Irrigation implementation promotes increases in salinity and nitrate concentration in the lower reaches of the Cidacos River (Navarre, Spain)

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Abstract

The shift from rainfed to irrigated agriculture is associated with a change in the fertilization rates due to increases in expected production and the fact of growing more N demanding crops. In addition, the circulation of irrigation return flows (IRF) mobilizes soluble salts stored in soils or geological materials. As a consequence, it implies severe modifications in the dynamics and total amount of soluble salts and nitrogen exported, especially in semi-arid watersheds. In this study, long-term data collected by the regional authorities was used to assess the effects of irrigation implementation on salinity (using electrical conductivity, EC, as a proxy) and nitrate concentration (NO_3^-) after the transformation of ca. 77 km^2 from rainfed to irrigated agriculture in the Cidacos River (CR) watershed. The results indicate that water quality in the lower reaches of the CR was significantly modified after the diffuse incorporation of IRF. In contrast, neither EC nor NO_3^- were different in those monitoring stations whose contributing watersheds did not include transformed area. In addition, the temporal dynamics in the analysed variables shifted from a rainfed land signal typical in the region to an irrigated land signal, and the hydrochemical type of the CR shifted from mixed-to-Na^+-mixed-to-HCO_3^- to mostly Na^+-mixed type, typical of waters affected by irrigation return flows in the region. Groundwater EC and NO_3^- also increased in those wells located within the irrigated area. Although there are great uncertainties in the actual amount of salt and N reaching the CR via irrigation return flows, the expected contribution of waste water spilled into the CR is minor in comparison to other sources, mostly agricultural sources in the case of N. The observed changes have promoted the designation of the lower reaches of the CR as "affected" by NO_3^- pollution, and the whole CR watershed as a Nitrate Vulnerable Zone, with the emergent question about whether irrigation implementation as carried out currently in Spain is against the environmental objectives of the Water Framework Directive.

Keywords: Land Use Change; Irrigation Return Flows; Water Quality; Salinization of water bodies; Nitrate pollution.

1. Introduction

Irrigation is a major factor in agricultural intensification (e.g., Pretty, 2008). Consequently, the area equipped for irrigation in the world has increased from 170 to 333 Mha between 1965 and 2015 (FAO and IWMI, 2018). The increase has been observed mainly in Asia, although it is also relevant in countries such as the United States or Australia. This trend is expected to continue in order to meet the nutritional requirements of an increasing world population. In the European

Union, high shares of irrigable area are mainly found in regions of the Mediterranean countries where irrigation is an essential element in many agricultural systems (EU, 2016). Indeed, irrigable area increased by 13.4% from 2003 to 2013 in the European Union. In particular, in Spain the increase of irrigated surface is significant, with 11 % increase in the period 2007 – 2017 (MAPA, 2019).

There is worldwide evidence of a link between irrigated agriculture and salinization (Stigter et al., 2006; Duncan et al., 2008; Pulido-Bosch et al., 2018) and/or nitrate pollution (e.g., Muñoz-Carpena et al., 2002; Thayalakumaran et al., 2008; Stamatis et al., 2011; Dzurella et al., 2015) of water bodies, among other environmental problems (aquifer overexploitation, river diversion, or other polluting substances such as pesticides). Indeed, the leaching of salts is a requirement of irrigated agriculture (Letey et al., 2011) since its accumulation in soils can be deleterious for plants, decreasing productivity (Villalobos et al., 2016) and even forcing the abandonment of cultivation. A proportion of those leached salts will reach a water body, contributing to an increase in its salinity, what affects the quality of its water for a range of human uses: urban supply (WHO, 2011), industrial uses, irrigation, etc. In addition, an increase in salinity implies, in general, a decrease in biodiversity (Nielsen et al., 2003) among other impacts on ecosystems.

Nitrate pollution of water bodies is acknowledged as a threat to water quality for both human consumption (WHO, 2011) and for the increased risk of eutrophication in continental and coastal waters (e.g., Sutton et al., 2011). Although other production systems (such a rainfed agriculture, livestock, aquiculture, etc.) also contribute to nitrate pollution (e.g., Casalí et al., 2008; Menciò et al., 2016; FAO and IWMI, 2018), the intensification of agriculture associated with irrigation along with the concurrent presence of water implies, in general, a higher nitrate-nitrogen loading or yield (the amount of nitrogen leached per unit surface, typically reported in kg NO_3^--N ha^{-1} $year^{-1}$) in irrigated areas. For instance, nitrate yield of over 100 kg NO_3^--N ha^{-1} $year^{-1}$ has been reported for flood irrigated areas in Spain (Barros et al., 2012a; García-Garizábal et al., 2012), whereas values in the range of ca. 20 – 70 kg NO_3^--N ha^{-1} $year^{-1}$ have been reported for pressurized irrigation systems (Andrés and Cuchí, 2014; Cavero et al., 2003; Merchán et al., 2015a, 2018). In contrast, N yield for rainfed systems tend to be lower: 6 – 32, 10 – 40 and 16 – 37 kg N ha^{-1} $year^{-1}$ in Sweden, Estonia, and Spain, respectively (Kyllmar et al., 2014; Iital et al., 2014, Casalí et al., 2008). In addition, irrigation is preferably implemented in semi-arid regions, where the nitrate loading under rainfed conditions is normally lower than those values reported above (Merchán et al., 2018).

In this context, it may be expected that a transition from rainfed to irrigated agriculture will increase the salinity and nitrate concentration and exports in a watershed. Indeed, in a review about land use changes, Scanlon et al. (2007) reported several examples (mainly from China, India and the U.S.A.) of irrigation effects in water quantity and quality. However, most of the reported examples do not present a pre-irrigation data set or a sharp increase in irrigated surface, but rather long-term trends or hydrochemical evidence. In fact, to the best of our knowledge, only a few studies have conducted such pre- vs. post-irrigation implementation comparison. For instance, Merchán et al. (2013, 2015a, 2015b) reported modifications in water quantity and quality for a semi-arid small (ca. 7.3 km^2) watershed in Spain in which irrigation was implemented in ca. 50 % of the watershed. Although this kind of studies at the small watershed scale have many advantages (such as relatively homogeneous climate, geology or land uses, low influence of flood plains or groundwater; Buttle, 1998), the dynamics of salts and especially nitrogen are highly scale-dependant, being of great interest to understand how the processes observed at the small watershed scale-up. In New Zealand, the impacts of irrigation implementation in pastures used for dairy production were analysed in two large watersheds (210 and 675 km^2), finding significant increases in N and P concentrations and exports in the streams draining them (McDowel et al., 2011). A recent study conducted in China (Hu et al., 2019) reported a significant increase in drainage salinity after irrigation expansion in a watershed (221 km^2) with desert climate. Given the international context of irrigation expansion and the global tendency to protect water quality, more studies on the impacts of irrigation implementation are required in order to understand the processes involved and feed models that will contribute to the environmental impact assessment of new irrigated areas.

Navarre (north-east Spain) is one of the regions with a higher rate of expansion of irrigated land in relative terms. Around 65,000 ha of arable land were irrigated in the 1980s, being negligible the surface under pressurized irrigation. Between 2000 and 2017, the surface has increased up to 102,000 ha or 29 %, being pressurized systems those recently implemented (Government of Navarre, 2018). Another 21,500 ha are under planning stages and intended to be transformed during the next years (CANASA, 2014-2019). Among the surface recently transformed from rainfed to irrigated agriculture in Navarre, there are around 77 km^2 in the Cidacos River (CR) watershed (477 km^2). The transformation of this area spans for a few years (mostly 2009 – 2011). This process has been recorded by the water quality monitoring network of the regional government, with surface and groundwater quality data for the period 2000 – 2018, i.e., before and after irrigation implementation. In relation with other published studies, this work presents a new pre vs. post case study at a regional scale (> 100 km^2) that builds on previous research at the small watershed scale. In addition, the changes in the agricultural system in the CR were significantly more drastic than those reported in other studies, as the irrigation water was obtained from other watershed and the fact that the expansion of irrigated surface occurred in a watershed in which the previous amount of irrigated land was negligible.

The objectives of this study were: (a) to characterize the water quality of the CR in the period before irrigation implementation in terms of salinity and nitrate concentration in order to provide a baseline for comparison with the irrigated period; (b) to detect and estimate changes in salinity and nitrate concentration occurred as a consequence of irrigation implementation; (c) to gain insight in the hydrological processes promoting those changes.

2. Methods

2.1. Study site description

The Cidacos River (CR) is a tributary of the Aragón River, which is itself a tributary of the Ebro River (Fig. 1), one of the main rivers in the Iberian Peninsula. Its watershed (centre at ca. 42º 31' North; 1º 37' West) is fully located within the province of Navarre (northern Spain). The watershed covers 477 km^2, and spans approximately 35 km from north to south and 15 from west to east. The CR follows mainly a north to south direction, with the main tributaries originating in the east. Altitudes range from 300 m (in the point where it discharges into the Aragón River) to ca. 1100 m (in the northern mountains, "Sierra de Alaiz").

According to the available meteorological information (Government of Navarre, 2018), the climate in the watershed is humid to dry temperate Mediterranean (Papadakis classification) with annual average temperature from 14.2 ºC in the south to 12.2 ºC in the north. Monthly averages reach up to 23.7 ºC and 21.2 ºC in August (south and north, respectively) and down to 5.4 ºC and 4.7 ºC in January (idem). A similar but more severe gradient exist for precipitation, from ca. 430 mm of rainfall in the south and up to 800 mm in the north. Reference evapotranspiration (ET_0 computed following Penman-Monteith method, Allen et al., 1998) is ca. 1150 mm $year^{-1}$, with around 76 % of ET_0 occurred during the April – September period.

The CR watershed is located within the Ebro Depression Domain (DOPTC, 2003). The oldest geological materials are sandstone and mudstone of Paleocene age located in the northern mountains. Most of the watershed is covered by red clays with sandstone and mudstone of Miocene age. Finally, there are gravel, sand and silt deposits in alluvial or terrace deposits (Quaternary age) near the main axis of the CR. The later form a free aquifer (CR alluvial aquifer) of 61 km^2 of extension and up to 20 m depth with mainly gravel and sand with low degree of cementation at the bottom and silt and clay in the surface layers (DOPTC, 2003).

The predominant land use in the CR watershed is agriculture. According to data from 2012 (Corine Land Cover, available at SITNA, 2018), there are 260 km^2 cultivated (88% herbaceous, 12% woody crops). Other land uses include shrub lands (107 km^2), forests (92 km^2), and a minor proportion of unproductive areas (urban, industrial, etc; 18 km^2). Before irrigation implementation the irrigated surface was minor (i.e., below 5 km^2 of traditional flood irrigation in the river terraces and surrounding areas). When irrigation water from the so-called "Canal de Navarra" was made available, pressurized irrigation was implemented in ca. 77 km^2. "Canal de Navarra" provides high quality irrigation water from a reservoir located in northern Navarre, a region with a rather higher annual precipitation (ca. 900 – 1700 mm $year^{-1}$). It crosses the CR watershed from northwest to southeast (Figure 1). The new irrigated surface was implemented between the years 2009 and 2011 (INTIA, 2018).

The main crops under rainfed agriculture are winter cereals (wheat and barley mainly), and vineyards. Average annual fertilization rates under rainfed agriculture were ca. 80 – 130 kg N ha^{-1} for winter cereals and 40 – 50 kg N ha^{-1} for vineyards. The estimated irrigation volume applied in the CR watershed for the 2014 – 2018 period (AguaCanal, 2018) is 27.5 hm^3 $year^{-1}$ or ca. 3600 m^3 ha^{-1} $year^{-1}$. The diversity of crops increased after irrigation implementation, with the appearance of new crops such as maize, grasses, peas, a wide range of vegetables, and others. In addition, due to the different crops and the higher expected production, fertilization needs increased. Average fertilization rates increased under irrigated agriculture up to 135 – 155 kg N ha^{-1} $year^{-1}$ for winter cereals and 50 – 60 kg N ha^{-1} $year^{-1}$ for vineyards. Fertilization rates for the new crops were 295, 195 and 160 kg N ha^{-1} $year^{-1}$ for maize, tomatoes and onions, respectively, among others (data provided by extension service, INTIA).

2.2. Available hydrological data

The study reported herein is based mainly in information obtained in the monitoring networks of water quality implemented by the Government of Navarre (Department of Rural Development and Environment). Currently, there are six physic-chemical quality monitoring points in the CR, most of them in operation since the year 2000. The monitoring points are located in the proximities of several settlements, namely, Barásoain, Pueyo, Tafalla, Olite, Beire and Traibuenas, and are coded with those names in this study (Fig. 1, Table 1). In addition, there are five groundwater monitoring points located in the CR alluvial aquifer in which groundwater level and quality are recorded. These monitoring points are coded as Tafalla, Olite, Pitillas, Murillo 1 and Murillo 2 according with the nearest village (Fig. 1, Table 1). In these points, samples were collected and analysed following standardised procedures by proficient staff and certified laboratories, respectively. Data was provided by the public company Environmental Management of Navarre (GAN, from its Spanish acronym, www.gan-nik.es).

In the collected samples, some parameters were consistently measured whereas other are determined in a subset of samples. In-situ data included electrical conductivity corrected to 20 ºC (EC), pH, dissolved oxygen and redox potential (relative to Cl-AgCl electrode). Laboratory determination included consistently nitrate (NO_3^-), chloride (Cl^-), sulphate (SO_4^{2-}), and bicarbonate (HCO_3^-). Additionally, in 29 % of surface water and all of groundwater samples major cations (calcium, Ca^{2+}; magnesium, Mg^{2+}; sodium, Na^+, and potassium, K^+) were determined.

Discharge in the CR was measured downstream the town of Olite (Fig. 1) in a gauging station installed and maintained by the Government of Navarre in 1988. Discharge data was aggregated to daily averages for analysis in this study. It is important to note that Olite gauging station measures the discharge generated in approximately 54 % of the watershed, particularly that not affected by irrigation implementation as discussed in section 2.1.

Besides the water quality and quantity data, other data sources used in this work include: wastewater effluents information (quantity and quality) collected by NILSA, the public company in charge of, among other functions, wastewater treatment plants in Navarre (www.nilsa.com); and analytical determinations (EC and NO_3^-) of irrigation return flows collected in the drainage network of the irrigable area in the year 2016 by INTIA, the public company in charge of, among other functions, agricultural extension (www.intiasa.es).

2.3. Data treatment and statistical analysis

To obtain an estimation of the proportion of irrigated surface contributing to the CR throughout its course, we used georeferenced data available at IDENA (SITNA, 2018). In particular, using a DEM (5 m × 5 m) and a layer with polygonal information of irrigated surfaces, we obtained the contributing watershed to any specific point in the river course along with the irrigated surface.

For comparison purposes, we selected a subset of samples in each monitoring point with available information: hydrological years 2001 – 2008 (Oct 1st 2000 – Sep 30th 2008) as representative of pre-irrigation implementation conditions (PRE), and hydrological years 2012 – 2018 (Oct 1st 2011 – Sep 30th 2018) as representative of the period in which irrigation surface has stabilized (POST) after the sharp increase between 2009 and 2011. We compared median EC and NO_3^- between PRE and POST periods in each point through the Wilcoxon-Mann-Whitney Rank-Sum Test (Helsel and Hirsch, 2002). Similar tests were carried out to test differences in discharge in the station were data was available (Olite), or among different monitoring points within the same period to assess the evolution of the parameter within the watershed. In the case of significant differences, we estimated the difference in medians through the Lehman-Hodges estimator (Helsel and Hirsch, 2002). We used non-parametric tests and estimators as the requirements in the statistical distribution of the input data are minimal, allowing the use of the same test in different variables without any kind of transformations to achieve normality. The tests and estimators were performed using *ad hoc* spreadsheets.

The sampling frequency (ca. 8 samples per year) did not allow for an adequate seasonal representation of the variables. For this reason, all samples for a period (PRE or POST) and monitoring point were lumped together and used in a smoothing technique (locally estimated scatterplot smoothing, LOESS) using its ordinal date (i.e., January 1st is day 1 and December 31st is day 365) to obtain a representation of the seasonal cycle of the variable. LOESS was applied using R software (R Development Core Team, 2008).

Given the low availability of groundwater quality data during the period prior to irrigation implementation, the approach to assess the effects of irrigation in groundwater were different. In this case, we assessed and quantified trends in the common temporal series for all monitoring wells (ca. October 2013 – September 2018) using the non-parametrical Mann-Kendall test, and quantified slopes by Sen's method (Helsel and Hirsch, 2002). We computed the statistical test and estimated the slope using *ad hoc* spreadsheets.

Finally, the loads of total dissolved solids (TDS) and dissolved inorganic nitrogen (DIN) in Olite gauging station were computed. First, we estimated TDS for those samples in which major ions had been determined and obtained a relationship between TDS and EC (which was determined for every sample). Second, we computed the DIN as the sum (in mg N L^{-1}) of nitrate, nitrite and ammonium concentrations. Although other form of N were not considered, this approach includes the most abundant N forms in nutrient-enriched waters (e.g., Durand et al., 2011). Then, we computed the load assigning the closest available sample to the daily average discharge measured in the gauging station.

3. Results

The upper reaches of the CR presented negligible irrigated area in its contributing watershed, and this irrigated areas date back from before the year 2000, i.e., they were not implemented during the transformation assessed in this study. Around 25 km above the point in which the CR discharge in the Aragón River, the proportion of irrigated surface begins being significant (Fig.

2). There are several points at which a tributary whose watershed has a high proportion of irrigated surface joins the CR, increasing the share of irrigated surface in the later significantly (kilometre points 24.5, 17.5, 14.0, 10.0, 8.5, 3.5; Fig. 2). Considering the river mouth, around 17 % of the CR basin surface is under irrigation. The transformed surface is located in the southern half of the watershed. In fact, most of the new irrigated surface is located in the watershed area that the CR gains between an intermediate monitoring point (town of Olite, watershed area 258 km^2) and the CR outlet where it reaches the Aragón River. This implies that ca. 35% of the gained watershed area (i.e., 77 out of 219 km^2) were transformed to irrigation between 2009 and 2011.

3.1. Surface water before irrigation implementation (PRE)

Median discharge in Olite gauging station during the hydrological years 2001 – 2008 was 0.10 $m^3 s^{-1}$ (inter-quartile range, IQR: 0.03 – 0.63 $m^3 s^{-1}$). Monthly median values ranged from 0.02 $m^3 s^{-1}$ in August or September up to 1.25 $m^3 s^{-1}$ in March. River contribution ranged from 1.0 hm^3 in the hydrological year 2002 to 51.9 hm^3 in 2007, averaging 22.9 hm^3 for the period before irrigation implementation.

The salinity of water increased and nitrate concentration decreased from headwaters in downstream direction, as depicted in Fig. 2 and Fig. 3 (note that data covering the PRE period are depicted in the first box-plot in each pair). Median EC increased from 650 µS cm^{-1} (IQR: 590 – 760 µS cm^{-1}, n = 76) in Pueyo (ca. 34 km from the CR mouth) up to 900 µS cm^{-1} (IQR: 710 – 1270 µS cm^{-1}, n = 75) in Traibuenas (3 km upstream from the CR mouth). In fact, there were significant differences among monitoring points in the upper reaches (Pueyo and Tafalla) of the river and those in the lower ones (Beire and Traibuenas, $p < 0.001$). In contrast, median NO_3^- decreased from 50 mg L^{-1} (IQR: 34 – 64 mg L^{-1}, n = 76) in Pueyo down to 29 mg L^{-1} (IQR: 12 – 42 mg L^{-1}, n = 75) in Traibuenas. Again, there were significant differences ($p < 0.001$) among monitoring points in the upper reaches (Pueyo) of the river and those in the lower ones (Beire and Traibuenas).

EC presented recognisable seasonal cycles, as it tends to reach minimum values in late winter or spring, whereas it reaches its maximum values in late summer. However, this pattern is more easily recognizable in the upper reaches of the CR (Barásoain, Pueyo, Olite; example of the later in Fig. 4) than in the lower half (Beire, Traibuenas; example of the later in Fig. 4). NO_3^- was rather more variable, and consistent seasonal cycles were not detected (example for Olite in Fig. 4), although in particular monitoring points there were apparent cycles (example for Traibuenas in Fig. 4).

Among water major chemical constituents (example in Fig. 5), Ca^{2+} was the predominant cation and HCO_3^- the predominant anion throughout the CR, although the contribution of Ca^{2+} and HCO_3^- decreased while that of Na^+, SO_4^{2-} and Cl^- increased in the lower reaches of the river. Thus, CR water hydrochemical type was mainly of Ca^{2+}-HCO_3^- type in the upper reaches and of mixed-to-Na^+-mixed-to-HCO_3^- in the lower reaches.

3.2. Surface water after irrigation implementation (POST)

Median daily averaged discharge in Olite during the hydrological years 2012 – 2018 was 0.29 $m^3 s^{-1}$ (IQR: 0.09 – 1.17 $m^3 s^{-1}$). Monthly median ranged from 0.07 $m^3 s^{-1}$ in September up to 3.00 $m^3 s^{-1}$ in March. River contribution ranged from 3.3 hm^3 in the hydrological year 2012 to 86.0 hm^3 in 2013, averaging 39.4 hm^3 for the period after irrigation implementation. The discharge in the post period was significantly higher ($p < 0.001$) than in the pre-irrigation period for almost any particular month, with estimated differences in the monthly medians ranging from 0.0 $m^3 s^{-1}$ in December to 1.8 $m^3 s^{-1}$ in March.

The modification in EC and NO_3^- are depicted in Fig. 2 and Fig. 3 (note that data covering the POST period are depicted in the second box-plot in each pair). Despite the significant differences in discharge, the EC and NO_3^- of the CR water only changed significantly from the PRE to the POST period ($p < 0.001$) in Traibuenas, i.e., that monitoring point downstream of most of the new irrigated area. Indeed, in this monitoring point median EC increased from 900 to 1340 μS cm^{-1} and median NO_3^- increased from 29 to 49 mg L^{-1}. In Beire (upstream from Traibuenas but still in the area of influence of irrigation implementation), median EC and NO_3^- were higher in the POST period than in the PRE-irrigation period, although in this case there were no significant differences (p = 0.13 and 0.14 for EC and NO_3^-, respectively). In the remaining points (those no or slightly affected by irrigation) both median EC and median NO_3^- were lower for the post period, although again no significant differences were founded ($p > 0.05$ in all the cases).

In Traibuenas, EC and NO_3^- not only increased, but also their seasonal cycles were severely modified (Figs. 4). Before irrigation implementation, maximum and minimum EC was usually reached in December-January and September-October, respectively. During the period in which irrigation was implemented, this shifted to maximum during August-September and minimum during February-March. Indeed, there were no significant differences during the non-irrigated season (as depicted by overlapping 95%-confidence intervals in Fig. 4), but significant differences during July-November, that is, the irrigated season along with the following months. Similar but even clearer patterns were observed for NO_3^-, with significant differences from middle May to December (Fig. 4). Seasonality in these water quality variables did not significantly change in other monitoring points, as depicted for Olite (Fig. 4), although the effect of irrigation return flows in nitrate concentration begins being apparent at this point, with lower NO_3^- throughout the non-irrigated season and higher in the end of the irrigation season (Fig. 4).

As observed for the total amount of salts (depicted by EC), there were no differences in the dissolved major constituents of water in the upper reaches of the CR between pre- and post-irrigation implementation. However, the share of Na^+ and Mg^{2+} (among cations) and Cl^- and SO_4^{2-} (among anions) increased in the lower reaches, especially in Traibuenas but also in Beire and

Olite (example in Fig. 5). Indeed, the water hydrochemical type shifted from mixed-to-Na^+-mixed-to-HCO_3^- to mostly Na^+-mixed type.

3.3. Groundwater trends throughout the study period

Only two groundwater monitoring points had available data for the PRE period. In Tafalla (n = 55), median EC was 1120 µS cm^{-1} (IQR: 1070 – 1180 µS cm^{-1}) and median NO_3^- was 61 mg L^{-1} (IQR: 53 – 69 mg L^{-1}). In Pitillas, available data cover only ca. half of the PRE period (hydrological years 2005 – 2008, n = 28). Median EC was 880 µS cm^{-1} (IQR: 820 – 950 µS cm^{-1}) and median NO_3^- was 52 mg L^{-1} (IQR: 47 – 59 mg L^{-1}).

During the POST period, median EC was 1160 µS cm^{-1} (IQR: 1110 – 1190 µS cm^{-1}) and median NO_3^- was 51 mg L^{-1} (IQR: 46 – 55 mg L^{-1}) in Tafalla (n = 78), with no significant differences for EC but significantly lower values ($p < 0.001$) in the POST-period (the estimated difference in median NO_3^- is 8 mg L^{-1}). As mentioned above, no statistical comparisons were performed in the remaining monitoring points due to the scarcity (Pitillas) or lack of samples (Olite, Murillo-1 and Murillo-2) during the PRE period. In Pitillas (n = 77), median EC was 1330 µS cm^{-1} (IQR: 1090 – 1570 µS cm^{-1}) and median NO_3^- was 49 mg L^{-1} (IQR: 42 – 61 mg L^{-1}). In Olite (n = 77), median EC was 1100 µS cm^{-1} (IQR: 1060 – 1150 µS cm^{-1}) and median NO_3^- was 60 mg L^{-1} (IQR: 52 – 67 mg L^{-1}). In Murillo-1 (n = 65), median EC was 2160 µS cm^{-1} (IQR: 2090 – 2300 µS cm^{-1}) and median NO_3^- was 62 mg L^{-1} (IQR: 51 – 67 mg L^{-1}). Finally, in Murillo-2 (n = 83), median EC was 2670 µS cm^{-1} (IQR: 2610 – 2760 µS cm^{-1}) and median NO_3^- was 91 mg L^{-1} (IQR: 41 – 154 mg L^{-1}).

Regarding the estimated trends during the period POST-irrigation implementation, those monitoring wells in the area affected by it presented significant trends for both EC and NO_3^- (Table 2). For those with low affection, no significant trends or even decreasing trends were detected (NO_3^- in Olite, Table 2). Pitillas presented a significant increasing trend for EC but a decreasing trend for NO_3^-, being the highest detected trend in EC of all the groundwater monitoring points. The southernmost point (Murillo-2) presented the highest NO_3^- trend (43 mg L^{-1} $year^{-1}$), reaching values over 300 mg L^{-1} in the year 2018.

3.4. Complementary water quality data

Irrigation Return Flows: During the 2016 irrigation campaign, sixteen collectors draining the irrigated area were sampled in April, August and November (that is, before, during and after the irrigation season). Median NO_3^- in these samples was 60.6 mg L^{-1} (IQR: 35.6 – 98.6 mg L^{-1}). There were no significant differences among sampling campaigns ($p = 0.5$) or a clear pattern within a sampling location (paired tests, $p > 0.05$). Median EC was 2180 µS cm^{-1} (IQR: 1610 – 4190 µS cm^{-1}), with no significant differences among sampling campaigns ($p = 0.13$), although in this case the paired tests indicate lower EC values during the irrigated season than those recorded either before ($p = 0.024$) or after it ($p < 0.001$). Unfortunately, major constituents were not determined in these samples and therefore there is no information about the water types.

Wastewater effluents: Three treatment plants discharge wastewater into the CR (Table 3). As a whole, ca. 2.15 hm^3 of wastewater, 2600 Mg of total dissolved solids and 34 Mg of N are spilled into the river each year, at a rather constant rate (no significant differences in the served town's population throughout the year).

3.5. Estimation of the mass of salts and nitrogen in Olite gauging station

The obtained regression equations between EC and TDS (1) was:

$$\text{TDS (mg L}^{-1}\text{)} = 0.803 \times \text{EC [µS cm}^{-1}\text{]} + 42 \qquad (n = 40;\ R^2 = 0.95)\ \text{[Eq. 1]}$$

For the period 2001 – 2018, the estimated annual load of TDS was ca. 16,900 Mg, or around 660 kg ha^{-1}, and its inter-annual variation was considerable (coefficient of variation, CV = 90 %).

Similarly, the average annual load of DIN was 220 Mg (CV = 90 %), or around 8.6 kg N ha^{-1}. In both cases, the load was transported mainly in winter months, with 58 % and 60 % of the load transported in January – March for TDS and DIN, respectively, and only ca. 3 % of the load transported during summer.

4. Discussion

4.1. Patterns of variation in water quality before irrigation implementation

In general, the Cidacos River water increased its salinity (as indicated by EC) and the share of Ca^{2+} and HCO_3^- in its TDS decreases from upper to lower reaches. This observation is justified by the significant gradient in climatic conditions along with a change in the geological materials in the watershed contributing runoff (more salt-providing materials such as gypsum and more saline soils in the south). This general pattern is consistent with the main controlling factors of river salinity (geology and climate, Milliman and Farnsworth, 2011). In most of the stations, a clear seasonal cycle is detected with high salinity during low waters (summer months) and low salinity during high waters (January to March), which is consistent with what has been described in Navarre for non-irrigated small watersheds (Merchán et al., 2019). The amount of exported TDS up to Olite (ca. 660 kg ha^{-1} $year^{-1}$) is lower than reported for rainfed watersheds in Navarre at smaller scales (ca. 1100 kg ha^{-1} $year^{-1}$; Merchán et al., 2019), what is justified mainly by the lower runoff in the CR (ca. 115 mm) than in the small scale rainfed watersheds (222 – 250 mm).

Before irrigation implementation, the CR may be considered as significantly impaired by diffuse pollution from agricultural sources, as depicted by the high NO_3^- observed in the groundwater and the upper reaches of the CR, and the DIN load estimated in Olite. Rainfed crops (winter cereals such as barley or wheat) dominate the agricultural production in this area. Nitrate-Nitrogen (NO_3^--N) exports from rainfed crops have previously been reported as significant in Navarre (Casalí et al., 2008; Lassaleta et al., 2010). Indeed, recent estimates from small watersheds (169 and 207 ha) representative of rainfed areas in Navarre were between 17 and 32 kg NO_3^--N ha^{-1} $year^{-1}$ (Hernández et al., in prep.), with a seasonal distribution similar to that observed in Olite (i.e., more than 50 % of the load in winter months). The value estimated for Olite (8.6 kg N ha^{-1} $year^{-1}$) is consistent with the values estimated at the small watershed scale given the differences in land uses in Olite (ca. 47 % of the surface under croplands vs. 90-95 % in the small watersheds) and the feasible attenuation processes (denitrification and N assimilation in the stream, riparian areas and floodplains; Burt et al., 2007; Antigüedad et al., 2017). With regards to the spatio-temporal dynamics, although NO_3^- values are rather similar during winter throughout the CR, during summer NO_3^- decreases in the lower reaches of the watershed. This observation may be explained by two processes: a) the afore mentioned attenuation processes; and b) a lower N loading contribution from the lower half of the watershed as a consequence of the low productive rainfed areas (Merchán et al., 2018).

4.2. Expansion of irrigation in the lower part of the watershed

The expansion of irrigated land in the lower half of the watershed implied an increase in river salinity and nitrate concentration in the monitoring points affected by the land use change. No significant changes were detected in the monitoring points in the upper reaches, where despite the non-significant differences, median EC and NO_3^- was lower for the irrigated period. This observation probably responds to a higher river discharge (wetter years) during the POST period as these variables usually respond to more humid years with lower concentrations (Milliman and Farnsworth, 2011; Burt, 2001). In contrast, a dramatic increase in EC and NO_3^- were observed in Traibuenas (Fig. 2 and 3), that is, the monitoring point in which the irrigation return flows of ca. 7000 ha have been incorporated. In addition, although non-significant in a yearly basis, the effect of irrigation in both Beire and Olite seems apparent from the seasonal cycle during the POST period. Beire is the first monitoring point in which the median actually increases and there are clear seasonal effects in both this point and the one upstream, Olite (Fig. 4). Although no compositional data were available for IRF in this study, the shift in the hydrochemical water type was consistent with that observed in other stream significantly affected by IRF under relatively similar climatic, geologic and agronomic conditions (Merchán et al., 2015c).

It is worthy to mention that, while surface water monitoring points integrate what happens in the contributing watershed, groundwater monitoring points are greatly affected by local factors (such as local flow directions, geological materials in the vicinity, or land use in the proximities of the well). Despite this fact, the effect of irrigation land use in the CR watershed seems apparent also in the groundwater monitoring points in this particular case. Indeed, significant increasing trends were detected for both EC and NO_3^- (Table 2) in those monitoring points in the irrigated area (Murillo-1 and Murillo-2) whereas no significant trends were detected for those monitoring points slightly or no affected by irrigation implementation (Tafalla and Olite). Increasing trends in groundwater NO_3^- under irrigated areas has been widely observed (several examples available in a review by Scanlon et al., 2007). Regarding groundwater salinity, the casuistic is more complex, as it depends on the source of irrigation water, the salinity of the soils, geological materials, irrigation water and the management (irrigation systems, efficiencies achieved, etc.). Thus, a combination of these factors may account for the dynamics observed in this study for the well Pitillas (increasing salinity and decreasing NO_3^-), which is located in an area with high soil salinity. In most of the cases, a long-term salinization of groundwater may be expected in areas under intensive irrigation (Foster et al., 2018; Pulido-Bosch et al., 2018). However, exceptions do exist. For instance, Stigter et al. (2006) reported how the substitution of the water source from locally extracted groundwater to regionally supplied surface water triggered freshening of the aquifer.

The increase in salinity and NO_3^- in the CR is consistent with the higher export usually reported for irrigated areas in comparison with rainfed ones (e.g., Merchán et al., 2018). Indeed, salt and nitrate export increased two-fold and three-fold after the implementation of irrigation in ca. 50 % of the surface in a study conducted at the small (7.3 km^2) watershed scale (Merchán et al., 2015a). Regarding water salinity, a decrease in headwaters salinity has been reported for recently transformed irrigated areas (Merchán et al., 2013, 2018). In these small watershed, salts that had accumulated during prevailing semi-arid conditions are washed out and leached by irrigation waters, being the rate of salt export higher than the possible weathering and dissolution of new salts. As a consequence the salinity in the stream decreased over time. In contrast, at a larger watershed scale the opposite effect is observed, i.e., an increase in river salinity as a consequence of the salt-enriched irrigation return flows (CHE, 2006; Hu et al., 2019). The results of the present study are in agreement with the later. A question emerges on where the threshold would be for decreased salinity at small watershed scale and increased salinity at large scale, what will depend on the salinization controlling factors at different spatial scales. Hydrological response (both quantitative and qualitative) at the small watershed scale is controlled mostly by soils and hillslope processes (e.g., Buttle, 1998). In the case of salinization, the net effect is the wash out of soluble salts after irrigation implementation. Consequently, the salinity of drainage water in small watersheds tend to decrease due to less available salts after irrigation implementation (Merchán et al., 2013). At a higher spatial scale, the influence of soils and hillslopes diminish as stream processes and the mix of different water sources become the main controlling factor (Tiwari et al., 2017). Indeed, this issue is rather complex, since equilibrium in the new conditions probably has not been reached after the recent land use change, and it may take long periods to be reached (even millennia at the large watershed scale; Thayalakumaran et al., 2007).

The dynamics of nitrate concentration after irrigation implementation are, in contrast with those of salinity, rather straightforward. An increase of NO_3^- is observed at a range of scales (CHE, 2006; Lassaletta et al., 2009; McDowel et al., 2011; Merchán et al., 2013), although the rate of change normally decreases with increasing watershed size. This fact is justified by the decreasing share of irrigation land use as the size of the watershed increases (Merchán et al., 2013), along with the aforementioned attenuation processes of NO_3^- pollution (Antigüedad et al., 2017). Indeed, this pattern of variation from small to large watersheds has also been reported for non-irrigated watersheds affected by diffuse pollution from agricultural sources (Howden and Burt, 2008).

4.3. Modifications in the water budget

Since irrigation does not only modify water quality but also water quantity in a watershed, to adequately assess the effect of irrigation return flows in CR water quality, not only concentrations but also loads should be taken into account (Aragües, 2011). Indeed, several studies show lower concentration but higher loads (and therefore impact in receiving water bodies) in low efficient irrigation systems (e.g., traditional flood irrigation schemes) than in pressurized ones (García-Garizabal et al., 2012, 2014; Barros et al., 2012a, 2012b).

Unfortunately, no gauging station was available in the lower reaches of CR during the study period, so load estimations in this study are limited to those of the upper reaches (not affected by irrigation implementation). From estimated efficiencies by design of the irrigation system (90 – 95 %; INTIA, 2018), it is expected that 1.3 – 2.7 hm^3 of irrigation return flows will reach the CR, mostly during the irrigation season and the following months. In September 2016, a campaign of river discharge measurements was performed to assess, in periods without rainfall in the previous weeks, the contribution of IRF to the CR (GAN, personal communication). This assessment indicated an unaccounted increase of discharge in the CR of ca. 0.12 $m^3\ s^{-1}$, which are expected to be mainly IRF. In 2017, discharge monitoring began in the lower reaches through a pressure sensor and a series of direct discharge measurements in the stream in order to obtain a rating curve (INTIA, 2018). A partial water balance between Olite and Traibuenas suggested that IRF could contribute 0.6 – 1.8 hm^3, or 0.06 – 0.17 $m^3\ s^{-1}$ during the assessed period (June – September 2017), which is consistent in order of magnitude with the punctual estimations reported above. However the uncertainties of these approaches are considerable and more work is needed to fully assess the effect of IRF in the discharge and water quality of the CR.

Although the impact or IRF in water quantity and quality happens mainly during the irrigated season, some lag in the hydrological response is expected as a consequence of storage in soils and/or phreatic layers (Andrés and Cuchí, 2014; Merchán et al., 2015b). Under non-irrigated conditions under Mediterranean climate, low runoff generation is expected during the first half of autumn in rainfed areas (e.g., Giménez et al., 2012). A higher soil moisture content in the soils along with higher water table in aquifers imply a higher runoff proportion and base flow in streams after the first rainfall events in autumn.

4.4. Expected influence of wastewater effluents

The average contribution of TDS and N loads from the wastewater treatment plants located in the upper reaches of the watershed (Barasoain-Garinoain, Table 3) accounts for ca. 0.5 % of salt load and 1.5 % of N load estimated in Olite. Therefore, even neglecting any process of natural attenuation that N may be exposed to, these loads seems to be minimal in comparison with the load carried by the CR. Although salt loads in the CR have a mixed origin (both natural and anthropogenic), agricultural land use is expected to enhance weathering and provide soluble constituents (fertilizers), and thus contribute significantly to the dissolved load (Merchán et al., 2019). In contrast, N load measured at Olite is expected to be mainly from agricultural origin, as background levels of NO_3^- rarely exceed 5 mg L^{-1} in pristine watersheds (Durand et al., 2011).

In the lower reaches of CR there is more uncertainty on the expected influence of wastewater, since the lack of a gauging station does not allow for adequate load estimation. Therefore, we cannot compare the estimated flux of salts and N with that of the CR. However, the estimated annual amount of salts and N provided by wastewater in the whole watershed (2600 and 34 Mg, respectively) is ca. 15 % of that annually flowing through Olite (16900 and 220 Mg, respectively), that is, around 60 % of the watershed surface. Considering that the CR increase its watershed in ca. 220 km^2 and estimations of its increase in water contribution average 20.3 hm^3 (Sacramento model simulation performed by Government of Navarre staff, personal communication) i.e., an increase of ca. 60 % with respect to that measured in Olite, the share of salts and N originated in wastewater treatment plants is expected to be below the 10 % figure.

It is worthy to mention that, although of minor importance considering annual figures, the contribution of wastewater may be significant in specific low-flow periods as the CR present a marked seasonal cycle in discharge and wastewater effluents are relatively constant throughout the year.

4.5. Implications related to Water Framework Directive

In this section the main implications of irrigation implementation in the context of the environmental objectives of the Water Framework Directive (WFD; Directive 2000/60/EC) are discussed. Although this section applies mostly to European countries, the discussed ideas are transferable, at least partially, to other legislative frameworks (e.g., total maximum daily loads system in the U.S.A.).

Given the relevance of agricultural land uses in NO_3^- pollution, in 1991 the Council of the European Communities dictated the Nitrates Directive (ND; Directive 91/676/EEC), concerning the protection of waters against pollution caused by NO_3^- from agricultural sources. According to the ND, those water bodies with a NO_3^- concentration higher than 50 mg L^{-1} are to be declared as "affected" by NO_3^- pollution, and the area contributing to this pollution must be designated as a Nitrate Vulnerable Zone (NVZ), where best management practices are promoted and action plans are implemented in order to minimize the possibilities of NO_3^- leaching. The instruments included in this directive were later incorporated in other norms. Indeed, all water bodies in the European Union were expected to comply with a set of environmental objectives by 2015 according with the WFD. These objectives include a good *ecological* and *chemical* status for surface water bodies and *quantitative* and *chemical* status for groundwater bodies. In both cases, one of the parameters to comply with the "good chemical status" is NO_3^- content. In addition, NO_3^- affected water bodies are considered "protected areas" within the WFD, that is, water bodies in which the achievement of the environmental objectives does not allow for exceptions or delays. This consideration does not only affect farmers, but also other users (for instance, higher prices for spilling wastewaters under the Spanish legislation).

According with the study case presented herein, the implementation of irrigation as carried out currently is against the fulfilment of WFD and ND in those water bodies receiving the IRF. Indeed, the CR watershed has recently been designated as a NVZ (Orden Foral 247/2018). Besides, despite the scarcity of data in many cases, irrigation is presumably linked to the designation of several NVZ in southern Europe countries. For instance, in the Ebro River Basin, 19 out of 36 affected groundwater bodies and 6 out of 11 affected surface water bodies are located within irrigated areas (CHE, 2016). In addition, agriculture has been reported as one of the sectors with less contribution to another of the charges of the WFD, "cost recovery principle" (e.g., Expósito, 2018) or the necessity to charge users with all the costs (infrastructures, water, social, environmental…) of their water use or the pollution generated by their activities.

5. Conclusions

The shift from rainfed to irrigated agriculture is associated with a change in the fertilization rates due to increases in expected production and the fact of growing more N demanding crops. In addition, the circulation of irrigation return flows mobilizes soluble salts stored in soils or geological materials. As a consequence, it implies severe modifications in the dynamics and total amount of soluble salts and nitrogen exported, especially in semi-arid watersheds.

In this study, long-term data collected by the regional authorities was used to assess the effects of irrigation implementation on salinity (using EC as a proxy) and nitrate concentration (NO_3^-) after the transformation of ca. 77 km^2 from rainfed to irrigated agriculture in the Cidacos River (CR) watershed. The results indicate that water quality in the lower reaches of the CR was significantly modified after the diffuse incorporation of IRF. For instance, in the last surface monitoring point in the CR, median EC and NO_3^- increased by ca. 450 µS/cm and 20 mg/L ($p <$ 0.001), respectively, in the period 2012-2018 (irrigated) in comparison with 2001-2008 (pre-irrigation). In contrast, neither EC nor NO_3^- were different in those monitoring station whose contributing watershed did not include transformed area ($p > 0.1$). In addition, the temporal dynamics in the analysed variables shifted from a rainfed land signal typical in the region to an irrigated land signal, and the hydrochemical type of the CR shifted from mixed-to-Na^+-mixed-to-HCO_3^- to mostly Na^+-mixed type, typical of waters affected by irrigation return flows in the region. Groundwater EC and NO_3^- also increased in those wells located within the irrigated area. Although there are great uncertainties in the actual amount of salt and N reaching the CR via irrigation return flows, the expected contribution of waste water spilled into the CR is minor in comparison to other sources, mostly agricultural sources in the case of N. The observed changes have promoted the designation of the lower reaches of the CR as “affected” by NO_3^- pollution, and the whole CR watershed as a Nitrate Vulnerable Zone, with the emergent question about whether irrigation implementation as carried out currently in Spain is against the environmental objectives of the Water Framework Directive.

Despite the valuable information collected, more data is required to adequately assess the effects of irrigation, as not only the concentrations but also the river discharge has been significantly modified in the lower reaches of the river, especially during summer months. Despite preliminary assessments carried out recently, the lack of a gauging station in the lower reaches does not allow for adequate estimations of salts and nitrate-N exports to downstream water bodies, what would complete the picture and allow for a mass balance approach. Nevertheless, the available data and the assessment presented herein may prove useful in the calibration and validation of hydrological models to estimate with a higher confidence the contribution of irrigated agriculture to salts and nutrients exports.

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Table 1. Information regarding the available water quality information in Government of Navarra database (as of October 2018).

Monitoring points	Sampled since	Samples	Code	Distance*
Surface water				
Barásoain	Feb 9th 2012	54	93406000	42 km
Pueyo	Jan 12th 2000	166	93401000	36 km
Tafalla	May 8th 2001	153	93405000	31 km
Olite	Jan 12th 2000	167	93403000	21 km
Beire	Jan 12th 2000	167	93404000	17 km
Traibuenas	Jan 12th 2000	166	93402000	3 km
Groundwater				
Tafalla	Feb 1st 2000	173	25097023	26 km
Olite	Mar 3rd 2009	111	251030003	22 km
Pitillas	Aug 24th 2004	141	251030007	12 km
Murillo-1	May 12th 2009	106	251030017	8 km
Murillo-2	Jun 26th 2013	83	251070008	6 km

* Distance to the point where the Cidacos River discharges into the Aragón River. In the case of groundwater monitoring points, the distance refers to the nearest point of the river.

Table 2. Trends in electrical conductivity (EC, µS cm^{-1} $year^{-1}$) and nitrate concentration (NO_3^-, mg L^{-1} $year^{-1}$) in groundwater monitoring points for the period October 2013 – September 2018.

Monitoring points	EC		NO_3^-	
Tafalla	+2	(p = 0.7)	0	(p = 0.9)
Olite	-4	(p = 0.5)	-2.5	(p = 0.013)
Pitillas	+125	(p = 0.009)	-3.0	(p = 0.016)
Murillo-1	+50	(p < 0.001)	+3.2	(p = 0.001)
Murillo-2	+70	(p < 0.001)	+ 42.6	(p < 0.001)

Table 3. Volume (V), electrical conductivity (EC), total nitrogen (TN) and estimated load of total dissolved solids (TDS Load) and nitrogen (N Load) from wastewater effluents in the Cidacos River watershed (source: NILSA, 2019).

Treatment Plant [served towns]	**V** [hm^3 $year^{-1}$]	**EC** [µS cm^{-1}]	**TN** [mg L^{-1}]	**TDS* Load** [Mg $year^{-1}$]	**N Load** [Mg $year^{-1}$]
Barásoain-Garinoain	0.11	1070	30	94	3.3
Tafalla-Olite	1.84	1510	12.4	2280	27.8
Beire-Pitillas	0.18	1550	18.6	220	3.2

* TDS (mg L^{-1}) estimated from EC (µS cm^{-1}) as: TDS = 0.80 · EC.

Figure 1. Main land uses in the Cidacos River watershed, with emphasis in the irrigated areas existing before and after irrigation implementation in ca. 77 km^2 (blue and green areas, respectively). The surface and groundwater monitoring points used in this study are depicted, along with the Olite gauging station subwatershed (grey contour).

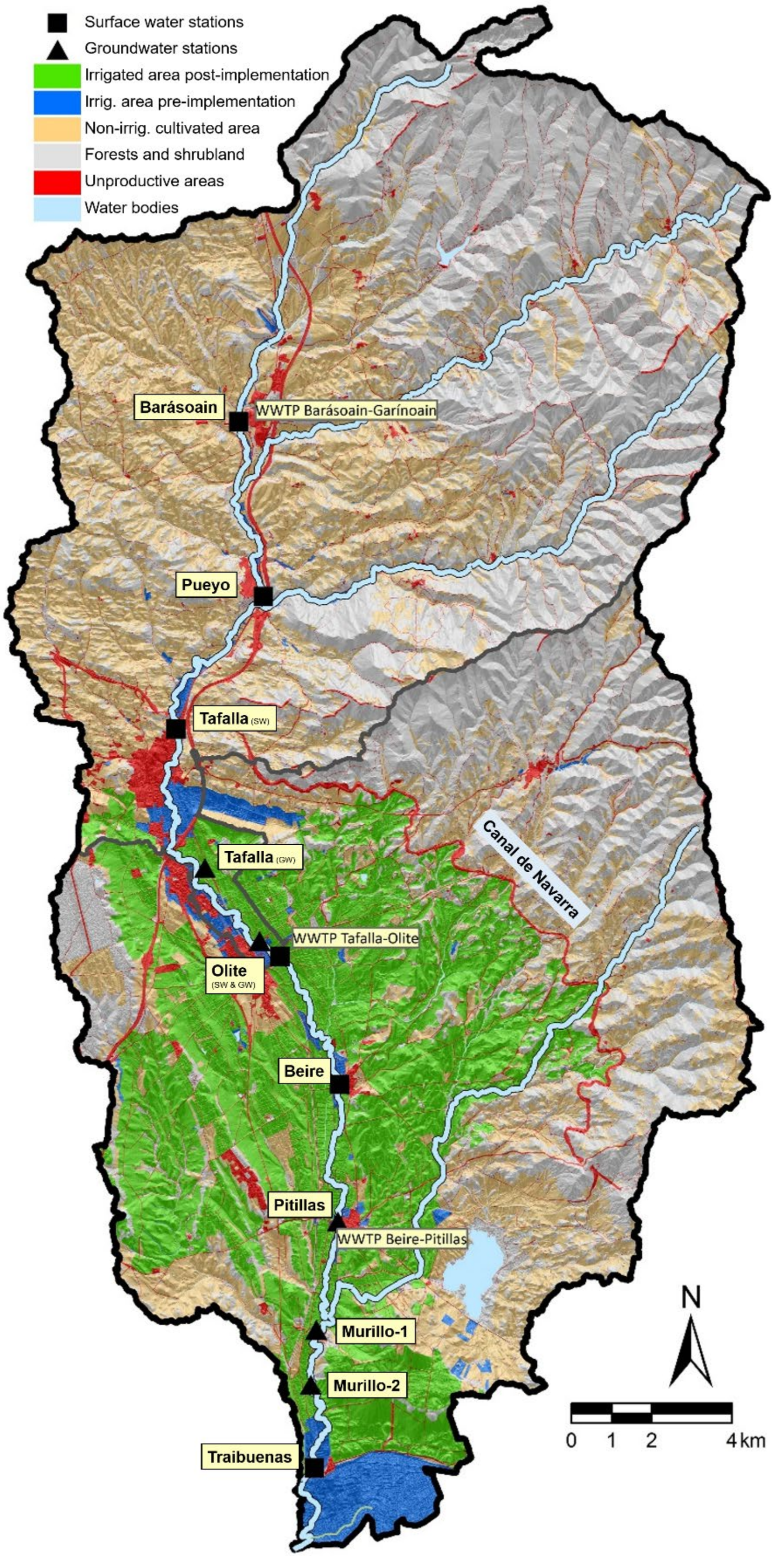

Figure 2. Box-plots showing the electrical conductivity (EC) of water samples collected during the hydrological years 2001-2008 (PRE-irrigation implementation) and 2012-2018 (POST-irrigation implementation) in the available monitoring points according to their distance to the river mouth. The green line indicates the proportion of the watershed at that particular point under irrigated agriculture and so does the dashed blue line for the PRE-period.

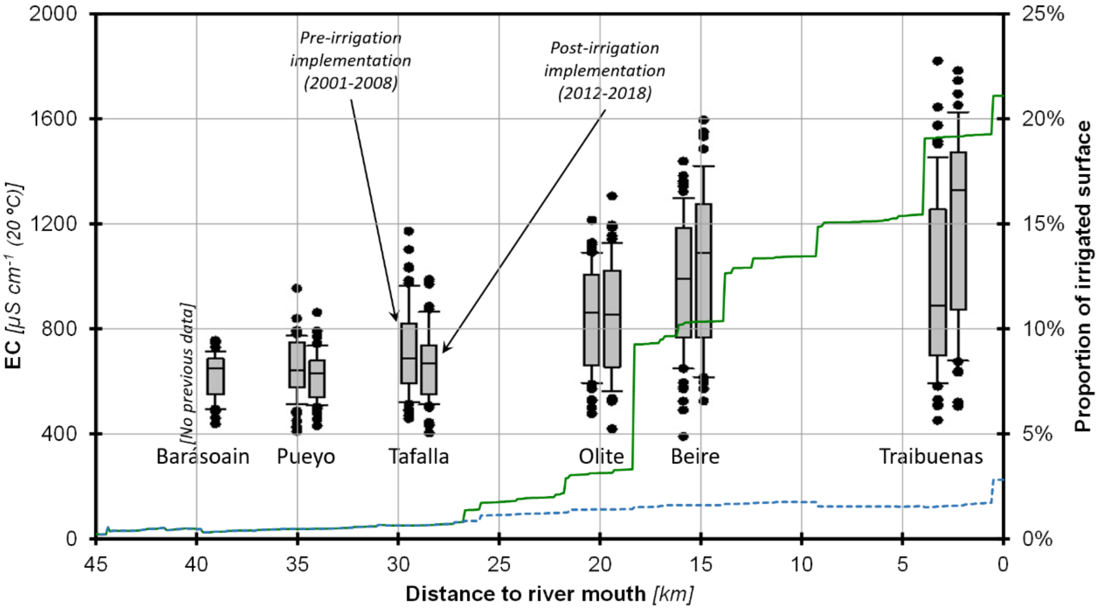

Figure 3. Box-plots showing the nitrate concentration (NO_3^-) of water samples collected during the hydrological years 2001-2008 (PRE-irrigation implementation) and 2012-2018 (POST-irrigation implementation) in the available monitoring points according to their distance to the river mouth. The green line indicates the proportion of the watershed at that particular point under irrigated agriculture and so does the dashed blue line for the PRE-period.

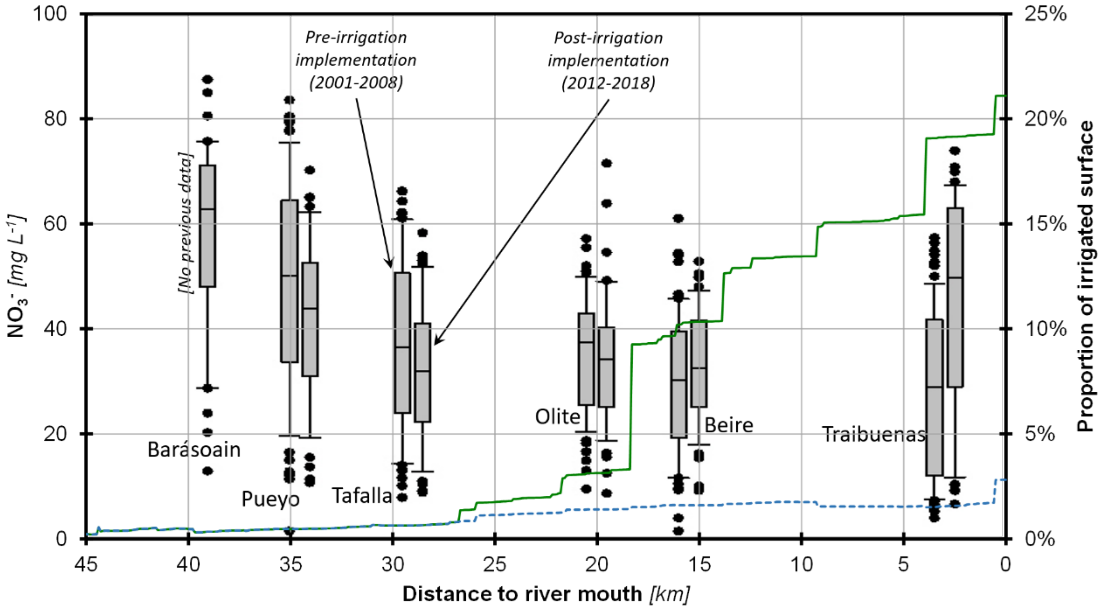

Figure 4. Seasonal dynamics of electrical conductivity (EC, left) and nitrate concentration (NO_3^-, right) in Olite (top) and Traibuenas (bottom) monitoring points during the period 2001-2008 (PRE-irrigation implementation, dashed blue lines) and 2012-2018 (POST-irrigation implementation, green lines). Lines and 95% confidence intervals obtained through LOESS smoothing technique.

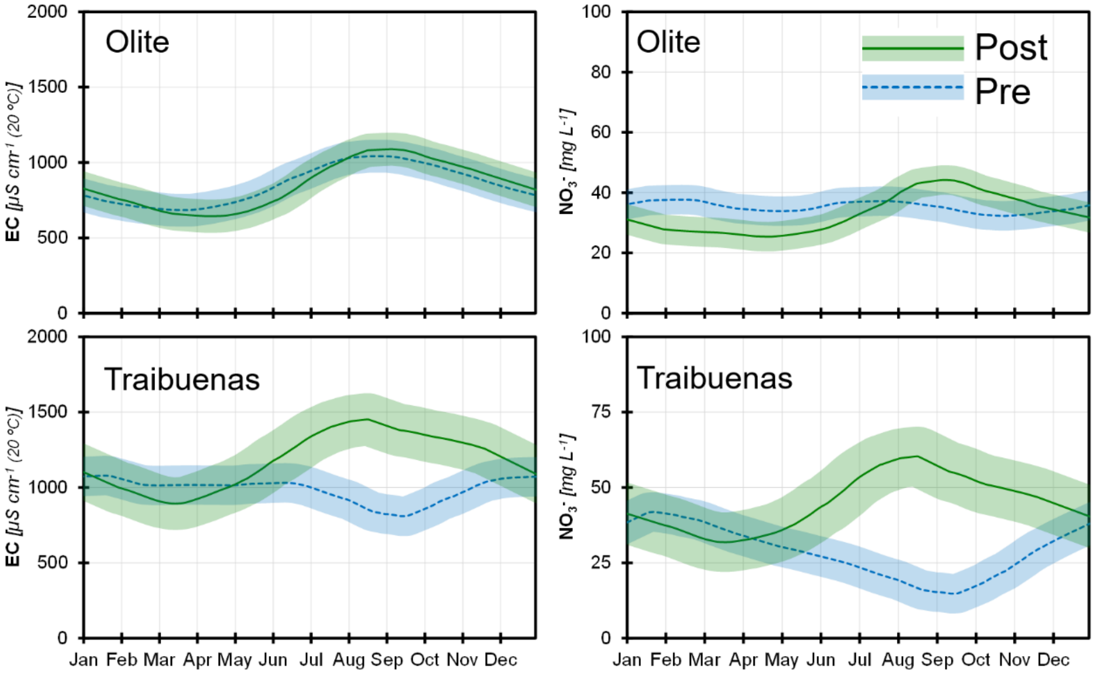

Figure 5. Major constituents of surface water along the Cidacos River main axis in August 31st 2004 (as an example of a pre-irrigation implementation sampling date) and September 25th 2017 (as an example of a post-irrigation implementation sampling date).