* ($T^i = \sum_{u=1 \in i}^{u=j \in i} T^u$)

$$SG^i = \sum_{u=1 \in i}^{u=j \in i} \frac{T^u}{T^i} * SI^u \tag{23}$$

The water allocation index (*WAI)* to estimate the system capacity to supply consumptive water demands is determined as a multi-annual average of the volumetric reliability (Equation 24). High values indicate a larger capacity to meet water demands.

$$WAI^u = \frac{(\sum_{t=1}^{n} X_t^u / T_t^u)}{n} \tag{24}$$

A WAI_{use}^{i} value can be determined for each water use sector in each sub-basin i, by aggregating the WAI^{u} values from all water demand units of a given sector within sub-basin *i,* as a weighted average (Equation 25).

$$WAI_{use}^{i} = \sum_{u=1 \in i}^{u=j \in i} \frac{T^{u}}{T_{use}^{i}} * WAI^{u} \tag{25}$$

2.8 Supporting tools to determine scarcity indices values

The significant data demands of indices such as the WEI+ and WPI, along with those of water accounting methodologies, have encouraged the use of mathematical models to supplement and enhance ground-based observations, satellite data, and global databases.

Hydrological models estimate key components of the natural water cycle, including evapotranspiration, groundwater recharge, and streamflow. Meanwhile, water allocation and management models offer insights into reservoir storage, water withdrawals, consumption, and return flows—contributing to a more complete understanding of water distribution and use.

Remote sensing products play a crucial role in complementing traditional monitoring systems, providing high-resolution spatial and temporal data for key variables needed to calculate water scarcity indices. These include precipitation, evapotranspiration, surface and groundwater storage, irrigated area, and irrigation water demand. Many of these datasets are readily accessible through existing global databases.

Examples of these applications are Pedro-Monzonís et al. (2016b) who presented a tool for integrating hydrological and water resources models within the framework of the System of Environmental-Economic Accounting for Water (SEEA-W) to construct water balances for a river basin. Delavar et al. (2020, 2022) linked the Water Accounting Plus (WA+) framework with the Soil Water Assessment Tool (SWAT) model to investigate water supply and demand in the semi-dry and enhance water management strategy evaluations. Ghorbanpour et al. (2022) assessed water use and crop water productivity in the western region of Lake Urmia by applying the Water Accounting Plus framework, using remote sensing data and SPHY hydrological model. Dembélé et al. (2023) developed a comprehensive modelling framework that integrates the mesoscale Hydrologic Model (mHM) with climate change scenarios and the Water Accounting Plus (WA+) tool to address future water resource challenges in the transboundary Volta River Basin (VRB) in West Africa. Rostami et al., 2024) use global databases for assessing water accounting in data-scarce river basins.

At IST, Sondermann and Oliveira (2021, 2022a, 2022b) applied a distributed hydrological model (APA/Bluefocus/Nemus/Hidromod, 2021) alongside AQUATOOL to assess water scarcity conditions in the Tagus River basin and evaluate climate adaptation strategies. This methodology was later extended to develop a water accounting framework for the same basin (Sondermann and Oliveira, 2025; APA/Bluefocus/Nemus/Hidromod, 2021) used the Mike Basin model to produce a detailed WEI+ map for Portugal and its transboundary river basins.

3 Discussion

The limitations of these water scarcity indexes have been extensively discussed in the literature.

The WSI and CR indices were developed to facilitate rapid comparisons between countries or large river basins without extensive data collection efforts. However, their inherent simplicity overlooks many critical details, resulting in significant limitations.

The WSI and CR indices do not account for the effects of seasonality on water availability and demand, factors that are particularly critical in river basins with Mediterranean climates. Additionally, these indices overlook flow redistribution provided by hydraulic infrastructures (Brown and Matlock, 2011; Damkjaer and Taylor, 2017; Liu et al., 2017; Savenije, 2000) and ignore inter-basin transfers. They also fail to consider alternative water resources such as desalination and wastewater reuse, which are crucial for water supply in certain regions (Rijsberman, 2006). As a result, water scarcity conditions in highly regulated basins may be inaccurately assessed by these indices.

Conversely, these indices overlook flow redistribution provided by hydraulic infrastructures (Damkjaer and Taylor, 2017; Liu et al., 2017; Savenije, 2000) and ignore inter-basin transfers.

They are also insensitive to the importance of green water which is available in the unsaturated zone (soil moisture) to plants and argued for an expansion of these indexes to cover this extra component of water availability and water use (Rockström and Falkenmark, 2015; Savenije, 2000; Vanham et al., 2018). They also usually fail to consider alternative water resources such as desalination and wastewater reuse, which are crucial for water supply in certain regions (Rijsberman, 2006).

Moreover, WSI and WEI typically do not account for water scarcity induced by water quality degradation. While historical and future trends influence water availability and demand, urbanization and agricultural activities have increasingly polluted water resources, thus limiting their safe use. To address this issue, Zeng et al. (2013) introduced a Quantity-Quality Water Scarcity Index, composed of a blue water scarcity index (the ratio of water withdrawals to freshwater resources) and a grey water scarcity index (the ratio of grey water footprint to freshwater resources). Here, grey water represents the volume of freshwater necessary to dilute and assimilate pollutants. Expanding on this concept, Liu et al. (2016) proposed a Quantity-Quality-EFR Water Scarcity Index, incorporating environmental flow requirements into water availability calculations. Wang et al. (2023) further demonstrated the significance of quality-induced scarcity, showing that in 2010, 984 sub-basins experienced scarcity based solely on quantity, whereas 2,517 sub-basins faced scarcity due to combined quantity and quality factors.

Concerning the water use component, the use of water abstraction as a synonym for water use is also a major criticism, as it does not consider water returns from consumptive uses, nor instream water uses (Liu et al., 2017; Rijsberman, 2006; Vanham et al., 2018). However, uncertainties are added when considering net abstractions, because the exact return volume and the exact location of return are usually unknown.

As a result, water scarcity conditions in highly regulated basins may be inaccurately assessed by these indices.

WEI+ offers added value to other water quantitative scarcity indexes. By considering storage capacity in reservoir and aquifer systems, seasonality and returns from water use, the index can present a more accurate picture of water scarcity in highly regulated river basins with significant seasonality. WEI+ values highlight hydrological information at smaller spatial and temporal scales and reveal a water-scarce situation that is underestimated when the results are presented on an annual or spatial aggregated basis (Pedro-Monzonís et al., 2016; Rijsberman, 2006).

However, the estimation of WEI+ is not trivial and, in the absence of comprehensive monitoring records on water availability and uses which rarely exist, requires the detailed modelling of water management and allocation in the river basin, including groundwater systems.

The improvements included in WEI+ demand more data and add difficulties to its calculation, especially in complex river basin systems with aquifer-river interactions and major hydrological alteration arising from reservoir storage, water abstractions and returns and inter-basin transfers (Bariamis et al., 2016; RBMP-Cyprus, 2016). The lack of data can be partially overcome with different techniques, such as satellite data, remote sensing and hydrological modelling(Contreras and Hunink, 2015), as discussed in chapter 2.8.

In addition to higher data requirements, the WEI+ definition also has certain limitations. While calculating water scarcity at a sub-basin level offers a more precise representation by minimizing estimation errors due to spatial aggregation, the spatial discretization itself can influence WEI+ values. This discretization may obscure dependencies of sub-basins on inflows from external basins, potentially resulting in misinterpretations of actual water scarcity conditions. Such inaccuracies are particularly significant in sub-basins with upstream-downstream dynamics, where increased upstream water consumption exacerbates scarcity downstream. This constitutes a critical issue in transboundary river basins, such as the international Iberian River basins.

The inclusion of environmental flow requirements within WEI+ computation enables its use by policymakers and stakeholders to better manage water allocation while respecting the environmental status of all water bodies. However, WEI+ is significantly dependent on the assumptions or decisions regarding environmental flow requirements, which vary widely. A clear understanding of this dependence is needed when evaluating WEI+ values.

Among the methodologies that have been proposed in the literature, hydrologically based methodologies, which define environmental flow requirements as a percentage of monthly flow, are the most used due to their simplicity and low cost. However, assumptions vary across river regimes (Vicente et al., 2019). For instance, Richter et al. (2012) suggest 80% of monthly runoff, while Pastor et al. (2014) use between 25% and 46% of mean annual flows (Vanham et al., 2018).

In addition, the computation of WEI+ may lead to values greater than one when $Qenv_t^i$ close to RWR_t^i and to negative values when $Qenv_t^i$ is greater than RWR_t^i. These situations are not unusual in southern European river basins with highly variable streamflow regimes, when $Qenv_t^i$ is defined as a fixed percentage of mean monthly flows. ${WEI^+}_t^i$ was set to one in these cases, depicting a situation where all renewable water resources are being used, even though not all demands are being fully satisfied.

The inclusion of environmental flow requirements within WEI+ computation enables its use by policymakers and stakeholders to better manage water allocation while respecting the environmental status of all water bodies. However, WEI+ is significantly dependent on the assumptions or decisions regarding environmental flow requirements, which vary widely. A clear understanding of this dependence is needed when evaluating WEI+ values.

Among the methodologies that have been proposed in the literature, hydrologically based methodologies, which define environmental flow requirements as a percentage of monthly flow, are the most used due to their simplicity and low cost. However, assumptions vary across river regimes (Vicente et al., 2019). For instance, Richter et al. (2012) suggest 80% of monthly runoff, while Pastor et al. (2014) use between 25% and 46% of mean annual flows (Vanham et al., 2018).

In addition, the computation of WEI+ may lead to values greater than one when $Qenv_t^i$ close to RWR_t^i and to negative values when $Qenv_t^i$ is greater than RWR_t^i. These situations are not unusual in southern European river basins with highly variable streamflow regimes, when $Qenv_t^i$ is defined as a fixed percentage of mean monthly flows. ${WEI^+}_t^i$ was set to one in these cases, depicting a situation where all renewable water resources are being used, even though not all demands are being fully satisfied.

Finally, the use of WEI+ to evaluate potential pressures on water bodies may clearly miss poor conditions resulting from significant changes in the hydrological regime induced by hydropower plants. WEI+ only considers water allocated to consumptive demands as water uses. Instream uses such as hydropower are reflected in the reservoir operating policies, but this is insufficient to measure the impacts that hydropower systems, with a large storage capacity, impose on the water dependent ecosystems and other uses.

The water accounting framework also has its limitations. Water accounting primarily relies on average estimates of stocks and flows, neglecting crucial concepts such as water supply reliability, vulnerability, and resilience. These factors are particularly important in basins with a Mediterranean climate, where water availability is highly variable, and acute water scarcity is common. While water managers rely on reservoir and aquifer storage to meet demands, dry periods frequently result in demands being only partially met. Effective water management requires a more comprehensive understanding of variability and risk to ensure sustainable and reliable resource use.

However, the water accounting framework has notable limitations. Water accounting primarily relies on average estimates of stocks and flows, neglecting crucial concepts such as water supply reliability, vulnerability, and resilience. These factors are particularly important in basins with a Mediterranean climate, where water availability is highly variable, and acute water scarcity is common. While water managers rely on reservoir and aquifer storage to meet demands, dry periods frequently result in demands being only partially met. Effective water management requires a more comprehensive understanding of variability and risk to ensure sustainable and reliable resource use.

Finally, the three indexes do not consider water quality issues which often diminish actual water availability. Zeng et al. (2013) addressed this concern by proposing a Quantity-Quality water scarcity index, determined by the sum of a blue water scarcity index (ratio of water withdrawals to freshwater resources) and a grey water scarcity index (ratio of grey water footprint to freshwater resources), where grey water is the freshwater required to assimilate the load of pollutants. Based on this index, Liu et al. (2016) proposed a Quantity-Quality-EFR water scarcity index, by deducting environmental flow requirements from water availability.

3.1 Global Assessment

Global freshwater use has increased significantly since 1950 due irrigation and improved life conditions, but it seems it has plateaued since 2020 (Figure 13). Although irrigated area went on growing, gains from improved efficient seem to have compensate this growth (FAO, 2024). From 2015 to 2021 global water use has marginally decreased by 0.1% and agriculture water use as decrease 0.6%. Water use efficiency, measured by gross value added by cubic meter, rose 19.3% from 2015 to 2021, with agriculture showing the highest percentage change, followed by the industrial sector and the service sector (FAO, 2024). Nevertheless, renewable freshwater resources have decreased, mainly due to population growth (Figure 14)

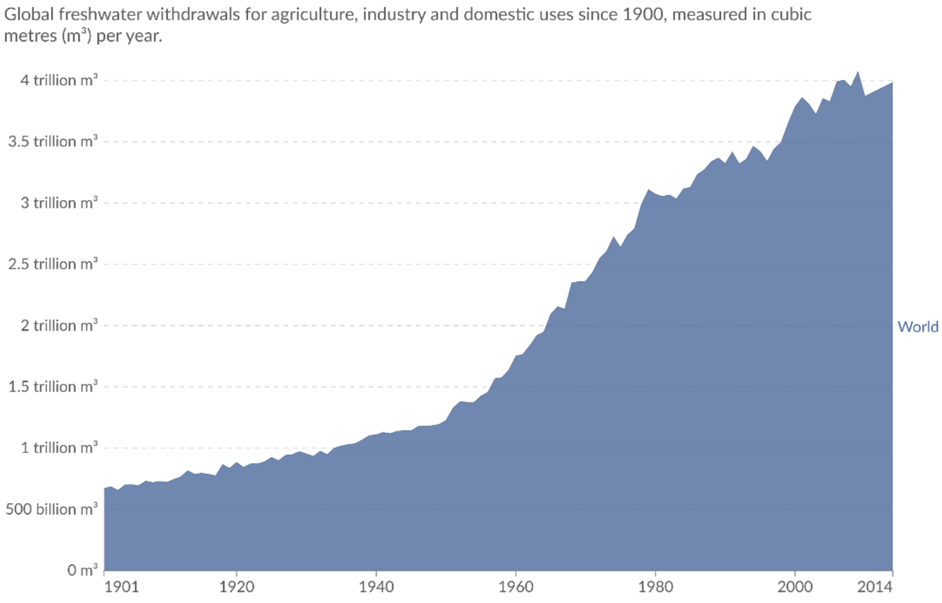

Figure 13 – Global freshwater use over the long run (source: Global International Geosphere-Biosphere Programme (IGB), processed by Our World in Data).

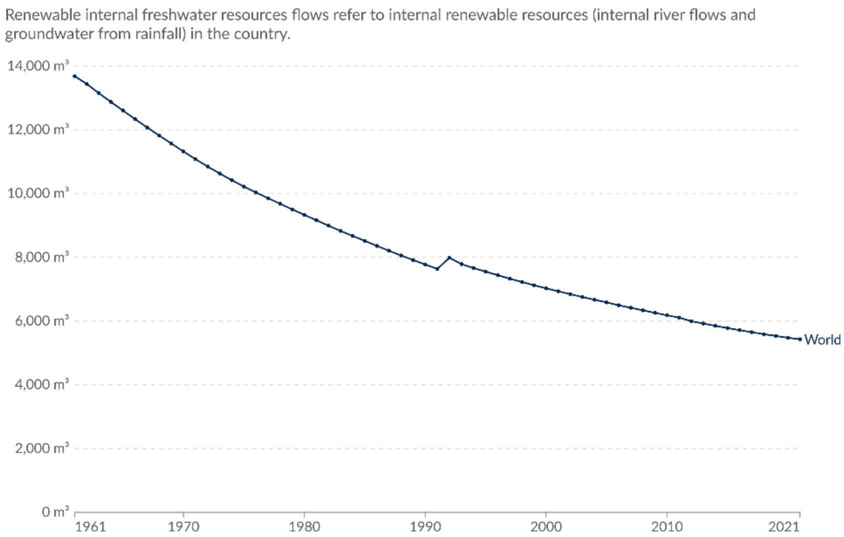

Figure 14 – Renewable freshwater resources per capita (source: Food and Agriculture Organization (via World Bank), processed By Our Word in Data (2025).

In Europe, the picture is clearer. Land equipped for irrigation has plateaued since 1985 (Figure 15). Water use efficiency has increased in agriculture, electricity production, industry, mining, public water supply and tourism, as Figure 16 shows (EEA 2021). Since 2000, water abstractions and consumption have decreased, particularly the former. Water consumption by these sectors was 16 % lower in 2017 (the last year for which EU-wide statistics are available) than in 1995 (baseline), while production in these sectors grew by 20 % in terms of net value added.

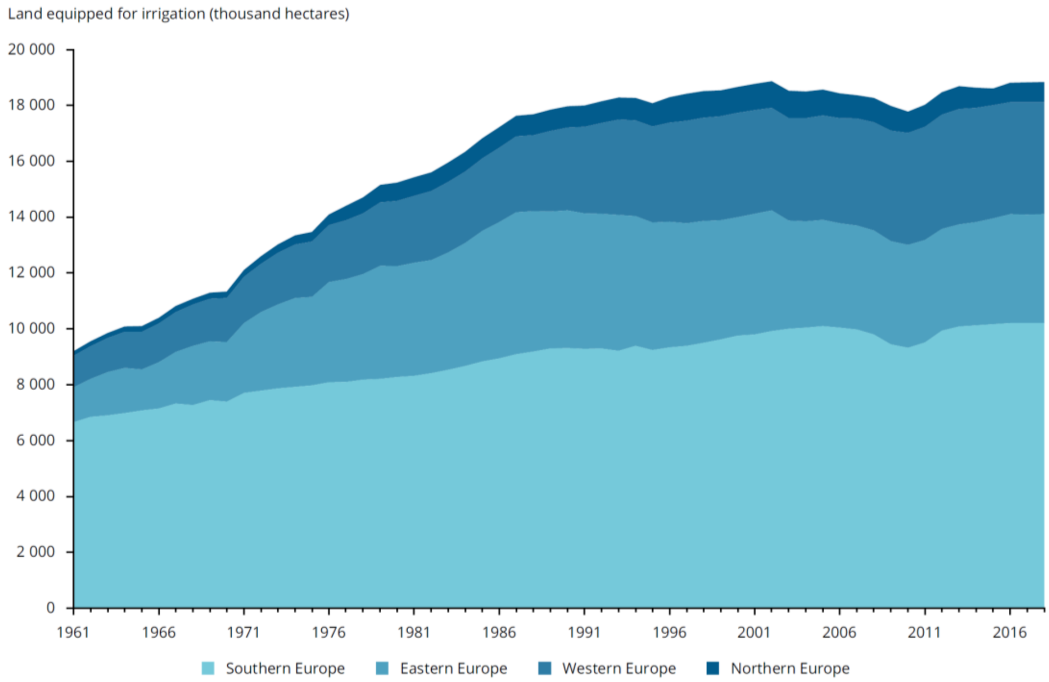

Figure 15 – Irrigable area in the EU-27 and the UK since 1961 (source: FAO)

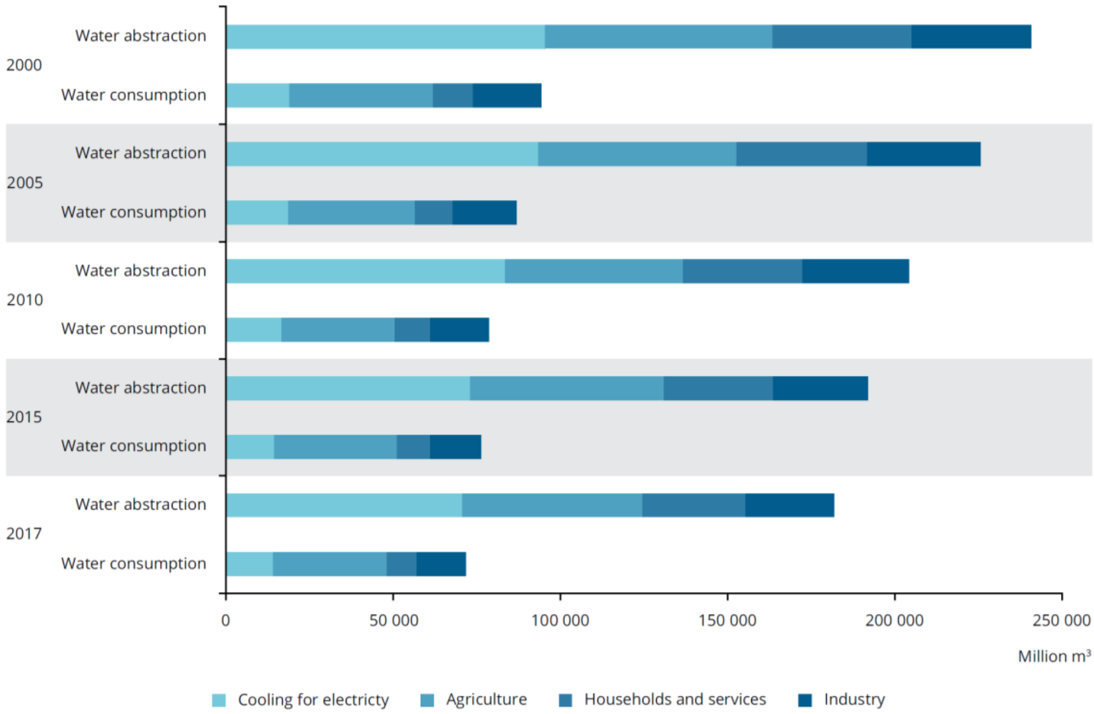

Figure 16 – Trends in water abstraction and water consumption in Europe (EEA-28 and the UK) (source: EEA (2019, 2021), Eurostat, 2020)

These recent trends in water abstraction and improvements in water-use efficiency provide valuable guidance for addressing the growing challenges of water scarcity. The indicators and approaches discussed in this lecture can be utilized to evaluate current conditions, set clear objectives, and identify specific regions or basins requiring targeted actions. Additionally, they can serve as tools for monitoring progress toward achieving these objectives. Among the reviewed approaches, the WEI+ and water accounting offer the most comprehensive analysis by considering the role of reservoirs and inter-basin transfers, which is particularly critical in highly regulated basins. Their study should be complemented with other elements of analysis to overcome their weaknesses.

3.2 The Tagus River basin

The Tagus River basin is one of the most heavily regulated and vital river systems in the region. Situated on the Iberian Peninsula in Southern Europe, this transboundary basin is shared between Spain, the upstream country, and Portugal, the downstream country. It supports major metropolitan areas such as Madrid and Lisbon, important agricultural hubs, and meets demands outside the basin in southern Spain, supplied through the Tagus-Segura aqueduct (APA, 2016; CHT, 2018). Additionally, the basin hosts significant hydropower production capacity, totalling 2700 MW, including 915.2 MW from the Alcantara power plant, located near the border between Spain and Portugal. The basin's total storage capacity is 14 000 hm^3, with 11 150 hm^3 in Spain and 2 850 hm^3 in Portugal (Sabater et al., 2009; Sondermann and Oliveira, 2021).

The Albufeira Convention, signed in 1998 and revised in 2008 and in 2024, establishes the rules for governing and protecting the shared river basins of the Iberian Peninsula. While detailed studies on environmental flow requirements are not completed for the Tagus River basin, a provisional regime is in place, specifying minimum annual, quarterly, weekly, and daily flows at the border and upstream of the estuary.

Situated in a semi-arid region, the river basin shows a high seasonality and inter-annual variability of flows, with high flows usually occurring in the winter, especially from November to March and low flows from June to September (APA/Nemus/Blefocus/Hidromod, 2021). Large floods usually occur in December and January (Azevêdo et al., 2004; Benito et al., 2003; Sabater et al., 2009). Water use also has a strong variability, with 80% of the abstractions to meet water demands for consumptive use occurring from April to September. This imbalance in time triggers significant stress on freshwater resources during summer. Figure 17shows the Tagus River basin, its hydraulic infrastructures, and the disparity of water resources in natural conditions and water uses.

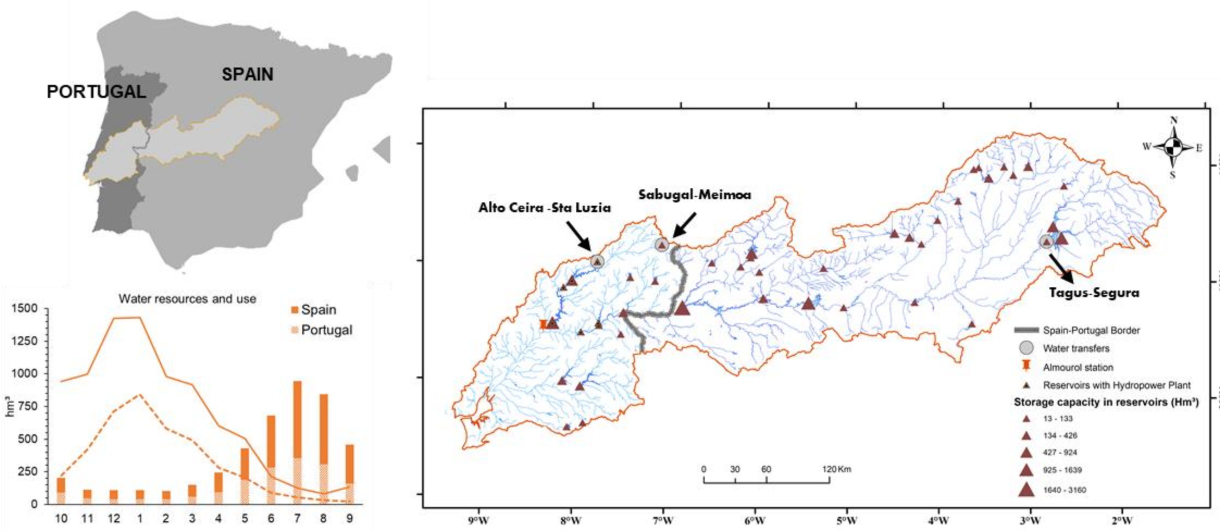

Figure 17 - The Tagus River basin and its hydraulic infrastructure, internal water resources and uses in the Spanish and Portuguese parts of the basin (source: Sondermann and Oliveira, 2025)

The Tagus River basin frequently comes under scrutiny due to the water scarcity challenges faced by both Spain and Portugal. These issues contribute to hydrological droughts and ongoing conflicts over water use between the riparian countries (Lobanova et al., 2017; López-Moreno et al., 2009; Lorenzo-Lacruz et al., 2010; Milanes, 2017; Sondermann and Oliveira, 2022a; Sondermann and Oliveira, 2022b). Climate change is expected to exacerbate water scarcity and reduce the ability to meet water demand, particularly for non-priority uses. This raises significant concerns about the long-term sustainability of water resource management (Lobanova et al., 2017; Pellicer-Martínez and Martínez-Paz, 2018; Sondermann and Oliveira, 2022a). This underscores the importance of studies that analyse water resource systems and support decision-makers in defining appropriate management strategies to address water scarcity conditions effectively.

The WSI values for the Spanish territory and for the whole Tagus River basin are also below 1700 m^3/hab, highlighting the pressures human occupation wields on the existing water resources. The WSI value for the Portuguese part of the Tagus River basin is significantly above this threshold (3625 m^3/hab), as flows from Spain are judged as available in Portugal.

The analysis of the Criticality Ratio leads to similar conclusions. The Spanish part of the Tagus River basin and the overall river basin, with a CR value close to 0.3, respectively, are classified as water-scarce ($0.2 < CR \leq 0.4$), according to Alcamo et al. (2000) reference values. The Portuguese part of the river basin, receiving flows generated in Spain, is considered a non-stressed territory ($CR \leq 0.2$).

The analysis of WEI+ shows higher water scarcity levels, in accordance with WEI+ thresholds defined by Faergemann (2012). Both the Spain and Portugal are classified as water-stressed ($0.2 < WEI+ \leq 0.4$) and the entire Tagus River basin as severe water stress conditions ($0.4 < WEI+ \leq 0.6$). The higher value obtained for the entire river basin (higher than the WEI+ values of Portugal and Spain) reflects the fact that the WEI+ value for Portugal, if computed with resources solely generated in Portugal, would be close to 0.7. The actual WEI+ for Portugal considers the inflows released from Spain, estimated on average as 6 734 hm3/year (Sondermann and Oliveira, 2022a)

Under the WA+ framework, the Resource Base Sheet provides information on the available water resources and their relationship with the main hydrological processes. It also includes information on water already committed to consumptive and non-consumptive uses. Figure 17 illustrates the current situation for the Tagus River basin. During an average year, *Precipitation* volumes that fall in Spain and Portugal are, respectively, 32.5 km^3 and 14.8 km^3, reflecting the larger area of the basin in Spain and the higher precipitation rate in Portugal. Green Water flows can be evaluated through Landscape Evapotranspiration and its distribution across various land use classes. Landscape Evapotranspiration represents the largest portion of Precipitation, accounting for approximately 78% of the average annual precipitation in Spain and 84% in Portugal.

The Blue Water Fraction is described by the Exploitable Water component, a part of Net Inflow which receives contributions from the Precipitation, External Inflows and Storages Changes in surface and groundwater systems. As a downstream country, Portugal Net Inflow also includes flows from Spain in addition to precipitation and storage changes, estimated to be 4.9 km3 in an average year. There are also two minor external water transfers from other Portuguese river basins, namely the Douro (Sabugal-Meimoa transfer) and the Mondego (Alto Ceira – Sta. Luzia transfer), totalling 0.13 km3 in an average year. Exploitable Water accounts for 22% of Net Inflow in Spain and 37% in Portugal, the rest being Landscape Evapotranspiration.

The Reserved Flow for hydropower and environmental purposes in an average year is estimated at 4.9 km^3 in Spain and 5.7 km^3 in Portugal. These estimates are based on historical hydropower production and environmental flow requirements, assumed to be 15% of average flows. It is important to note that environmental flows can also be utilized for hydropower production, and navigation in the Tagus River basin is not significant. Given the importance of hydropower in the Tagus River basin, the Reserved Flow is an important fraction of Exploitable Water, ranging from 63% to 70% in Spain, and from 47% to 78% in Portugal.

Available water can be allocated to existing uses within the basin (Utilised Flow) or made available to future or downstream uses (Utilisable Flow). Utilised Flow shows slight variations across different water year types. In an average year, it reaches 2.1 km^3 in Spain and 1.0 km^3 in Portugal. During dry years, and wet years, Utilised Flow varies significantly posing significant water scarcity challenges in the river basin.

The Utilisable Water from Spain (depicted at right hand side of Figure 18) represents the flow at the Spain-Portugal border, which contributes significantly to Portugal's inflows (shown at left hand side of Figure 18). This inflow is essential for meeting downstream water demands and ensuring compliance with committed environmental flow requirements. On average, the Utilisable Water amounts to approximately 0.1 km^3 annually, with significant variability, ranging from 0 km^3 in dry years to 2.7 km^3 in wet years. Furthermore, a major outflow from the Spanish Tagus sub-basin is the Tagus-Segura transfer, which diverts water from the headwaters of the Tagus River to southern Spain. This transfer accounts for an external outflow of about 0.3 km^3 during average and dry years, increasing to approximately 0.5 km^3 in wet years.

The Utilisable Water from Portugal corresponds to the flow at the outlet of the Tagus River basin. It reaches 0.6 km3 in an average year and varies from 0 km3 in a dry year to 9.9 km3 in a wet year.

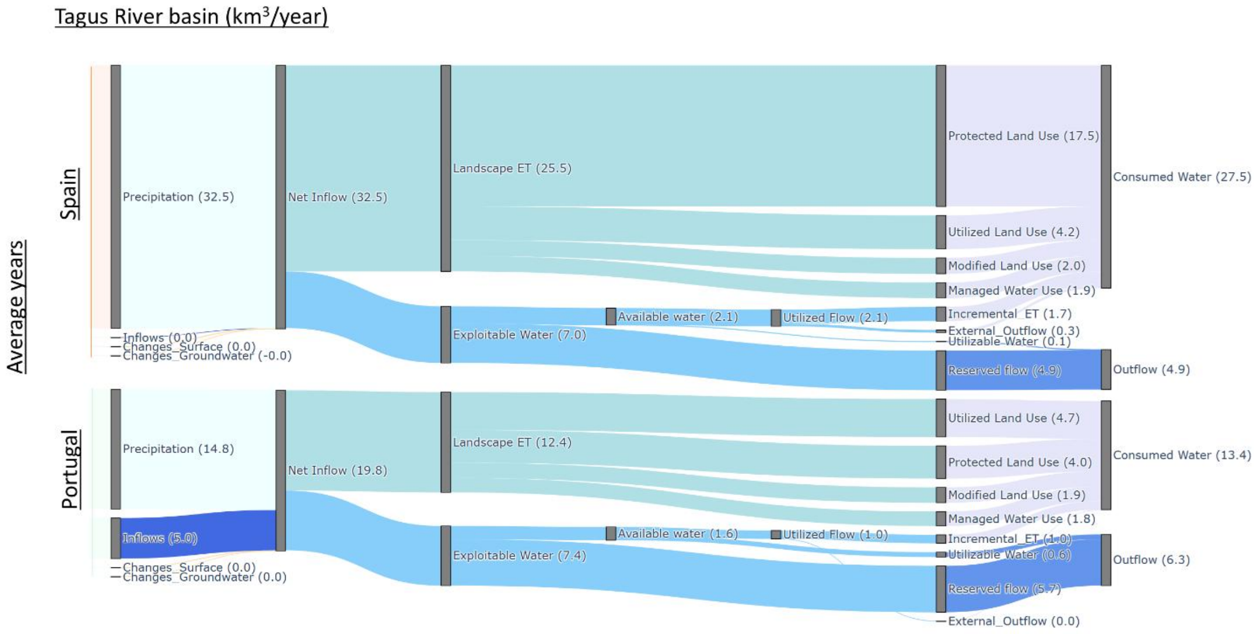

Figure 18 - Resource base diagram for the entire Tagus River basin under the current situation, for an average year

Under the WA+ framework, the *Withdrawal Sheet* provides information on consumptive water use. Withdrawal diagrams were developed for the entire Tagus River basin, covering the period from 1990 to 2015, with the basin divided into Spanish and Portuguese territory. Figure 19 presents an overview of internal water allocation under the current situation.

During an average year, surface and groundwater *Withdrawal's* amount to 2.8 km^3 in Spain and 1.6 km^3 in Portugal. *Water Diverted* to meet demand remains relatively stable across the three categories of years, as reservoir and aquifer storage helps to mitigate water shortages caused by reduced precipitation and natural flow.

Surface Water Resources accounts for approximately 97% of gross water withdrawals in Spain and around 63% in Portugal. Although there are uncertainties in evaluating the contributions of groundwater to meet demands, these estimates reveal the importance of significant aquifer systems in Lower Tagus Valley in Portugal, namely the Tagus-Sado aquifer system.

Agricultural Use accounts for approximately 59% of total gross withdrawals in the Tagus River basin, followed by *Urban Use* (22%), *Reservoir* replenishment (11%), and *Industrial Use* (8%). *Water Depletion*, represented by *Incremental Evapotranspiration* (ET), accounts for approximately 63% of the total water diverted in the entire Tagus River basin.

In Spain, 57% of the total *Return Flows* are directed to surface water resources during average years, with the remaining 43% contributing to groundwater recharge. In Portugal, about 53% of *Return Flows* are to surface water during average years, with the rest contributing to groundwater.

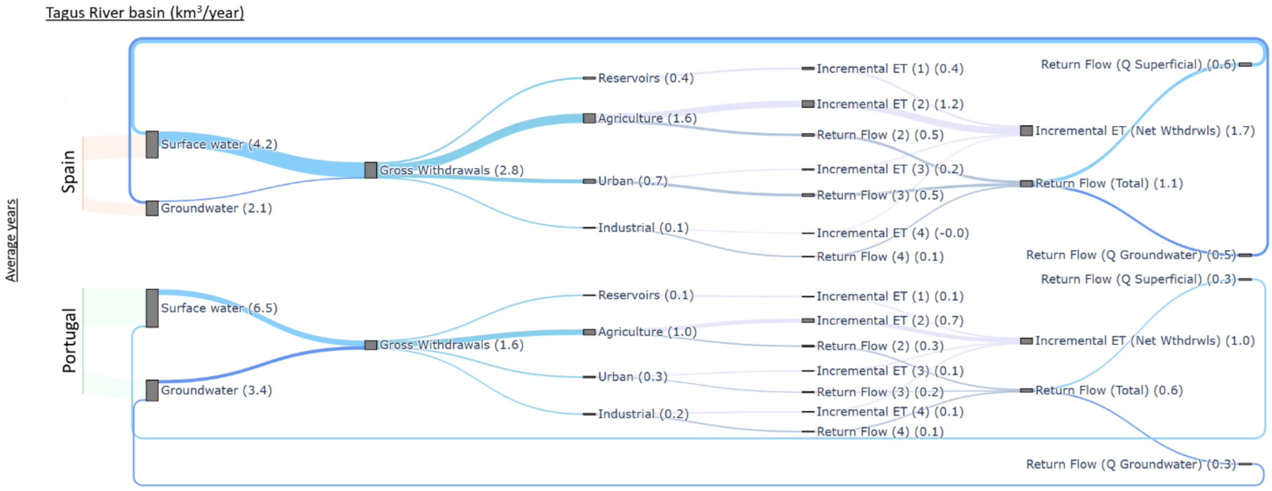

Figure 19 - Withdrawal diagram for the entire Tagus River basin under the current situation, for an average year.

The Tagus River basin faces a significant challenge with water scarcity, as resources are intensively utilised to meet different water needs and growing demands, particularly during dry years. Water accounting offers a streamlined and user-friendly framework for reporting water availability and allocation within a river basin by consolidating and presenting water resource data in a structured and organized manner that facilitates a comprehensive understanding of how water is distributed and utilized across various sectors and regions. Sondermann and Oliveira (2025) explore the water accounting approach for wet and dry years, in addition to the dry years described herein.

This approach is invaluable in identifying critical management challenges such as water scarcity, inefficiencies, and imbalances in allocation, as it supports informed decision-making by providing actionable insights that promote sustainable water management practices and equitable distribution among stakeholders. This comprehensive understanding is particularly useful in transboundary river basins, where water resources are shared among multiple countries that must collaborate effectively. By offering a common platform for understanding water availability, allocation, and usage, water accounting fosters transparency and builds trust among stakeholders. A shared perspective on key issues helps create a unified vision of challenges and opportunities, which is essential for negotiating agreements on cooperative actions to achieve sustainable and equitable management of transboundary water resources.

4 Final comments

Growing population demands and the impacts of climate change are placing increasing pressure on water resources, affecting both their availability and quality. At the same time, farmers are shifting toward more profitable, irrigated permanent crops, which reduce water management flexibility and further strain water supplies. With irrigated farming being cornerstone of local and regional economies, particularly in sparsely populated areas where agriculture dominates economic activity, there is a strong political pressure to construct new dams to enhance water storage and regulate resources, or to transfer water from wetter regions to drier ones.

Reservoirs have traditionally played a pivotal role in addressing water scarcity, particularly in Mediterranean climates where irrigation is vital for agriculture. These regions often endure extended dry seasons, making dependable water storage essential for maintaining crop production.

Despite their benefits, reservoirs and dams come with notable environmental and economic costs. Their construction can disrupt ecosystems, lead to biodiversity loss, and displace communities, while requiring substantial financial investments and ongoing maintenance. These factors highlight the need for rigorous cost-benefit analyses when considering new reservoir projects, ensuring that their advantages outweigh their negative impacts and that alternative solutions are also evaluated.

When evaluating new reservoir projects or large-scale water transfers, it is essential to conduct comprehensive cost-benefit analyses to ensure that the anticipated benefits clearly outweigh potential negative impacts. These analyses must rigorously consider economic, environmental, and social dimensions, including the socio-economic value of the activities supported. Key aspects include assessing reservoir operations under various climate change scenarios and examining the risk of future water shortages in donor regions involved in transfers. Equally important is a thorough evaluation of alternative solutions, to ensure that reservoir construction is genuinely the most advantageous and sustainable option.

Additional measures contribute significantly to control water scarcity effectively beyond reservoirs, if framed within a diverse, yet coherent, set of initiatives targeting water demand, supply, and quality. This approach requires active involvement from governments, user associations, the private sector, and local communities. It embodies the concept of Integrated Water Resources Management (IWRM), formally introduced during the International Conference on Water and the Environment in Dublin, Ireland, in 1992.

Some measures are widely agreed upon, such as improving water use efficiency and effectively reducing physical water losses from supply and distribution networks in the short term. Taking stock on the improved water use efficiency of the recent decades, additional gains in water productivity and in the preservation of soil moisture are essential. To achieve this, it is crucial to expand traditional farmers' access to micro-irrigation technologies, climate-resilient seed varieties, and sustainable cropping practices (GCEC, 2024).

Another priority area involves enhancing groundwater monitoring to prevent overuse and increasing water availability by integrating the management of surface and groundwater resources.

Within water supply measures, source diversification should be promoted. The reuse of treated wastewater and seawater desalination are emerging strategies in some parts of the world, although expanding these solutions globally presents certain challenges. Typically, desalination plants and wastewater treatment facilities are located near coastal areas and major urban centres. Therefore, using this water in inland regions requires transporting it over potentially significant distances, adding transportation and storage costs to the already substantial treatment and desalination expenses. Furthermore, utilizing this water for agricultural purposes poses additional challenges due to aligning a relatively steady water supply with the seasonal variability of agricultural demand. Thus, treated or desalinated water should ideally complement other water sources or be linked to robust water storage infrastructure.

A comprehensive strategy aligned with integrated water resources management (IWRM) principles should be implemented. By integrating reservoirs with complementary solutions and prioritizing water efficiency, decision-makers can create a more resilient and sustainable approach to water management. This holistic strategy not only addresses current water scarcity challenges but also supports the economic and ecological needs of future generations.

Finally, managing water demand will require the use of economic instruments that internalize water-related costs in decision-making processes. The Global Commission on the Economics of Water has called for a new economics of water to value water properly to reflect its scarcity and the multiple benefits it provides as the Earth's most precious resource, to tackle externalities caused by the misuse and pollution of water and to spur a wave of innovations, capacity-building and investments (GCEW, 2024). The commission stresses that the widespread underpricing of water today encourages its profligate use across the economy and unwisely skew the locations of the most water-intensive crops to areas most at risk of water stress.

The introduction of intelligent tariff schemes reflecting ideal or recommended water allocations per usage category can encourage responsible water consumption across all sectors. While incremental increases in water pricing may be feasible, provisions must ensure affordability for essential needs and public interest scenarios. A core shift is to correctly price water and allocate subsidies to achieve both its efficient use and access for all.

5 References

APA, 2016. Plano de Gestão de Região Hidrográfica do Tejo e Ribeiras do Oeste (RH5). Agência Portuguesa do Ambiente, Lisboa, Portugal.

Alcamo, J., Henrichs, T., Rösch, T., 2000. World Water in 2025 Global modeling and scenario analysis for the World Commission on Water for the 21st Century. Kassel, Germany.

APA/Bluefocus/Nemus/Hidromod, 2021 Avaliação das disponibilidades hídricas por massa de água e aplicação do índice de escassez WEI+. Agência Portuguesa do Ambiente, Lisbon, Portugal.

Arnell, N.W.; van Vuuren, D.P.; Isaac, M, 2011. The Implications of Climate Policy for the Impacts of Climate Change on Global Water Resources. *Glob. Environ. Chang.* **2011**, *21*, 592–603, doi:10.1016/j.gloenvcha.2011.01.015

Azevêdo, T.M., Nunes, E., Ramos, C., 2004. Some Morphological Aspects and Hydrological Characterization of the Tagus Floods in the Santarém Region, Portugal. Natural Hazards 31, 587–601.

Bariamis, G., Panagos, D., Zachos, A., Baltas, E., 2016. Use of Freshwater Resources in Europe 2002-2012. Supplementary document to the European Environment Agency's core set indicator 018. ETC/ICM Technical Report 1/2016, Magdeburg: European Topic Centre on inland, coastal and marine waters.

Benito, G., Díez-Herrero, A., Fernández De Villalta, M., 2003. Magnitude and frequency of flooding in the Tagus basin (Central Spain) over the last millennium. Clim Change 58, 171–192. https://doi.org/10.1023/A:1023417102053

Bisselink, B., Bernhard, J., Gelati, E., Jacobs, C., Mentaschi, L., Roo, A. De, 2018. Impact of a changing climate , land use , and water usage on water resources in the Danube river basin. Publications Office of the European Union, Luxembourg. https://doi.org/10.2760/89828

Brown, A., Matlock, M.D., 2011. A Review of Water Scarcity Indices and Methodologies, The Sustainability Consortium. Arkansas, United States.

BoM, 2010. Bureau of Meteorology. Australian Water Accounting Standard. Canberra: Commonwealth of Australia Brown and Matlock, 2011;

Casadei, S., Peppoloni, F., Pierleoni, A., 2020. A New Approach to Calculate the Water Exploitation Index (WEI +). Water. https://doi.org/10.3390/w12113227

CHT, 2015. Plan hidrológico de la parte española de la Demarcación Hidrográfica del Tajo. Confederación Hidrográfica del Tajo, Madrid, Spain.

CHT, 2018. Plan Hidrológico de la Demarcación Hidrográfica del Tajo Revisión de tercer ciclo (2021-2027) Documentos iniciales. Programa, calendario, estudo general sobre la Demarcación y fórmulas de consulta Memoria 19 de octubre de 2018. Confederación Hidrográfica del Tajo, Madrid, Spain.

Contreras, S., Hunink, J., 2015. Water accounting at the basin scale: Water use and supply (2000-2010) in the Segura River Basin using the SEEA framework., Future Water Technical Report. Cartagena (Murcia), Spain.

Craswell, E., Bonell, M., Bossio, D., Demuth, S., van de Giesen, N., 2007. Integrated Assessment of Water Resources and Global Change, Springer. ed, Integrated Assessment of Water Resources and Global Change. Springer Netherlands, Dordrecht, The Netherlands. https://doi.org/10.1007/978-1-4020-5591-1

Damkjaer, S., Taylor, R., 2017. The measurement of water scarcity: Defining a meaningful indicator. Ambio A J. Hum. Environ. 46, 513–531. https://doi.org/10.1007/s13280-017-0912-z

De Roo, A., Bisselink, B., Beck, H., Bernhard, J., Burek, P., Reynaud, A., Pastori, M., Lavalle, C., Jacobs-Crisioni, C., Baranzelli, C., Zajac, Z., Dosio, A., 2016. Modelling water demand and availability scenarios for current and future land use and climate in the Sava River Basin. Joint Research Centre, Ispra, Italy.

Delavar, M., Eini, M.R., Kuchak, V.S., Zaghiyan, M.R., Shahbazi, A., Nourmohammadi, F., Motamedi, A., 2022. Model-based water accounting for integrated assessment of water resources systems at the basin scale. Science of The Total Environment 830, 154810. https://doi.org/10.1016/j.scitotenv.2022.154810

Dembélé, M., Salvadore, E., Zwart, S., Ceperley, N., Mariéthoz, G., Schaefli, B., 2023. Water accounting under climate change in the transboundary Volta River Basin with a spatially calibrated hydrological model. J Hydrol (Amst) 626. https://doi.org/10.1016/j.jhydrol.2023.130092

EEA, 2010. Water exploitation index (WEI). — European Environment Agency [WWW Document]. URL https://www.eea.europa.eu/data-and-maps/figures/water-exploitation-index-wei-3 (accessed 9.17.21).

EEA, 2019. Indicators. European Environment Agency. https://www.eea.europa.eu/data-and-maps/indicators/#c0=30&c12-operator=or&b_start=0&c6=wei. Accessed 5 Nov 2020

EEA 2021, Water resources across Europe —confronting water stress: an updated assessment, EEA Report 12/2021, European Environment Agency.

EEA, 2024, Europe's state of water 2024: The need for improved water resilience, EEA Report 7/2024, European Environment Agency.

Esen, S.E., Hein, L., 2020. Development of SEEA water accounts with a hydrological model. Science of the Total Environment 737. https://doi.org/10.1016/J.SCITOTENV.2020.140168

Faergemann, H., 2012. Faergemann, H., 2012. — European Environment Agency [WWW Document]. URL https://www.eea.europa.eu/data-and-maps/indicators/use-of-freshwater-resources-3/faergemann-h-2012 (accessed 2.2.21).

Falkenmark, M., Lundqvist, J., Widstrand, C., 1989. Macro-scale water scarcity requires micro-scale approaches. Aspects of vulnerability in semi-arid development. Nat. Resour. Forum 13, 258–267. https://doi.org/10.1111/J.1477-8947.1989.TB00348.X

FAO, 2019. Step-by-step monitoring methodology for indicator 6.4.2. Food and Agricultural Organization of the United Nations, Rome.

GCEW, 2024, The economics of water: Valuing the hydrological cycle as a global common good, Global Commission on the Economics of Water, OECD Environment Directorate: Climate, Biodiversity and Water Division.

Godfrey, J.M., Chalmers, K., 2012. Water Accounting: International Approaches to Policy and Decision-making. Water Account. Int. Approaches to Policy Decis. 1–318. https://doi.org/10.4337/9781849807500

Hashimoto, T., Stedinger, J.R, Loucks, D.P., 1982a, Reliability, Resiliency, and Vulnerability Criteria for Water Resource System Performance Evaluation, *Water Resources Research*, September 1982.

Hashimoto, T., Loucks, D.P., Stedinger, J.T., 1982b, Robustness of Water Resource Systems, *Water Resources Research*, September 1982.

Hoekstra, A.Y., Mekonnen, M.M., Chapagain, A.K., Mathews, R.E., Richter, B.D., 2012. Global monthly water scarcity: Blue water footprints versus blue water availability. PLoS One 7. https://doi.org/10.1371/journal.pone.0032688

Hussain, Z., Wang, Z., Wang, J., Yang, H., Arfan, M., Hassan, D., Wang, W., Azam, M.I., Faisal, M., 2022. A comparative Appraisal of Classical and Holistic Water Scarcity Indicators. Water Resour. Manag. 36, 931–950. https://doi.org/10.1007/S11269-022-03061-Z/TABLES/2

IPCC, 2022. Climate Change 2022: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [H.-O. Pörtner, D.C. Roberts, M. Tignor, E.S. Poloczanska, K. Mintenbeck, A. Alegr. Cambridge University Press, Cambridge, UK and New York, NY, USA, 3056 pp., doi:10.1017/9781009325844.

Karabulut, A., Egoh, B.N., Lanzanova, D., Grizzetti, B., Bidoglio, G., Pagliero, L., Bouraoui, F., Aloe, A., Reynaud, A., Maes, J., Vandecasteele, I., Mubareka, S., 2016. Mapping water provisioning services to support the ecosystem-water-food-energy nexus in the Danube river basin. Ecosyst. Serv. 17, 278–292. https://doi.org/10.1016/j.ecoser.2015.08.002

Karimi, P., Bastiaanssen, W.G.M., Molden, D., 2013. Water Accounting Plus (WA+)-a water accounting procedure for complex river basins based on satellite measurements. Hydrol. Earth Syst. Sci 17, 2459–2472. https://doi.org/10.5194/hess-17-2459-2013

Karimi, P., Bastiaanssen, W.G.M., Molden, D.J., Cai, X., 2012. Water accounting to assess use and productivity of water: Evolution of a concept and new frontiers. https://doi.org/10.4337/9781849807494.00012

Lawrence, P., Sullivan, C, 2022, The water poverty index: an international comparison, Keele Economics Research Papers.

Liu, J., Liu, Q., Yang, H., 2016. Assessing water scarcity by simultaneously considering environmental flow requirements, water quantity, and water quality. Ecol. Indic. 60, 434–441. https://doi.org/10.1016/j.ecolind.2015.07.019

Liu, J., Yang, H., Gosling, S.N., Kummu, M., Flörke, M., Pfister, S., Hanasaki, N., Wada, Y., Zhang, X., Zheng, C., Alcamo, J., Oki, T., 2017. Water scarcity assessments in the past, present, and future. Earth's Futur. 5, 545–559. https://doi.org/10.1002/2016EF000518

Lobanova, A., Liersch, S., Tàbara, J.D., Koch, H., Hattermann, F.F., Krysanova, V., 2017. Harmonizing human-hydrological system under climate change: A scenario-based approach for the case of the headwaters of the Tagus River. J. Hydrol. 548, 436–447. https://doi.org/10.1016/j.jhydrol.2017.03.015.

López-Moreno, J.I., Vicente-Serrano, S.M., Begueria, S., Garcia-Ruiz, J.M., Portela, M.M., Almeida, A.B., 2009. Dam effects on droughts magnitude and duration in a transboundary basin: The lower river tagus, pain and Portugal. Water Resour. Res. 45. https://doi.org/10.1029/2008WR007198

Lorenzo-Lacruz, J., Vicente-Serrano, S.M., López-Moreno, J.I., Beguería, S., García-Ruiz, J.M., Cuadrat, J.M., 2010. The impact of droughts and water management on various hydrological systems in the headwaters of the Tagus River (central Spain). J. Hydrol. 386, 13–26. https://doi.org/10.1016/j.jhydrol.2010.01.001

Loucks DP, van Beek, E, 2017. Water resource systems planning and management: An introduction to methods, models, and applications, Springer. Deltares and UNESCO-IHE, New York, USA

Lyu, F., Zhang, H., Dang, C., Gong, X., 2023. A novel framework for water accounting and auditing for efficient management of industrial water use. J Clean Prod 395, 136458. https://doi.org/10.1016/J.JCLEPRO.2023.136458

Mckee, T.B., Doesken, N.J., Kleist, J., 1993. The relationship of drought frequency and duration to time scales. Eighth Conf. Appl. Climatol. 17–22.

Mekonnen, M.M., Hoekstra, A.Y., 2016. Sustainability: Four billion people facing severe water scarcity. Sci. Adv. 2. https://doi.org/10.1126/SCIADV.1500323

Milanes, M., 2017. Transboundary Water Management Along the Tagus River Basin in the Iberian Peninsula: Sustainable Water Allocation of the Aqueduct Tagus-Segura, in: Management of Transboundary Water Resources under Scarcity: A Multidisciplinary Approach. World Scientific Publishing Company, New Jersey, pp. 237–275.

Milano M, Ruelland D, Dezetter A, et al (2013) Modeling the current and future capacity of water resources to meet water demands in the Ebro basin. J Hydrol 500:114–126. https://doi.org/10.1016/j.jhydrol.2013.07.010

Mishra, A.K., Singh, V.P., 2010. A review of drought concepts. J. Hydrol. 391, 202–216. https://doi.org/10.1016/J.JHYDROL.2010.07.012

Molden, D., 1997. Accounting for water use and productivity, SWIM Paper 1. Colombo, Sri Lanka: International Irrigation Management Institute.

Molden, D., Sakthivadivel, R., 2010. Water Accounting to Assess Use and Productivity of Water. http://dx.doi.org/10.1080/07900629948934 15, 55–71. https://doi.org/10.1080/07900629948934

Momblanch, A., Andreu, J., Paredes-Arquiola, J., Solera, A., Pedro-Monzonís, M., 2014. Adapting water accounting for integrated water resource management. The Júcar Water Resource System (Spain). J Hydrol (Amst) 519, 3369–3385. https://doi.org/10.1016/J.JHYDROL.2014.10.002

Momblanch, A., Pedro-Monzonís, M., Solera, A., Andreu, J., 2018. Water Accounting for Integrated Water Resources Management: Experiences and Recommendations. Advances in Chemical Pollution, Environmental Management and Protection 3, 63–96. https://doi.org/10.1016/BS.APMP.2018.08.001

Ohlsson, L., 2000. Water conflicts and social resource scarcity. Phys. Chem. Earth, Part B Hydrol. Ocean. Atmos. 25, 213–220. https://doi.org/10.1016/S1464-1909(00)00006-X

Palmer, W.C., 1965. Meteorological Drought, US. Department of Commerce. U.S. Weather Bureau, Washington, D.C.

Pedro-Monzonís, M., Jiménez-Fernández, P., Solera, A., Jiménez-Gavilán, P., 2016a. The use of AQUATOOL DSS applied to the System of Environmental-Economic Accounting for Water (SEEAW). J Hydrol (Amst) 533, 1–14. https://doi.org/10.1016/j.jhydrol.2015.11.034

Pedro-Monzonís, M., Solera, A., Ferrer, J., Andreu, J., Estrela, T., 2016b. Water accounting for stressed river basins based on water resources management models. Science of the Total Environment 565, 181–190. https://doi.org/10.1016/j.scitotenv.2016.04.161

Pedro-Monzonís, M., Solera, A., Ferrer, J., Estrela, T., Paredes-Arquiola, J., 2015. A review of water scarcity and drought indexes in water resources planning and management. J. Hydrol. 527, 482–493. https://doi.org/10.1016/j.jhydrol.2015.05.003

Pedro-Monzonís M, Solera A, Ferrer J, et al (2016) Water accounting for stressed river basins based on water resources management models. Sci Total Environ 565:181–190. https://doi.org/10.1016/j.scitotenv.2016.04.161

Pérez-Blanco, C.D., Essenfelder, A.H., Gutiérrez-Martín, C., Pérez-Blanco, C.D., Essenfelder, A.H., Gutiérrez-Martín, C., 2020. A tale of two rivers: Integrated hydro-economic modeling for the evaluation of trading opportunities and return flow externalities in inter-basin agricultural water markets. J. Hydrol. 584. https://doi.org/10.1016/J.JHYDROL.2020.124676

Perry, C., 2010. Accounting for water use: Terminology and implications for saving water and increasing production. Agric Water Manag 98, 1840–1846. https://doi.org/10.1016/j.agwat.2010.10.002

Postel, S.L., 2000. Entering an era of water scarcity: The challenges ahead. Ecol. Appl. 10, 941–948. https://doi.org/10.1890/1051-0761(2000)010[0941:EAEOWS]2.0.CO;2

Raskin, P., Gleick, P., Kirshen, P., Pontius, G., Strzepek, K., 1997. Comprehensive Assessment of the Freshwater Resources on the World. Water Futures: Assessment of Long-range Patterns and Problems. Stockholm Environment Institute, Sweden.

Rijsberman, F.R., 2006. Water scarcity: Fact or fiction? Agric. Water Manag. 80, 5–22. https://doi.org/10.1016/j.agwat.2005.07.001

Rockström, J., Falkenmark, M., 2015. Agriculture: Increase water harvesting in Africa. Nature 519, 283–285. https://doi.org/10.1038/519283A

Sabater, S., Feio, M.J., Graça, M.A.S., Muñoz, I., Romaní, A.M., 2009. The Iberian Rivers, in: Rivers of Europe. Elsevier Ltd, pp. 113–149. https://doi.org/10.1016/B978-0-12-369449-2.00004-7

Saft, M., Western, A.W., Zhang, L., Peel, M.C., Potter, N.J., 2015. The influence of multiyear drought on the annual rainfall-runoff relationship: An Australian perspective. Water Resour. Res. 51, 2444–2463. https://doi.org/10.1002/2014WR015348

Sandoval-Solis S, McKinney DC, Loucks DP (2011) Sustainability index for water resources planning and management. J Water Resour Plan Manag 137:381–390. https://doi.org/10.1061/(ASCE)WR.1943-5452.0000134

Savenije, H.H.G., 2000. Water scarcity indicators; the deception of the numbers. Phys. Chem. Earth, Part B Hydrol. Ocean. Atmos. 25, 199–204. https://doi.org/10.1016/S1464-1909(00)00004-6

Schulte, P., Morrison, J., 2014. Driving harmonization of water-related terminology. Pacific Institute, Oakland, CA.

Shukla, S., Wood, A.W., 2008. Use of a standardized runoff index for characterizing hydrologic drought. Geophys. Res. Lett. 35. https://doi.org/10.1029/2007GL032487

Sondermann, M.N., Oliveira, R.P. de, 2021. A Shared Vision on the Transboundary Water Management Challenges of the Tagus River Basin. Water Resources Management 35, 4647–4664. https://doi.org/10.1007/S11269-021-02973-6

Sondermann, M.N., Oliveira, R.P., 2022a. Using the WEI+ index to evaluate water scarcity at highly regulated river basins with conjunctive uses of surface and groundwater resources. Science of The Total Environment 836, 155754. https://doi.org/10.1016/J.SCITOTENV.2022.155754

Sondermann, M.N., Oliveira, R.P., 2022b. Climate Adaptation Needs to Reduce Water Scarcity Vulnerability in the Tagus River Basin. Water (Basel) 14. https://doi.org/10.3390/W14162527

Sondermann, M.N., Oliveira, R.P., 2025 (submitted). A Water Accounting Perspective on Water Scarcity in the Transboundary Tagus River Basin

Sullivan, C., 2002. Calculating a Water Poverty Index. World Dev. 30, 1195–1210. https://doi.org/10.1016/S0305-750X(02)00035-9

Sun, C., Zhou, X., 2020. Characterizing Hydrological Drought and Water Scarcity Changes in the Future: A Case Study in the Jinghe River Basin of China. Water 12. https://doi.org/10.3390/w12061605

UN, 2012. SEEA-Water|System of Environmental-Economic Accounting for Water asdf. United Nations.

UNCCD, 2024. The Global Threat of Drying Lands: Regional and global aridity trends and future projections, United Nations Convention to Combat Desertification.

Vanham, D., Hoekstra, A.Y., Wada, Y., Bouraoui, F., de Roo, A., Mekonnen, M.M., van de Bund, W.J., Batelaan, O., Pavelic, P., Bastiaanssen, W.G.M., Kummu, M., Rockström, J., Liu, J., Bisselink, B., Ronco, P., Pistocchi, A., Bidoglio, G., 2018. Physical water scarcity metrics for monitoring progress towards SDG target 6.4: An evaluation of indicator 6.4.2 "Level of water stress." Sci. Total Environ. 613–614, 218–232. https://doi.org/10.1016/J.SCITOTENV.2017.09.056

Vicente-Serrano, S.M., Beguería, S., López-Moreno, J.I., 2010. A multiscalar drought index sensitive to global warming: The standardized precipitation evapotranspiration index. J. Clim. 23, 1696–1718. https://doi.org/10.1175/2009JCLI2909.1

Vicente, D.J., Rodríguez-Sinobas, L., Garrote, L., Sánchez, R., 2019. Applying water accounting methods through statistical data and simulation models: The Douro transboundary watershed. Adv. Chem. Pollution, Environ. Manag. Prot. 3. https://doi.org/https://doi.org/10.1016/bs.apmp.2018.06.001

Wang, M., Bodirsky, B.L., Rijneveld, R., Beier, F., Bak, M.P., Batool, M., Droppers,B., Popp, A., van Vliet, M.T.H, Strokal, M. 2024. A triple increase in global river basins with water scarcity due to future pollution, Nature Communications, https://doi.org/10.1038/s41467-024-44947-3.

WEF, 2020. The Global Risks Report 2020. World Economic Forum, Geneva, Switzerland.

Wu, J., Miao, C., Tang, X., Duan, Q., He, X., 2018. A nonparametric standardized runoff index for characterizing hydrological drought on the Loess Plateau, China. Glob. Planet. Change 161, 53–65. https://doi.org/10.1016/J.GLOPLACHA.2017.12.006

Zal, N., Zachos, A., Bariamis, G., Baltas, E., 2017. Use of Freshwater Resources in Europe - An assessment based on water quantity accounts. European Topic Centre on Inland, Coastal and Marine Waters – ETC/ICM, Magdeburg, Germany.

Zeng, Z., Liu, J., Savenije, H.H.G., 2013. A simple approach to assess water scarcity integrating water quantity and quality. Ecol. Indic. 34, 441–449. https://doi.org/10.1016/j.ecolind.2013.06.012

Zou, L., Xia, J., She, D., 2018. Analysis of Impacts of Climate Change and Human Activities on Hydrological Drought: a Case Study in the Wei River Basin, China. Water Resour. Manag. 32. https://doi.org/10.1007/s11269-017-1877-1