



Using water footprint concepts for water security assessment of a basin under anthropogenic pressures



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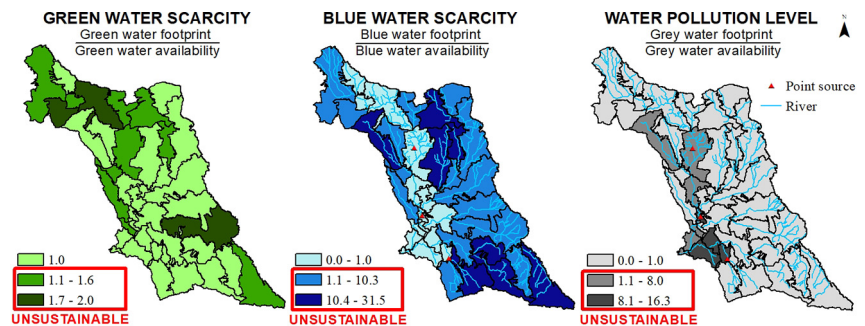
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HIGHLIGHTS

- Water security is analysed by using the water footprint (WF) sustainability approach.
- The WF is assessed for agriculture and treated wastewater disposal.
- High spatio-temporal variability of the WF within a basin is considered.
- Pollution is the main factor affecting surface water security.
- Response formulations to achieve water security are identified.

GRAPHICAL ABSTRACT



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ABSTRACT

The evaluation of water shortages and pollution levels is crucial for watershed management and sustainable development. This paper proposes a water footprint (WF) sustainability assessment approach to analyse water security in a river basin under human pressures. The methodology involves a comprehensive assessment of the current water security at different spatial and temporal levels, and identifies suitable response formulations to achieve sustainability. Field surveys and measurements (streamflow, water quality) were carried out, and the Soil and Water Assessment Tool model was used for assessing water balance components and water quality. The study was carried out in the Canale d'Aiedda river basin (Taranto, Italy), which is part of the 'area of environmental crisis' of Taranto, which requires remediation of surface water, groundwater, soil and subsoil. Considering all the anthropogenic activities in the basin, including agriculture and the treated effluent disposed of via wastewater treatment plants (WWTPs), the average WF was $213.9 \text{ Mm}^3 \text{ y}^{-1}$, of which 37.2%, 9.2% and 53.6% comprised respectively for WF_{green} , WF_{blue} and WF_{grey} . The WF sustainability assessment revealed that pollution was the main factor affecting surface water security. In particular, point sources contributed with 90% to the total WF_{grey} , and lower pollutant thresholds should be fixed for effluent from WWTPs in order to increase water quality of the receiving water body. In addition, for assuring water security the extension of the natural areas should be increased to support biodiversity in the river basin and soil management strategies should be improved to allow more water to be retained in the soil and to reduce nutrients in surface runoff. This study demonstrates that the WF sustainability assessment is a feasible approach for integrated water resources management, as well as offering a much broader perspective on how water security can be achieved in a Mediterranean basin affected by multiple anthropogenic stressors.

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1. Introduction

Fresh water is essential for human life, for a thriving economy and for supporting natural ecosystems (Oki and Kanai, 2006). Climate change, increasing demand, and pollution could put the availability of water resources for future generations at risk (De Girolamo et al., 2017a; Pophare et al., 2014; Rosa et al., 2018). International organisations, aware of this risk, have introduced the concept of water security (GWP, 2019; UNESCO and UNESCO i-WSSM, 2019). This has received increasing attention in policy and scientific debates in recent decades (Cook and Bakker, 2012). Water security is defined as “the availability of an acceptable quantity and quality of water for health, livelihoods, ecosystems and production, coupled with an acceptable level of water-related risks to people, environments and economies” (Grey and Sadoff, 2007). The term refers to economic welfare, social equity, long-term sustainability and water-related risks (Hoekstra et al., 2018). Water security is a conceptual framework for analysing water systems that is able to integrate multidisciplinary concerns across spatial scales (from local to global). The main challenges of this broad conceptual approach are the implementation and management processes (Cook and Bakker, 2012).

Water security assessments based on the concept of the water footprint (WF) are becoming globally largely accepted (Giri et al., 2018; Hoekstra, 2017; Veetil and Mishra, 2016). The WF quantifies the direct and indirect volume of water required by human activities in a specific geographical area during a unit of time (Hoekstra et al., 2011). It is partitioned into three components: green, blue and grey. The green WF (WF_{green}) refers to the amount of precipitation temporarily accumulated on vegetation and/or in soil, and that is available for evapotranspiration through the vegetation and soil (Falkenmark, 2003). The blue WF (WF_{blue}) is the volume of fresh water provided along a supply chain of products and/or services (Veetil and Mishra, 2016). The grey WF (WF_{grey}) refers to the degree of water pollution induced by human activities, and is defined as the volume of fresh water required to dilute a load of pollutants so that the receiving water body can assimilate those loads without compromising the achievement or maintenance of specific water-quality standards (Franke et al., 2013).

In recent decades, the WF concept has been used to evaluate the sustainability of water resources management (Mikosch et al., 2020), and the common spatial unit considered in such evaluations is the river basin (Liu et al., 2017; Zang et al., 2012). The WF sustainability assessment is a reliable way to estimate current water shortages and pollution apportionment, from point sources (PSs) and diffuse sources (DSs), and for identifying local hotspots in a river basin (De Girolamo et al., 2019; Giri et al., 2018; Pellicer-Martínez and Martínez-Paz, 2016; Salmoral et al., 2017). In addition, the WF sustainability assessment can be used to compare different cropping systems (Mikosch et al., 2020; Pellegrini et al., 2016) and wastewater treatment plant designs (Morera et al., 2016), in order to identify suitable response formulations to achieve water security (D'Ambrosio et al., 2020; Lovarelli et al., 2018) and to develop climate change adaptation strategies (Haida et al., 2019; Veetil and Mishra, 2018).

The present study is a continuation of previous work that tested a methodological approach for assessing the sustainability of water use at the basin scale through WF indicators (D'Ambrosio et al., 2020). Those authors pointed out that operating at the basin scale and on a yearly basis constitutes the first step of a sustainability assessment, and that further analysis at a higher spatial and temporal resolution is also needed. Indeed, despite the broad applicability of the WF concept, several studies have highlighted that improving the spatial and temporal accuracy of sustainability assessments is necessary in order to determine differences within a basin and to identify local-scale, unsustainable situations (Liu et al., 2017; Pellicer-Martínez and Martínez-Paz, 2016). Salmoral et al. (2017) assessed the water-related impacts of agriculture with a spatiotemporal WF assessment to evaluate the current status of streamflow, soil water and the assimilability of sediments by streamflow.

Mikosch et al. (2020) suggest using a subbasin scale and monthly resolution for meaningful water security assessment. However, to the best of our knowledge, there are no case studies in the literature that have provided a complete WF sustainability assessment at high spatial and temporal resolution (i.e. subbasin scale, monthly time scale) or studies that have included all the anthropogenic activities in operation in a basin.

PSs have been recognised as ‘critical pressures’, especially in intermittent rivers due to their limited dilution capacity in dry seasons (De Girolamo and Lo Porto, 2020). Studies focusing on the WF_{grey} of treated wastewater, disposed of into a river network, are scarce because it is often assumed that treated wastewater complies with the water-quality standards associated with the receiving water body, and thus that pollution from these is negligible (Johnson and Mehrvar, 2019). Conversely, wastewater treatment plant (WWTP) effluent's pollutant thresholds are higher than the concentrations of nutrients allowed by the Water Framework Directive (European Parliament and Council of the European Union, 2000) for the purpose of maintaining a good environmental water-quality standard for the receiving surface water body. Consequently, the WF_{grey} concept cannot neglect the PSs of pollutants, especially in basins with temporary river systems.

This study aimed to define a methodological approach for analysing water security at a high spatial and temporal scale within a basin that receives both PSs and DSs of pollutants. The proposed integrated modelling framework was defined specifically for areas with an intermittent river network. It integrates the WF sustainability assessment, as proposed by the Water Footprint Network (Franke et al., 2013; Hoekstra et al., 2011), with a hydrological and water-quality model. The integrated modelling framework was applied to a Mediterranean basin (Canale d'Aiedda Basin, Italy), where the hydrological regime has been altered, and the water quality impaired, by agricultural activities and treated wastewater disposal (D'Ambrosio et al., 2019). The objectives were: i) to record the spatial and temporal variability of WF_{green} , WF_{blue} and WF_{grey} by applying the Soil and Water Assessment Tool (SWAT) model; ii) to assess the water security by means of WF sustainability indicators; and iii) to identify local hotspots and suitable strategies to achieve surface-water security across the basin (i.e. response formulation).

The results of the study offer a much broader perspective on how water security can be achieved in a Mediterranean basin affected by multiple anthropogenic stressors. Hence, the developed, integrated analytical framework constitutes a useful guide for improving sustainable usage, protecting water resources and supporting the making of suitable water policies for semi-arid environments.

2. Study area

The study area included part of the Canale d'Aiedda Basin (Apulia Region, SE Italy) (Fig. 1). The whole basin covers an area of 360 km²; however, the area that effectively contributes to surface runoff, which was considered as the study area, is 222 km². The northern and eastern parts are karstic (i.e. fractured limestone) areas, which recharge a deep limestone aquifer, and not contribute to surface runoff (D'Ambrosio et al., 2019).

The study area is characterised by 12 different soil types, varying from silty clay to sandy loam, which were divided into three hydrological soil groups (B, C, D) (Regione Puglia, 2001). The elevation ranged from 0 to 517 m a.s.l., with a mean value of 168 m a.s.l., and a mean slope of 2.7°.

The climate, land cover and management practices are typical of the Mediterranean region. The climate is characterised by wet winters and warm, dry summers. Mean monthly temperatures (2000–2013) ranged between 8.1 °C in January and 27.9 °C in August. In general, precipitation occurs in the autumn/winter months (October–March); in spring/summer, it is mostly concentrated in a few events of short duration and high intensity. The mean annual rainfall (2000–2013) ranged

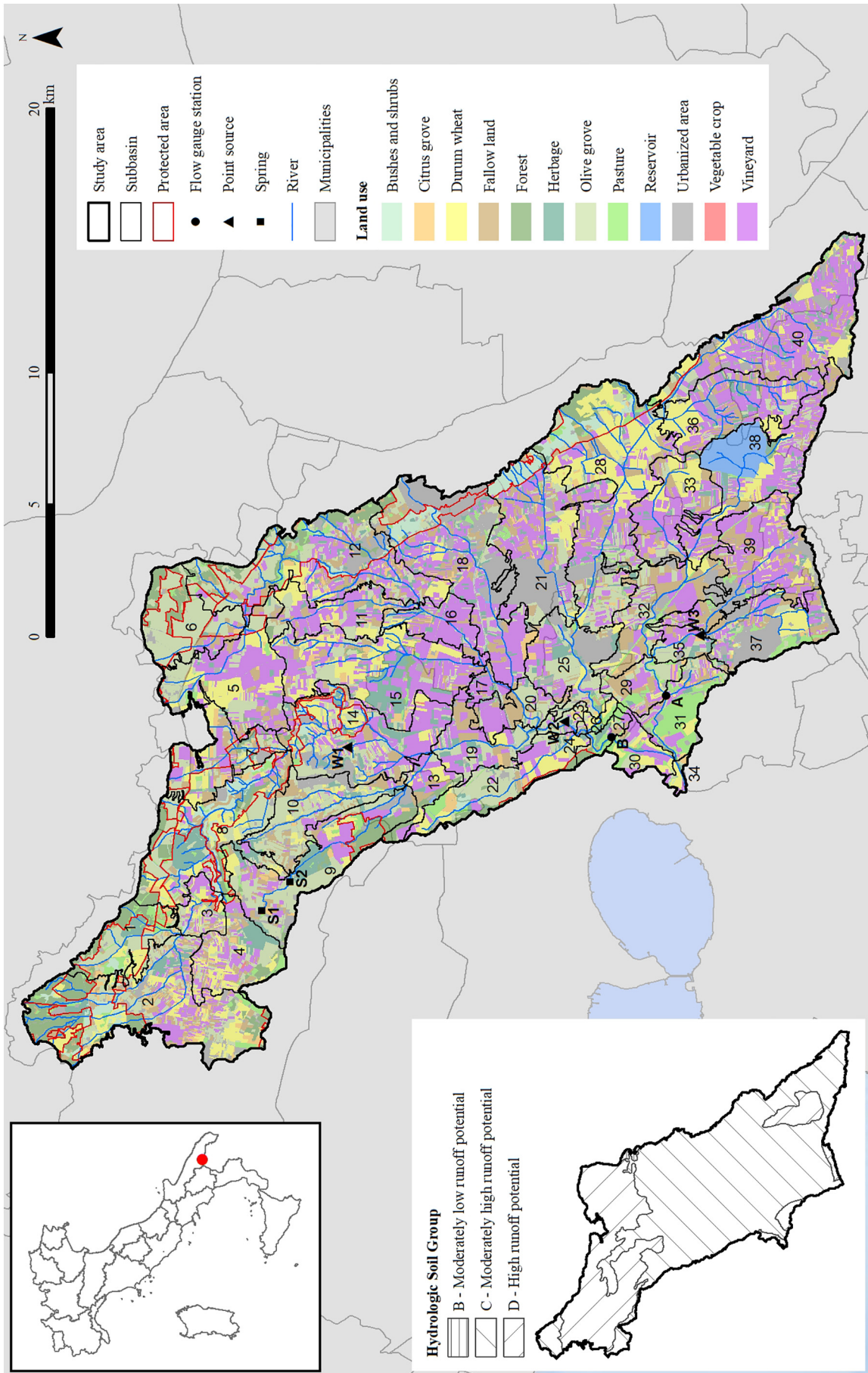


Fig. 1. Location, land use and hydrological soil groups identified in the study area. The thick black line delimits the area contributing to the surface runoff, which was divided into 40 subbasins (thin black line) for the purpose of applying the SWAT model. Regional Park and Forest Reserve are identified (thin red line).

from 601 to 865 mm and exhibits spatial variability throughout the basin.

In the basin, the main economic activity is extensive agriculture. A limited number of breeding farms is mostly localised in the north of the study area. Vineyards (36.3%), olive groves (24.5%) and durum wheat in rotation with herbage and fallow land (28.1%) constitute the main land uses (Fig. 1). Citrus groves and vegetables represent minority land uses. These latter, in addition to the vineyards and olive groves, are irrigated. The main irrigation source is deep groundwater. The urban area accounts for 3.9% of the basin. Deciduous (e.g. holm oak) and coniferous forest are present (2.7%), as well as pasture (e.g. Mediterranean maquis) (2.4%) and bushes and shrubs (e.g. myrtus, mastic bush) (0.9%). These natural areas mainly fall within Regional Park and Forest Reserve (i.e. Terra delle Gravine, Bosco delle Pianelle).

The river network is heavily modified, with concrete river banks being present (D'Ambrosio et al., 2019). In addition, the hydrological regime has been altered by the discharge from three WWTPs – Montemesola (W1), Monteiasi (W2) and San Giorgio Ionico (W3) – the contributions from which assure permanent downstream streamflow (Fig. 1). Meanwhile, an intermittent regime is recognisable upstream.

The river network flows into the second inlet of the Mar Piccolo, which is an inner sea basin (20.72 km²) considered to be a Site of Community Importance (IT9130004) due to its ecological relevance, as well as a Contaminated Site of National Interest due to the high pollution level (National Law no. 426/98). Indeed, mussel farming, fisheries, industry, agricultural activities, WWTP effluent, landfill locations, ports and a shipyard are a great threat to the Mar Piccolo's ecology. Interventions involving environmental requalification are planned to reduce the anthropogenic impact in the Mar Piccolo. These interventions concern an inland area, draining into the sea basin, designated an 'area of environmental crisis' (564 km²), which partially includes the study area.

3. Material and methods

The integrated modelling framework proposed for water security assessment consists of three steps: 1) evaluating the water balance components, water quality and source apportionment through modelling; 2) assessing the WF sustainability; and 3) formulating responses to achieve water security in hotspot areas. The second and third steps are based on the WF concepts of green water scarcity (WS_{green}), blue water scarcity (WS_{blue}) and water pollution level (WPL) (Hoekstra et al., 2011), which were analysed at the subbasin and basin scales, considering monthly and yearly time scales. Meanwhile, the hydrological and water-quality model was used to evaluate the variables (i.e. water consumption, nutrient load added to the river for each crop) necessary for the WS_{green} , WS_{blue} and WPL calculations (Fig. 2). Each step is explained below. The procedure was applied to the Canale d'Aiedda Basin.

3.1. Modelling hydrology and water quality

Several catchment-scale hydrological and water-quality models have been used, at different geographical scales worldwide, to model water-balance components (e.g. precipitation, evapotranspiration, infiltration, percolation, surface runoff), crop growth, and sediment and nutrient loads added to surface and groundwater (Pulighe et al., 2019). The SWAT (Arnold et al., 2012), applied in the present study, is one of the most used models (Curk et al., 2020; Fu et al., 2019). This model has also been successfully applied in Mediterranean basins (Brouziyne et al., 2017; De Girolamo and Lo Porto, 2020; Ricci et al., 2020) to assess the spatiotemporal variation of WF components (D'Ambrosio et al., 2020; De Girolamo et al., 2019; Salmoral et al., 2017).

SWAT is a semi-distributed, process-based model that divides the whole watershed into subbasins, based on topographic data, so that the flow within each subbasin is directed to a single point, known as

the subbasin outlet (Arnold et al., 2012). Each subbasin is then further subdivided into hydrological response units (HRUs), which have unique land-use, soil and slope features. The HRU is the basic unit for estimating water-balance components, and sediment and nutrient loads in model computations. In the current study, thresholds for land use (10%), soils (10%) and slopes (20%) were considered for HRU delineation (Neitsch et al., 2009), which resulted in 271 HRUs distributed over 40 subbasins. Twenty-one, 11 and three classes were identified for the land use, soil type and slope, respectively. Each land-use class was associated to a land cover/plant types within the SWAT "land cover/plant growth database", in order to model water balance components considering the development stage of the plant (Neitsch et al., 2009). The outputs of the SWAT model were used to assess the WF sustainability (Fig. 2).

The Hargreaves and Soil Conservation Service Curve Number methods were used to evaluate the potential evapotranspiration and surface runoff, respectively. Interviews with farmers were performed to determine the management practices (i.e. timing of operations, fertiliser application rate, irrigation supply and grazing) for each land use.

The input data required by the SWAT, and used in the current study for setting up the model, are provided in Table 1. Details on discharge volumes and the main pollutant concentrations are provided in Table 2 for each PS (i.e. treated effluent from WWTPs, springs).

The model was run at a daily time step from January 2008 to March 2018, with a 3-year (2008–2010) warm-up period used to initialise the model parameters. Model calibration was performed between August 2017 and March 2018. Daily streamflow (Q), total suspended solids (TSS), total nitrogen (TN) and total phosphorus (TP) loads, measured at gauge stations A and B (Fig. 1), were used for the calibration. The full measured time series was used for the calibration, since the limited available data could not be meaningfully split into two subsets to be used for the calibration and validation, respectively. Indeed, Arsenault et al. (2018) pointed out the need to use the full time-series in the calibration step and to disregard the validation aspects under certain conditions. In this case, to make a robust calibration, a split-in-space strategy was adopted by using measured data in two gauging stations (A and B in Fig. 1). Details on the monitoring activities (from which the data for the model calibration were obtained) are reported in D'Ambrosio et al. (2019, 2020). Table 3 provides the mean results from gauges A and B.

The Sequential Uncertainty Fitting (SUFI-2) procedure in SWAT-CUP (Rouholahnejad et al., 2012) was used to auto-calibrate 104 model parameters for both gauges (D'Ambrosio et al., 2020). The calibration protocol indicated by Abbaspour (2015) was applied. The model performance was evaluated using the coefficient of determination (R^2), the Nash-Sutcliffe efficiency (NSE) coefficient and the percent deviation (PBIAS), with the criteria of acceptability being fixed as suggested by Moriasi et al. (2007).

Following the calibration, the SWAT was run without crop irrigation and WWTP input, in order to calculate the WS_{green} and WS_{blue} , respectively (Fig. 2). Other simulations were executed that considered different WWTP effluent conditions in order to identify a response formulation towards achieving sustainability, where necessary.

3.2. Water footprint sustainability assessment

3.2.1. Green water scarcity

Although Savenije (2000) has already underscored the importance of the WF_{green} sustainability assessment, most of the studies on water scarcity have focused only on blue water resources (Hoekstra et al., 2019). According to Hoekstra et al. (2011), the WF_{green} sustainability assessment is based on the evaluation of WS_{green} , which is defined as the ratio of the total WF_{green} associated with crop production to the green water availability (WA_{green}) within a certain period. In the study area, vineyard, olive grove, citrus groves, durum wheat and herbage were the productive crops. The total WF_{green} represents the volume of green

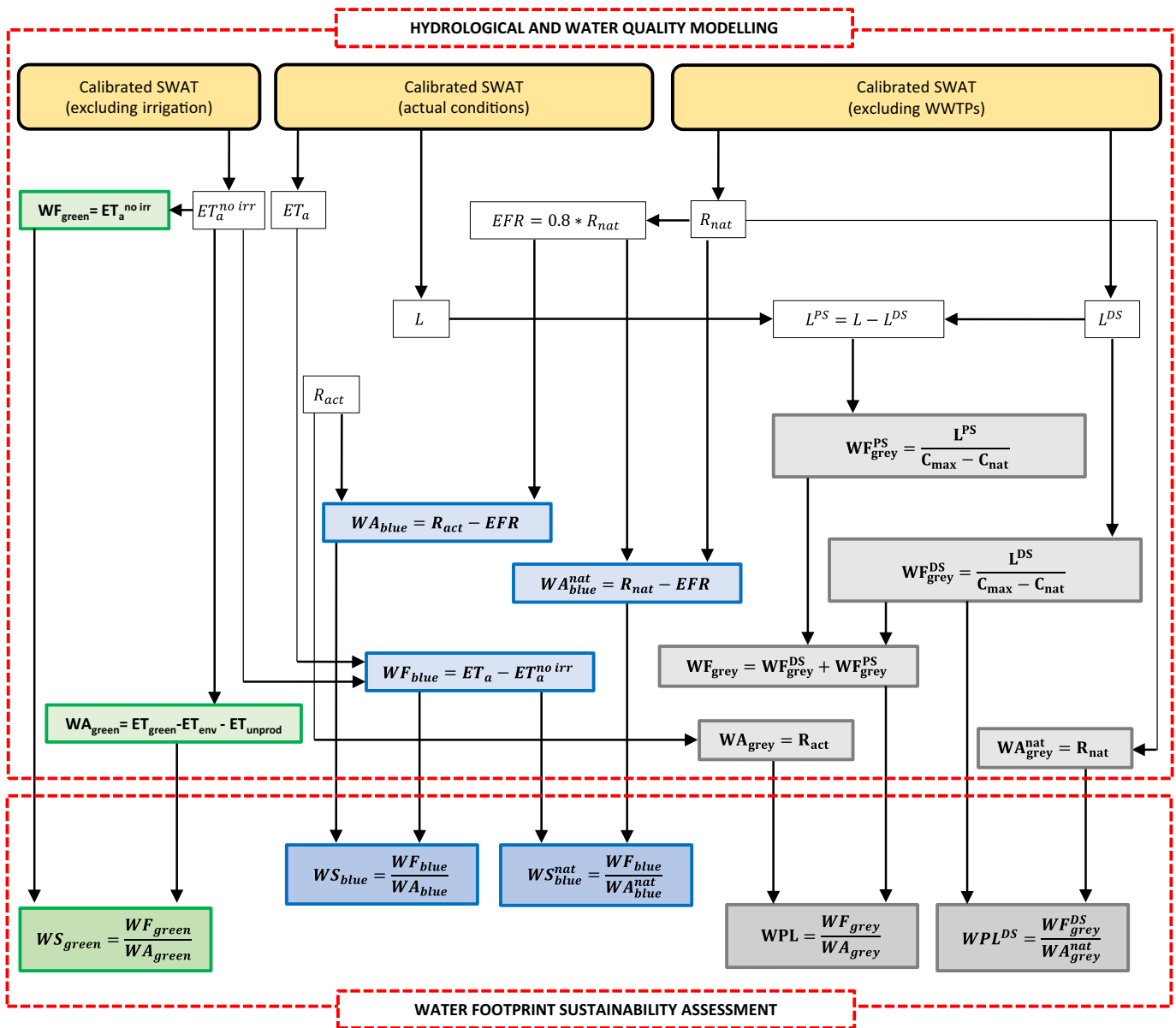


Fig. 2. Scheme of the methodological approach used in the water security assessment.

water consumed through evapotranspiration processes during the growing period by the productive crops cultivated in the analysed spatial unit (Pellicer-Martínez and Martínez-Paz, 2016). It is equal to the actual crop evapotranspiration of productive crops, assuming that the soil does not receive any irrigation ($ET_a^{no\ irr}$) (Hoekstra et al., 2011). To estimate the $ET_a^{no\ irr}$, the SWAT model was run for the current land use using the best model parameters' values identified with the calibration procedure ("calibrated SWAT model") without crop irrigation.

In the study area, durum wheat and herbage are the only temporary crops. The former is sown in late autumn (1st December in the SWAT simulation) and harvested in early summer (30th June in the SWAT simulation), the latter sown in autumn (1st November) and also harvested in early summer (30th June). Thus, only those months (from December to June and from November to June) were considered in the WF_{green} assessment of durum wheat and herbage, respectively. In the remaining months, the soil was bare. Meanwhile, vineyard (harvesting date: 15th August), olive grove (harvesting date: 15th October), and citrus groves (harvesting date: 15th October), contributed throughout the whole year in determining the WF_{green} , being permanent trees.

The WA_{green} was calculated by:

$$WA_{green} = ET_{green} - ET_{env} - ET_{unprod} \quad (1)$$

ET_{green} is the volume of green water available at the basin or sub-basin scale (actual evapotranspiration for the current land use excluding irrigation). Similarly to the flow requirements to support aquatic ecosystems, the ET_{env} is defined as the green water flow to be reserved for nature, needed to support biodiversity. In the present work, land reserved for nature was very limited (6%), thus, it was assumed to achieve the Aichi Biodiversity Target (ABT) 11, which entails expanding the protected area network to at least 17% of the terrestrial world by 2020 (www.cbd.int/sp/targets).

ET_{env} was computed as the actual evapotranspiration of natural vegetation present in the area (i.e. bushes and shrubs, forest and pasture) that was incremented in order to achieve the ABT 11 target (17% of the catchment area). In particular, it was supposed to convert part of arable lands (i.e. durum wheat and herbage) falling within Regional Park and Reserve (Fig. 1) into forests. The computation of ET_{env} was performed by using the SWAT model assuming the default values of crop

Table 1
Source and spatial resolution of input data used for SWAT model set-up.

Input	Source	Resolution
Digital Terrain Model	Puglia Region (http://www.sit.puglia.it)	8 × 8 m
Land use map	Puglia Region (http://www.sit.puglia.it) National Agricultural Census (http://censimentoagricoltura.istat.it/index.php?id=73)	1:5000
Soil map and database	Puglia Region (2001)	–
	JRC-ESDAC (https://esdac.jrc.ec.europa.eu/resource-type/datasets)	1:100000
Point sources	Apulian Water Authority (Personal communication) (W1, W2, W3)	500 × 500 m
	Regional Agency for Environmental Protection (http://www.arpa.puglia.it/web/guest/depuratori) (W1, W2, W3)	–
	Sampling and chemical analysis of treated effluents (W2, W3)	–
	Puglia Region (http://www.sit.puglia.it) (S1, S2)	–
Meteorological data	Civil Protection Service - Puglia Region (https://protezionecivile.puglia.it/)	–
	Regional Agency for Irrigation and Forestry Activities (http://www.agrometeopuglia.it/)	–
	National Agency for New Technologies, Energy and Sustainable Economic Development (http://www.solaritaly.enea.it/)	–
Agricultural practices	Interviews with farmers and agricultural advisors (D'Ambrosio et al., 2019).	–

Table 2
Mean (January 2012–March 2018) volumes (V_{eff}), total suspended solids (TSS), total nitrogen ($C_{\text{eff}}^{\text{TN}}$) and total phosphorous concentrations ($C_{\text{eff}}^{\text{TP}}$) in effluent disposed of in the river network by wastewater treatment plants (W1, W2, W3) and springs (S1, S2). N is the number of water samples analysed to determine the mean TSS, TN and TP concentrations. The limits fixed by national legislation for wastewater treatment plant discharge into temporary streams (i.e. $C_{\text{eff}}^{\text{TN}^a}$, $C_{\text{eff}}^{\text{TP}^a}$) are also provided.

Parameter	V_{eff} ($\text{m}^3 \text{d}^{-1}$)	TSS (mg L^{-1})	$C_{\text{eff}}^{\text{TN}}$ (mg L^{-1})	$C_{\text{eff}}^{\text{TP}}$ (mg L^{-1})	N	$C_{\text{eff}}^{\text{TN}^a}$ (mg L^{-1})	$C_{\text{eff}}^{\text{TP}^a}$ (mg L^{-1})
W1	743	5.2 ± 5.0	15.2 ± 7.3	2.7 ± 2.3	75	15	2
W2	6444	55.2 ± 123.7	18.7 ± 12.2	3.0 ± 2.7	89	15	2
W3	4182	39.4 ± 80.2	19.2 ± 12.8	2.1 ± 2.7	89	15	2
S1 ^a	172.8	–	14.4	–	–	–	–
S2 ^a	86.4	–	9.2	–	–	–	–

^a Measures performed by the National Research Institute for Hydrogeological Protection (Istituto di Ricerca per la Protezione Idrogeologica, 2014).

coefficient for bushes and shrubs, forest and pasture (garrigue). Meanwhile, ET_{unprod} is the evapotranspiration from unproductive land (i.e. urban areas, reservoirs), which is not available for cultivation, and it also includes fallow land that is essential for restoring and improving soil fertility for next cultivation period.

Fig. 3 shows the allocation of ET_{green} , WF_{green} , ET_{unprod} and ET_{env} from the study area.

Being a ratio, the WS_{green} indicates the degree of green water use throughout the analysed spatial unit. A $WS_{\text{green}} > 1$ means that the available green water is fully consumed by productive crops (Hoekstra et al., 2011), which should be partially converted in natural areas to support biodiversity; < 1 (or = 1) indicates a suitable WS_{green} , where there is enough green water to meet the productive and natural crop demands.

3.2.2. Blue water scarcity

The environmental sustainability of the WF_{blue} is evaluated by means of the WS_{blue} , which is the ratio of the total WF_{blue} to the blue water availability (WA_{blue}) (Hoekstra et al., 2011). Based on data availability, this study focused only on the surface water, ignoring the sustainability of groundwater use. In addition, only the agricultural water use was considered as water for industrial, commercial and domestic uses was imported from outside the Canale d'Aiedda Basin.

The total WF_{blue} is the amount of fresh water consumed in crop production (i.e. irrigation). This fresh water is 'consumed' because it does not return to the source in the form of return flow, being used up via

Table 3
Observed mean streamflow (Q), total suspended solid (TSS), total nitrogen (TN) and total phosphorous (TP) (August 2017–March 2018) values used for SWAT model calibration.

Parameter	Station A	Station B
Q ($\text{m}^3 \text{d}^{-1}$)	2710.64 ± 1663.85	6062.78 ± 6488.36
TSS (t d^{-1})	0.23 ± 0.37	2.67 ± 6.18
TN (Kg d^{-1})	23.22 ± 23.73	82.09 ± 106.12
TP (Kg d^{-1})	3.79 ± 5.27	8.01 ± 4.80

evapotranspiration, incorporated into products and/or returned to another catchment or to the sea. In the study area, only the olive groves, vineyards and citrus groves are irrigated. The WF_{blue} is obtained from the difference between the actual crop evapotranspiration, considering current management practices (ET_a) and $ET_a^{\text{no irr}}$ (Hoekstra et al., 2011). Thus, the calibrated SWAT model output was used to obtain the ET_a .

The WA_{blue} is the maximum volume of water that can be abstracted from a river without harming its ecosystems (Bonamente et al., 2017). This was assessed through the following equation (Hoekstra et al., 2011):

$$WA_{\text{blue}} = R_{\text{act}} - \text{EFR} \quad (2)$$

R_{act} is the actual runoff from the basin or subbasin, which was obtained from the calibrated SWAT model output (including WWTP discharge). EFR is the environmental flow requirement, needed to preserve the ecological integrity of downstream aquatic ecosystems. Depending on data availability (Tharme, 2003), this can be assessed by a multitude of methods that can be divided into four categories (i.e. hydrological, hydraulic, habitat simulation and holistic). The 'presumptive environmental flow standard' proposed by Richter et al. (2012), and earlier applied in many WF sustainability assessment studies (e.g. de Miguel et al., 2015; Kaur et al., 2019; Veetil and Mishra, 2018), was used here. According to this standard, the EFR accounts for an 80% share of the natural flow regime (R_{nat}). Thus, the calibrated SWAT model was run without considering WWTP discharge, in order to obtain the R_{nat} and then the EFR (=0.8 R_{nat}).

The $WS_{\text{blue}}^{\text{nat}}$ was assessed without including PS discharge, by substituting R_{nat} with R_{act} in Eq. (2) to give the $WA_{\text{blue}}^{\text{nat}}$.

According to de Miguel et al. (2015), the WS_{blue} can be divided into four levels of water scarcity—negligible (< 1), moderate (1–1.5), significant (1.5–2) and severe (> 2).

3.2.3. Water pollution level

The WPL is the fraction of the waste assimilation capacity consumed at the analysed spatial scale (Hoekstra et al., 2011). It is quantified by dividing the total WF_{grey} by the grey water availability (WA_{grey}). The

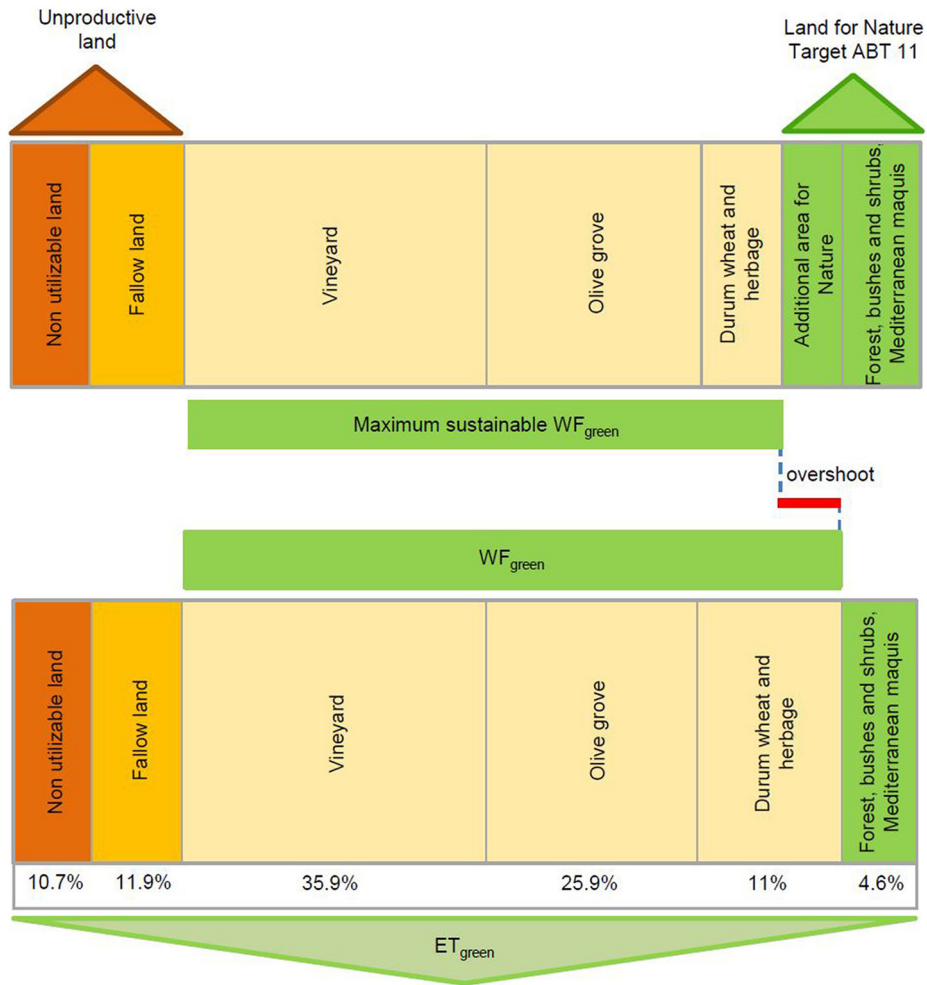


Fig. 3. Allocation of the volume of green water available at the basin scale (ET_{green}), green water footprint (WF_{green}), evapotranspiration from unproductive land (ET_{unprod}) and actual evapotranspiration from current natural vegetation and additional area to be reserved for Nature to achieve the ABT 11 target (ET_{env}). The size of the rectangles is linked to the contribution to determine the ET_{green} of each land use, of which the percentage is provided.

WF_{grey} refers to the amount of fresh water needed to dilute the pollutant load discharged into a receiving water body such that the quality of the water body remained within the range of a fixed water-quality standard (Franke et al., 2013). Meanwhile, the WA_{grey} is the amount of fresh water that could be polluted without affecting the ecosystems of the receiving water body (Bonamente et al., 2017). Due to a lack of data, this study focused on the surface water, ignoring any pollution in the groundwater.

In the study area, the surface water quality is impaired by both agriculture (i.e. DS) and WWTP discharge (i.e. PS) (D'Ambrosio et al., 2019). Thus, the total WF_{grey} was calculated as the sum of the WF_{grey} associated with the DSs (WF_{grey}^{DS}) and PSs (WF_{grey}^{PS}).

WF_{grey}^{DS} (m^3) was calculated thus (Hoekstra et al., 2011):

$$WF_{grey}^{DS} = \frac{L^{DS}}{C_{max} - C_{nat}}, \quad (3)$$

where L^{DS} (kg) is the diffuse pollution load added to the river network, and C_{max} and C_{nat} ($kg\ m^{-3}$) are the maximum allowed and natural concentrations of the pollutant in the water body, respectively.

In Eq. (3), the loads (L^{DS}) of TN ($L^{DS,TN}$) and TP ($L^{DS,TP}$) added from the HRUs (unique land-use, soil and slope features) to the stream by means of surface runoff and lateral flow were estimated by running

the calibrated SWAT model and excluding the WWTP discharge as reported in the scheme of the methodology (Fig. 2). For each crop production, inputs of TN and TP from fertilizers used in the model set up were derived from interviews with farmers and agricultural advisors (D'Ambrosio et al., 2019). After calibrating the model, the SWAT outputs (simulated Q, TSS, TN and TP loads) were in good agreement with the observed loads (Q, TSS, TN and TP) and the SWAT model was able to simulate different conditions such as the absence of irrigation and WWTPs required for assessing $ET_a^{no\ irr}$ (WF_{green}), L^{DS} and R_{nat} (Fig. 2). National legislation (Ministero dell'Ambiente e della Tutela del Territorio e del Mare, 2010), implementing the Water Framework Directive (European Parliament and Council of the European Union, 2000), has provided five different thresholds (from poor to high water quality) for N-NH₄, N-NO₃ and TP, as supporting chemical parameters for ecological status evaluation. In this study, the 'good' TP environmental water-quality standard was adopted ($C_{max}^{TP} = 0.0001\ kg\ m^{-1}$). Meanwhile, the standard proposed by Liu et al. (2017) was used for TN ($C_{max}^{TN} = 0.003\ kg\ m^{-3}$), since the national legislation has not fixed a threshold for that.

Several values of C_{nat} have been reported in the literature for both TN (C_{nat}^{TN}) and TP (C_{nat}^{TP}) due to huge differences in their environmental characteristics (Hejzlar et al., 2009; Meybeck, 1982; Smith et al., 2003). In this study, since it was not possible to measure C_{nat} because

of the human disturbance, C_{nat}^{TN} and C_{nat}^{TP} were assumed to equal $0.00065 \text{ kg m}^{-3}$ and $0.000013 \text{ kg m}^{-3}$, respectively (D'Ambrosio et al., 2020).

WF_{grey}^{DS} was calculated separately for TN ($WF_{grey}^{DS, TN}$) and TP ($WF_{grey}^{DS, TP}$), and the highest value was assumed to be the final value.

WF_{grey}^{PS} (m^3) was assessed using:

$$WF_{grey}^{PS} = \frac{L^{PS}}{C_{max} - C_{nat}}, \quad (4)$$

where L^{PS} (kg) is the pollution load resulting from the WWTP effluent discharged within the river network, which was provided by the calibrated SWAT model, considering the measured concentrations of effluent (C_{eff}^{TN} , C_{eff}^{TP}) and the regulatory limits fixed by national legislation ($C_{eff}^{TN^*}$, $C_{eff}^{TP^*}$) (Table 2). The L^{PS} was obtained by subtracting the L^{DS} from the total pollution load (L , L^{TN} , L^{TP}) provided by the model, which was run considering both the DS and PS. Thus, $L^{PS, TN}$, $L^{PS, TP}$, L^{PS, TN^*} and L^{PS, TP^*} were considered in Eq. (4).

The WF_{grey}^{PS} was calculated separately for TN ($WF_{grey}^{PS, TN}$, WF_{grey}^{PS, TN^*}) and TP ($WF_{grey}^{PS, TP}$, WF_{grey}^{PS, TP^*}), and the highest values were assumed to be the final values (i.e. WF_{grey}^{PS} , $WF_{grey}^{PS^*}$). Thus, the WF_{grey}^{DS} ($=WF_{grey}^{DS} + WF_{grey}^{PS}$) and WF_{grey}^* ($=WF_{grey}^{DS} + WF_{grey}^{PS^*}$) were assessed, as well as the WF_{grey}^{TN} ($=WF_{grey}^{DS, TN} + WF_{grey}^{PS, TN}$) and WF_{grey}^{TP} ($=WF_{grey}^{DS, TP} + WF_{grey}^{PS, TP}$), in order to assess the contribution of each pollutant.

The WA_{grey} is equal to the R_{act} (Hoekstra et al., 2011). Finally, the WPL ($=WF_{grey}^* R_{act}^{-1}$) and WPL^* ($=WF_{grey}^* R_{act}^{-1}$) were quantified. In addition, in the hypothetical absence of PSs, in order to evaluate the sustainability of agricultural crop production, the WPL^{DS} was determined, by dividing the WF_{grey}^{DS} by the R_{nat} ($=WA_{grey}^{nat}$).

The WPL can have values from 0 to >1 (Liu et al., 2012). A $WPL >1$ indicates that the river's capacity to assimilate the existing pollutant load has been surpassed, and therefore the WF_{grey} is unsustainable. Conversely, WPL values <1 indicate that there is enough water to dilute the pollutant load to below the maximum acceptable concentration at the basin and/or subbasin scale.

In order to achieve a sustainable WPL, iterations of the calibrated SWAT model were run several times, considering different values of the pollutant concentrations from WWTP effluent to identify suitable TN and TP concentrations ($C_{eff}^{TN, S}$, $C_{eff}^{TP, S}$) for W1, W2 and W3 disposal. The WPL was thus recalculated for each subbasin by considering, as a model input, the obtained values of $C_{eff}^{TN, S}$ and $C_{eff}^{TP, S}$ for W1, W2 and W3 (WPL^S).

4. Results

4.1. Hydrological and water-quality model results

The model performances ranged from acceptable to very good (Moriasi et al., 2007) (Table 4), depending on the gauge (A, B in Fig. 1) and the calibrated variables (Q, TSS, TN, TP). In particular, a very good performance rating was recorded for the TN load simulations at gauges A and B. A good performance rating was assigned to the Q and TSS at A,

Table 4
SWAT model performances. The coefficient of determination (R^2), the Nash–Sutcliffe efficiency (NSE) coefficient and the percent deviation (PBIAS) are shown for mean daily streamflow (Q), total suspended solid (TSS), total nitrogen (TN) and total phosphorous (TP) loads at the gauges A and B.

	Gauging station A (drainage area 36.2 km ²)			Gauging station B (drainage area 180.0 km ²)		
	R^2	NSE	PBIAS	R^2	NSE	PBIAS
Q	0.94	0.90	12.06	0.96	0.57	-39.55
TSS	0.99	0.95	-28.09	0.98	0.93	-30.83
TN	0.91	0.84	9.07	0.89	0.78	17.88
TP	0.95	0.50	53.89	0.59	0.34	23.38

while the TP load model at A and TSS load model at B were satisfactory. The relatively poorest (i.e. acceptable) model performances were found for the Q and the TP load at B. However, obtaining a relatively poor (i.e. acceptable) model performance is a common issue, especially for temporary streams, where the natural streamflow is intermittent and WWTP discharge constitutes a large part of the daily streamflow and pollution loads (D'Ambrosio et al., 2019, 2020; De Girolamo et al., 2017b).

Indeed, the input data for effluent disposed in the river network by WWTPs were affected by significant uncertainties due to the variability in the plants' treatment efficiency. Additional details can be found in D'Ambrosio et al. (2020).

The simulation results at the basin scale indicated that the actual crop evapotranspiration ($ET_a^{no\ irr}$) was the dominant outflow component of the water balance because it constituted 80.4% of the mean (2011–2017) annual rainfall, which was 617.6 mm (137.1 Mm³). Specifically, the $ET_a^{no\ irr}$ contribution ranged between 68.4% (2015) and 91.2% (2017). The specific crop contributions were analysed in Section 4.2. Potential evapotranspiration was about 1200 mm y^{-1} . Meanwhile, the R_{nat} , on average, accounted for a 12.0% share of the annual rainfall. Its annual volume varied between 3.7 Mm³ (2017) and 24.4 Mm³ (2015), with a mean value of 17.2 Mm³. The mean annual contribution of the WWTP effluent disposed of in the river network was estimated to be 3.7 Mm³, which resulted in an average increase of 21.7% in the streamflow at the basin outlet. In particular, the actual streamflow (R_{act}) ranged from 6.6 Mm³ (2017) to 28.2 Mm³ (2015) (mean 20.9 Mm³).

The main components of the annual average water balance simulated by the SWAT model for the current land use are shown in Fig. 4. In the graph, the components are allocated among the users. R_{act} is differentiated in EFR, WWTPs contribution and amount available for human uses (R_{nat} -EFR), meanwhile the deep aquifer recharge is differentiated in water abstraction for irrigation and net recharge. Total actual evapotranspiration is differentiated for productive land (including irrigated crops), unproductive land (non-utilizable and fallow land) and natural areas.

On average, at the basin outlet, the WWTP effluents contributed 90.1% and 89.9% to the total annual load associated with TN (L^{TN}) and TP (L^{TP}), respectively. In particular, the $L^{PS, TN}$ and $L^{PS, TP}$ were 58,981.56 kg y^{-1} and 8810.6 kg y^{-1} , respectively. Meanwhile, the contribution of DSs was 6479.6 kg y^{-1} for TN ($L^{DS, TN}$) and 994.7 kg y^{-1} for TP ($L^{DS, TP}$). Indeed, the model results suggested that the average concentration of TN and TP without WWTP discharge would be 0.39 mg L^{-1} and 0.06 mg L^{-1} at the basin outlet. Hence, without WWTP effluent and considering the current agricultural activities, the water quality of the Canale d'Aiedda Basin would be below the threshold limits fixed for supporting good ecological status (Ministero dell'Ambiente e della Tutela del Territorio e del Mare, 2010), both estimated values being less than the C_{max}^{TN} and C_{max}^{TP} .

Considering the DSs, durum wheat and vineyards were the main crops, contributing 27.3% and 19.9% to the $L^{DS, TN}$ at the basin scale. Meanwhile, 63.0% of the $L^{DS, TP}$ was due to phosphorus fertilisation of the vineyards. The specific average $L^{DS, TN}$ and $L^{DS, TP}$ (2011–2017) exported from the basin were estimated to be 0.75 kg $ha^{-1} y^{-1}$ (ranging from 0.34 to 1.65 kg $ha^{-1} y^{-1}$) and 0.04 kg $ha^{-1} y^{-1}$ (from 0.02 to 0.10 kg $ha^{-1} y^{-1}$), respectively. The higher specific values were associated with bushes and shrubs, with a mean value of 4.1 kg $ha^{-1} y^{-1}$ for TN and 0.1 kg $ha^{-1} y^{-1}$ for TP. This was due to their proximity to the river network and to grazing activities.

4.2. Water footprint sustainability assessment results

The annual total water footprint of the Canale d'Aiedda Basin, which was obtained by summing the total WF_{green} , WF_{blue} and WF_{grey} , ranged between 120.6 and 259.3 Mm³ y^{-1} from 2011 to 2017. The average value was 213.9 Mm³ y^{-1} , of which 37.2%, 9.2% and 53.6% came from

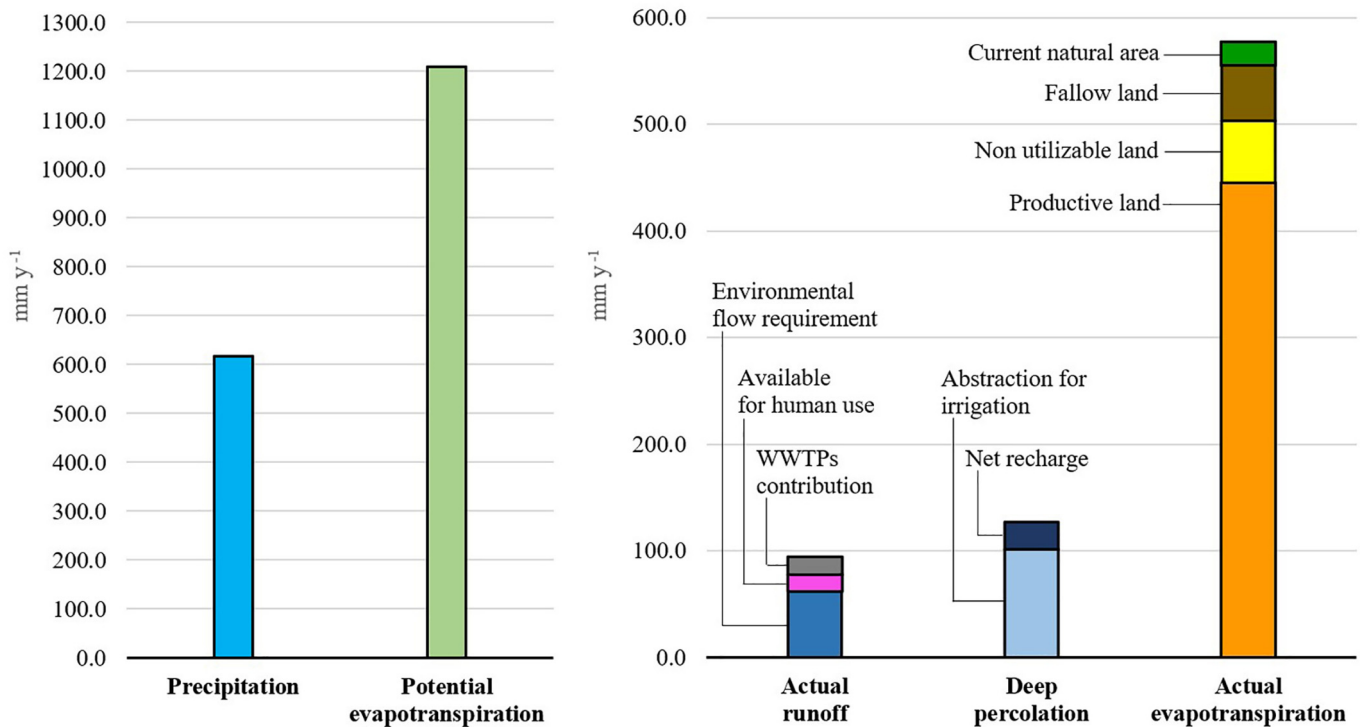


Fig. 4. Average annual water balance components at the basin scale.

the WF_{green} , WF_{blue} and WF_{grey} , respectively. Therefore, the WF_{grey} played the most important role in the Canale d'Aiedda Basin, as a result of the agricultural activities and, more significantly, WWTP effluent. Indeed, the WF_{grey}^{DS} and WF_{grey}^{PS} were, on average, 11.4 and 103.1 $Mm^3 y^{-1}$, contributing 10.0% and 90% to the total WF_{grey} , respectively. If the regulatory limits fixed by national legislation ($C_{eff}^{TN^*}$, $C_{eff}^{TP^*}$) for the WWTP treated effluent were respected, the contribution of pollutant PSs would decrease to 78.8 $Mm^3 y^{-1}$ ($WF_{grey}^{PS^*}$). For most of the study period, TP was the critical pollutant for both the PSs and DSs.

Fig. 5 shows the mean annual water-use allocation at the basin scale. Among the crops, the vineyards and olive groves consumed the greatest amounts of green and blue water. Durum wheat and herbage consumed a lesser amount of green water resources. Considering the $WF_{grey}^{TN^*}$ and $WF_{grey}^{TP^*}$, the WWTP contribution was predominant, with vineyard TP fertilisation being about 6.4% of the $WF_{grey}^{TP^*}$.

The mean monthly variations in the WF components and WF sustainability assessment indicators are shown in Fig. 6.

The mean monthly WF_{green} ranged between 2.5 Mm^3 (July) and 14.3 Mm^3 (April), depending on climate and vegetation cover (Fig. 6a). In particular, temporary crops (i.e. durum wheat and herbage) from July to October did not contribute to the WF_{green} , since that ground was fallow in those months. Also, the ET_{green} and WA_{green} showed a marked seasonality, with higher values being recorded in spring. Meanwhile, the monthly variability of the ET_{env} was low, being between 0.9 Mm^3 (December) and 2.9 Mm^3 (April).

At the basin scale, the mean (2011–2017) annual WS_{green} was 1.2. However, when the temporal variability was assessed at the monthly scale, its values fluctuated between 1.12 (May) and 1.45 (August).

With respect to the mean monthly WF_{blue} , the results were >0 from May to September, when the olive groves and vineyards received irrigation. The lowest value was registered in September (880.4 m^3), the highest in July (6 Mm^3). Fig. 6b shows the monthly variability of the WF_{blue} , R_{act} , R_{nat} , EFR, WS_{blue} , and WS_{blue}^{nat} .

The R_{act} , R_{nat} and EFR varied little over the months. Specifically, on a monthly time scale, they averaged between 0.9 and 3.0 Mm^3 , 0.6 and

2.6 Mm^3 , and 0.5 and 2.1 Mm^3 , respectively, following the precipitation regime.

Conversely, the WS_{blue} and WS_{blue}^{nat} were characterised by a relative discrepancy between the summer months and the rest of the year. Indeed, the WS_{blue} varied between 0 and 13.19 (August) at the basin scale and at a monthly temporal scale. The maximum value increased to 100.67 for the WS_{blue}^{nat} . In August, throughout the studied period, the WS_{blue} varied from 6.88 (2012) to 27.65 (2017), while the WS_{blue}^{nat} ranged from 16.48 to 518.10, respectively. In terms of the large interannual variability, the standard deviation associated with August was the highest for both indicators (Fig. 6b). The mean (2011–2017) annual WS_{blue} and WS_{blue}^{nat} were found to be 3.0 and 8.46, respectively. Therefore, if irrigation were provided from surface water, the Canale d'Aiedda Basin would experience severe blue water scarcity. However, deep groundwater constituted the main source for irrigation in the study area.

The mean monthly WF_{grey} varied from 6.8 Mm^3 (October) to 14.3 Mm^3 (March) (Fig. 6c). Specifically, the highest value was registered when peak pollutant concentrations in discharged effluent overlapped with the runoff from fertiliser applied in February to the durum wheat and vineyards. Meanwhile, the WF_{grey}^{DS} and WF_{grey}^{PS} ranged between 0.03 Mm^3 (August) and 4.6 Mm^3 (February), and between 6.0 Mm^3 (September) and 12.0 Mm^3 (March), respectively. From April to August, more than 99% of the WF_{grey} was due to WF_{grey}^{PS} . If the limits fixed by national legislation for WWTP treated effluent (i.e. $C_{eff}^{TN^*}$, $C_{eff}^{TP^*}$) were adhered to, the $WF_{grey}^{PS^*}$ would remain at the same level (4.5–6.7 Mm^3) for almost the whole year.

The mean (2011–2017) annual WPL, WPL* and WPL^{DS} were 5.97, 4.75, 0.70, respectively, at the basin scale. The mean monthly values ranged from 3.69–9.96, 2.73–7.74 and 0.04–2.30, respectively. Thus, WWTP discharge constituted a significant pollutant source in the study area throughout the year, even if the TN and TP concentration thresholds were respected. Meanwhile, agricultural activities hampered the water quality in February, September, November and December, due to fertiliser runoff.

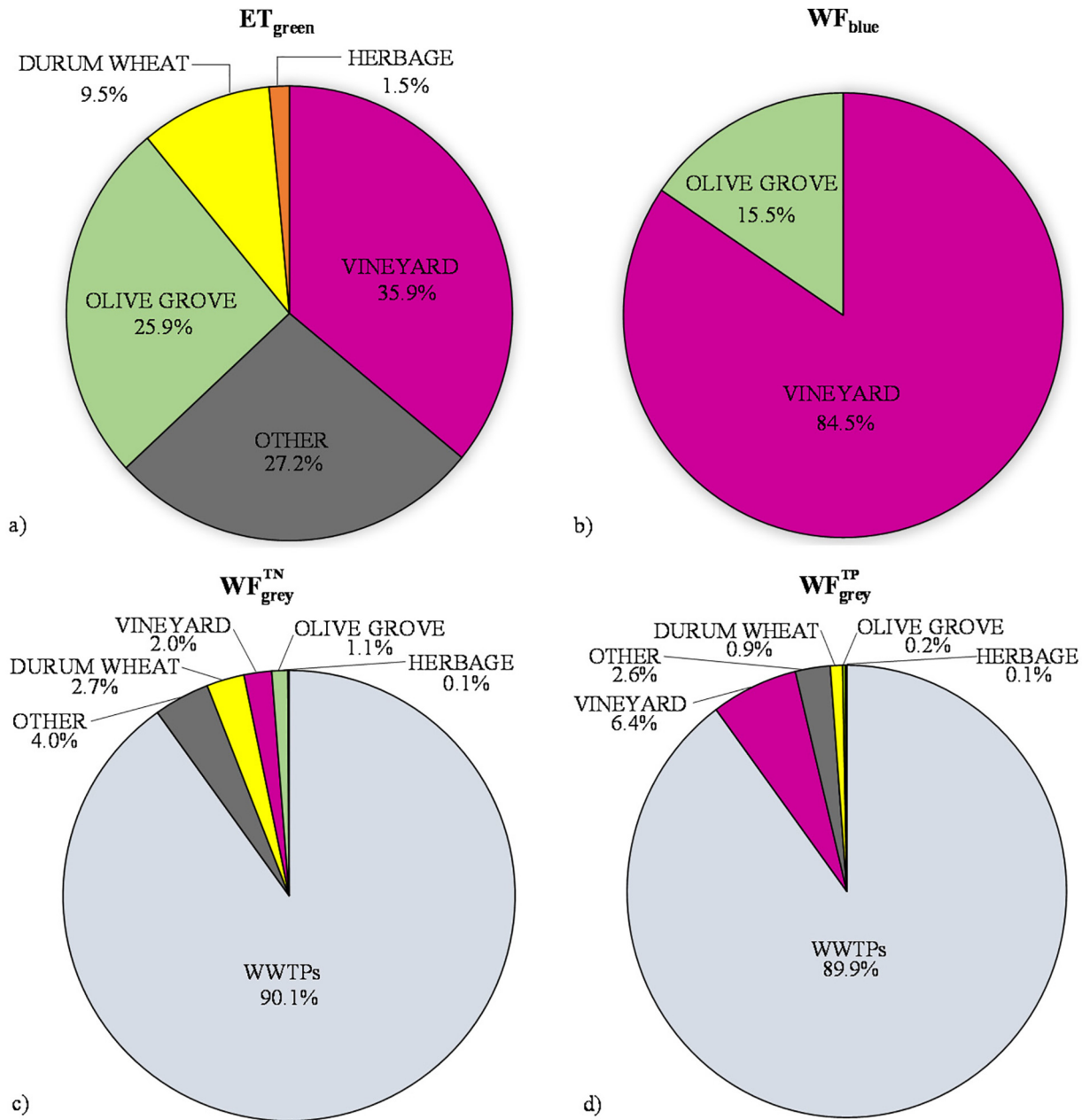


Fig. 5. Mean (2011–2017) annual allocation of ET_{green} , WF_{blue} , WF_{grey}^{TN} and WF_{grey}^{TP} at the basin scale. The part of the pie chart labelled 'other' refers to urbanised land, bushes and shrubs, fallow land, forest, pasture and reservoir. The contribution of citrus groves was negligible.

The spatial distribution of the analysed WF indicators is shown in Fig. 7 and Table 5. The annual mean (2011–2017) WS_{green} varied from 0.61 to 1.66. Unsustainable green water use ($WS_{green} > 1$) was registered in those subbasins where the agricultural land use comprised only permanent crops (i.e. vineyards and/or olive groves) and natural areas were absent (Fig. 7a). The WS_{blue} and WS_{blue}^{nat} spatially ranged from 0 to 31.5. Also, without considering the contribution of WWTP effluent, the annual irrigation-water requirement could be satisfied by the WA_{blue}^{nat} in 15 subbasins (Fig. 7b).

With respect to the WPL, the agricultural activities were sustainable at the yearly average, and also when the subbasin scale was considered. Only Subbasin 9 had a mean annual value of $WPL^{DS} > 1$ (Fig. 7c, Table 5). The WWTP effluent consumed the capacity to assimilate the pollutant load in downstream subbasins (i.e. Subbasins 14, 19, 23, 24, 27, 29, 30, 31, 34 and 35 in Fig. 1), considering both C_{eff}^{TN} and C_{eff}^{TP} (WPL in Fig. 7d) and C_{eff}^{TN*} and C_{eff}^{TP*} (WPL^* in Fig. 7e). Depending on the month, TN or TP were alternatively found to be the critical pollutant

for both the DS and PS. In order to obtain a sustainable pollution level (WPL^S in Fig. 7f), the discharge from W1, W2 and W3 would have to respect the desirable TN and TP concentrations ($C_{eff}^{TN,S}$, $C_{eff}^{TP,S}$) obtained from the iterative procedure described in Section 3.2.3. These concentrations are reported in Table 6.

5. Discussion

5.1. Water security assessment and response formulation

The integrated modelling framework proposed in the current study provided an effective method for assessing water security in a basin altered by agricultural activities and WWTP treated effluent discharge.

The results showed that pollution was the main factor affecting surface-water security in the Canale d'Aiedda Basin. In particular, surface-water security was hampered by PSs that exerted a mean WF_{grey}^{PS} of $103.1 \text{ Mm}^3 \text{ y}^{-1}$, and contributed 90% to the total WF_{grey} . A

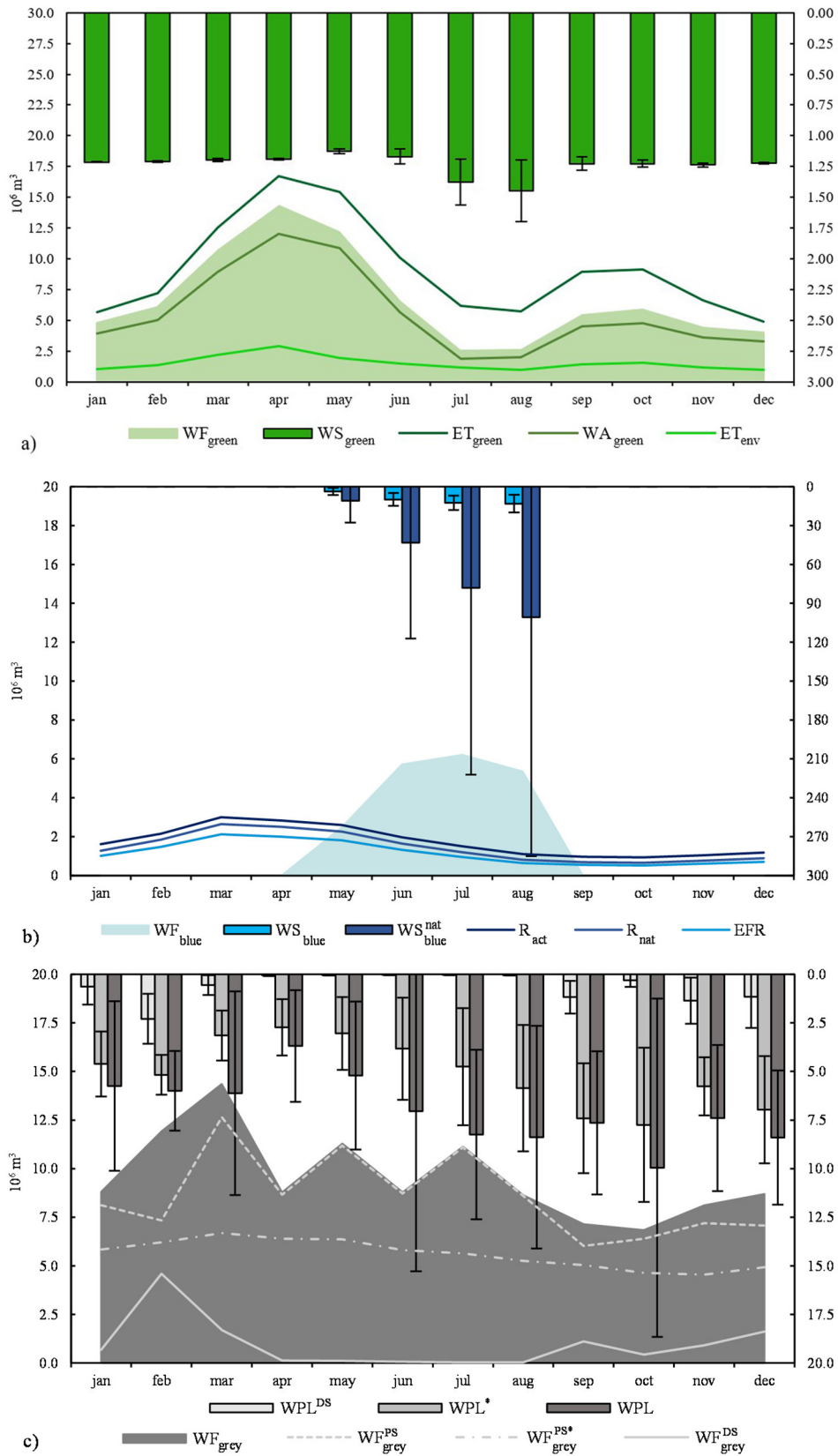


Fig. 6. Basin-scale mean monthly ET_{green} , WF_{green} , WA_{green} , ET_{env} and WS_{green} (a); WF_{blue} , EFR , R_{act} , R_{nat} , WS_{blue} and WS_{nat_blue} (b); WF_{grey} , WF_{grey}^{DS} , WF_{grey}^{PS} , WF_{grey}^{PS*} , WPL , WPL^* and WPL^{DS} (c). The whiskers indicate the standard deviation associated with the WF sustainability assessment indicators (secondary vertical axis).

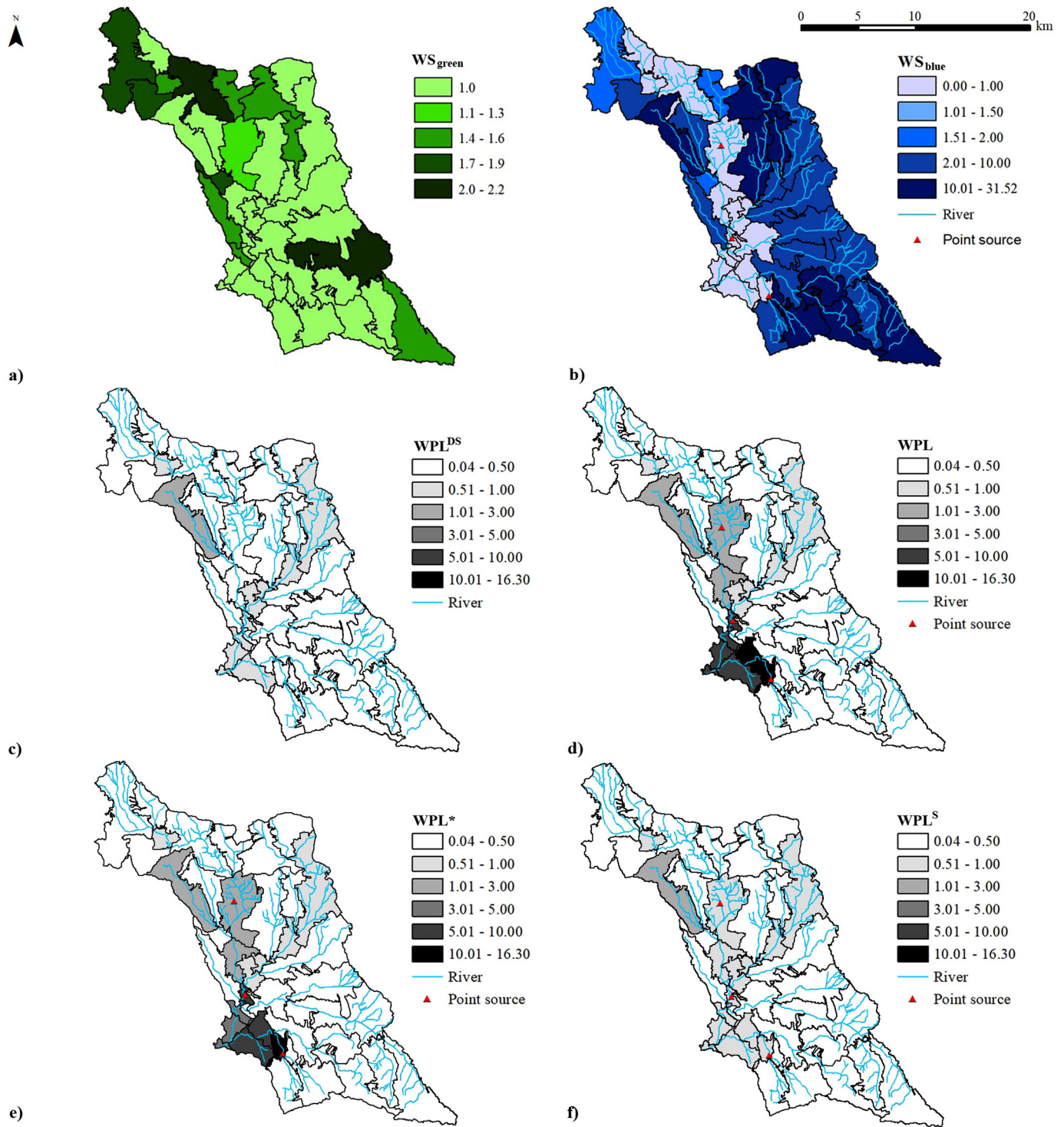


Fig. 7. Variation in WF sustainability assessment indicators at the subbasin scale. a) WS_{green} ; b) WS_{blue} ; c) WPL considering only the DS (WPL^{DS}); d) WPL considering the DS and WWTP effluent discharged into the river network with actual (measured) concentrations (i.e. C_{em}^{TN} , C_{em}^{TP}) (WPL); e) with the regulatory limits fixed by national legislation (i.e. C_{em}^{TN*} , C_{em}^{TP*}) (WPL^*); and f) with the identified desirable concentrations for WF_{grey} sustainability (i.e. $C_{em}^{TN,S}$, $C_{em}^{TP,S}$) (WPL^S).

comparison with the WF_{grey}^{PS} obtained in previous studies (Johnson and Mehrvar, 2019; Morera et al., 2016) was not feasible because its value depends on the WWTP size and the technology used, and on the values of C_{max} and C_{nat} that were considered in its computation (D'Ambrosio et al., 2020).

The estimated values for WPL, with respect to different spatial and temporal scales, highlighted that the water quality was not in a safe range. Indeed, the average monthly WPL varied between 3.69 and

9.96, and the annual mean value was 5.97 for the whole basin. At the subbasin scale, the mean annual WPL ranged from 0.04 to 16.26. Subbasins located downstream of WWTP discharge showed higher values. Even if the effluent pollutant concentrations had respected national thresholds, the WPL^* would not have improved. This is because the threshold limits C_{em}^{TN*} and C_{em}^{TP*} were fixed for perennial rivers with a high assimilation capacity (D'Ambrosio et al., 2020). Meanwhile, a temporary stream, such as the Canale d'Aiedda, has a lower potential

Table 5

Mean (2011–2017) annual WS_{green} , WS_{blue} , WS_{blue}^{nat} , WPL^{DS} , WPL , WPL^* and WPL^S associated with each subbasin. Unsustainable values (>1) for the WS_{green} , WS_{blue} , WS_{blue}^{nat} , WPL^{DS} , WPL , WPL^* and WPL^S are in bold.

Subbasin	WS_{green}	WS_{blue}	WS_{blue}^{nat}	WPL^{DS}	WPL	WPL^*	WPL^S
1	1.00	0.79	0.79	0.04	0.04	0.04	0.04
2	1.75	1.95	1.95	0.08	0.08	0.08	0.08
3	1.42	0.25	0.25	0.79	0.79	0.79	0.79
4	1.84	2.58	2.58	0.06	0.06	0.06	0.06
5	1.37	21.23	21.23	0.10	0.10	0.10	0.10
6	1.00	10.35	10.35	0.22	0.22	0.22	0.22
7	1.36	1.74	1.74	0.10	0.10	0.10	0.10
8	2.23	0.27	0.27	0.47	0.47	0.47	0.47
9	1.00	15.98	15.98	1.13	1.13	1.13	1.13
10	1.00	2.77	2.77	0.08	0.08	0.08	0.08
11	1.30	12.81	12.81	0.38	0.38	0.38	0.38
12	1.00	5.22	5.22	0.72	0.72	0.72	0.72
13	1.64	1.59	1.59	0.39	0.39	0.39	0.39
14	1.28	0.97	1.36	0.41	1.47	1.25	0.91
15	1.00	14.51	14.51	0.31	0.31	0.31	0.31
16	1.00	2.45	2.45	0.60	0.60	0.60	0.60
17	1.00	0.58	0.58	0.56	0.56	0.56	0.56
18	1.00	4.27	4.27	0.25	0.25	0.25	0.25
19	1.00	0.38	0.52	0.40	1.18	1.01	0.77
20	1.00	0.42	0.42	0.67	0.67	0.67	0.67
21	1.00	2.57	2.57	0.18	0.18	0.18	0.18
22	1.50	5.49	5.49	0.20	0.20	0.20	0.20
23	1.00	0.02	0.05	0.54	8.01	5.17	0.92
24	1.00	0.00	0.01	0.68	7.77	5.08	1.00
25	1.00	0.55	0.55	0.28	0.28	0.28	0.28
26	1.00	0.01	0.01	0.38	0.38	0.38	0.38
27	1.51	0.01	0.02	0.60	5.70	3.75	0.83
28	2.17	2.87	2.87	0.38	0.38	0.38	0.38
29	1.00	0.12	0.99	0.17	11.55	8.44	0.69
30	1.00	0.04	0.10	0.65	5.63	3.72	0.86
31	1.00	0.15	0.65	0.94	8.93	6.74	0.92
32	1.00	4.38	4.38	0.07	0.07	0.07	0.07
33	1.00	31.52	31.52	0.08	0.08	0.08	0.08
34	1.00	0.01	0.02	0.70	5.93	4.21	0.85
35	1.00	0.11	1.35	0.24	16.26	12.20	0.96
36	1.00	6.50	6.50	0.07	0.07	0.07	0.07
37	1.00	3.03	3.03	0.40	0.40	0.40	0.40
38	1.00	25.96	25.96	0.05	0.05	0.05	0.05
39	1.00	14.16	14.16	0.09	0.09	0.09	0.09
40	1.31	12.79	12.79	0.11	0.11	0.11	0.11

dilution capacity; therefore, lower pollutant thresholds need to be fixed for WWTP effluent in order to increase the water quality of the receiving water body and assuring water security (Morera et al., 2016). To this end, Zhang et al. (2014) emphasised that the regulatory framework related to PSs and DSs could be improved if formulated around the WPL concept. Thus, in the current study, the SWAT model was run several times in order to identify those values of C_{eff}^{TN-S} and C_{eff}^{TP-S} for each WWTP, which was able to ensure a sustainable WPL (WPL^S) in all the subbasins. It was found that these concentrations ranged from 4 to 15 $mg L^{-1}$ for TN, and from 0.1 to 1.2 $mg L^{-1}$ for TP, respectively. These concentrations of nutrients in effluent could be achieved by improving WWTP efficiency, or by adopting different systems, such as lagoons, phytodepurations or groundwater recharge (Ceschin et al., 2020; Crini and Lichtfouse, 2019). Alternatively, the reuse of treated wastewater for irrigation purposes could be promoted in a circular-economy context, since this would also promote a reduction in fertiliser

Table 6

Desirable TN (C_{eff}^{TN-S}) and TP (C_{eff}^{TP-S}) concentrations in WWTP effluent (W1, W2, W3) required to obtain a sustainable water pollution level (WPL^S) throughout the study area.

Parameter	C_{eff}^{TN-S} ($mg L^{-1}$)	C_{eff}^{TP-S} ($mg L^{-1}$)
W1	15	1.2
W2	8	0.1
W3	4	0.1

usage, as well as the partial satisfaction of irrigation demand (De Girolamo and Lo Porto, 2020).

The suitable thresholds (C_{eff}^{TN-S} , C_{eff}^{TP-S}) identified in the current study improve on those obtained by a previous study in the same study area (D'Ambrosio et al., 2020). Indeed, D'Ambrosio et al. (2020) identified desirable concentrations of TN and TP equal to 8.1 and 0.3 $mg L^{-1}$, respectively, that were evaluated at the basin scale.

Concerning agricultural activities at the basin scale, February, September, November and December were found to be critical months, with the WPL^{DS} being 2.30, 1.18, 1.36 and 1.16, respectively. In those months, fertilisation was carried out on the vineyards (February, November), durum wheat (February, December) and herbage (April, September) throughout the basin. If the yearly average of the WPL^{DS} at the subbasin scale is considered, low unsustainability ($WPL^{DS} = 1.1$) was associated with only Subbasin 9 (Figs. 1, 7c). This is due to the contribution of a plurality of factors, such as the high runoff potential of the soil, land use (29.2% olive, 16.5% vineyard, 10.0% herbage) and slope (about 3.7°). Hence, fertiliser strategies aimed at minimising nutrient runoff (i.e. precision agriculture) should be adopted, especially in hotspot subbasins and in critical months (D'Ambrosio et al., 2018a; De Girolamo et al., 2017c). Replacing part of the chemical fertiliser with mycorrhizal fungi (Liu et al., 2020) or with treated, reusable wastewater (De Girolamo and Lo Porto, 2020) could be effective ways to reduce the WF_{grey}^{DS} . The involvement of local farmers and dealers, as well as detailed economic analyses, are fundamental in supporting any decisions about the agricultural practices to be promoted and the pursuit of win-win solutions (Lovarelli et al., 2016; Mekonnen and Hoekstra, 2010).

Water security assessments should not neglect green water resources since these are critical and limited; as with, blue water, they should explicitly be taken into account because both are related to the amount of precipitation, which is limited in time and space (Hoekstra et al., 2019; Schyns et al., 2019). The limited use of the WS_{green} indicator in watershed management is due to the lack of a standardised methodology for computing the WA_{green} (D'Ambrosio et al., 2020; Quinteiro et al., 2018). Different formulas have been proposed and used for its assessment in the few studies that have evaluated it (Giri et al., 2018; Kaur et al., 2019; Pellicer-Martínez and Martínez-Paz, 2016; Salmoral et al., 2017; Schyns et al., 2019; Veetil and Mishra, 2018), which has resulted in comparisons becoming meaningless.

The conceptual approach of the WS_{green} developed by Schyns et al. (2019) and adopted in the present work refers to the competition for green water resources, which should support both the natural ecosystem and crop production. The WS_{green} indicator quantifies the degree of human appropriation of the green water flow. In their study, the authors quantified the maximum sustainable WA_{green} considering agro-ecological suitability and accessibility of land, biophysical constraints to intensifying land use, and biodiversity conservation needs (ABT 11). In that approach, increasing WS_{green} means that reduced green water resources remain for nature. At the basin scale, if WS_{green} is >1 means that the target fixed for supporting the biodiversity was not achieved and the WS_{green} is defined "unsustainable".

Veetil and Mishra (2016, 2018) and Giri et al. (2018) adopted a different approach for estimating the WA_{green} , which was the amount of initial soil water content and it is computed as the difference between soil moisture at the root zone and wilting point. In this approach, there is no target to be achieved for biodiversity.

The final aim of the study should guide in choosing the most suitable methodology. Thus, the second approach (Veetil and Mishra, 2016) could be more suitable in studies aimed at assessing the impact of land-use or climate change on the WA_{green} at a local scale. Meanwhile, the studies evaluating water security using the concept of the WF should privilege the approach developed by Schyns et al. (2019). However, an integration of the two approaches is desirable.

At the basin scale, the mean annual WS_{green} is 1.20. This value indicates that reduced green water resources remain for nature. The value is in line with the overshoot of the WA_{green} identified for the study

area by Schyns et al. (2019) and is higher than the value estimated at the national level (0.9–1.0). At the subbasin scale, some subbasins face water insecurity where WS_{green} is > 1 . These subbasins have tree crops (i.e. olive groves and/or vineyards) as the main land cover, and all the WA_{green} has been allocated to human (agricultural) use. Meanwhile in the subbasins with the presence of natural areas WF_{green} equals WA_{green} . The Canale d'Aiedda basin has no potential remaining to increase rain-fed agricultural production. In order to locally increase the WA_{green} , a conversion of part of the agricultural land in natural areas is proposed in this study in order to enlarge the existing protected areas. However, additional investigations are needed to explore in detail this possibility. Indeed, protected areas should be integrated into a wider region considering ecological connectivity and networks and should be established and managed in collaboration with local communities, with regional and national authorities. Soil management practices should be improved to enable more water to be retained in the soil, which can then be taken up by crops and recycled via transpiration (Schyns et al., 2019). The best management practices would also reduce the WF_{blue} because less water would be necessary for irrigation to prevent the plants from experiencing water stress. In particular, no- and reduced-tillage systems (which increase carbon sequestration in soil and improve soil organic matter), or the application of mulch, could limit soil evaporation and increase transpiration efficiency, thereby contributing to biomass growth. However, an optimal irrigation scheduling and new efficient technologies and equipment should be adopted for improving water use.

The WS_{blue} exhibited different trends, which were mainly determined by the seasonality of irrigation water consumption. Indeed, in the study area, irrigation was predominantly provided to the vineyards and olive groves from May to September. Thus, at the basin scale, that indicator resulted in values > 0 only in those months, increasing progressively from 3.73 to 13.19. These values ranged from 10.90 to 100.67 when the WWTP effluent contribution was neglected (WS_{blue}^{nat}). Therefore, surface water (WA_{blue} , WA_{blue}^{nat}) was not sufficient to satisfy the irrigation requirements of the study area, and using deep groundwater remained necessary because there is no irrigation board in this area. However, surface water could be used for reducing the quantitative pressures on groundwater resources (Casella et al., 2019). Specifically, in some subbasins, the annual irrigation water requirement could be completely satisfied by both WA_{blue} and WA_{blue}^{nat} , even when taking their monthly variability into account. In these subbasins, water conservancy infrastructure could be used to accumulate surface water during flood events (thereby also reducing the flood risk) to be released in dry periods to fulfil irrigation-water requirements (Cai et al., 2020).

In summary, the implementation of strategies aimed at reducing the three WF components is fundamental to increasing water security in the Canale d'Aiedda Basin. Given that the WF_{grey} constitutes more than 50% of the total WF, it is clear that improving wastewater treatment and management, and reducing nutrient runoff from agricultural land, is of paramount importance.

5.2. Advantages related to the assessment of water security using water footprint indicators and hydrological and water-quality models

Water security is broadly recognised as an important and urgent policy challenge, which can be achieved by balancing human needs while safeguarding ecosystems and biodiversity (Bakker, 2012). In order to address this challenge, it is fundamentally important to quantify water security (Hoekstra et al., 2018). In this sense, indicators can be a powerful tool. They can help to define the current situation as well as changes over time, prioritise actions and investments and measure their effectiveness, and stimulate policy-makers because they can be shown a direct link between actions and results (Jensen and Wu, 2018). Indicators are considered meaningful and feasible for in policy-making only if they are credible, valid and salient (Hoekstra et al., 2018). In other words, they need to be scientifically valid and technically robust to be

legitimised and accepted by stakeholders and recognised by the policy-makers who should be aware of them. In addition, the assessment of such indicators should be based on data that are available from trusted sources and that can be collected within time and budgetary constraints (Jensen and Wu, 2018).

WF indicators are characterised by the features mentioned above and have become the main international reference for evaluating the sustainability of water use since the ISO 14046 norm was adopted (Pellicer-Martínez and Martínez-Paz, 2016). In addition, WF concepts can be used for understanding the complexity and temporal-spatial variability of water security through the adoption of a system-dynamic perspective based on the pressure-state-impact-response schematisation of social-environmental systems (Hoekstra et al., 2018). They allow the evaluation of water supplies and water-use efficiency, as well as aspects of quality. Thus, the traditional idea of water security assurance, which has historically focused on water-quantity items, ignoring the fact that pollution can compromise water use (David da Silva et al., 2020), can be improved through use of the WF sustainability concept (Liu et al., 2020). To guarantee water security, collective actions should be taken to effectively reduce the WF of human activities, moving them towards sustainable levels (Hoekstra et al., 2019). Interventions via different actors along the supply chains, and at different spatial and temporal scales, are essential, and the use of modelling approaches at the watershed scale can enable water-resource managers to address the challenge of water security by enabling them to foresee the effects of particular response formulations that have been identified as being able to reduce the multiple pressures that may currently affect the analysed water system (Giri et al., 2018).

Hydrological and water-quality models (such as SWAT) are robust tools for WF assessment, since processing the model outputs makes it possible to calculate all the WF components, together with the maximum sustainable values of the WF at the different spatial and time scales. This allows a WF sustainability analysis to be performed by direct comparison (Pellicer-Martínez and Martínez-Paz, 2016). Models can be used to address socio-environmental issues relating to actual land-use management, and/or for predicting the effects on water security of alternative scenarios and the implementation of best management practices (Giri et al., 2018; Malagó et al., 2017; Pulighe et al., 2019). This issue is particularly pressing for Mediterranean basins, considering that climate change is expected to have a great impact on the water balance, with an increase in temperature, a reduction in rainfall and changes in river hydrological regimes, all of which will likely negatively impact water security (D'Ambrosio et al., 2018a; De Girolamo et al., 2017a).

For all these reasons, the integrated modelling framework proposed in this study could have great utility in integrated water-resources management, since it allows a better understanding of how (hydrological and water-quality processes), where (environmental hotspots) and when (time) water scarcity may occur across a basin, considering both the current condition and different future scenarios and WF mitigation strategies (Giri et al., 2018).

5.3. Limitations of the study and future improvements

The concept of water security embraces multidisciplinary concerns related to economic welfare, social equity, sustainability and risk (Hoekstra et al., 2018), and thus presents a way of thinking about all water-related issues, such as safe drinking water, water scarcity, water pollution and flooding, in the context of an integrated framework (Sun et al., 2016). To capture the different facets of water security, a very large number of indicators have been developed. Jensen and Wu (2018) grouped these indicators into four main classes, related to resource availability (e.g. WF indicators, water storage capacity), access (e.g. water poverty index, drinking-water safety, water tariffs, water treatment capacity and coverage), risk (e.g. flood frequency and damage indicators) and institutional management capacity appraisal (e.g.

strategic planning, disaster management regulation). Thus, water security could be achieved by increasing resource availability and access, minimising water-related risks and enhancing management capacity. In addition, major pressures affecting water security can be divided into environmental and socioeconomic categories (Cai et al., 2020; Hoekstra et al., 2018). Environmental pressures relate to the hydrological and geographical conditions of the study area (i.e. climate change, flooding, drought, sea-level rise, the unbalanced distribution of water resources), while, socioeconomic pressures refer to anthropogenic impacts, such as population growth, water shortages, pollution and land-use changes.

This study only considered the socioeconomic characteristics of water security and assessed the resource availability issues related to the current situation. Industrial, commercial, and domestic water uses were not included in this study since to satisfy those uses water was imported from outside the Canale d'Aiedda Basin. Thus, further analyses will be necessary in order to evaluate all the different facets of water security in the Canale d'Aiedda Basin. In addition, this study focused only on the surface water, neglecting the groundwater due to the lack of data. However, the SWAT is an extremely versatile model that can be coupled with other tools (such as MODFOW and RT3D), which can simulate hydrogeological processes and nutrient percolation (Wei et al., 2019; Yu et al., 2019). Thus, there is room for further modelling to obtain a complete assessment of fresh water resources in the study area.

Another limitation of the current study was that the agricultural management practices were assumed to have remained the same throughout the study area. If an agent-based model was coupled with the model presented here, it would be possible to generate land-use patterns that took stochasticity and uncertainty into account (Giri et al., 2018). Using such a model could help to mimic the complexity of land-use systems in future applications.

Other limitations affecting the WPL assessment resulted from only considering TN and TP as pollutants. The combined effect of coexistent pollutants, such as pesticides (Gil et al., 2017) and human and veterinary pharmaceuticals (Wöhler et al., 2020), should be analysed so as to better evaluate the actual WF_{grey} .

Similarly, the WPL assessment was limited to agricultural activities and WWTP effluent, which are the main activities in the basin; however, as a general procedure, all the anthropogenic activities carried out across the whole basin, such as by the industrial and domestic sectors (Liu et al., 2017), should be considered in order to enable an evaluation of the actual environmental sustainability level of the study area.

In addition, it is necessary to highlight that the WF sustainability assessment procedures needed to be improved in several aspects. Although the WF concept can be traced back to Hoekstra and Hung (2002), uncertainties remain concerning the standardisation of a single procedure to produce a consistent sustainability assessment. Indeed, the literature provides various methodologies for assessing WA_{green} , EFR and WF_{grey} , which makes it difficult to compare the results of different studies. Schyns et al. (2019) recommended that future work focus on better estimating WA_{green} , especially with regard to ET_{env} and ET_{unprod} . Meanwhile, in the current study, the procedure proposed by Richter et al. (2012) for EFR accounting was adopted. However, a large number of methods exist for estimating EFR, and these can give significantly different results (Arthington, 2012). Finally, for the WF_{grey} assessment, the assumptions related to C_{max} and C_{nat} significantly influenced the results and needed to be standardised (D'Ambrosio et al., 2018b; Liu et al., 2017). Further research on these issues is needed in order to standardise the methodology for WF sustainability assessments.

Finally and importantly, one of the major challenges is the translation of research results into coherent mechanisms of governance and ways of intervention, along supply chains and at different levels (Hoekstra et al., 2019). Constructive synergies between researchers, decision-makers and practitioners are fundamental.

6. Conclusions

The concept of water security throughout a holistic approach aims to achieve sustainable development and human well-being. From a theoretical standpoint, water security has been well defined; however, an operational strategy is still in a critical phase, and several difficulties exist in integrating the concept into different levels of governance (at global, national and local scales). Indeed, the achievement of water security needs not only coordinated policies and good governance, but also interdisciplinary approaches, and models able to integrate its multi-sectoral aspects.

This study has provided a contribution that helps in the development of a procedure for implementing the water security concept at the local scale through a case study. The proposed approach was intended to be an operational tool for high spatial and temporal resolutions. It was defined by integrating the WF sustainability assessment methodology proposed by the Water Footprint Network with a hydrological and water-quality model and field surveys. Coupling modelling activities with the WF sustainability assessment procedure has proven effective in responding to water-security challenges because both quantitative and qualitative aspects can be evaluated.

The outcomes of this work have revealed critical issues in water security at the basin scale. While this spatial scale is appropriate as the basic unit for water resources management, this study has shown that a subbasin scale is needed to identify where and when management options have to be implemented in order to deliver water security. In the Canale d'Aiedda Basin—a basin representative of the Mediterranean climate—pollution is the main factor affecting water security, and so lower pollutant thresholds for WWTP effluent were identified as a means of increasing water quality. In addition, the natural areas should be increased to locally increase the WA_{green} as well as agricultural management practices should be improved to enable the retention of more water in the soil and reduce nutrient runoff. Currently, surface-water availability is not sufficient to satisfy the irrigation requirements across the study area, and extracting water from deep groundwater remains necessary because there is no irrigation board operating in this area. This study also highlighted that the WF sustainability assessment procedure needed to be improved in several aspects.

The integrated framework developed in this study may have great utility in integrated water-resources management because it could support the implementation of a suitable water policy that would ensure water security in basins under human pressures. The biggest challenge remaining is the translation of the study outcomes into coherent mechanisms of governance that would require adequate financial resources.

CRedit authorship contribution statement

Ersilia D'Ambrosio: Conceptualization, Methodology, Software, Data curation, Writing - original draft, Formal analysis. **Giovanni Francesco Ricci:** Data curation, Writing - original draft. **Francesco Gentile:** Resources, Validation. **Anna Maria De Girolamo:** Writing - review & editing, Supervision, Validation.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Glossary

Abbreviation	Description	Unit of measure
$C_{\text{eff}}^{\text{TN}}$	Concentration of TN in the effluent discharged by the wastewater treatment plant	mg L^{-1}
$C_{\text{eff}}^{\text{TN}^*}$	Threshold concentration of TN allowed to wastewater treatment plants' discharges in temporary rivers by national legislation	mg L^{-1}
$C_{\text{eff}}^{\text{TP}}$	Concentration of TP in the effluent discharged by the wastewater treatment plant	mg L^{-1}
$C_{\text{eff}}^{\text{TP}^*}$	Threshold concentration of TP allowed to wastewater treatment plants' discharges in temporary rivers by national legislation	mg L^{-1}
$C_{\text{eff}}^{\text{TN}_{\text{S}}}$	Suitable concentration of TN in the effluent discharged by the wastewater treatment plant	mg L^{-1}
$C_{\text{eff}}^{\text{TP}_{\text{S}}}$	Suitable concentration of TP in the effluent discharged by the wastewater treatment plant	mg L^{-1}
C_{max}	Maximum concentration of the pollutant in surface water bodies	mg L^{-1}
$C_{\text{max}}^{\text{TN}}$	Threshold concentration of TN in surface water associated with a good ecological status	mg L^{-1}
$C_{\text{max}}^{\text{TP}}$	Threshold concentration of TP in surface water associated with a good ecological status	mg L^{-1}
C_{nat}	Natural concentration of the pollutant in surface water	mg L^{-1}
$C_{\text{nat}}^{\text{TN}}$	Natural concentration of TN in surface water	mg L^{-1}
$C_{\text{nat}}^{\text{TP}}$	Natural concentration of TP in surface water	mg L^{-1}
DS	Diffuse pollution source (i.e., agriculture)	–
EFR	Environmental flow requirement	$\text{m}^3 \text{y}^{-1}$
ET_a	Actual crop evapotranspiration considering current management practices (i.e., including irrigation)	mm d^{-1}
$\text{ET}_a^{\text{no irr}}$	Actual crop evapotranspiration assuming that the soil does not receive any irrigation	mm d^{-1}
ET_{env}	Environmental green water requirement	mm d^{-1}
$\text{ET}_{\text{unprod}}$	Evapotranspiration from land that cannot be made productive in crop production (e.g., fallow land and non-utilizable land)	mm d^{-1}
HRU	Hydrologic Response Unit	–
L	Total pollution load estimated by SWAT	$\text{kg ha}^{-1} \text{y}^{-1}$
L^{TN}	Total TN load adducted to the river and estimated by SWAT	$\text{kg ha}^{-1} \text{y}^{-1}$
L^{TP}	Total TP load adducted to the river and estimated by SWAT	$\text{kg ha}^{-1} \text{y}^{-1}$
L^{DS}	Diffuse pollution load	$\text{kg ha}^{-1} \text{y}^{-1}$
$L^{\text{DS,TN}}$	Diffuse TN load	$\text{kg ha}^{-1} \text{y}^{-1}$
$L^{\text{DS,TP}}$	Diffuse TP load	$\text{kg ha}^{-1} \text{y}^{-1}$
L^{PS}	Pollution load resulting from the wastewater treatment plants' effluent discharged within the river network	$\text{kg ha}^{-1} \text{y}^{-1}$
$L^{\text{PS,TN}}$	TN load resulting from the wastewater treatment plants' effluent discharged within the river network	$\text{kg ha}^{-1} \text{y}^{-1}$

(continued)

Abbreviation	Description	Unit of measure
$L^{\text{PS,TP}}$	TP load resulting from the wastewater treatment plants' effluent discharged within the river network	$\text{kg ha}^{-1} \text{y}^{-1}$
$L^{\text{PS,TN}^*}$	TN load resulting from the wastewater treatment plants' effluent discharged within the river network considering $C_{\text{eff}}^{\text{TN}^*}$	$\text{kg ha}^{-1} \text{y}^{-1}$
$L^{\text{PS,TP}^*}$	TP load resulting from the wastewater treatment plants' effluent discharged within the river network considering $C_{\text{eff}}^{\text{TP}^*}$	$\text{kg ha}^{-1} \text{y}^{-1}$
Mm^3	Million of cubic meter (10^6 m^3)	–
NSE	Nash-Sutcliffe Efficiency coefficient	–
PBIAS	Percent deviation	–
PS	Point pollution sources (e.g., WWTPs' effluent)	–
R_{act}	Actual runoff from the catchment	$\text{m}^3 \text{y}^{-1}$
R_{nat}	Natural runoff in the catchment	$\text{m}^3 \text{y}^{-1}$
R^2	Coefficient of determination	–
TN	Total nitrogen	–
TP	Total phosphorous	–
TSS	Total suspended solid	–
V_{eff}	WWTPs' effluent volume	$\text{m}^3 \text{y}^{-1}$
W1	Treated effluent discharged into the river network by Monteiasi wastewater treatment plant	–
W2	Treated effluent discharged into the river network by Montemesola wastewater treatment plant	–
W3	Treated effluent discharged into the river network by San Giorgio Ionico wastewater treatment plant	–
WA_{blue}	Blue surface-water availability in actual conditions	$\text{m}^3 \text{y}^{-1}$
$WA_{\text{blue}}^{\text{nat}}$	Blue surface-water availability in natural conditions (absence of human pressures)	$\text{m}^3 \text{y}^{-1}$
WA_{green}	Green water availability	$\text{m}^3 \text{y}^{-1}$
WA_{grey}	Grey water availability	$\text{m}^3 \text{y}^{-1}$
WF	Total water footprint	$\text{m}^3 \text{t}^{-1}$
WF_{blue}	Blue water footprint	$\text{m}^3 \text{t}^{-1}$
WF_{green}	Green water footprint	$\text{m}^3 \text{t}^{-1}$
$WF_{\text{grey}}^{\text{DS}}$	Grey water footprint ($WF_{\text{grey}}^{\text{DS}} + WF_{\text{grey}}^{\text{PS}}$)	$\text{m}^3 \text{t}^{-1}$
$WF_{\text{grey}}^{\text{PS}}$	Grey water footprint ($WF_{\text{grey}}^{\text{DS}} + WF_{\text{grey}}^{\text{PS}}$)	$\text{m}^3 \text{t}^{-1}$
$WF_{\text{grey}}^{\text{DS}}$	Grey water footprint associated with diffuse source	$\text{m}^3 \text{y}^{-1}$
$WF_{\text{grey}}^{\text{PS}}$	Grey water footprint associated with WWTPs' discharges	$\text{m}^3 \text{y}^{-1}$
$WF_{\text{grey}}^{\text{PS}^*}$	Grey water footprint associated with WWTPs' discharges considering $C_{\text{eff}}^{\text{TN}^*}$ or $C_{\text{eff}}^{\text{TP}^*}$	$\text{m}^3 \text{y}^{-1}$
WPL	Water pollution level due to agricultural activities (DS) and WWTPs' discharges (PS)	–
WPL*	Water pollution level due to agricultural activities (DS) and WWTPs' discharges (PS), considering $C_{\text{eff}}^{\text{TN}^*}$ and $C_{\text{eff}}^{\text{TP}^*}$	–
WPL ^{DS}	Water pollution level due to agricultural activities (DS)	–
WPL ^S	Water pollution level considering $C_{\text{eff}}^{\text{TN}_{\text{S}}}$ and $C_{\text{eff}}^{\text{TP}_{\text{S}}}$	–
WS_{green}	Green water scarcity	–
WS_{blue}	Blue water scarcity	–
$WS_{\text{blue}}^{\text{nat}}$	Blue water scarcity without including PS discharge	–
WWTP	Waste water treatment plant	–

References

- Abbaspour, K.C., 2015. SWAT-CUP: SWAT Calibration and Uncertainty Programs - A User Manual. Eawag, Dübendorf, Switzerland.
- Arnold, J.G., Moriasi, D.N., Gassman, P.W., Abbaspour, K.C., White, M.J., Srinivasan, R., Santhi, C., Harmel, R.D., Van Griensven, A., Liew, M.W. Van, Kannan, N., Jha, M.K., Harmel, D., Member, A., Liew, Michael W. Van, Arnold, J.-F.G., 2012. SWAT: model use, calibration, and validation. *Trans. ASABE* 55, 1491–1508.
- Arsenault, R., Brissette, F., Martel, J.-L., 2018. The hazards of split-sample validation in hydrological model calibration. *J. Hydrol.* 566, 346–362. <https://doi.org/10.1016/j.jhydrol.2018.09.027>.
- Arthington, A.H., 2012. Environmental Flows: Saving Rivers in the Third Millennium. Environmental Flows: Saving Rivers in the Third Millennium. University of California Press <https://doi.org/10.1080/2325548x.2014.919157>.
- Bakker, K., 2012. Water security: research challenges and opportunities. *Science* 337 (6097), 914–915. <https://doi.org/10.1126/science.1226337>.
- Bonamente, E., Rinaldi, S., Nicolini, A., Cotana, F., 2017. National water footprint: toward a comprehensive approach for the evaluation of the sustainability of water use in Italy. *Sustainability* 9, 1341. <https://doi.org/10.3390/su9081341>.
- Brouziyne, Y., Abouabdillah, A., Bouabid, R., Benaabidate, L., Oueslati, O., 2017. SWAT manual calibration and parameters sensitivity analysis in a semi-arid watershed in North-western Morocco. *Arab. J. Geosci.* 10, 1–13. <https://doi.org/10.1007/s12517-017-3220-9>.

- Cai, J., He, Y., Xie, R., Liu, Y., 2020. A footprint-based water security assessment: an analysis of Hunan province in China. *J. Clean. Prod.* 245, 118485. <https://doi.org/10.1016/j.jclepro.2019.118485>.
- Casella, P., De Rosa, L., Salluzzo, A., De Gisi, S., 2019. Combining GIS and FAO's crop water productivity model for the estimation of water footprinting in a temporary river catchment. *Sustain. Prod. Consum.* 17, 254–268. <https://doi.org/10.1016/j.spc.2018.11.002>.
- Ceschin, S., Crescenzi, M., Iannelli, M.A., 2020. Phytoremediation potential of the duckweeds *Lemna minuta* and *Lemna minor* to remove nutrients from treated waters. *Environ. Sci. Pollut. Res.*, 1–9 <https://doi.org/10.1007/s11356-020-08045-3>.
- Cook, C., Bakker, K., 2012. Water security: debating an emerging paradigm. *Glob. Environ. Chang.* 22, 94–102. <https://doi.org/10.1016/j.gloenvcha.2011.10.011>.
- Crini, G., Lichtfouse, E., 2019. Advantages and disadvantages of techniques used for wastewater treatment. *Environ. Chem. Lett.* 17, 145–155. <https://doi.org/10.1007/s10311-018-0785-9>.
- Curk, M., Glavan, M., Pintar, M., 2020. Analysis of nitrate pollution pathways on a vulnerable agricultural plain in Slovenia: taking the local approach to balance ecosystem services of food and water. *Water* 12, 707. <https://doi.org/10.3390/w12030707>.
- D'Ambrosio, E., De Girolamo, A.M., Rulli, M.C., 2018a. Assessing sustainability of agriculture through water footprint analysis and in-stream monitoring activities. *J. Clean. Prod.* 200, 454–470. <https://doi.org/10.1016/j.jclepro.2018.07.229>.
- D'Ambrosio, E., De Girolamo, A.M., Rulli, M.C., 2018b. Coupling the water footprint accounting of crops and in-stream monitoring activities at the catchment scale. *MethodsX* 5, 1221–1240. <https://doi.org/10.1016/j.mex.2018.10.003>.
- D'Ambrosio, E., De Girolamo, A., Spanò, M., Corbelli, V., Capasso, G., Morea, M., Velardo, R., Abdelwahab, O., Lonigro, A., Milillo, F., Ricci, G., Romano, G., Calabrese, A., Casale, B., Mauro, R., Pappagallo, G., Gentile, F., 2019. A spatial analysis to define data requirements for hydrological and water quality models in data-limited regions. *Water* 11, 267. <https://doi.org/10.3390/w11020267>.
- D'Ambrosio, E., Gentile, F., De Girolamo, A.M., 2020. Assessing the sustainability in water use at the basin scale through water footprint indicators. *J. Clean. Prod.* 244, 118847. <https://doi.org/10.1016/j.jclepro.2019.118847>.
- David da Silva, D., Pereira, S.B., Vieira, E. de O., 2020. Integrated water resources management in Brazil. *Integrated Water Resource Management*. Springer International Publishing, pp. 13–26 https://doi.org/10.1007/978-3-030-16565-9_2.
- De Girolamo, A.M., Lo Porto, A., 2020. Source apportionment of nutrient loads to a Mediterranean River and potential mitigation measures. *Water* 12, 577. <https://doi.org/10.3390/w12020577>.
- De Girolamo, A.M., Bouraoui, F., Buffagni, A., Pappagallo, G., Lo Porto, A., 2017a. Hydrology under climate change in a temporary river system: potential impact on water balance and flow regime. *River Res. Appl.* 33, 1219–1232. <https://doi.org/10.1002/rra.3165>.
- De Girolamo, A.M., Barca, E., Pappagallo, G., Lo Porto, A., 2017b. Simulating ecologically relevant hydrological indicators in a temporary river system. *Agric. Water Manag.* 180, 194–204. <https://doi.org/10.1016/j.agwat.2016.05.034>.
- De Girolamo, A.M., D'Ambrosio, E., Pappagallo, G., Rulli, M.C., Lo Porto, A., 2017c. Nitrate concentrations and source identification in a Mediterranean river system. *Rend. Lincei* 28, 291–301. <https://doi.org/10.1007/s12210-016-0593-8>.
- De Girolamo, A.M., Miscioscia, P., Politi, T., Barca, E., 2019. Improving grey water footprint assessment: accounting for uncertainty. *Ecol. Indic.* 102, 822–833. <https://doi.org/10.1016/j.ecolind.2019.03.040>.
- de Miguel, Á., Kallache, M., García-Calvo, E., 2015. The water footprint of agriculture in Duero River Basin. *Sustainability* 7, 6759–6780. <https://doi.org/10.3390/su7066759>.
- European Parliament and Council of the European Union, 2000. *Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for community action in the field of water policy*. *Off. J. Eur. Communities Brussels* 43, 1–73.
- Falkenmark, M., 2003. Freshwater as shared between society and ecosystems: from divided approaches to integrated challenges. *Philos. Trans. R. Soc. Lond. B Biol. Sci.* 358 (1440), 2037–2049. <https://doi.org/10.1098/rstb.2003.1386>.
- Franke, N.A., Boyacioglu, H., Hoekstra, A.Y., 2013. *Grey Water Footprint Accounting: Tier 1 Supporting Guidelines*. Unesco-IHE, Delft.
- Fu, B., Merritt, W.S., Croke, B.F.W., Weber, T.R., Jakeman, A.J., 2019. A review of catchment-scale water quality and erosion models and a synthesis of future prospects. *Environ. Model. Softw.* 114, 75–97. <https://doi.org/10.1016/j.envsoft.2018.12.008>.
- Gil, R., Bojacá, C.R., Schrevens, E., 2017. Uncertainty of the agricultural grey water footprint based on high resolution primary data. *Water Resour. Manag.* 31, 3389–3400. <https://doi.org/10.1007/s11269-017-1674-x>.
- Giri, S., Arbab, N.N., Lathrop, R.G., 2018. Water security assessment of current and future scenarios through an integrated modeling framework in the Neshan River Watershed. *J. Hydrol.* 563, 1025–1041. <https://doi.org/10.1016/j.jhydrol.2018.05.046>.
- Grey, D., Sadoff, C.W., 2007. Sink or swim? Water security for growth and development. *Water Policy* 9, 545–571. <https://doi.org/10.2166/wp.2007.021>.
- GWP, 2019. Mobilising for a water secure world. <https://www.gwp.org/globalassets/global/about-gwp/strategic-documents/gwp-strategy-summary.pdf> (Accessed 7 April 2020).
- Haida, C., Chapagain, A.K., Rauch, W., Riede, M., Schneider, K., 2019. From water footprint to climate change adaptation: capacity development with teenagers to save water. *Land Use Policy* 80, 456–463. <https://doi.org/10.1016/j.landusepol.2018.02.043>.
- Hejzlar, J., Anthony, S., Arheimer, B., Behrendt, H., Bouraoui, F., Grizzetti, B., Groenendijk, P., Jeuken, M.H.J.L., Johnson, H., Lo Porto, A., Kronvang, B., Panagopoulos, Y., Siderius, C., Silgram, M., Venohr, M., Žaloudík, J., 2009. Nitrogen and phosphorus retention in surface waters: an inter-comparison of predictions by catchment models of different complexity. *J. Environ. Monit.* 11, 584–593. <https://doi.org/10.1039/b901207a>.
- Hoekstra, A.Y., 2017. Water footprint assessment: evolution of a new research field. *Water Resour. Manag.* 31, 3061–3081. <https://doi.org/10.1007/s11269-017-1618-5>.
- Hoekstra, A.Y., Hung, P.Q., 2002. Virtual water trade, a quantification of virtual water flows between nations in relation to international crop trade. *Value of Water Research Report Series No. 11*. UNESCO-IHE, Delft, The Netherlands.
- Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M., 2011. *The Water Footprint Assessment Manual*. Earthscan, London – Washington DC.
- Hoekstra, A.Y., Buurman, J., Van Ginkel, K.C.H., 2018. Urban water security: a review. *Environ. Res. Lett.* 13, 053002. <https://doi.org/10.1088/1748-9326/aa5a52>.
- Hoekstra, A.Y., Chapagain, A.K., van Oel, P.R., 2019. Progress in water footprint assessment: towards collective action in water governance. *Water* 11, 1070. <https://doi.org/10.3390/w11051070>.
- Istituto di Ricerca per la Protezione Idrogeologica, 2014. Progetto "Il Mar Piccolo di Taranto: approfondimento tecnico scientifico sulle interazioni tra il sistema ambientale ed i flussi di contaminanti da fonti primarie e secondarie. Conoscenze di geologia tecnica e idrogeologia a supporto della caratterizzazione dei corpi idrici sotterranei". *Relazione e Tavole (in italian)*.
- Jensen, O., Wu, H., 2018. Urban water security indicators: development and pilot. *Environ. Sci. Pol.* 83, 33–45. <https://doi.org/10.1016/j.envsci.2018.02.003>.
- Johnson, M.B., Mehrvar, M., 2019. An assessment of the grey water footprint of winery wastewater in the Niagara Region of Ontario, Canada. *J. Clean. Prod.* 214, 623–632. <https://doi.org/10.1016/j.jclepro.2018.12.311>.
- Kaur, B., Shrestha, N., Daggupati, P., Rudra, R., Goel, P., Shukla, R., Allataifeh, N., 2019. Water security assessment of the Grand River watershed in southwestern Ontario, Canada. *Sustainability* 11, 1883. <https://doi.org/10.3390/su11071883>.
- Liu, C., Kroeze, C., Hoekstra, A.Y., Gerbens-Leenes, W., 2012. Past and future trends in grey water footprints of anthropogenic nitrogen and phosphorus inputs to major world rivers. *Ecol. Indic.* 18, 42–49. <https://doi.org/10.1016/j.ecolind.2011.10.005>.
- Liu, W., Antonelli, M., Liu, X., Yang, H., 2017. Towards improvement of grey water footprint assessment: with an illustration for global maize cultivation. *J. Clean. Prod.* 147, 1–9. <https://doi.org/10.1016/j.jclepro.2017.01.072>.
- Liu, X., Shi, L., Engel, B.A., Sun, S., Zhao, X., Wu, P., Wang, Y., 2020. New challenges of food security in Northwest China: water footprint and virtual water perspective. *J. Clean. Prod.* 245, 118939. <https://doi.org/10.1016/j.jclepro.2019.118939>.
- Lovarelli, D., Bacenetti, J., Fiala, M., 2016. Water footprint of crop productions: a review. *Sci. Total Environ.* 548–549, 236–251. <https://doi.org/10.1016/j.scitotenv.2016.01.022>.
- Lovarelli, D., Ingrao, C., Fiala, M., Bacenetti, J., 2018. Beyond the water footprint: a new framework proposal to assess freshwater environmental impact and consumption. *J. Clean. Prod.* 172, 4189–4199. <https://doi.org/10.1016/j.jclepro.2016.12.067>.
- Malagó, A., Bouraoui, F., Vigiak, O., Grizzetti, B., Pastori, M., 2017. Modelling water and nutrient fluxes in the Danube River Basin with SWAT. *Sci. Total Environ.* 603–604, 196–218. <https://doi.org/10.1016/j.scitotenv.2017.05.242>.
- Mekonnen, M.M., Hoekstra, A.Y., 2010. A global and high-resolution assessment of the green, blue and grey water footprint of wheat. *Hydrol. Earth Syst. Sci.* 14, 1259–1276. <https://doi.org/10.5194/hess-14-1259-2010>.
- Meybeck, M., 1982. Carbon, nitrogen, and phosphorus transport by world rivers. *Am. J. Sci.* 282, 401–450. <https://doi.org/10.2475/ajs.282.4.401>.
- Mikosch, N., Becker, R., Schelter, L., Berger, M., Usman, M., Finkbeiner, M., 2020. High resolution water scarcity analysis for cotton cultivation areas in Punjab, Pakistan. *Ecol. Indic.* 109, 105852. <https://doi.org/10.1016/j.ecolind.2019.105852>.
- Ministero dell'Ambiente e della Tutela del Territorio e del Mare, 2010. *D.M. 08 novembre 2010, n. 260, G.U. n. 30 del 7/02/2011, Suppl. Ordinario n. 31 (in italian)*.
- Morera, S., Corominas, L., Poch, M., Aldaya, M.M., Comas, J., 2016. Water footprint assessment in wastewater treatment plants. *J. Clean. Prod.* 112, 4741–4748. <https://doi.org/10.1016/j.jclepro.2015.05.102>.
- Moriassi, D.N., Arnold, J.G., Liew, M.W.V., Bingner, R.L., Harmel, R.D., Veith, T.L., 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Trans. ASABE* 50 (3), 885–900. <https://doi.org/10.13031/2013.23153>.
- Neitsch, S.L., Arnold, J.G., Kiniry, J.R., Williams, J.R., 2009. *Soil and Water Assessment Tool Theoretical Documentation*. Version 2009. Texas A&M University System, College Station.
- Oki, T., Kanae, S., 2006. Global hydrological cycles and world water resources. *Science* 313 (5790), 1068–1072. <https://doi.org/10.1126/science.1128845>.
- Pellegrini, G., Ingrao, C., Campoese, S., Tricase, C., Contò, F., Huisinger, D., 2016. Application of water footprint to olive growing systems in the Apulia region: a comparative assessment. *J. Clean. Prod.* 112, 2407–2418. <https://doi.org/10.1016/j.jclepro.2015.10.088>.
- Pellicer-Martínez, F., Martínez-Paz, J.M., 2016. The water footprint as an indicator of environmental sustainability in water use at the river basin level. *Sci. Total Environ.* 571, 561–574. <https://doi.org/10.1016/j.scitotenv.2016.07.022>.
- Pophare, A.M., Lamsoge, B.R., Katpatal, Y.B., Nawale, V.P., 2014. Impact of over-exploitation on groundwater quality: a case study from WR-2 watershed, India. *J. Earth Syst. Sci.* 123, 1541–1566. <https://doi.org/10.1007/s12040-014-0478-0>.
- Pulighe, G., Bonati, G., Colangeli, M., Travero, L., Lupia, F., Altobelli, F., Dalla Marta, A., Napoli, M., 2019. Predicting streamflow and nutrient loadings in a semi-arid Mediterranean watershed with ephemeral streams using the SWAT model. *Agronomy* 10 (1), 2. <https://doi.org/10.3390/agronomy10010002>.
- Quinteiro, P., Ridoutt, B.G., Arroja, L., Dias, A.C., 2018. Identification of methodological challenges remaining in the assessment of a water scarcity footprint: a review. *Int. J. Life Cycle Assess.* 23, 164–180. <https://doi.org/10.1007/s11367-017-1304-0>.
- Regione Puglia, 2001. *Studio per la caratterizzazione agronomica della Regione Puglia e la classificazione del territorio in funzione della potenzialità produttiva. Progetto ACIA 2, P.O.P. Puglia, pp. 94–99 Regione Puglia, Bari. (in italian)*.
- Ricci, G.F., Jeong, J., De Girolamo, A.M., Gentile, F., 2020. Effectiveness and feasibility of different management practices to reduce soil erosion in an agricultural watershed. *Land Use Policy* 90, 104306. <https://doi.org/10.1016/j.landusepol.2019.104306>.
- Richter, B.D., Davis, M.M., Apse, C., Konrad, C., 2012. A presumptive standard for environmental flow protection. *River Res. Appl.* 28, 1312–1321. <https://doi.org/10.1002/rra.1511>.

- Rosa, L., Rulli, M.C., Davis, K.F., Chiarelli, D.D., Passera, C., D'Odorico, P., 2018. Closing the yield gap while ensuring water sustainability. *Environ. Res. Lett.* 13, 104002. <https://doi.org/10.1088/1748-9326/aadeef>.
- Rouholahnejad, E., Abbaspour, K.C., Vejdani, M., Srinivasan, R., Schulin, R., Lehmann, A., 2012. A parallelization framework for calibration of hydrological models. *Environ. Model. Softw.* 31, 28–36. <https://doi.org/10.1016/j.envsoft.2011.12.001>.
- Salmoral, G., Willaarts, B.A., Garrido, A., Guse, B., 2017. Fostering integrated land and water management approaches: evaluating the water footprint of a Mediterranean basin under different agricultural land use scenarios. *Land Use Policy* 61, 24–39. <https://doi.org/10.1016/j.landusepol.2016.09.027>.
- Savenije, H.H.G., 2000. Water scarcity indicators; the deception of the numbers. *Phys. Chem. Earth B Hydrol. Oceans Atmos.* 25 (3), 199–204. [https://doi.org/10.1016/S1464-1909\(00\)00004-6](https://doi.org/10.1016/S1464-1909(00)00004-6).
- Schyns, J.F., Hoekstra, A.Y., Booij, M.J., Hogeboom, R.J., Mekonnen, M.M., 2019. Limits to the world's green water resources for food, feed, fiber, timber, and bioenergy. *Proc. Natl. Acad. Sci. U. S. A.* 116, 4893–4898. <https://doi.org/10.1073/pnas.1817380116>.
- Smith, R.A., Alexander, R.B., Schwarz, G.E., 2003. Natural background concentrations of nutrients in streams and rivers of the conterminous United States. *Environ. Sci. Technol.* 37, 3039–3047. <https://doi.org/10.1021/es020663b>.
- Sun, F., Staddon, C., Chen, M., 2016. Developing and applying water security metrics in China: experience and challenges. *Curr. Opin. Environ. Sustain.* 21, 29–36. <https://doi.org/10.1016/j.cosust.2016.10.006>.
- Tharme, R.E., 2003. A global perspective on environmental flow assessment: emerging trends in the development and application of environmental flow methodologies for rivers. *River Res. Appl.* 19, 397–441. <https://doi.org/10.1002/rra.736>.
- UNESCO, UNESCO i-WSSM, 2019. Water security and the sustainable development goals (series I). <https://digitallibrary.un.org/record/3807832> (Accessed 7 April 2020).
- Veettil, A.V., Mishra, A.K., 2016. Water security assessment using blue and green water footprint concepts. *J. Hydrol.* 542, 589–602. <https://doi.org/10.1016/j.jhydrol.2016.09.032>.
- Veettil, A.V., Mishra, A.K., 2018. Potential influence of climate and anthropogenic variables on water security using blue and green water scarcity, Falkenmark index, and freshwater provision indicator. *J. Environ. Manag.* 228, 346–362. <https://doi.org/10.1016/j.jenvman.2018.09.012>.
- Wei, X., Bailey, R.T., Records, R.M., Wible, T.C., Arabi, M., 2019. Comprehensive simulation of nitrate transport in coupled surface-subsurface hydrologic systems using the linked SWAT-MODFLOW-RT3D model. *Environ. Model. Softw.* 122, 104242. <https://doi.org/10.1016/j.envsoft.2018.06.012>.
- Wöhler, L., Niebaum, G., Krol, M., Hoekstra, A.Y., 2020. The grey water footprint of human and veterinary pharmaceuticals. *Water Res.* X 7, 100044. <https://doi.org/10.1016/j.wroa.2020.100044>.
- Yu, X., Moraetis, D., Nikolaidis, N.P., Li, B., Duffy, C., Liu, B., 2019. A coupled surface-subsurface hydrologic model to assess groundwater flood risk spatially and temporally. *Environ. Model. Softw.* 114, 129–139. <https://doi.org/10.1016/j.envsoft.2019.01.008>.
- Zang, C.F., Liu, J., van der Velde, M., Kraxner, F., 2012. Assessment of spatial and temporal patterns of green and blue water flows under natural conditions in inland river basins in Northwest China. *Hydrol. Earth Syst. Sci.* 16, 2859–2870. <https://doi.org/10.5194/hess-16-2859-2012>.
- Zhang, G., Mathews, R.E., Frapporti, G., Mekonnen, M.M., 2014. *Water Footprint Assessment for the Hertfordshire and North London Area. Report RESE000335.* Environment Agency, London, UK.