Ecological Indicators 98 (2019) 19–28

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Original Articles

Sustainability assessment of the agricultural water footprint in the Cachapoal River basin, Chile

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A R T I C L E I N F O

Keywords:
Scarcity
Unsustainability
Water availability
Climate variability
Watershed
Sustainability indicators
Environmental flow

A B S T R A C T

Amid the growing population, climate change and decreasing water availability, water resources planning faces the challenge of achieving efficient and equitable water use, mainly in food production, its greatest consumer. The sustainability of agricultural water consumption in the Cachapoal River basin (34°S 71°O) is assessed using the blue (WFS_{blue}) and gray water footprint sustainability (WFS_{gray}) indicators under conditions of climate variability. The WFS_{blue} indicator of agricultural water consumption showed unsustainability in the upper basin in the dry year (2007), indicating severe water scarcity in autumn and low scarcity in spring, and in the wet year (2005), in which moderate water scarcity was observed only in spring, as a result of the decrease in blue water availability (WA_{blue}) and the increase in the blue water footprint (WF_{blue}), exceeding environmental flow requirements and the environmental carrying capacity. An opposite situation was found in the lower basin, where no water scarcity was observed in the analyzed years. Surface water quality, based on the concentration of fertilizers applied to crops, was observed to be more affected and unsustainable (WFS_{gray}) in the upper basin in the autumn and winter period of the dry year (2007). Currently, the indicators can be applied to provide useful information on the sustainability of water use in basins in order to establish and meet water resources protection objectives.

1. Introduction

Water is the backbone of community development due to the range of ecosystem services that water resources provide. Therefore, it is vital that water management allow efficient, equitable and sustainable allocation of water in order to ensure poverty reduction, economic growth and environmental protection (Zhineng et al., 2016). However, human consumption has altered the global water cycle, modifying speeds and residence times in various storage reservoirs, *i.e.*, oceans, the atmosphere, surface water, groundwater, snow and ice (Keys et al., 2016).

Agriculture is the main consumer of water resources worldwide, accounting for ~70% of total direct water extractions and ~90% of indirect consumption (water evapotranspired and not returned to the system), affecting infiltration rates, soil moisture patterns and runoff generation (Konar et al., 2011; Russo et al., 2014). It is projected that by 2050 agricultural water demand will grow by 55%, with a particular increase in emerging economies and developing countries that currently exhibit low water stress (Gerten et al., 2011; Wang et al., 2016). Agriculture is in turn extremely vulnerable to climate change phenomena (Challinor et al., 2014; Mubako and Lant, 2013). Changes in rainfall patterns increase the probabilities of harvest risks (Hunter et al., 2015), as do increases in temperatures, drought periods and frosts (Zwiers et al., 2011).

The intensification of climate variability is often a determining factor of water scarcity (Pfister et al., 2009; Mubako and Lant, 2013). Irrigation provides a water supply buffer during dry periods (Hunter et al., 2015), in which water use sometimes exceeds available resources (Herath et al., 2013; Chartzoulakisa and Bertaki, 2015). In order to reach sustainable levels of water consumption in agriculture, water use

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https://doi.org/10.1016/j.ecolind.2018.10.048
Received 18 January 2018; Received in revised form 5 October 2018; Accepted 22 October 2018
1470-160X/

must not exceed its natural replenishment rate, making it necessary to consider the water balance and environmental carrying capacity in the area of influence (Chartzoulakisa and Bertaki, 2015; Keys et al., 2016).

Therefore, one of the main challenges worldwide is achieving water sustainability (Zhineng et al., 2016), which is understood as the satisfaction of current water demands for all users without impairing the future supply, while contributing to the objectives of society and maintaining the environment (Russo et al., 2014; Chouchane et al., 2018). There is water sustainability in a basin only if it is possible to ensure the water supply in an equitable and efficient manner in terms of quantity and quality over time (Wang et al., 2016), preserving the hydrological, ecological, biological and chemical functions of ecosystems (Pfister et al., 2009).

The foregoing demonstrates the importance of water governance, the effectiveness, implementation and monitoring of which depend on the use of indicators that establish current and future water security (Cortés et al., 2012). Therefore, a water resources assessment that reflects natural variability and the intake flows and/or demands that compete for their use is necessary (Hejazi et al., 2014); the concept of the water footprint (WF) meets these criteria.

The WF as a sustainability indicator has begun to include greater complexity in its geographic models, developed at different time resolutions and spatial scales, in order to estimate water consumption limits in the environment (Richter et al., 2011; Gerten et al., 2011; Shrestha et al., 2017). This indicator is especially useful in areas that are sensitive to water variability, as is the case in basins with mediterranean climates, where water demand for irrigation increases in periods of lower precipitation, reducing runoff and downstream flow, as described in studies carried out in Spain (Cazcarro et al., 2015), Latin America (Mekonnen et al., 2015), California (Fulton et al., 2014), China (Zeng et al., 2012) and Australia (Stoeglehner et al., 2011).

Water unsustainability in Chile is observed using environmental protection instruments such as declarations of surface water depletion, reserve decrees and restrictions or prohibitions on groundwater extraction in affected areas (DGA, 2016). Nonetheless, these measures have been insufficient, since there were serious water scarcity problems in 2015 and 2016 in the Atacama, Coquimbo, Valparaíso and O'Higgins regions despite the decrees (Sánchez and Carvacho, 2013; Cabrales et al., 2014; DGA, 2016). Thus, conditions have been placed on the water supply for food production in a large portion of the country's farming regions, including the O'Higgins Region (34°S 71°O) (Oyarzun et al., 2008; Divakar et al., 2011; DGA, 2016).

The Cachapoal River basin accounts for 78% of the farmland in the O'Higgins Region (34°S 71°O). There have been reports of water deficits in areas of the interior and Central Valley, leading to water use and availability restrictions (Sánchez and Carvacho, 2013; Cabrales et al., 2014). Further problems are the over-granting of water rights, unsuitable agricultural practices and the severely eroded state of the soil (Sánchez and Carvacho, 2013). In addition, the intensive use of fertilizers, *i.e.*, nitrogen, phosphorus, and sulfur, which are necessary to make crops profitable, promotes the degradation of water (Pizarro et al., 2010; Cortés et al., 2012; Liu et al., 2012).

As the sustainability of agricultural water use in a given area depends not only on runoff and surface water availability, but also on water consumption and quality, with current basin-level water management practices in Chile making food production dependent on the type of irrigation and climate characteristics, the following questions arise: Is the WF_{blue} of the crops sustainable considering the availability of surface water and climatic variability in the Cachapoal River basin? In what periods and sections does the WF_{blue} of agriculture become unsustainable? Does climate variability affect the sustainability of surface water quality? Is the sustainability of the WF_{blue} and WF_{gray} of the crops affected according to the area of influence of the agricultural practices? Therefore, through the determination of the water consumption of the main crops ($WF_{agricultural}$), carried out in the Cachapoal river basin, considering scenarios of climatic variability (dry, wet and normal year). The objective of this work was to evaluate the sustainability of the $WF_{agricultural}$ resulting from water extractions and identify unsustainable periods as shown by the WFS_{blue} and WFS_{gray} indicators.

2. Methods

The WF is defined as a water resources use indicator that allows the volume of water consumed to be characterized (Hoekstra et al., 2016). It accounts for surface or groundwater used for production or irrigation purposes, called the "blue water footprint" (WF_{blue}); use of precipitation water stored as soil moisture, referred to as the "green water footprint" (WF_{green}); and the quantity of water needed to dilute point- or disperse-source pollution to an acceptable concentration, known as the "gray water footprint" (WF_{gray}) (Liu et al., 2012; de Miguel et al., 2015; Mekonnen et al., 2015).

The WF takes into account the impacts that arise from the cumulative effect of all activities, with the understanding that the agricultural water footprint ($WF_{agricultural}$) is the sum of the WF of each crop grown in an area, in order to identify the periods in which extractions are unsustainable (Fulton et al., 2014; Mekonnen et al., 2015; Lovarelli et al., 2016). In the case of blue water footprint sustainability (WFS_{blue}), there is a hotspot when use exceeds surface water availability, surpassing the environmental flow necessary to maintain ecosystems (Zeng et al., 2012; de Miguel et al., 2015).

The sustainability hotspot of the green water footprint (WFS_{green}) occurs when it exceeds the availability of green water, defined as the total evapotranspiration of rainwater from land, minus the evapotranspiration from land reserved for natural vegetation and evapotranspiration from land that cannot be made productive (Hoekstra et al., 2012, 2016). The sustainability hotspot of the gray water footprint (WFS_{gray}) occurs when the waste assimilation capacity is fully consumed (Konar et al., 2011; Liu et al., 2012).

3. Study area

The Cachapoal River basin (34°S 70°W) spans a surface area of 6370 km^2 and has a length of 170 km, covering 18 communes with a total of 584,000 inhabitants, 30% of whom work in agriculture-related activities. It has a temperate mediterranean climate with variations due to the topography and drastic thermal and rainfall modifications generated by the relief. The Cachapoal River rises from the Andes Mountains and is fed by first-order streams such as the Pangal and Coya rivers, and then the Cadena and Claro rivers in the Central Valley. Finally, approaching the Rapel reservoir, it is joined by Zamorano Stream (Fig. 1).

The sustainability analysis of the $WF_{agricultural}$ in the Cachapoal basin was carried out, in accord with Chilean practice, using the administrative division of the basin, *i.e.*, 1st section, upper basin; 2nd section, middle basin; and 3rd section, lower basin (Fig. 1). In addition, agricultural information such as cultivated area, crop type and irrigation techniques in the different basin sections, which was obtained from the agricultural census (INE, 2007), was used (Table 1).

The meteorological input data were obtained from 9 meteorological and rainfall stations maintained by the General Water Directorate (DGA, for its initials in Spanish), with homogeneous data over 34 years. Three conditions were covered: wet year (2005), normal year (2006) and dry year (2007), according to previous estimates in Novoa et al. (2016), in which climatic variability was determined with the Mass Curve method and rain behavior by means of box plots (Guenni et al., 2008). Then the years were classified using the percentile method that defines 5 categories: very dry (0–20th percentile), dry (20–40th percentile), normal (40–60 percentile), humid (60th percentile) -80) and very humid (80–100th) (Valiente, 2001) (Fig. 2).

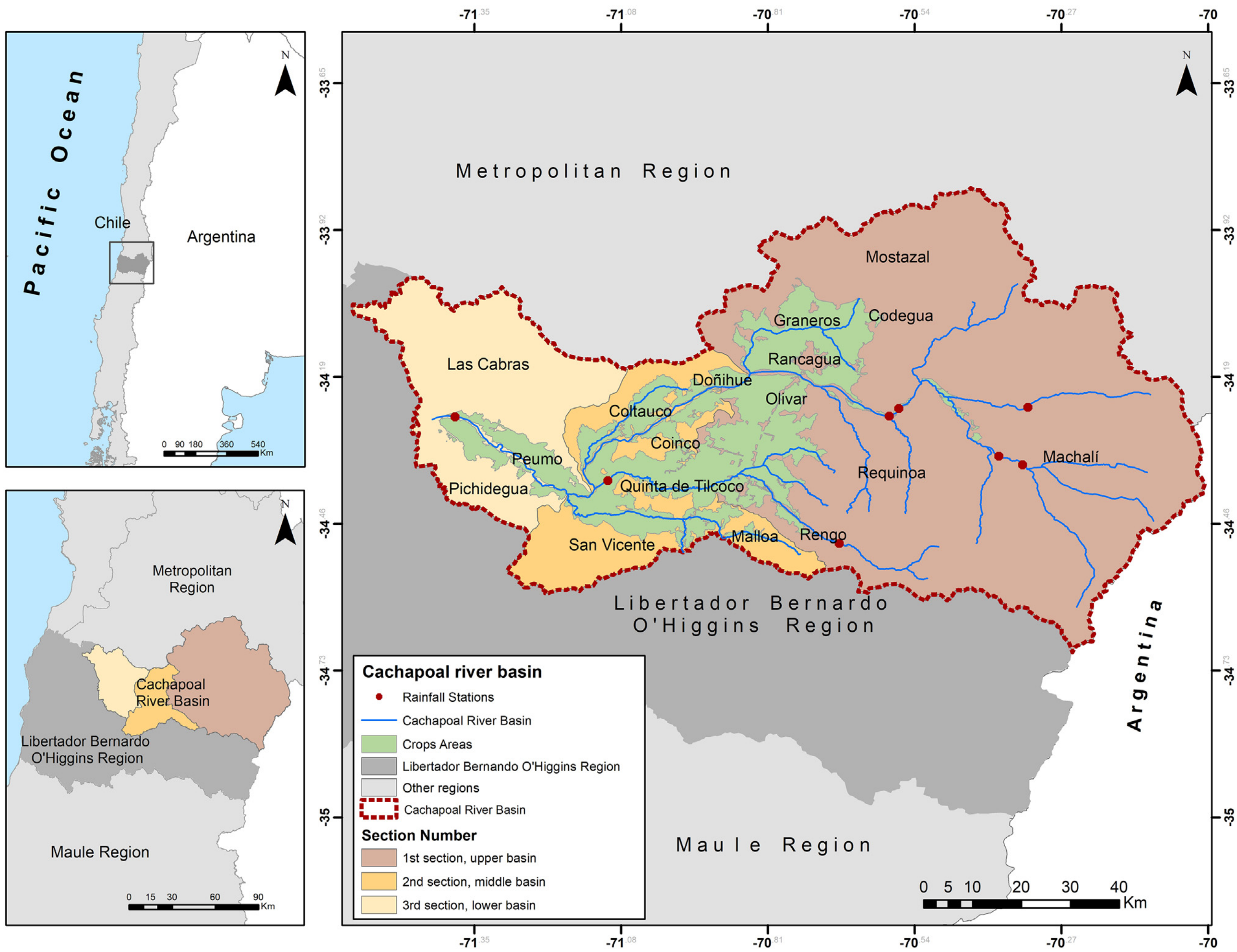

Fig. 1. Cachapoal River basin. The gauging stations used in the environmental flow analysis are indicated.

3.1. Calculation of the agricultural water footprint ($WF_{agricultural}$)

The calculation of the agricultural water footprint included climatic variables with geo-referenced information such as total and effective precipitation, temperature, solar radiation, wind, insolation, reference evapotranspiration and humidity, as well as soil properties, according to the type of soil, such as humidity, infiltration rate, and root depth. In addition, crop data extracted from the Agricultural Studies and Policies Office (ODEPA, 2013a) such as Kc values, stages, and critical depletion fraction by development period, along with estimates of crop evapotranspiration and irrigation requirements, were used.

The methodology proposed by Hoekstra et al. (2011) was employed,

Table 1
Summary of agricultural activity information: crop types, cultivated area and irrigation techniques in the Cachapoal River basin. Source: INE, 2007.

		Family	Species	Upper basin 34°45′31 S 70°45′24O	Middle basin 34°17′30 S 71°04′52O	Lower basin 34°17′29 S 71°24′24O
Cultivated area (ha)	Vegetables	Solanaceae	Industrial tomato	934.2	428.2	271.8
			Tomato	145.2	590.8	218.7
		Cucurbitaceae	Melon	18.8	847.6	467.1
		Amarilidacea	Onion	29.7	947.6	88.4
	Total vegetable crops			1128	2814	1046
	Fruit trees	Rutaceae	Citrus	194.1	1023.6	2441.3
		Rosaceae	Apple	4758.4	1639.5	39.9
		Rosaceae	Peach	7068.4	2730.7	914.6
		Lauraceae	Avocado	62.9	873.4	1724.1
		Vitaceae	Grape	5204.1	1434.5	4178.5
		Oleaceae	Olive	222.1	207	16.1
	Total fruit tree crops			17,510	7909	9315
	Cereals	Poaceae	Corn	7473	12,264	9975
	Total area evaluated			26,111	22,986	20,335
Irrigated surface (%)	Gravity-fed irrigation			75.7	82.6	62.8
	Mechanical irrigation			0.4	0.8	1.2
	Micro-irrigation			24	16.7	36.4

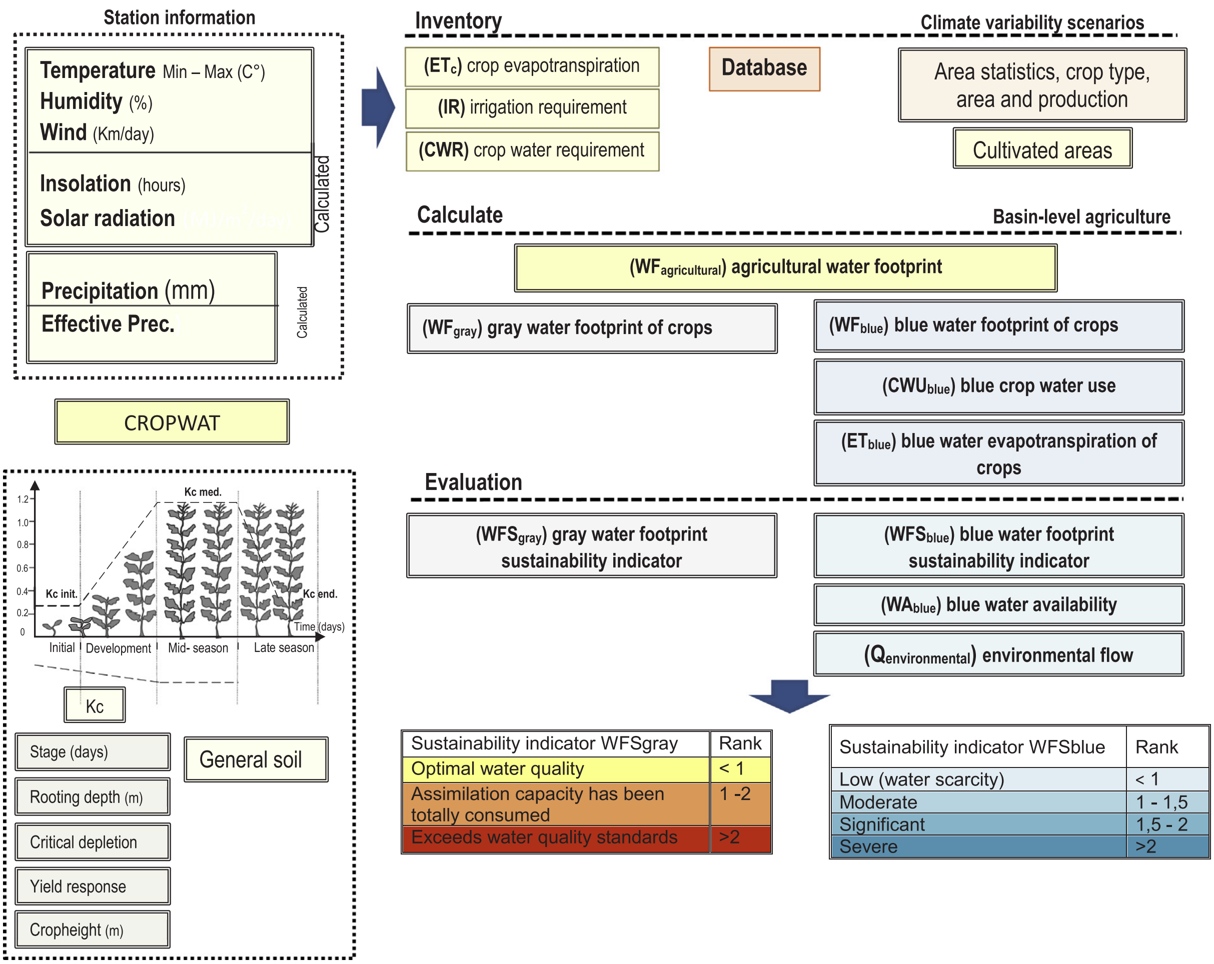

Fig. 2. Methodology summary diagram.

and the blue and gray components of the analyzed crops considered.

3.2. Calculation of the blue water footprint of crops (WF_{blue}) (m^3 $month^{-1}$)

The component of the blue water footprint (WF_{blue}) of the crops within a geographical area [volume/time] was determined using the following equations:

$$WF_{blue} = CWU_{blue} * Y [m^3 month^{-1}]$$

where: Y (ha^{-1}) = area of each total crop in each section of the basin (Table 1), and CWU_{blue} (m^3 ha^{-1}) = crop blue water use.

3.3. Calculation of crop blue water use (CWU_{blue}) (m^3 ha^{-1})

The following relationships were used:

$$CWU_{blue} = 10 \times \sum_{d=1}^{lgp} ET_{blue} [volume \ \ area^{-1}]$$

where: Σ: crop growth cycle, that is, from sowing (day 1) to harvest, Igp: length, days in each stage of the cycle, and ET_{blue}: blue water evapotranspiration (mm $month^{-1}$). The factor 10 is meant to convert water depths in millimeters into water volumes per land surface area in m^3/ha.

3.4. Estimation of blue water evapotranspiration of crops (ET_{blue}) (mm/period)

The water demand of the crops was determined from the crop water requirement (CWR) using CROPWAT 8.0 software. Under ideal growing conditions, it is assumed that crop water requirements are fully met, such that actual crop evapotranspiration (ETc) will be equal to the crop water requirement: ETc = CWR. The ETc calculation was performed according to the irrigation requirement (IR). This methodology assumes that losses due to irrigation remain and return to the basin. ET_{azul} was estimated from:

$$ET_{blue} = IR [mm \ \ month^{-1}]$$

$$IR = ET_C - Peff [mm \ \ month^{-1}]$$

Where Effective Precipitation (Peff) was calculated by the program according to the USDA S.C method; when the effective rainfall is greater than the total crop evapotranspiration, ET_{blue} is equal to zero. ETc was estimated with a ten-day time step and over the total growing season, using the following equation:

$$ET_c = Kc * ET_o [mm \ \ month^{-1}]$$

where Kc = crop coefficient, which incorporates crop characteristics and averaged effects of evaporation from the soil, ETo = reference evapotranspiration (mm $month^{-1}$), which was calculated with the CROPWAT 8.0 program using the Penman-Monteith method, and climate data based on the latitude and analyzed period obtained from the DGA (2016).

3.5. Calculation of the gray water footprint of crops (WF_{gray}) (m^3 $month^{-1}$)

The gray water footprint is defined as the volume of freshwater that is required to assimilate the pollutant load based on natural background concentrations and existing environmental water quality standards (Hoekstra et al., 2011), expressed in a geographical area:

$$\mathrm{Wfgray} = \mathrm{WFgray} * \mathrm{Y}[\mathrm{m^3 month^{-1}}]$$

where: Y (ha) = total area of each crop in each section of the watershed (Table 1), and WF_{gray} (m^3 ha^{-1}) = gray water footprint, estimated with:

$$\mathrm{WFgray} = \frac{\frac{(\alpha * AR)}{(Cmax - Cnat)}}{\mathrm{Y}} [\text{volume} \quad \text{area}^{-1}]$$

The gray component of the water footprint (WF_{gray}) of crops was calculated as the fertilizer application rate in the field per hectare (AR, kg/ha) times the leaching-runoff fraction (α), expressed as a percentage, divided by the maximum acceptable concentration (cmax, kg m^{-3}), defined by quality standards, minus the natural concentration of the pollutant considered (cnat, kg m^{-3}) and then divided by crop yield, formulated as fertilization time per hectare of crops (Y, t).

The agent used to measure WF_{gray} was nitrogen, with (AR) being the reference dose of nitrogen for each agricultural crop in the region, taken from ODEPA (2013b). The leaching fraction for this agent was 10% and the C_{max} was 0.015 kg m^{-3}, based on DS. 90 of the Chilean Issuance Standard. The C_{nat} of the Cachapoal River basin was established at 0.00001 kg m^{-3} by evaluating the best scenario according to the methodology of Hoekstra et al. (2011).

3.6. Calculation of the agricultural blue water footprint sustainability indicator (WFS_{blue})

The estimate was established as the relationship between the ΣWF_{blue} of all the analyzed crops in the basin and blue water availability (WA_{blue}) (Hoekstra et al., 2012), according to the following equation:

$$\mathrm{WFS_{blue}}[\mathrm{x,\ t}] = \frac{\Sigma \mathrm{WF_{blue}}[\mathrm{x,\ t}]}{\mathrm{WA_{blue}}[\mathrm{x,\ t}]}[-]$$

ΣWF_{blue} (m^3 $month^{-1}$) was calculated as the sum of the monthly component WF_{blue} volumes of the main agricultural crops analyzed in the Cachapoal River basin (x) in a certain period (t). The methodology proposed by Hoekstra et al. (2011) was used.

WA_{blue} (blue water availability) was estimated using the following expression:

$$\mathrm{WA_{blue}}[\mathrm{x,\ t}] = \mathrm{QMM}[\mathrm{x,\ t}] - \mathrm{Q_{environmental}}[\mathrm{x,\ t}][\text{volume/time}]$$

where QMM (m^3/month) corresponds to the runoff or natural flow of the basin (x) and $Q_{environmental}$ to the requisite environmental flow in a time (t) (m^3/month).

The QMM daily and monthly flow input data were collected from the Cachapoal in Puente Termas de Cauquenes station, at an elevation of 700 m (34°16′S, 70°34′ W), representative of the upper basin, and the Cachapoal in Puente Arqueado station, at an elevation of 115 m (34° 16′S, 70° 22′ W), representative of the lower basin, both of which are managed by the DGA.

3.7. Calculation of environmental flow ($Q_{environmental}$)

$Q_{environmental}$, or environmental flow, is used in Chile as an instrument of environmental management, preservation and protection. Its regulation is described in the Water Code, the rules for the determination of the minimum ecological flow and the Environmental Impact Assessment System (SEIA, for its initials in Spanish) (General Environmental Framework Law No. 19.300).

It was developed in order to establish a minimum amount of water that must flow in a certain surface water course (Water Code article 149 N° 7) to avoid deterioration that may occur due to water extraction, implying a restriction on the exercise of water use rights.

The environmental flow was estimated using as a formula 50% of the 95% exceedance probability flow for each month, with the following restrictions: a) when the monthly calculation is < 10% of the annual average flow, the minimum environmental flow is 10% of the annual average flow, b) when the calculation is > 10% of the average annual flow and < 20% of the average annual flow, the minimum environmental flow is 50% of the 95% exceedance probability flow, c) when the calculation is > 20% of the average annual flow, the minimum environmental flow is 20% of the annual average flow (Decree 71 Art. 1 A, Regulation for the determination of the minimum environmental flow).

A 25-year span of daily and monthly flow data was used, according to the methodology established by the DGA (Decree 71, Art. A). The missing values from the upper and lower zones were estimated via basin transposition.

The WFS_{blue} indicator was classified according to four levels: < 1 low; between 1 and 1.5 moderate; between 1.5 and 2 significant and > 2 severe, indicating water scarcity. In addition, the number of months in which the WF_{blue} exceeded water availability was estimated (Hoekstra et al., 2012).

3.8. Calculation of the agricultural gray water footprint sustainability indicator (WFS_{gray})

The monthly water contamination indicator (WFS_{gray}), established as the consumed waste assimilation capacity, was calculated according to the following equation:

$$\mathrm{WFS_{gray}}[\mathrm{x,\ t}] = \frac{\Sigma \mathrm{WF_{gray}}[\mathrm{x,\ t}]}{\mathrm{QMM}[\mathrm{x,\ t}]}[-]$$

where (ΣWF_{gray}) (m^3/month) corresponds to the sum of the WF_{gray} of the crops grown in the basin (x) assessed according to nitrogen input, QMM corresponds to the runoff or natural flow of the basin in a time (t) (m^3/month) (Hoekstra et al., 2012).

The WFS_{gray} indicator was classified according to two levels: between 1 and 2 when its assimilation capacity has been completely consumed and > 2 when limits set by environmental water quality standards are exceeded. Both indicators were calculated for the upper and lower sections of the basin using data from stations representative of each. There are no gauging stations in the middle basin; therefore, the indicators could not be estimated there (Hoekstra et al., 2012).

4. Statistical analyses

Correlation analyses were performed for the hydrological variables according to the Pearson coefficient using the STATISTICA 8.0 program. To establish the WF_{blue} and WA_{blue} trend over the years a decomposition analysis was carried out, with time series decomposed into a seasonal component (S_t) and a trend component (T_t).

The trend was calculated with the deterministic decomposition method, which consists of fitting a regression function to the series that explains the trend, where $yt = \mu t + at$, $t = 1, \ldots, n$, with μt being the deterministic level of the series at time t and at the random error at time t. It also uses a model with additive components and $t = \mu t + st + at$, $t = 1, \ldots, n$, where μt is the deterministic trend and st is the deterministic seasonal component, which has a periodic function, where the period is ℓ, $st = st - \ell$, for monthly series, with annual seasonality, $\ell = 12$.

5. Results

5.1. Agricultural blue water footprint sustainability (WFS_{blue})

The water sustainability indicator (WFS_{blue}) for the wet (2005),

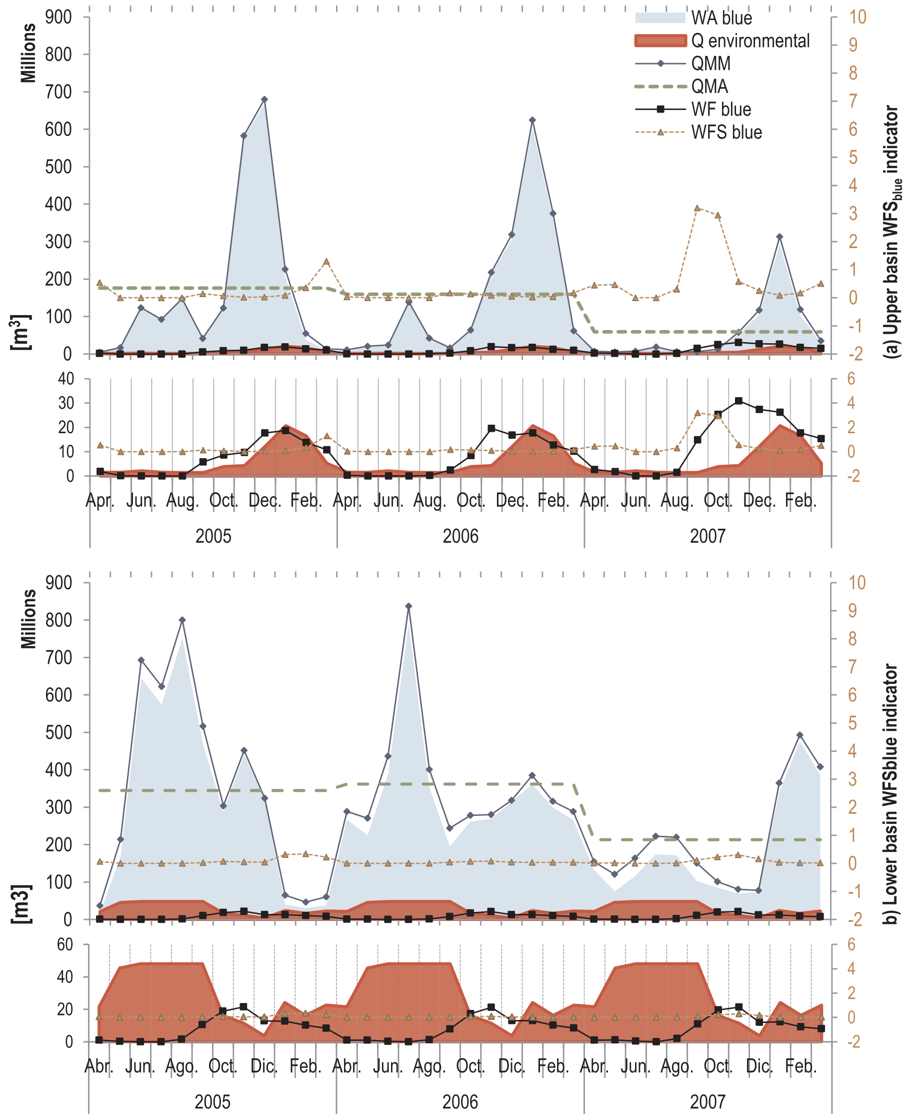

Fig. 3. Blue water footprint sustainability indicator (WFSblue) of the Cachapoal River basin: a) upper basin, b) lower basin.

normal (2006) and dry (2007) scenarios is shown in Fig. 3. It is observed that the mean monthly flows (QMM) and mean annual flows (QMA) of the Cachapoal River were greater in the lower basin than in the upper basin. In the lower basin the QMA was greatest in 2006, at 361.9 MMm^3/year, and in the upper basin it was greatest in 2005, at 175.7 MMm^3/year. In addition, in the upper basin the QMM exhibited the greatest water volume in the months of November and December, while in the lower basin the greatest QMM was observed in the month of July. As a result, the environmental flow ($Q_{environmental}$) proved to be greatest in the summer months in the upper basin and in the winter months in the lower basin, characteristic of the nivo-pluvial influence (Fig. 3).

The calculated water availability (WA_{blue}) was 70% lower in the upper basin and 38% lower in the lower basin in the dry year (2007) than in the wet year (2005) (Fig. 4).

Fig. 4 shows that independent of the scenario, *i.e.*, dry, normal or wet year, or the analyzed basin section, *i.e.*, upper or lower, a minimum in WA_{blue} (available water) was observed in the months of April and May in the Cachapoal River basin.

Regarding the deseasonalized component (St) of the WA_{blue} in the Cachapoal River basin, the trend series projected a linear component (Tt), obtained from a linear regression fit, in which the trend was decreasing WA_{blue} over time in both sections (upper basin, $R^2 = 0.20$;

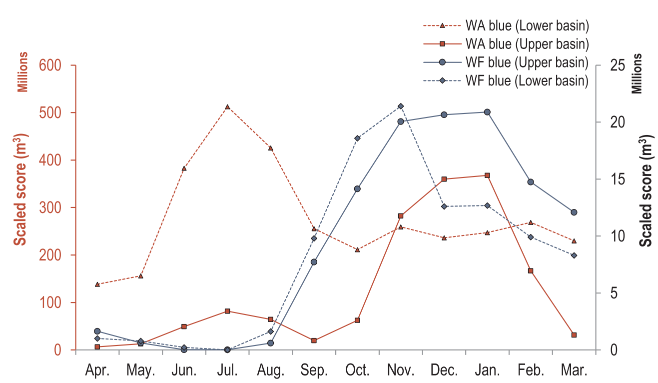

Fig. 4. Seasonal component of WAblue and WFblue (St) in the Cachapoal River basin. Estimated independently of the nature of the year (wet, normal or dry).

slope = −45,97,043; P = 0.0055) and (lower basin, $R^2 = 0.14$; slope = −58,00,324; P = 0.0237) (Fig. 5).

The agricultural WF_{blue} in the Cachapoal River was greatest in the upper basin in the dry year (2007), reaching 163.6 MMm^3, an amount twice that of the other years analyzed, while in the lower basin the greatest WF_{blue} occurred in the dry (2007) and wet (2005) years, with

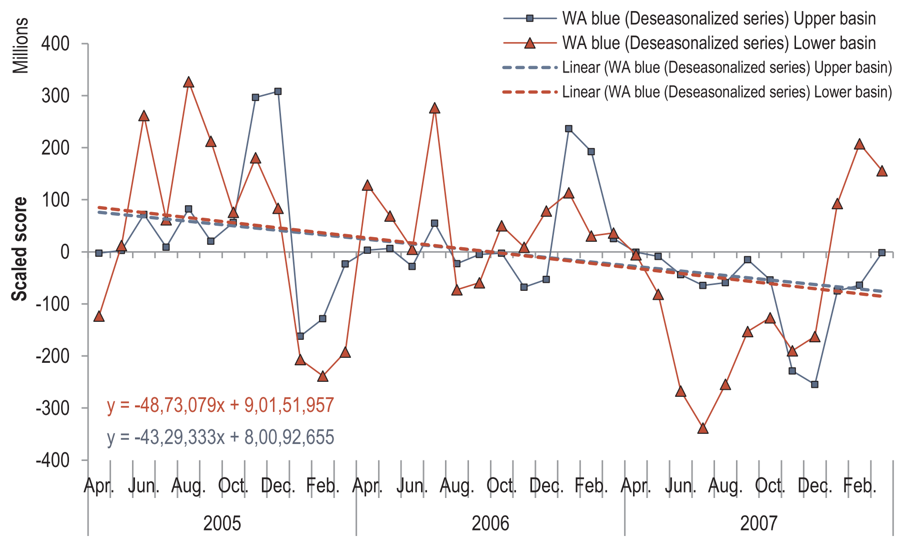

Fig. 5. Trend component (Tt) and deseasonalized series (St) of WAblue consumption in the Cachapoal River basin: a) upper basin, b) lower basin.

volumes of 98.1 MMm^3 and 97.9 MMm^3, respectively (Fig. 3). The seasonal component of the WF_{blue} shows the monthly behavior of this variable (St), in which the WF_{blue} of the crops is highest in the spring and summer months (Fig. 4).

A linear component was identified in the deseasonalized series, according to the linear regression fit. The trend was increasing WF_{blue} over time in the upper basin ($R^2 = 0.34$; slope = 235950.3; $P = 0.0002$) and remaining constant in the lower basin ($R^2 = 0.0002$; slope = −781.6; $P = 0.9322$) (Fig. 6).

Thus, the WFS_{blue} indicator classified the upper basin as unsustainable in 2007 (dry), indicating a severe shortage in the months of September and October and low scarcity in the months of November and March. In 2005, scarcity was classified as moderate in March and low in April. Meanwhile, in the lower basin no water unsustainability was identified in the years studied (Fig. 3).

5.2. Agricultural gray water footprint sustainability (WFS_{gray})

The greatest WF_{gray} was observed in 2007, with a water volume of 85.01 MMm^3/year in the upper basin and 68.2 MMm^3/year in the lower basin (Fig. 7).

The WFS_{gray} sustainability indicator assessed for nitrogen input to cropland classified the upper basin as unsustainable in the month of September 2006, with a consumed assimilation capacity. Similarly, the months of May, June and August of 2007 were classified as having a consumed assimilation capacity, with the limits set by environmental quality standards exceeded in the month of September. Meanwhile, in the lower basin there was no water unsustainability (Fig. 7).

6. Discussion

In this study of the water footprint in the Cachapoal River basin, spatial and seasonal variations in the basin were incorporated as input data as proposed by Hoekstra et al. (2016). The analyses of scenarios with wet, normal and dry years, selected from a period of 34 years of observations (DGA, 2016), were useful for visualizing uncertainties associated with the climate and land-use conditions that effect the efficiency of water resources (Dong et al., 2013; Donoso et al., 2016). Approaches based on these climate scenarios have been applied in order to explore management strategies and support water managers and decision makers, as identifying the critical periods of water consumption in agriculture makes it possible to propose solutions, as reported de

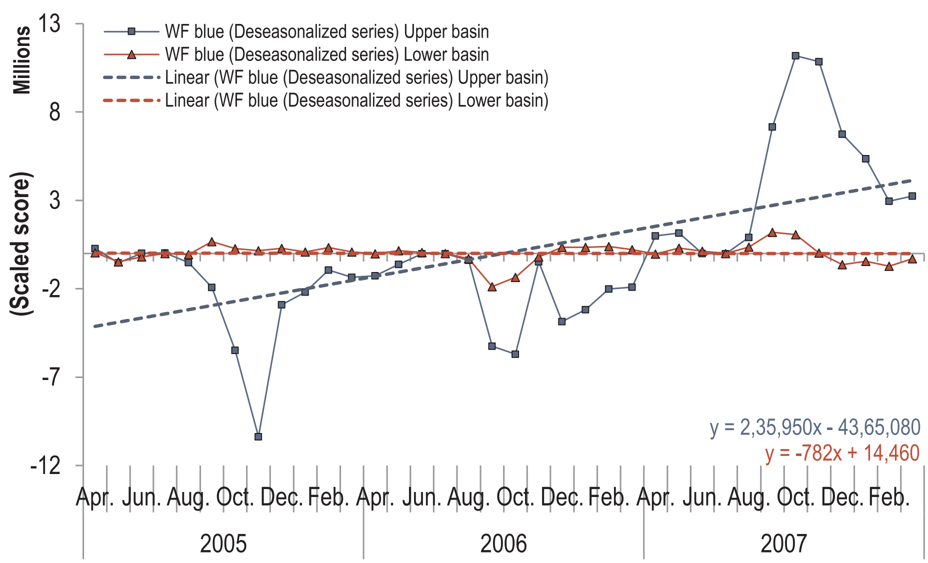

Fig. 6. Trend component (Tt) and deseasonalized series of the WFblue of the Cachapoal River basin: a) upper basin, b) lower basin.

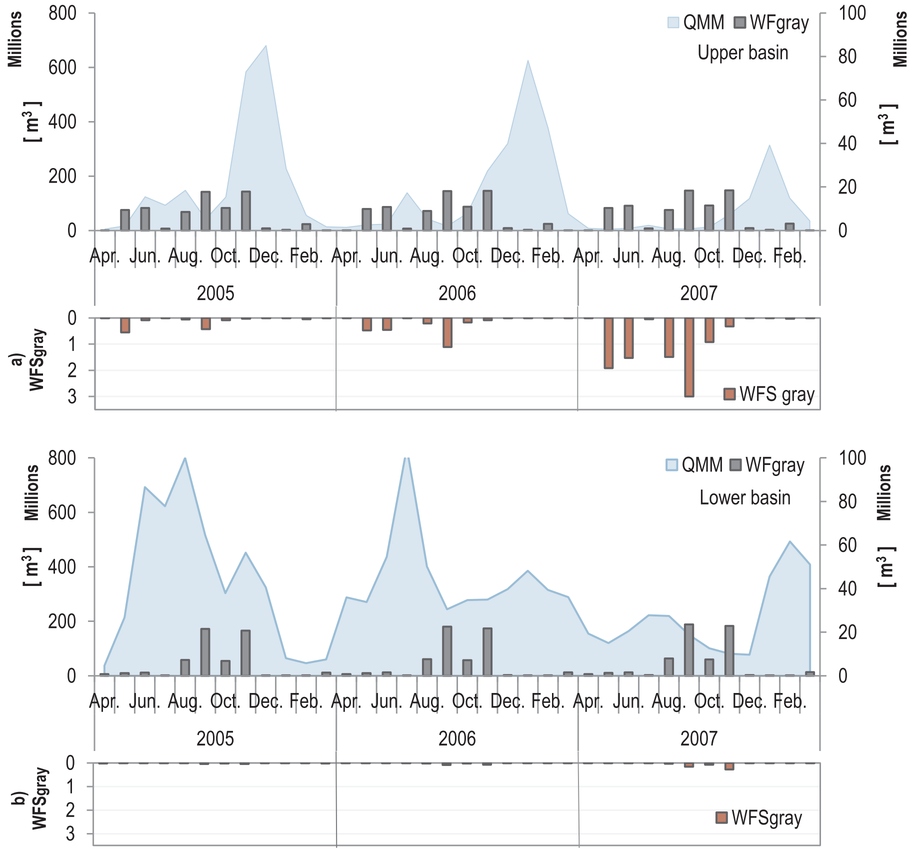

Fig. 7. Gray water footprint sustainability (WFSgray) of the Cachapoal River basin: a) upper basin, b) lower basin.

Miguel et al. (2015) and Fulton et al. (2014) in basins in Spain and California, respectively.

The sustainability of water use in agriculture is an important line of research worldwide, with most research conducted in countries such as China, the United States, Australia, India and Germany. However, there have been insufficient studies in Latin American countries (Velasco-Muñoz et al., 2018). The contribution of this work is an analysis of water consumption in a region located in the Southern Hemisphere, Chile, where, in spite of its geographical peculiarities (Aguilera et al., 2019), high vulnerability to climate change (Sánchez and Carvacho, 2013; Challinor et al., 2014; Henríquez et al., 2016; Yevenes et al., 2016) and potential for high-value, high-quality agricultural exports (Fernández et al., 2018), water use has not yet been adequately planned, even though agricultural exports in 2017 were estimated at US $ 5183 million, a decrease of 7.1% from the previous year reported by the National Customs Service.

The calculation of environmental flows has also recently taken on importance when assessing the water sustainability of a basin (Richter et al., 2011). Chilean regulations do not distinguish between natural and anthropogenic fluctuations in watercourse volumes; however, it is advisable in the Cachapoal River basin that minimum flows, with which the long-term environmental sustainability of the basin is guaranteed, be established, as proposed by Jaramillo and Destouni (2015), and that an ongoing preventive watch program be implemented, since according to observations of the blue water footprint sustainability indicator (WFS_{blue}) the volume of water allocated for maintaining the environment in critical periods is minimal (Fig. 3) (Hejazi et al., 2014).

The environmental damage in the Cachapoal River basin resulting from unsuitable agricultural practices is directly related to surface water quality. In this basin, primarily in the upper section, WFS_{gray} presented the most months with unsustainability in the dry period (Fig. 3). Nitrogen is a fertilizer that is used intensely in agriculture, since it acts as the limiting nutrient of crop growth and ensures production (Arumí et al., 2005; Pizarro et al., 2010). The spatial variation of nitrogen application in a basin is directly related to land use, precipitation and topography (*i.e.*, moisture, hydrologic group and erosion level) (Liu et al., 2012). Studies carried out by Yevenes et al. (2016) in basins in the VII and VIII regions of Chile reveal that farmland accounts for 60% of the variation in river water quality.

The WFS_{blue} indicated unsustainability mainly in the upper Cachapoal River basin. In the wet year this situation was identified only in spring, while in the dry year it was observed in autumn and spring (Fig. 3), which coincided with decreased water input in natural conditions (surface water availability) (Figs. 5 and 6), resulting in the failure to meet environmental flow requirements. In these months water demand is met thanks to groundwater extraction (DGA, 2016). In addition, the calculated values confirm that the climate variability in the area can result in a failure to meet environmental flow requirements due to the effect of the basin type, *i.e.*, nivo-pluvial regime (Wang et al., 2016).

In the upper basin greater water availability (WA_{blue}) was observed in the summer months, contrary to that which occurred in the lower basin, where water availability was greater in the winter months, which is typical of a mediterranean climate (Novoa et al., 2016). Therefore, the increase in temperatures, climate variability and the decrease in flows in the summer period could alter the water supply and regional demand and thus exacerbate the conditions of water scarcity in the basin (Wang et al., 2016; D'Ambrosio et al., 2018).

As a preventive measure the weighting of these indicators could be more restrictive. The study of de Miguel et al. (2015) indicates that levels over 0.5 are unsustainable (those of this study were > 1), changing the environmental carrying capacity of the Cachapoal River basin.

The effect of the increase in the irrigable area in the basin due to greater food demand was reflected in the upper basin in a higher WF_{blue}

over time. However, it is important to stress that in the lower basin, the trend is a stable WF_{blue}, a determining condition for maintaining the sustainability of this area; therefore, in environmental terms, it should not be significantly affected (Fig. 6). For efficient allocation of water, facilitating a balance between supply and demand, the WFS_{blue} indicator must be continuously monitored, since it allows the pressure that human activities exert on local water resources as a function of their seasonal availability to be visualized (Divakar et al., 2011; Gerten et al., 2011; Donoso et al., 2016).

A promising future depends on agriculture establishing sustainable water management and consumption (Mubako and Lant, 2013; Dong et al., 2013; Iglesias and Garrote, 2015). These measures require information and education programs in order to promote water and land conservation (Russo et al., 2014; Fulton et al., 2014; Chartzoulakisa and Bertaki, 2015). The water footprint indicator also contributes to water supply planning from an environmental perspective, since it reveals water availability and/or scarcity and the carrying capacity in the studied area (Cabrales et al., 2014; Jaramillo and Destouni, 2015; Xinchun et al., 2017). In general, areas with scarcity and pollution in their watercourses tend to incur an increase in water supply and treatment costs, which sometimes outweigh the benefits, with problems resolved from the supply side (Wang et al., 2016). $WF_{agricultural}$ reduction strategies and measures are in most cases oriented toward the application of technology to irrigation and intensive farming that consumes less water (Challinor et al., 2014, Cazcarro et al., 2015, Le Roux et al., 2016). In the Cachapoal basin, it is suggested that planting calendars be revised, taking into account the months with the greatest water unsustainability (Fig. 3) and the comparative advantages of growing in the lower basin, where the water supply is greater (Fig. 4).

7. Conclusions

1. Analysis of the water footprint in a basin with a mediterranean climate such as that of the Cachapoal River reveals that the seasonal factor exerts significant pressure on water consumption, with a severe scarcity level exacerbated by periods of drought observed. Nonetheless, maintaining efficient use of water resources is a determining factor in sustaining the carrying capacity. In addition, the vulnerability of water quality is manifested in land uses, given that agricultural activity intensifies environmental degradation, just as the decrease in water availability is directly related to the unsustainability of the water footprint.
2. As Chile has experienced a prolonged drought since 2007, a water sustainability indicator is a substantial contribution to improving water management and monitoring, determining extraction limits and establishing geographically appropriate agricultural land-use planning foundations, as in the 69,433 ha in which the primary crops of the Cachapoal River basin are grown.
3. The multifactor analysis showed that climate variations determine the crop water requirements, relating natural flow to the various forms of consumption. The greatest WF_{blue} was estimated in drought conditions in both sections of the basin, as well as in the wet year in the lower basin, where the importance of both precipitation amounts and the period in which it occurs is demonstrated.
4. The climate variability of a mediterranean area, added to the effect of supply type (snow-rain), influences estimated blue water availability (WA_{blue}), which decreases in both sections over time, and is greater in the wet year and lower in the dry year, with the situation becoming critical in the months of April and March.
5. The WFS_{blue} indicator identified unsustainability in the upper basin in the dry and wet years, with severe and moderate water scarcity resulting from the decrease in WA_{blue} (with greater snow influence) along with the increase in WF_{blue} consumption. This was contrary to the situation in the lower basin, where no water scarcity was observed due to a greater QMM volume, rainfall input of WA_{blue} and a constant WF_{blue}.
6. In the Cachapoal River basin, an agricultural area, there is significant human influence due to nitrogen input via fertilization to ensure crop yields. Surface water contamination (WF_{gray}) was seen mainly in the upper section in the dry year, with a consumed assimilation capacity and exceedance of the limits set by environmental quality standards, where the analyzed crop area was larger, *i.e.*, 26,111 ha.
7. The foregoing suggests the need to amend and design public policies that ensure efficiency and equity in water resources. In Chile, the way in which water is managed is defined in the Water Code (DFL 1.122). Essentially, it is a distribution system in which freely transferable use rights are conceded. This allocation reflects a unitary legal system, independent of the local ecosystem and current abundance of natural resources. This has led to over-granting of water rights in a large portion of the agricultural regions of the central macro-zone (30°–35°S), where quantitative indicators of resource unsustainability, which are not currently taken into account, are needed for planning and management.

Acknowledgments

The authors express their gratitude to the Conicyt National Commission for Scientific and Technological Research for the funding provided through the CRHIAM Conicyt/Fondap/15130015 project, the Conicyt Advanced Human Capital Program of the Government of Chile, the National Doctorate Grant-2011, the 11150424 and 1150459 Project FONDECYT. Dr. Ahumada-Rudolph express their gratitude to the Post-Doctorate program of the Direction of investigation, University of Bío-Bío.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2018.10.048.

References

Aguilera, M.A., Aburto, J.A., Bravo, L., Broitman, B.R., García, R.A., Gaymer, C.F., Gelcich, S., López, B.A., Montecino, V., Pauchard, A., Ramos, M., Rutllant, J.A., Sáez, C.A., Valdivia, N., Thiel, M., 2019. Chapter 29 – Chile: Environmental Status and Future Perspectives. In: Sheppard, C. (Ed.), World Seas: an Environmental Evaluation, second ed. Academic Press, Europe, the Americas and West Africa, pp. 673–702 vol. I.
Arumí, J.L., Oyarzún, R.A., Sandoval, M., 2005. Natural protection against groundwater pollution by nitrates in the Central Valley of Chile. Hydrol. Sci. J. 50 (2), 331–340.
Cabrales, G., Fernando, L., Nespolo, C.M., 2014. Racionamiento del agua ante fluctuaciones de disponibilidad: Una discusión teórica para el caso de Chile. Idesia 32 (1), 129–137.
Cazcarro, I., Duarte, R., Martín-Retortillo, M., Pinilla, V., Serrano, A., 2015. How sustainable is the increase in the water footprint of the Spanish agricultural sector? a provincial analysis between 1955 and 2005–2010. Sustainability 7 (3), 5094–5119.
Challinor, A.J., Watson, J., Lobell, D.B., Howden, S.M., Smith, D.R., Chhetri, N., 2014. A meta-analysis of crop yield under climate change and adaptation. Nat. Clim. Change 4 (2), 287–291.
Chartzoulakisa, K., Bertaki, M., 2015. Sustainable water management in agriculture under climate change. Agric. Agric. Sci. Procedia 4 (2), 88–98.
Cortés, A.E., Oyarzún, R., Kretschmer, N., Chaves, H., Soto, G., Soto, M., Amézaga, J., Oyarzún, J., Rötting, T., Señoret, M., Maturana, H., 2012. Application of the watershed sustainability index to the Elqui river basin, North-Central Chile. Obras y Proyectos 12 (2), 57–69.
Chouchane, H., Krol, M.S., Hoekstra, A.Y., 2018. Virtual water trade patterns in relation to environmental and socioeconomic factors: a case study for Tunisia. Sci. Total Environ. 613, 287–297.
D'Ambrosio, E., De Girolamo, A.M., Rulli, M.C., 2018. Assessing sustainability of agriculture through water footprint analysis and in-stream monitoring activities. J. Cleaner Prod. 200, 454–470.
De Miguel, A., Hoekstra, A.Y., García-Calvo, M., 2015. Sustainability of the water footprint of the Spanish pork industry. Ecol. Ind. 57, 465–474.
DGA, Dirección general de agua. (2016). Atlas del agua, Chile: DGA. Serie de Estudios Básicos DGA, S.E.B. N° 6. Santiago, Chile. http://www.dga.cl/DGADocumentos/Atlas2016parte1-17marzo2016b.pdf (Accessed 21 December 2017).
Divakar, L., Babel, M.S., Perret, S.R., Das Gupta, A., 2011. Optimal allocation of bulk water supplies to competing use sectors based on economic criterion – an application to the Chao Phraya River Basin, Thailand. J. Hydrol. 401 (2), 22–35.

Dong, C., Gerrit, S., Van de Giesen, N., 2013. Scenario development for water resource planning and management: a review. Technol. Forecast. Soc. Chang. 80 (4), 749–761.
Donoso, G., Franco, G., Blanco, E., Lira, J., 2016. Water footprints and irrigated agricultural sustainability: the case of Chile. Int. J. Water Resour. Dev. 32 (5), 738–748.
Fernández, F.J., Blanco, M., Ponce, R.D., Vásquez-Lavín, F., Roco, L., 2018. Implications of climate change for semi-arid dualistic agriculture: a case study in Central Chile. Reg. Environ. Chang. https://doi.org/10.1007/s10113-018-1380-0.
Fulton, J., Cooley, H., Gleick, P., 2014. Water footprint outcomes and policy relevance change with scale considered: evidence from California. Water Resour. Manage. 28 (11), 3637–3649.
Gerten, D., Heinke, J., Hoff, H., Biemans, H., Fader, M., Waha, K., 2011. Global water availability and requirements for future food production. J. Hydrometeorol. 12 (1), 887–898.
Guenni, L., Degryze, E., Alvarado, K., 2008. Análisis de la tendencia y la estacionalidad de la precipitación mensual en Venezuela. Rev. Colomb. Estad. 31 (1), 41–65.
Hejazi, M.I., Edmonds, J.L., Clarke, P., Kyle, E., Davies, V., Chaturvedi, M., Wise, P., Pate, J., Eom, L., Calvin, K., 2014. Integrated assessment of global water scarcity over the 21st century under multiple climate change mitigation policies. Hydrol. Earth Syst. Sci. 18 (2), 2859–2883.
Herath, I., Green, S.R., Horne, D., Singh, R., Clothier, B.E., 2013. Water footprinting of agricultural products: evaluation of different protocols using a case study of New Zealand wine. J. Cleaner Prod. 44 (3), 156–167.
Henríquez, C., Aspee, N., Quense, J., 2016. Zonas de catástrofe por eventos hidrometeorológicos en Chile y aportes para un índice de riesgo climático. Rev. Geogr. Norte Grande. https://doi.org/10.4067/S0718-34022016000100003.
Hoekstra, A.Y., Chapagain, A.K., Zhang, G.P., 2016. Water footprints and sustainable water allocation. Sustainability 8 (20), 1–6.
Hoekstra, A.Y., Mekonnen, M.M., Chapagain, A.K., Mathews, R.E., Richter, B.D., 2012. Global monthly water scarcity: blue water footprints versus blue water availability. PLoS One 7 (3), 32–68.
Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M., 2011. The Water Footprint Assessment Manual: Setting the Global Standard. Earthscan, London, UK.
Hunter, C., Gironás, J., Bolster, D., Karavitis, C., 2015. A dynamic, multivariate sustainability measure for robust analysis of water management under climate and demand uncertainty in an arid environment. Water 7 (11), 5928–5958.
Iglesias, A., Garrote, L., 2015. Adaptation strategies for agricultural water management under climate change in Europe. Agric. Water Manag. 155, 113–124.
INE, 2007. Instituto Nacional de Estadísticas Chile. Censo Agropecuario y Forestal. INE, Santiago, Chile.
Jaramillo, F., Destouni, G., 2015. Local flow regulation and irrigation raise global human water consumption and footprint. Science 350 (3), 1248–1250.
Keys, P.W., Wang-Erlandsson, L., Gordon, L.J., 2016. Revealing invisible water: moisture recycling as an ecosystem service. PLoS One 11 (3), 32–40.
Konar, M., Dalin, C., Suweis, S., Hanasaki, N., Rinaldo, A., Rodriguez-Iturbe, I., 2011. Water for food: the global virtual water trade network. Water Resour. Res. 47 (4), 1–17.
Le Roux, B., Van der Laan, M., Vahrmeijer, T., Bristowa, K.L., Annandale, J.G., 2016. Establishing and testing a catchment water footprint framework to inform sustainable irrigation water use for an aquifer under stress. Sci. Total Environ. 56, 1119–1129.
Liu, C., Kroezea, 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. Ind. 18 (4), 42–49.
Lovarelli, D., Bacenetti, J., Fiala, M., 2016. Water Footprint of crop productions: a review. Sci. Total Environ. 5 (1), 236–251.
Mekonnen, M.M., Pahlow, M., Aldaya, M.M., Zarate, E., Hoekstra, A.Y., 2015. Sustainability, efficiency and equitability of water consumption and pollution in Latin America and the Caribbean. Sustainability 7 (2), 2086–2112.
Mubako, S., Lant, 2013. Agricultural virtual water trade and water footprint of United States. Ann. Assoc. Am. Geogr. 103 (2), 385–396.
Novoa, V., Rojas, O., Arumí, J.L., Ulloa, C., Urrutia, R., Rudolph, A., 2016. Variabilidad de la huella hídrica del cultivo de cereales, río Cachapoal, Chile. Tecnol. Ciencias Agua 2 (2), 35–50.
ODEPA, Oficina de estudios y políticas agrarias, 2013a. Cambio climático Impacto en la Agricultura Heladas y Sequía. ODEPA, Santiago, Chile.
ODEPA, Oficina de estudios y políticas agrarias, 2013b. Evolución de las exportaciones silvoagropecuarias de Chile. ODEPA, Santiago, Chile.
Oyarzun, R., Arumí, J.L., Álvarez, P., Rivera, D., 2008. Water use in the Chilean agriculture: current situation and areas for research development. In: Sørensen, M.L. (Ed.), Agricultural Water Management Research Trend. Nova Science Publisher, New York, pp. 213–236.
Pfister, S., Koehler, A., Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Technol. 43 (11), 4098–4104.
Pizarro, J., Vergara, P., Rodríguez, J., Sanhueza, J., Castro, J., 2010. Nutrients dynamics in the main river basins of the centre-southern region of Chile. J. Hazard. Mater. 175 (2), 608–613.
Richter, B.D., Davis, M.M., Apse, C., Konrad, C.A., 2011. A presumptive standard for environmental flow protection. River Res. Appl. 8 (2), 1312–1321.
Russo, T., Alfredo, K., Fisher, J., 2014. Sustainable water management in urban, agricultural, and natural systems. Water 6 (12), 3934–3956.
Sánchez, M., Carvacho, L., 2013. Aproximación a la determinación del déficit hídrico en la Región del Libertador General Bernardo O'Higgins, Chile, a partir de imágenes MODIS y datos meteorológicos. Rev. Geogr. Norte Grande 55 (2), 109–124.
Stoeglehner, G., Edwards, P., Daniels, P., Narodoslawsky, M., 2011. The water supply footprint (WSF): a strategic planning tool for sustainable regional and local water supplies. J. Cleaner Prod. 19 (15), 1677–1686.
Shrestha, S., Chapagain, R., Babel, M.S., 2017. Quantifying the impact of climate change on crop yield and water footprint of rice in the Nam Oon Irrigation Project, Thailand. Sci. Total Environ. 599, 689–699.
Valiente, O., 2001. Sequía: definiciones, tipologías y métodos de cuantificación. Investigaciones Geográficas 26, 59–80.
Velasco- Muñoz, J.F., Aznar-Sanchez, J.A., Belmonte-Urena, L.J., Roman-Sanchez, I.M., 2018. Sustainable water use in agriculture: a review of worldwide research. Sustainability 10 (4), 1084. https://doi.org/10.3390/su10041084.
Wang, X., Zhang, J., Shahid, S., Guan, E., Wu, Y., Gao, J., He, R., 2016. Adaptation to climate change impacts on water demand. Mitig. Adapt. Strat. Glob. Change 21 (1), 81–99.
Yevenes, M., Arumí, J.L., Farías, L., 2016. Unravel biophysical factors on river water quality response in Chilean Central-Southern watersheds. Environ. Monit. Assess. 188, 264. https://doi.org/10.1007/s10661-016-5235-1.
Xinchun, C., Mengyang, W., Xiangping, G., Yalian, Z., Yan, G., Nan, W., Weiguang, W., 2017. Assessing water scarcity in agricultural production system based on the generalized water resources and water footprint framework. Sci. Total Environ. 609, 587–597.
Zeng, Z., Liu, J., Koeneman, P.H., Zarate, E., Hoekstra, A.Y., 2012. Assessing water footprint at river basin level: a case study for the Heine River Basin in northwest China. Hydrol. Earth Syst. Sci. 16 (4), 2771–2781.
Zhineng, H., Yazhen, C., Liming, Y., Weib, C., Chaozhi, L., 2016. Optimal allocation of regional water resources: from a perspective of equity–efficiency tradeoff. Resour. Conserv. Recycl. 109, 102–113.
Zwiers, F.W., Zhang, X., Feng, Y., 2011. Anthropogenic influence on long return period daily temperature extremes at regional scales. J. Clim. 24 (3), 881–892.