Appraisal of the water footprint of irrigated agriculture in a semi-arid area: The Segura River Basin

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Abstract

Irrigated agriculture is a key activity in water resources management at the river basin level in arid and semi-arid areas, since this sector consumes the largest part of the water resources overall. The current study proposes a methodology to evaluate the water footprint (WF) of the irrigated agriculture sector at the river basin level, through a simulation of the anthropised water cycle combining a hydrological model and a decision support system. The main difference from the approaches that have already been used is that the new methodology includes the limitations of the system for the exploitation of water resources where the irrigated areas are located, and it considers the hydrological principles governed by the law of continuity of mass. Water footprint accounting was carried out for the Segura River Basin (South-eastern Spain), applying the methodology proposed and another that is usually applied. The results of the two methodologies were compared, revealing significant differences in the values of the WF, basically due to the blue component. The methodology that is usually applied over-estimated the WF of the agriculture in the basin since supply deficits were not taken into account, providing results that would only be possible if there were no spatial or temporal restrictions to water use. So, in order to make the WF indicator useful in water resources management plans, it is necessary to adapt the computations to the main characteristics of the water exploitation system of the whole basin under study, respecting the hydrological principles of the water cycle: regulation and transport infrastructure, the real water resources available and the priority of access to water between concurrent water uses.

Introduction

The scarcity of and/or pollution in continental fresh water is currently one of the main issues regarding natural water resources management at the global level [1], particularly in semi-arid areas with low availability of resources and concurrent water uses [2]. The assessment of these available water resources and their use is a priority for the authorities in charge of water
allocation, which is generally developed at the basin level [3]. Agriculture is, by far, the sector that demands the largest water volume at the global level, accounting for no less than 70% of the water used in the world [4]. Therefore, the analysis of water use in agriculture is imperative for water resources management policies.

The water footprint (WF) is an indicator that measures the use of fresh water as a production factor at different levels (goods, services, businesses, geographical areas and so on). It takes into consideration water sources, giving rise to three components represented by colours: green, blue and grey [5]. The green component is the consumption of the precipitation that is stored in the root zone of the soil by plants (WF\textsubscript{Green}). The majority of this consumed water is lost from the plants by evapotranspiration, although a small part can be incorporated into the plants. The blue component is water that has been sourced from surface or groundwater resources and either evaporates, is incorporated into a product or taken from one water body and returned to another or is returned to the same basin but in a time period different from that of the analysis being carried out (WF\textsubscript{Blue}). The grey component is the amount of fresh water that would be required to assimilate any pollution and meet specific quality standards (WF\textsubscript{Grey}) [6]; this volume does not represent actual consumption. Thus, the WF quantifies the pressure of human activity on water resources, providing results in terms of fresh water volume (homogeneous unit), and can be employed as an indicator of the sustainability of the spatial-temporal management of this resource at the local or regional scale [7]. Therefore, the WF is a multi-dimensional indicator that is particularly adequate for the comprehensive assessment of water use in agricultural activities [8].

In fact, due to the important role of irrigation in the use of water resources and the potential of the WF indicator, a large number of practical applications of WF accounting have been developed [9, 10]. Studies generally use a soil water balance model, in which the value of crop evapotranspiration due to precipitation is the WF\textsubscript{Green}, whereas the irrigation water consumed by crops (evapotranspiration) is the WF\textsubscript{Blue} [11, 12]. In some cases, the water applied is considered instead of the consumption by the plant, and the leaks and leachates which are not consumed by crops have been called the white water footprint [13]. Finally, the WF\textsubscript{Grey} which has not been considered in most studies [14, 15] until recently, is generated by the agrochemicals (fertilisers, pesticides, etc.) that reach and hence pollute surface and groundwater [16].

Despite the large number of studies on the WF in agriculture, there are criticisms of the methodologies applied and its usefulness in the management of water resources [17–19]. These derive from the limitations of the methodologies that normally are applied to determine this indicator (see [14, 20], among others). For local or regional cases, the following stand out: it is assumed that there are always blue water resources for an optimal supply and that these resources always have hydraulic infrastructure that enables them to reach the irrigated areas. So, the blue and green components are evaluated separately, ignoring the hydrological principles of continuity of mass. If these limitations are ruled out in a scenario with competition for the same blue water resources, this results in an overestimated value of WF\textsubscript{Blue}, even more so when considering that irrigation has a lower priority than the urban water supply [19], as is usually the case in resource allocation policies [21].

So, all this can lead to a WF value that does not represent reality and cannot be used as a water resources management indicator. In this sense, the main purpose of the current work is to develop a methodology for the accounting of the WF of irrigated agriculture in a specific geographical area that deals with the above-mentioned limitations. This WF accounting is structured on the results from the simulation of the anthropised water cycle of a river basin [22], which links modelling of the natural hydrology to the exploitation system of water resources (hydraulic infrastructure, available water resources, supply sources, priority between water uses, etc.). Therefore, although this WF accounting takes into consideration the influence of the soil
on plants, it is done from the engineering perspective of water resources management. In addition, the results obtained with the proposed methodology are contrasted with those obtained with one of the methodologies that has been applied elsewhere, in which there are no water deficits for irrigation.

Both methodologies were applied to the Segura River Basin (SRB), one of the most complex territorial units of water resources management in Europe [23]. This Mediterranean coastal area has a semi-arid climate, the annual average potential evapotranspiration being more than twice the average annual precipitation [24]. In addition, in this region there is intensive use of water, not only for agricultural irrigation purposes, but also for urban supply, tourism and industry, which demand ever larger volumes for their development [25]. The low availability of natural water resources in the SRB has led to a complex supply system and, most importantly, to the assignment of smaller volumes of water to some areas of irrigated crops that would receive greater volumes under optimal supply conditions [26]. Therefore, this case study, beyond its regional interest, is a clear example of irrigated agriculture located in a semi-arid area that is very productive in general terms but has water availability as one of its limiting factors [27].

**Materials and methods**

**Crop water footprint accounting**

The accounting of the water footprint of irrigated agriculture (WFIA) in a geographical area is based on the estimation of the WF of the crops existing in it for a given period [6]. First, the WF of the main processes that consume or pollute water is estimated for each crop (Eq 1). These processes are mainly the evapotranspiration of rainwater and irrigation water, the accumulation of water in plants and their products (which is not usually taken into account in the final calculation due to its relatively low importance) and the use of agrochemicals. Depending on the aim and scope of the study, and the baseline information, there are different approaches to WF accounting in the specialized literature, from the application of monthly modelling for large areas [28–30] to the study of small plots through daily monitoring [31]. However, in all of them, rainwater evapotranspiration is the green component of the WF (WF\textsubscript{Green} [volume/time]), the net irrigation water consumed or evapotranspired is the blue component (WF\textsubscript{Blue} [volume/time]) and the excess of fertilisers/pesticides that ends up in water bodies determines the grey part (WF\textsubscript{Grey} [volume/time]). Once the WF has been assessed for each crop, the results are aggregated for all of the area under analysis [8, 14, 20].

\[
WF = WF_\text{Green} + WF_\text{Blue} + WF_\text{Grey} [\text{volume/time}]
\]  

(1)

The green and blue components are usually evaluated using soil water balance models, such as CROPWAT, based on the approach of [32], AquaCrop [33] or CropSyst [34]. In these models, crop evapotranspiration under standard conditions (ET\textsubscript{c}) is calculated from climatic variables (ET\textsubscript{0}) and agronomic data (K\textsubscript{c}). If ET\textsubscript{c} is greater than the effective precipitation, then the effective precipitation is the green component of the water consumed (CWC\textsubscript{Green}) and coincides with the evapotranspiration under non-standard conditions (ET\textsubscript{c,adj}). But, if ET\textsubscript{c} is lower than the effective precipitation, then the CWC\textsubscript{Green} is ET\textsubscript{c} (Eq 2). When ET\textsubscript{c}—ET\textsubscript{c,adj} is higher than 0, there is a water deficit for the crops. This water deficit (also called the crop water requirement) can be compensated by irrigation and is the blue component of the water consumed (CWC\textsubscript{Blue}) (Eq 3) [6]. Water deficits can be totally covered if one assumes optimal conditions for each crop [14], and the sum of the green and blue components is equal to ET\textsubscript{c}. But, there is also a more realistic option when establishing an irrigation programme for each crop.
[12], and the sum of the green and blue components may be less than ETc [17].

\[
WF_{\text{Green}} = CWC_{\text{Green}} = \min(ETc, ET_{c,\text{adj}}) \text{[volume/time]}
\]  

(2)

\[
WF_{\text{Blue}} = CWC_{\text{Blue}} = \max(0, ETc - ET_{c,\text{adj}}) \text{[volume/time]}
\]  

(3)

The grey component of the WF (WF_{\text{Grey}}) is calculated by dividing the pollutant load (L [mass/time]) of each k pollutant by the difference between the ambient water quality standard for that pollutant (c_{\text{max}} [mass/volume]) and its natural concentration in the receiving water body (c_{\text{nat}} [mass/volume]). The variable c_{\text{max}} is the maximum or limit concentration of each k pollutant that the receiving water body is able to assimilate, and c_{\text{nat}} is the concentration of each k pollutant that would occur if there were no human disturbances in the catchment where the water body is located (Eq 4). The equation is evaluated for every pollutant, and the one that requires the greatest volume for its assimilation (\max[k]), known as the critical pollutant, determines the value of WF_{\text{Grey}} [6, 35].

\[
WF_{\text{Grey}} = \max[k] \frac{L[k]}{(c_{\text{max}}[k] - c_{\text{nat}}[k])} \text{[volume/time]}
\]  

(4)

**Water footprint of irrigated agriculture with full supply (WFIA-FS)**

The methodology defined, assuming optimal conditions for crops, and applied in this study is referred to as WFIA-FS (Water Footprint of Irrigated Agriculture with Full Supply) [14]. This methodology is usually applied for the crop pattern of a given year, and the soil balance is simulated using the climatic data for that year. In this case, the crop pattern of a given year was fixed and simulated for the climatic data of a period (several consecutive years). Thus, the climatic variability, which is a determining factor in the calculation of the WF in agriculture, is included in the results.

So, the soil water balance used to calculate the WF was defined with the distributed hydrological model SIMPA [36]. This model simulates, monthly, the water cycle along a period of time. The input data include climate series of precipitation (P) and the crop evapotranspiration under standard conditions (ETc), calculated from the reference crop evapotranspiration (ET0) and crop coefficients (Kc). The results are ETc,adj series distributed in cells. So, the CWC_{\text{Green}} and CWC_{\text{Blue}} (water deficit) series are calculated from the ETc and ETc,adj series of the cells where crops are located (Eqs 2 and 3).

The grey component of the WF (WF_{\text{Grey}}) is calculated by the following expression (Eq 5):

\[
WF_{\text{Grey}} = \max[k] \frac{z[k] \times AR[k]}{(c_{\text{max}}[k] - c_{\text{nat}}[k])} \text{[volume/time]}
\]  

(5)

This equation assumes that the pollutant load (L) reaching a water body is a percentage (z [%]) of the quantity of agrochemicals (AR [mass/time]) applied to the crops [16], as fertilisers/pesticides. For this case, the k pollutant loads considered are nitrate and phosphate due to fertilisers, taking into account the monthly pattern of their application. Although there are other pollutants in the return flows [37], these are the main source of contamination caused by irrigation in the water bodies [12]. This z percentage depends on the climate, soil, agricultural practices, slope and runoff [16, 34]. In this work, the variables considered to calculate this percentage were the slope of each cell and the amount of irrigation water applied in each cell every month [8, 29, 38]. The volume of water applied, instead of runoff, is used since it is irrigated agriculture. As for the water bodies that receive the pollutants, two possibilities have been considered: irrigated crops located in river plains that discharge directly into the river
(surface water bodies), or discharges of crops located far from a natural streamflow that end up polluting aquifers (groundwater bodies) (Fig 1).

**Water footprint of irrigated agriculture with exploitation system (WFIA-ES)**

The methodology proposed in this study is called Water Footprint of Irrigated Agriculture with Exploitation System (WFIA-ES). For this methodology, a crop pattern scenario is also set for a given year. Other uses of water are added to this scenario, such as urban and industrial supply, as well as the priority of water allocation. This scenario is simulated over a period of consecutive years, providing the value of the WF for the climate of different years.

Unlike the previous methodology (WFIA-FS), in the WFIA-ES the blue WF depends on the irrigation programme established for each crop. Specifically, irrigated areas have legal water allocations, which vary monthly, enabling access to a certain amount of water that depends on the crop mosaic within them. So, $WF_{\text{Green}} + WF_{\text{Blue}}$ usually is less than $ET_c$ since it depends

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Fig 1. Methodological scheme for WFIA-FS accounting.

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on the water allocations. If sufficient water resources are available, taking into account the current water uses and the existing hydraulic infrastructure, then the supplies for irrigation are complete and coincide with the water allocations. Otherwise, there are water deficits in some irrigated areas and the difference between WF\textsubscript{Green} plus WF\textsubscript{Blue} and ET\textsubscript{c} increases.

This methodology begins with the SIMPA distributed hydrological model. It provides temporal series of ET\textsubscript{c,adj}, run-off and recharge of the aquifers. The WF\textsubscript{Green} is calculated using the ET\textsubscript{c} and ET\textsubscript{c,adj} series of the crops (Eq 2), as in the WFIA-FS methodology. The run-off and recharge series of the aquifers are the blue water resources of the basin (natural water resources, Qn [volume/time]). These series of blue water resources are used in the following step: modelling of the integrated system of water resources. This kind of modelling is usually carried out with a Decision Support System (DSS), which simulates a water network with water resources inputs and water uses [39]. The DSSs represent the interrelations between the main elements of the water exploitation system: rivers, reservoirs, lakes, aquifers, intakes, uses, desalination plants, return flows from water uses, etc. [40]. The main results are the series of the supply to irrigation [volume/time] and their return flows [volume/time], which are used in this methodology to account the WF\textsubscript{Blue} and WF\textsubscript{Grey}, respectively. The water uses are located throughout the water network, as potential demands (water allocations). In other words, the maximum optimal values are defined and they are reached completely only when the resource is available in the area in which they are located. The natural water resources are incorporated into the water network at the locations where they are generated; for instance, inputs to reservoirs [41]. As stated previously, DSS models also allow the incorporation of non-conventional water sources (Qa [volume/time])—such as desalination and/or transfers—into the exploitation systems (Fig 2). Finally, the DSSs incorporate the return flows from the water uses into the same water network; therefore, when the topological conditions are appropriate, these water volumes can be reused in another water use (e.g. an irrigation area located downstream), preventing double-accounting in the WF\textsubscript{Blue} calculation [18].

The DSS used in this work is the so-called Optiges (Fig 2), the optimisation module of AquatoolDMA, a platform to simulate water exploitation systems that is widely used around the world due to its versatility [39, 42, 43, 44, 45]. This DSS has been tested sufficiently in the basin under study [46, 47], since it is able to establish in the exploitation system the order of priority among the different concurrent water uses according to the Spanish water planning law [48, 49]. This order of priority is as follows, from greater to lesser priority: environmental requirements, urban-tourism, irrigation and industrial demands. This scheme of priorities at the operational level allows the DSS to provide resources to only one use when high-priority uses are already covered with a guaranteed level specified by law for each one of them [49]. So, this DSS distributes water resources among different concurrent water uses on a monthly basis, taking into account the topology of the water network and following three criteria: i) environmental requirements (flows and consumption by wetlands) as established by the WFD [50]; ii) the supply of available water resources, following the established order of priority of uses so as to distribute deficits between demands in times of scarcity [51]; and iii) the storage of maximum water volumes in reservoirs once the two former criteria are met.

The WF\textsubscript{Blue} is determined as the series of water supplied less the series of return flows not consumed by crops, whereas the WF\textsubscript{Grey} needs to be calculated for each series of irrigation return flows, using the following expression (Eq 6):

$$WF_{\text{Grey}} = \max[k] \left[ \frac{Q_{\text{eff}} \cdot c_{\text{eff}}[k]}{(c_{\text{max}}[k] - c_{\text{sur}}[k])} \right] [\text{volume/time}]$$ (6)
The pollutant load is determined from a return flow ($Q_{\text{eff}}$ [volume/time]) and the concentration ($c_{\text{eff}}$ [mass/volume]) of the different pollutant $k$ substances that it contains. The Optiges DSS provides return flows ($Q_{\text{eff}}$) and specifies the water body into which they are incorporated, being able to distinguish between surface and groundwater. The concentration of each pollutant comes from measurements of crop leachates ($c_{\text{eff}}$). The maximum permissible concentrations for each pollutant ($c_{\max}$) are established by law for the studied river basin, whereas the natural concentrations ($c_{\text{nat}}$) are defined by measurements of un-altered water bodies in the
studied river basin [52]. Finally, the $WF_{\text{Grey}}$ value for each irrigation return flow is also determined by the critical pollutant [6, 35].

Case study
The Segura River Basin (SRB) district, which includes the SRB and other small coastal catchments without permanent streamflows, is located in South-eastern Spain and covers an area of 18 740 km$^2$ (Fig 3). The co-existence of good-quality soils, a semi-arid climate and water resources, both surface and groundwater, has fostered the development of one of the most...
productive irrigated agriculture systems in Europe [53]. Additionally, the horticultural sector is extremely advanced, with major exports, and there is also an important associated agro-industrial cluster. Currently, irrigated crops occupy about 262,393 ha, divided into 75 irrigation districts, called "Agrarian Demand Units". Irrigation uses over 85% of the available water resources [54]; regarding the method of irrigation, 73% is by drip irrigation, 25% by gravity and 2% by sprinkling. The origin of the water for irrigation is variable according to the hydrological year conditions, being, on average, 29% rainfall-runoff (superficial water), 20% inter-basin water transfer, 38% groundwater, 7% treated wastewater and 6% desalinated seawater. The second-greatest use by volume is the urban sector (14%), which supplies both the permanent population of around two million people and the strong tourism sector on the coast, especially in summer. Industry represents barely 1% of the water demands, although a large part of the industrial activity is directly connected to the urban distribution network [26].

The average precipitation is 400 mm/year [55], although with a strong variability in space and time: it can be over 1000 mm/year in the North-western areas and under 200 mm/year on the coast. Maximum precipitation occurs in the winter and spring months (December-May), whereas precipitation is rare in the summer (June-September). The climate conditions lead to a noticeable seasonal pattern in the natural water resources available in the SRB, which is the opposite of that of the water demands of agriculture and tourism (Fig 4). This intra-annual gap between resources and demands, together with the frequent droughts, has led to the construction of important hydraulic infrastructures, such as channels and reservoirs, to connect irrigated lands. The capacity of the reservoirs (over 1100 $10^6$ m$^3$) is larger than the mean annual surface water resources. This is very important with regard to accumulation of water for drought periods and in case of increasing demands, and to minimise the damage resulting

![Image](https://doi.org/10.1371/journal.pone.0206852.g004)

**Fig 4.** Average intra-annual variability of natural water resources for the period 1940–2010, and the water uses for the year 2015: irrigation demand, urban, industrial and tourism.

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from floods. However, the supply problems have not yet been solved because there is no continuous excess of water resources available to be stored. An important transfer system from the Tagus river basin started to operate in 1979, to provide the SRB with additional resources [56]. Also, seawater desalination has been implemented [57], together with wastewater treatment and its direct reuse in irrigation [58, 59]. Despite the measures put in place to increase the available resources and the numerous management plans aimed at saving resources, such as the modernisation of irrigation systems, the SRB is currently the only river basin with a structural deficit in Spain, as acknowledged by authorities and institutions [26, 47]. This implies occasional supply deficits that are only diminished by the over-exploitation of aquifers [60].

Data

The scenario for both methodologies was the year 2015. The crop pattern used to determine the crop evapotranspiration under standard conditions (ET$_c$) for both methodologies was the Corine Land Cover 2000 classification of soil uses [14, 61]. This distribution of crop irrigation is representative of that year (2015) since the irrigated area and the crop pattern have not changed substantially since 2000. In fact, due to the scarce resources available, it is not possible to increase the irrigated area, only to maintain it [26]. The distribution of the irrigated areas, grouped according to the productive orientations, is: citrus (29%), vegetables (28%), stone-fruit trees (16%), vineyards (8%), olive trees (7%), almond (5%), intensive horticultural crops (2%), winter cereals (2%) and others (3%) [26]. The climatic data used in both methodologies cover the period 1940–2010.

The SIMPA hydrological model considers this crop scenario and the range of climatic data specified, providing the WF$_{Green}$ series for both methodologies. However, regarding the blue water, WFIA-FS determines the WF$_{Blue}$ series as the water deficit in the irrigated areas, whereas WFIA-ES uses the modelling of the integrated system of water resources. The year 2015 is the reference year for the design of the water exploitation system in the WFIA-ES methodology [50]. The irrigation water demand is defined according to the crop groups in the Basin Plan [26], which is based on the calculations of the National Irrigation Plan [62]–this considers the theoretical gross demand (the result of the net needs of each crop and the irrigation efficiency) and the allocation index of the agrarian demand unit. The crops of the basin are slightly under-endowed, their average supply being 75–90% of the theoretical gross demand. The modelling of the water exploitation system also considers environmental requirements, and urban and industrial uses. The environmental requirements are the ecological flows and the water volumes destined to the conservation of wetlands [63, 64]. Urban uses include the urban supply and tourism, and industrial uses are those that are unconnected to the urban network (Table 1). Irrigation constitutes by far the largest volume in the case study (more than 80% of the water used). The available water resources include the natural ones (the mean of the results of the SIMPA model), the resources from transfers, those from seawater desalination and the return flows that likely can be reused [65].

The pollutants considered in both methodologies are nitrate and phosphate. For the WFIA-FS, the values of the application of fertilisers (AR) were established according to the practical guide on crop fertilisation in Spain [66], which establishes the mean values for each type of crop, taking into account the monthly pattern of the fertiliser application. The digital elevation map used to produce the slope map is available from the Spanish National Geographic Information Centre (http://www.ign.es/).

In the WFIA-ES methodology (Eq 6), the nitrate and phosphate loads employed were based on the study by [67], in which the pollutant concentrations in leachates of irrigated crops are discussed. These loads in the leachates showed high variability (as corroborated by other
studies, such as those of [68, 69, 70]), underlining the fact that the larger the return flow in relative terms, the lower the concentration of nutrients ($c_{\text{eff}}$) because they are more diluted. Therefore, this variability was introduced by a linear interpolation of pollutant concentrations for the range of percent returns in the basin: 5–15%. Thus, for the concentration of nitrate, the range of variability was established as: between 173 (mg/L), for return flows of 5%, and 28 (mg/L), for return flows of 15%. The same range of percent returns was used for the phosphate (see Table 2). These percentages were based on the results of [67] and on data published by the water board for previous investigation in different irrigation areas [26, 46, 47, 65].

Regarding $c_{\text{nat}}$, the measurements made in the unaltered surface and groundwater bodies showed that the nitrate and phosphate concentrations were around zero [71], which allowed the establishment of null values for the natural concentrations of both pollutants.

For aquifers, only nitrate was included in the calculation of the WF$_{\text{Grey}}$, since the current law does not contemplate a limitation for phosphates in aquifers, whereas both pollutants were analysed in surface water bodies (Table 2). In the latter case, the pollutant that needs a greater volume of water in order to be assimilated is the one that establishes the value of the grey water footprint (called the critical pollutant), since this volume is capable of diluting both pollutants [35, 37].
Results

The water footprint of irrigated agriculture (WFIA) results for the SRB are presented below, according to the methodology (WFIA-FS and WFIA-ES). In addition, a comparison of the two methodologies is included in the final section.

Water footprint of irrigated agriculture with full supply (WFIA-FS)

The WFIA-FS had a mean value of $4403 \times 10^6$ m$^3$/year, ranging between $3770$ and $4563 \times 10^6$ m$^3$/year. The breakdown of the WFIA-FS revealed that the blue component had the greatest weight (67%), more than double that of the green (28%), while the grey component accounted for barely 5% of the WFIA-FS (Table 3). These percentages are very similar to those obtained for the irrigated lands of the Guadiana river basin (close to the SRB, with a similar climate), to which the same methodology was applied, using the CROPWAT model for the soil water balance [14].

The standard deviation of the WFIA-FS was smaller than those of the green and blue components of the WFIA-FS (Table 3). In wet years there is more water available in the soil and the crop evapotranspiration under non-standard conditions ($ET_{c,adj}$) is greater (greater $WF_{\text{Green}}$). Therefore, water deficits are lower and $WF_{\text{Blue}}$ decreases. This is a consequence of the methodology applied, since the sum of $WF_{\text{Green}}$ and $WF_{\text{Blue}}$ has to be $ET_c$. So, this results in a negative correlation between these two components and the trade-off between them balances the aggregated value of WFIA-FS. $WF_{\text{Grey}}$ had a greater relative variability, resulting from the amount of irrigation applied, since the slope and the AR do not vary with time. In years with less precipitation $WF_{\text{Grey}}$ would increase, as the amount of irrigation water applied —that washes the fertilisers into the water bodies—would also increase (Fig 5). Finally, the WFIA-FS and its three components behaved like normal random variables (they derive from the same data series: evapotranspiration and precipitation), checked with the Shapiro-Wilk test and the Anderson-Darling test at 1% [72], and the results can be expressed in probabilistic terms or with confidence intervals. This information enables one to know the probability of obtaining a WFIA-FS higher than a given value; for example, there is a 95% probability of obtaining a WFIA greater than $4157 \times 10^6$ m$^3$/year.

Table 3. Main statistics of WFIA-FS, WFIA-ES and their respective components, and the relative values of WFIA-FS (%) with respect to WFIA-ES.

| WFIA-FS ($10^6$ m$^3$/year) | Main statistic | $WF_{\text{Green}}$ | $WF_{\text{Blue}}$ | $WF_{\text{Grey}}$ | WFIA-FS |
|-------------------------------|----------------|----------------------|----------------------|-------------------|---------|
| Average                       | 1214           | 2957.1               | 231.4                | 4402.5            |
| Maximum                       | 2036.6         | 3631.7               | 364.4                | 4563.4            |
| Minimum                       | 744.2          | 1929.8               | 29                   | 3769.8            |
| Standard deviation (Sd)       | 301.6          | 377.9                | 74.5                 | 125.1             |

| WFIA-ES ($10^6$ m$^3$/year) | Main statistic | $WF_{\text{Green}}$ | $WF_{\text{Blue}}$ | $WF_{\text{Grey}}$ | WFIA-ES |
|-------------------------------|----------------|----------------------|----------------------|-------------------|---------|
| Average                       | 1214           | 1404.8               | 254.9                | 2873.7            |
| Maximum                       | 2036.6         | 1422.1               | 245.8                | 3717              |
| Minimum                       | 744.2          | 1213.6               | 258.3                | 2216.6            |
| Standard deviation (Sd)       | 301.6          | 30.4                 | 1.88                 | 315               |

| Relative values of the WFIA-FS (%) | $WF_{\text{Green}}$ | $WF_{\text{Blue}}$ | $WF_{\text{Grey}}$ | WFIA |
|------------------------------------|----------------------|----------------------|-------------------|------|
| Average                            | -                    | 210%                 | 91%               | 153% |
| Maximum                            | -                    | 255%                 | 148%              | 123% |
| Minimum                            | -                    | 159%                 | 11%               | 170% |

In the relative values of the WFIA-FS, $WF_{\text{Green}}$ was not introduced into this analysis because it is the same in both approaches. Full data available at: WFIA-FS: https://doi.pangaea.de/10.1594/PANGAEA.892557; WFIA-ES: https://doi.pangaea.de/10.1594/PANGAEA.892558

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Water footprint of irrigated agriculture with exploitation system (WFIA-ES)

The WFIA-ES had a mean value of $2874 \times 10^6$ m$^3$/year, ranging from $2217 \times 10^6$ to $3717 \times 10^6$ m$^3$/year. The blue component's weight (49%) was greater than that of the green (42%), whereas the grey component accounted for 9% of the WFIA-ES (Table 3). The standard deviation of the WFIA-ES depended mainly on the green component; the standard deviations of the other two components (blue and grey) were much lower. The low variability of the blue WF is the result of the agricultural water management in the basin studied, and is basically due to the legal water allocations of the irrigated areas and the exploitation system included in this methodology through the DSS model. This model considered the regulation of surface water by reservoirs, the functioning of desalination plants in drought periods, and/or access to non-renewable groundwater in over-exploited aquifers of some irrigated lands. So, the WFIA-ES only had a positive, significant relationship with WF$_{\text{Green}}$ and a non-significant one with the other two components. Finally, WF$_{\text{Blue}}$ and WF$_{\text{Grey}}$ had a positive, significant relationship, because when supply increases, so do the return flows.

The WFIA-ES and the WF$_{\text{Green}}$ component behaved like normal variables, according to the Shapiro-Wilk and Anderson-Darling tests at 1% [69]. Since WF$_{\text{Blue}}$ and WF$_{\text{Grey}}$ did not fit a normal distribution, their fits to other distributions were tested: WF$_{\text{Blue}}$ fitted a Cauchy distribution, whereas WF$_{\text{Grey}}$ fitted a Gumbel-Min distribution best. Therefore, as with the previous methodology, the results can be presented in probabilistic terms and allow the performance of statistical inferences. But, unlike WFIA-FS, the blue and grey components had a distribution with a heavy left tail, representing the years with deficits. The statistical behaviour of the WF shows that WF$_{\text{Green}}$ is a random variable that depends essentially on natural phenomena whereas WF$_{\text{Blue}}$ and WF$_{\text{Grey}}$ are random variables that are influenced more by anthropic actions.
Discussion

Two methodologies were applied to the same case study. A comparison of the values obtained, in the form of index numbers, is presented in the last rows of Table 3. The first fact to be underlined is that the average value of the total WFIA-FS (4403 $10^6$ m$^3$/year) is 53% greater than for the methodology that incorporated the exploitation system (2874 $10^6$ m$^3$/year). The same comparison of the two methodologies shows an increased discrepancy, of 70%, for the minimum values, whereas the maximum values are closer, with a difference of only 23%.

The WF$_{Blue}$ was mainly responsible for these differences, as the methodology with full supply gave values for this component that clearly doubled (210%) those of the new methodology proposed. Representation of the WF$_{Blue}$ series for both methodologies and their comparison with the series of available natural water resources in the SRB over the same period shows that there would not be sufficient water to cover the WF$_{Blue}$ obtained when taking into account full supply, not even when considering over-exploitation of aquifers and/or alternative sources. It is clear then that the WF$_{Blue}$ values, and hence the WFIA, are overestimated with WFIA-FS, and the blue water consumption values provided are impossible in the SRB owing to the actual availability [17, 19]. This comparison demonstrates that the green and blue components of the WFIA have to be accounted by following the hydrological principles [17], as the WFIA-ES does provide WF$_{Blue}$ values that are compatible with the availability of resources in the SRB.

Fig 6 also shows the high variability of the blue component of the WFIA-FS and its negative relationship with the series of available natural water resources, which is hard to explain in physical terms. This contrasts with the relative stability of the WF$_{Blue}$ calculated using WFIA-ES—which, as stated previously, is the result of water allocations, reservoir regulation, desalination and aquifer over-exploitation.
The average values of WF\textsubscript{Grey} were alike for the two methodologies, being 9% higher in WFIA-ES, although the extreme values of WF\textsubscript{Grey} differed between the two approaches. Both methodologies, in accordance with the findings in previous studies [16], yielded a positive relationship between the WF\textsubscript{Grey} and WF\textsubscript{Blue} values: linear for WFIA-FS and non-linear for WFIA-ES. Regarding the critical pollutant, nitrate was responsible for 80% of the WF\textsubscript{Grey} in the WFIA-FS methodology, a value that increased to 97% when calculated by the WFIA-ES methodology. These differences can be negligible if the considerations adopted are taken into account: the fertiliser load applied does not depend on the year (WFIA-FS) and the concentrations and their variability in the returns are introduced in a simplified way (WFIA-ES). It should also be noted for the two methodologies used that not all the variability of the grey component is reflected in the final results, leaving out two important sources of uncertainty: the uncertainty of the data used and the maximum concentration (c\textsubscript{max}) that is set, which in this case depends on legal criteria. The choice of the c\textsubscript{max} is crucial since it can modify the WF\textsubscript{Grey} value substantially or even change the critical pollutant. In this sense, although the grey component serves to show the degree of contamination of the basin, it will always be fixed by the restrictive or lax limitations of the stipulated maximum concentrations.

Focusing the analysis on WFIA-ES, the proposed methodology addresses the appraisal of the WFIA at the level of the river basin from the hydrometeorological perspective, and considering the hydraulic characteristics of its water exploitation system. Therefore, it focuses on one of the components of scarcity, namely the water supplies. However, the social and productive structure of the river basin must not be forgotten, since it is the other component of the scarcity concept. Thus, droughts due to anomalous deficits in the supplies to water uses, known as operational droughts, are not produced exclusively by a decline in natural water resources (hydrological droughts), they are also produced by an excess of water demand or by inadequate water resources management, so that they are usually known as socio-economic droughts [73]. These considerations are relevant in river basins located in areas with low availability of water resources and concurrent water uses. They are especially pertinent when considering the likely increment in the frequency of socio-economic droughts in semi-arid areas caused by future population growth and climate change effects on natural water resources [74]. In this regard, the appraisal of the WFIA could be a useful tool in hydrological planning in the medium and long term, since irrigated agriculture will be the water use that suffers most directly the effects of the socio-economic droughts [22].

Finally, to address one of the limitations of this work, an improvement could be achieved by considering in more detail specific aspects of each crop in the irrigated areas, such as different irrigation strategies, the phenological stages of the crops and their yields. For this purpose, the water supplied would be disaggregated for the crop areas, obtaining a homogeneous indicator that would quantify the pressure on water resources exerted by each crop, regarding both quantity and quality. This information would be very useful with regard to knowing how the water availability affects the different crop yields, and could be used to support the design of land use policies.

**Conclusions**

This study evaluated the WF of irrigated agriculture in a geographical area (the Segura River Basin, South-eastern Spain), proposing a new approach for its calculation (the water footprint of irrigated agriculture with exploitation system: WFIA-ES) that addresses key aspects criticised in previous works [17–19]. This approach considers the actual availability of water resources, the exploitation system that distributes them and the legal criteria for water management in irrigation. Moreover, as the amount of water (blue and green) incorporated into the
modelling comes from the same simulation, and coincides with the precipitation that actually falls in the basin, it can be said that the methodology complies with the basic principles of hydrology (satisfying the law of continuity of mass).

The WFIA-ES methodology has been applied here together with the methodology that have been used generally in other works and that supposes a complete supply to match the irrigation demands (the water footprint of irrigated agriculture with full supply: WFIA-FS). Both methodologies used the same series of climatic data, the same land uses and the same soil balance model. So, the series of WF\textsubscript{Green} obtained are the same. The two approaches gave similar average values of WF\textsubscript{Grey}, despite the differences in the calculation processes. In addition, for both methodologies, the values obtained for the grey component are directly proportional to the volume that returns and to the contaminant load, as established in the proposed formulation. The main differences are in the blue component. For the methodology that considers full supply (WFIA-FS) the WF\textsubscript{Blue} value almost doubles that obtained using the methodology that recreates the anthropised water cycle (WFIA-ES). In fact, the value of WF\textsubscript{Blue} for WFIA-FS exceeds the sum of the water resources available in the basin, and it is also decoupled from the hydrology of the basin. Moreover, there is an inversely proportional relationship between resource availability and WF\textsubscript{Blue}, which is difficult to explain in physical terms. So, this value could never be obtained in practice for a semi-arid basin. However, the blue water consumption values for WFIA-ES are in line with the actual availability of water. These results reveal that the values obtained using WFIA-FS are highly over-estimated. This methodology simply provides a maximum reference value that the WFIA could achieve if spatial and temporal availability were not a limiting factor.

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References

1. Seckler D, Barker R, Amarasinghe U. Water scarcity in the twenty-first century. Towards an Agenda for Agricultural Research in Europe. Proceedings of a Conference held in Wageningen, The Netherlands, 2000. pp 147–160.

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

3. Fulton J, Cooley H, Gleick PH. Water Footprint Outcomes and Policy Relevance Change with Scale Considered: Evidence from California. Water Resour Manag. 2014; 28(11): 3637–3649. https://doi.org/10.1007/s11269-014-0692-1

4. Shiklomanov IA. Appraisal and Assessment of World Water Resources. Water Int. 2000; 25: 11–32. https://doi.org/10.1080/02508060008686794

5. Hoekstra AY. Virtual water trade: Proceedings of the International Expert Meeting on Virtual Water Trade, Delft, Holanda, 12–13 December 2002, Value of Water Research Report Series No.12, UNESCO-IHE, Delft, the Netherlands. 2003.

6. Hoekstra AY, Chapagain AK, Mekonnen MM. The Water Footprint Assessment Manual: Setting the Global Standard. Earthscan, London, UK. 2011.

7. Metulini R, Tamea S, Laio F, Riccoboni M. The Water Suitcase of Migrants: Assessing Virtual Water Fluxes Associated to Human Migration. PLoS ONE. 2016; 11(4): e0153982. https://doi.org/10.1371/journal.pone.0153982 PMID: 27124488

8. Liu J, Yang H. Spatially explicit assessment of global consumptive water uses in cropland: Green and blue water. J Hydrol. 2010; 384: 187–197. https://doi.org/10.1016/j.jhydrol.2009.11.024

9. Metulini R, Tamea S, Laio F, Riccoboni M. The Water Suitcase of Migrants: Assessing Virtual Water Fluxes Associated to Human Migration. PLoS ONE. 2016; 11(4): e0153982. https://doi.org/10.1371/journal.pone.0153982 PMID: 27124488

10. Aldaya MM, Martinez-Santos P, Llamas R. Incorporating the Water Footprint and Virtual Water into Policy: Reflections from the Mancha Occidental Region, Spain. Water Resour Manag. 2010; 24(5): 941–958. https://doi.org/10.1007/s11269-009-9480-8

11. Zeng Z, Liu J, Koemanen PH, Zarate E, Hoekstra AY. Assessing water footprint at river basin level: a case study for the Heihe River Basin in northwest China. Hydrol Earth Syst Sci. 2012; 15: 1557–1600. https://doi.org/10.5194/hess-15-1557-2011

12. Franke N, Hoekstra AY, Boyacioglu H. Grey Water Footprint Accounting: Tier 1 Supporting Guidelines. Unesco-IHE Institute for Water Education, Delft, the Netherlands. 2013. Available from: http://waterfootprint.org/en/about-us/news/news/new-guidelines-tier-1-grey-water-footprint-account/

13. Perr C. 2014. Water Footprints: path to enlightenment, or false trail? Agr Water Manage. 2014; 134:119–125. https://doi.org/10.1016/j.agwat.2013.12.004

14. Wichelns D. Virtual water and water footprints do not provide helpful insight regarding international trade or water scarcity. Ecol Indic. 2015; 52: 277–283. https://doi.org/10.1016/j.ecolind.2014.12.013

15. de Miguel A, Kallache M, Garcia-Calvo E. The Water Footprint of Agriculture in Duero River Basin. Sustainability. 2015; 6: 6759–6780. https://doi.org/10.3390/su7066759

16. Tabari MMR, Yazdi A. Conjunctive Use of Surface and Groundwater with Inter-Basin Transfer Approach: Case Study Piranshahr. Water Resour Manag. 2014; 28: 1887–1906. https://doi.org/10.1007/s11269-014-0578-2

17. Perry C. 2014. Water Footprints: path to enlightenment, or false trail? Agr Water Manage. 2014; 134:119–125. https://doi.org/10.1016/j.agwat.2013.12.004
22. Pellicer-Martínez F, Martínez-Paz JM. The Water Footprint as an indicator of environmental sustainability in water use at the river basin level. Sci Total Environ. 2016; 571: 561–574. https://doi.org/10.1016/j.scitotenv.2016.07.022 PMID: 27405519

23. Almansa C, Calatrava J, Martínez-Paz JM. Extending the framework of the economic evaluation of erosion control actions in Mediterranean basins. Land Use Policy. 2012; 29: 294–308. https://doi.org/10.1016/j.landusepol.2011.06.013

24. Gunkel A, Lange J. Water scarcity, data scarcity and the Budyko curve—An application in the Lower Jordan River Basin. Journal of Hydrology: Regional Studies. 2017; 12: 136–149. https://doi.org/10.1016/j.ejrh.2017.04.004

25. Grindlay AL, Zamorano M, Rodríguez MI, Molero E, Urrea MA. Implementation of the European Water Framework Directive: Integration of hydrological and regional planning at the Segura River Basin, southeast Spain. Land Use Policy. 2011; 28: 242–256. https://doi.org/10.1016/j.landusepol.2010.06.005

26. CHS. Memoria Plan Hidrológico de la Demarcación del Segura 2015/21. Confederación Hidrográfica del Segura (CHS). Ministerio de Agricultura, Alimentación y Medio Ambiente. Murcia. 2015; (In Spanish) Available from: https://www.chsegura.es/chs/planificaciony_dma/planificacion15-21/

27. Tian Y, Zheng Y, Wu B, Wu X, Liu J, Zheng C. Modeling surface water-groundwater interaction in arid and semi-arid regions with intensive agriculture. Environ Modell Softw. 2015; 63: 170–184. https://doi.org/10.1016/j.envsoft.2014.10.011

28. Dourte DR, Clyde WF, Uryasev O. WaterFootprint on AgroClimate: A dynamic, web-based tool for comparing agricultural systems. Agr Syst. 2014; 125: 33–41. https://doi.org/10.1016/j.agsy.2013.11.006

29. Miguel-Ayala L, van Eupen M, Guoping Z, Pérez-Soba M, Martorano LG, Lisboa LS, et al. Impact of agricultural expansion on water footprint in the Amazon under climate change scenarios. Sci Total Environ. 2016; 569–570: 1159–1173. https://doi.org/10.1016/j.scitotenv.2016.06.191 PMID: 27443460

30. Salmoral G, Willaarts BA, Garrido A, Gused B. Fostering integrated land and water management approaches: Evaluating the water footprint of a Mediterranean basin under different agricultural land use scenarios. Land Use Policy. 2017; 61: 24–39. https://doi.org/10.1016/j.landusepol.2016.09.027

31. D’Ambrosio E, De Girolamo AM, Rulli MC. Assessing sustainability of agriculture through water footprint analysis and in-stream monitoring activities. J Clean Prod. 2018; 200: 454–470. https://doi.org/10.1016/j.jclepro.2018.07.229

32. Allen RG, Pereira SL, Raes D, Smith M. Crop Evapotranspiration. Guidelines for computing crop water requirement FAO Irrigation and drainage paper 56, Rome. 1998. Available from: https://www.fao.org/docrep/X0490E/X0490E00.htm

33. Steduto P, Hsiao TC, Raes D, Fereres E. AquaCrop—The FAO crop model to simulate yield response to water: I. Concepts and underlying principles. Agron J. 2009; 101(3): 426–437. https://doi.org/10.2134/agronj2008.0139s

34. Stöckle C, Donatelli M, Roger N. CropSyst, a cropping systems simulation model. Eur J Agron. 2003; 18: 289–307. https://doi.org/10.1016/S1161-0301(02)00109-0

35. Wang L, Wu X. Careful considerations when reporting and evaluating the grey water footprint of products. Ecol Indic. 2014; 41: 131–132. https://doi.org/10.1016/j.ecolind.2014.01.039

36. García-Barrón L, Camarillo JM, Morales J, Sousa A. Temporal analysis (1940–2010) of rainfall aggressiveness in the Iberian Peninsula basins. J Hydrol. 2015; 525: 747–759. https://doi.org/10.1016/j.jhydrol.2015.04.036

37. Martínez-Alcalá I, Pellicer-Martínez F, Fernández-López C. Pharmaceutical grey water footprint: Accounting, influence of wastewater treatment plants and implications of the reuse. Water Res. 2018; 135: 278–287. https://doi.org/10.1016/j.watres.2018.02.033 PMID: 29482095

38. Velthof GL, Oudendag DA, Witzke HP, Asman WAH, Klimont Z, Oenema O. Integrated assessment of nitrogen emission losses from agriculture in EU-27 using MITERRA-EUROPE. J Environ Qual. 2009; 38: 1–16. https://doi.org/10.2134/jeq2009.0001e

39. Andrea J, Ferrer-Polo J, Perez MA, Solera A. Decision support system for Drought planning and management in the Jucar River Basin, Spain. 18th World Imacs congress and Modsim09; 2009. pp 3223–3229.

40. Pulido M, Alvarez-Mendiola E, Andreu J. Design of Efficient Water Pricing Policies Integrating Basin-wide Resource Opportunity Costs. J Water Res Plan Man. 2013; 139: 583–592. https://doi.org/10.1061/(ASCE)WR.1943-5452.0000262

41. Pellicer-Martínez F, Martínez-Paz JM. 2014. Assessment of interbasin groundwater flows between catchments using a semi-distributed water balance model. J Hydrol 519, 1848–1858. https://doi.org/10.1016/j.jhydrol.2014.09.067
42. Paredes-Arquiola J, Solera A, Martínez-Capell F, Mombianch A, Andreu J. Integrating water management, habitat modelling and water quality at the basin scale and environmental flow assessment: case study of the Tormes River, Spain. Hydrolog Sci J. 2014; 59: 878–889. https://doi.org/10.1080/02626667.2013.821573

43. Salla MR, Paredes-Arquiola J, Solera A, Andreu J, Pereira CE, Alamy JE et al. Integrated modeling of water quantity and quality in the Araguari River basin, Brazil. Lat Am J Aquat Res. 2014; 42: 224–244. doi:10.3856/vol42-issue1-fulltext-19

44. Pedro-Monzonis M, Jimenez-Fernandez P, Solera A, Jimenez-Gavilan P. The use of AQUATOOL DSS applied to the System of Environmental-Economic Accounting for Water (SEEAW). J Hydrol. 2016; 533: 1–14. https://doi.org/10.1016/j.jhydrol.2015.11.034

45. Hernandez-Bedolla J, Solera A, Paredes-Arquiola J, Pedro-Monzonis M, Andreu J, Sanchez-Quispe ST. The Assessment of Sustainability Indexes and Climate Change Impacts on Integrated Water Resource Management. Water. 2017; 9: 213. https://doi.org/10.3390/w9030213

46. CHS. Esquema provisional de Temas Importantes de la Demarcación Hidrográfica del Segura. Confederación Hidrográfica del Segura (CHS). Ministerio de Medio Ambiente y Medio Rural y Marino. Murcia. 2008; (In Spanish). Available from: https://www.chsegura.es/chs/planificaciony DMA/planificacion/proceso.html#eti

47. CHS. Memoria Plan Hidrológico de la Demarcación del Segura 2009/15. Confederación Hidrográfica del Segura (CHS). Ministerio de Agricultura, Alimentación y Medio Ambiente. Murcia. 2013; (In Spanish) Available from: https://www.chsegura.es/chs/planificacionyDMA/planificacion/

48. BOE. Real Decreto Legislativo 1/2001, de 20 de julio, por el que se aprueba el Texto Refundido de la Ley de Aguas. Ministerio de Medio Ambiente. Boletín Oficial del Estado (BOE). 2001; 176: 26791–26817. (In Spanish). Available from: https://www.boe.es/buscar/act.php?id=BOE-A-2001-14276

49. BOE. Orden APM/2656/2008, de 10 de septiembre, por la que se aprueba la instrucción de planificación hidrológica. Ministerio de Medio Ambiente y Medio Rural y Marino. Boletín Oficial del Estado (BOE). 2008; 71: 12820–12821. (In Spanish). Available from: https://www.boe.es/buscar/doc.php?id=BOE-A-2008-15340

50. EC. Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy, OJ L 327 n 22 December 2000. Available from: http://eur-lex.europa.eu/legal-content/ES/TXT/?uri=CELEX:32000L0060

51. Chavez-Jimenez A, Granados A, Garrote L, Martin-Carrasco F. Adapting Water Allocation to Irrigation Demands to Constraints in Water Availability Imposed by Climate Change. Water Resour Manag. 2015; 29: 1413–1430. https://doi.org/10.1007/s11269-014-0882-x

52. Pellicer-Martinez F, Martinez-Paz JM. Grey water footprint assessment at the river basin level: Accounting method and case study in the Segura River Basin, Spain. Ecol Indic. 2016; 60: 1173–1183. https://doi.org/10.1016/j.ecolind.2015.08.032

53. Martinez-Paz JM, Pemi A, Ruiz-Campuzano P, Pellicer-Martinez F. Valoración económica de los fallos de suministro en los regadíos de la cuenca del Segura. Revista Española de Estudios Agrosociales y Pesqueros. 2016; 244: 35–67. Available from: http://hdl.handle.unirioja.es/servlet/articulo?codigo=5674875

54. Alcon F, Tapsuwan S, Brouwer R, De Miguel MD. Adoption of irrigation water policies to guarantee water supply: A choice experiment. Environ Sci Policy. 2014; 44: 226–236. https://doi.org/10.1016/j.envsci.2014.08.012

55. Garcia-Galiano SG, Olmos-Gimenez P, Giraldo-Osorio JD. Assessing Nonstationary Spatial Patterns of Extreme Droughts from Long-Term High-Resolution Observational Dataset on a Semi-arid Basin (Spain). Water. 2015; 7: 5458–5473. https://doi.org/10.3390/w7105458

56. Kroll S, Ringer N, De las Heras J, Gomez-Alday J, Moratalla A, Briggs R. Analysis of anthropogenic pressures in the Segura Watershed (SE Spain), with a focus on inter-basin transfer. Ecohydrology. 2013; 6: 878–888. https://doi.org/10.1002/eco.1311

57. Lapuente E. Full cost in desalination. A case study of the Segura River Basin. Desalination. 2012; 300: 40–45. https://doi.org/10.1016/j.desal.2012.06.002

58. Alcon F, Martin-Ortega J, Arcas N, Alarcón JJ, de Miguel MD. The non-market value of reclaimed wastewater for use in agriculture: a contingent valuation approach. Span J Agric Res. 2010; 8: 187–196. https://doi.org/10.5424/sjar/201008S2-1361

59. Rodenas MA, Albacete M. The River Segura: reclaimed water, recovered river. J Water Reuse Desal. 2014; 1: 50–57. https://doi.org/10.2166/wrd.2013.044

60. Custodio E, Andreu-Rodes JM, Aragon R, Estrela T, Ferrer J, Garcia-Arostegui JL et al. Groundwater intensive use and mining in south-eastern peninsular Spain: Hydrogeological, economic and social aspects. Sci Total Environ. 2016; 559: 302–316. https://doi.org/10.1016/j.scitotenv.2016.02.107 PMID: 27065448
61. Reif J, Hanzelka J. Grassland winners and arable land losers: The effects of post-totalitarian land use changes on long-term population trends of farmland birds. Agr Ecosyst Environ. 2016; 223: 208–217. https://doi.org/10.1016/j.agee.2016.08.007

62. MAPA. Plan Nacional de Regadíos: Horizonte 2008. Ministerio de Agricultura, Pesca y Alimentación (MAPA). Gobierno de España. Madrid; 2001; (In Spanish). Available from: http://www.mapama.gob.es/es/developments/subjects/sustainable-regadios/plans-national-regadios/text-complete/

63. Perni A, Martínez-Paz JM. Measuring conflicts in the management of anthropized ecosystems: evidence from a choice experiment in a human-created Mediterranean wetland. J Environ Manage. 2017; 203: 40–50. https://doi.org/10.1016/j.jenvman.2017.07.049 PMID: 28778004

64. Perni A, Martinez-Paz J, Martínez-Carrasco F. Social Preferences and Economic Valuation for Water Quality and River Restoration: The Segura River, Spain. Water Environ J. 2012; 26: 274–284. https://doi.org/10.1111/j.1747-6593.2011.00286.x

65. CHS. Plan Hidrológico de la Cuenca del Segura. Confederación Hidrográfica del Segura. Ministerio de Medio Ambiente. Madrid; 1998 (In Spanish). Available from: https://www.chsegura.es/chs/planificacionydma/plandecuenca/documentoscompletos/

66. MARM. 2011. Guía práctica de la fertilización racional de los cultivos en España. Ministerio de Medio Ambiente y Medio Rural y Marino (MARM). Gobierno de España. Madrid; 2011; (In Spanish). Available from: http://www.mapama.gob.es/es/agricultura/publicaciones/Publicaciones-fertilizantes.aspx

67. Dechmi F, Claverís-Laborda I, Balcells-Oliván M, Isidoro-Ramírez D. 2013. La calidad de los retornos de riego en Riegos del Alto Aragón (Huesca, España). XXXI Congreso Nacional de Riegos, Orihuela, Alicante, 18 a 20 de junio de 2013. Available from: http://hdl.handle.net/10532/2311

68. Causape J, Quilez D, Araguees R. Irrigation efficiency and quality of irrigation return flows in the Ebro River Basin: An overview. Environ Monit Assess. 2006; 117: 451–461. https://doi.org/10.1007/s10661-006-0763-8 PMID: 16917723

69. Barros R, Isidoro D, Araguees R. Irrigation management, nitrogen fertilization and nitrogen losses in the return flows of La Violada irrigation district (Spain). Agr Ecosyst Environ. 2012; 155: 161–171. https://doi.org/10.1016/j.agee.2012.04.004

70. Gu B, Ge Y, Chang SX, Luo W, Chang J. Nitrate in groundwater of China: Sources and driving forces. Global Environ Chang. 2013; 23: 1112–1121. https://doi.org/10.1016/j.gloenvcha.2013.05.004

71. CHS. Monitoring and analysis: surface water quality (in Spanish). Murcia; 2017. Available in: http://www.chsegura.es/chs/cuenca/redesdecontrol/calidadenaguassuperficiales/ (accessed on 31 May 2017).

72. Greene WH. Econometric Analysis. Forth ed. Upper Saddle River, NJ: Prentice-Hall; 2000.

73. Mishra AK, Singh VP. A review of drought concepts. J Hydrol, 2010; 391: 202–216. https://doi.org/10.1016/j.jhydrol.2010.07.012

74. Pellicer-Martínez F. and Martínez-Paz J. M.: Climate change effects on the hydrology of the headwaters of the Tagus River: implications for the management of the Tagus-Segura transfer, Hydrol Earth Syst Sci. Discuss., https://doi.org/10.5194/hess-2018-258, in review, 2018.