agriculture

MDPI

Article

Evaluation of the Impact of Changing from Rainfed to Irrigated Agriculture in a Mediterranean Watershed in Spain

Brian Omondi Oduor [1,*], Miguel Ángel Campo-Bescós [1], Noemí Lana-Renault [2,3], Alberto Alfaro Echarri [4] and Javier Casalí [1]

[1] Department of Engineering, IS-FOOD Institute (Innovation & Sustainable Development in Food Chain), Public University of Navarre, Campus de Arrosadía, 31006 Pamplona, Spain
[2] Department of Human Sciences, University of La Rioja, 26006 Logroño, Spain
[3] Institute for Biodiversity and Ecosystem Dynamics (IBED), University of Amsterdam, 1098 XH Amsterdam, The Netherlands
[4] Navarre Institute of Agricultural and Food Technologies and Infrastructures (INTIA), Avenida Serapio Huici 22, 31610 Villava, Spain
* Correspondence: brianomondi.oduor@unavarra.es; Tel.: +34-644-65-27-62

Abstract: The conversion of cultivated areas from rainfed to irrigated agriculture alters the watershed's hydrology and could affect the water quality and quantity. This study examined how streamflow, nitrate load, and nitrate concentration changed after irrigation implementation in a Mediterranean watershed in Navarre, Spain. The Soil Water Assessment Tool (SWAT) model was applied in the Cidacos River watershed to simulate streamflow and nitrate load under rainfed conditions. The simulated outputs were then compared with the post-irrigation observed values from mid-2017 to 2020 at the watershed outlet in Traibuenas to determine the irrigation impact. The model calibration (2000–2010) and validation (2011–2020) results for streamflow (NSE = 0.82/0.83) and nitrate load (NSE = 0.71/0.68) were satisfactory, indicating the model's suitability for use in the watershed. A comparison of the rainfed and post-irrigation periods showed an average annual increase in streamflow (952.33 m^3 ha^{-1}, +18.8%), nitrate load (68.17 kg ha^{-1}, +62.3%), and nitrate concentration (0.89 mg L^{-1} ha^{-1}, +79%) at the watershed outlet. Irrigation also caused seasonal changes by altering the cropping cycle and increasing the streamflow and nitrate export during the summer and autumn when irrigation was at its peak. The increases in the post-irrigation period were attributed to the added irrigation water for streamflow and increased nitrogen fertilizer application due to changes in cropping for nitrate concentration and export. These findings are useful to farmers and managers in deciding the best nitrate pollution control and management measures to implement. Furthermore, these results could guide future development and expansion of irrigated lands to improve agricultural sustainability.

Keywords: irrigation; nitrate concentration; nitrate load; rainfed cultivation; streamflow; SWAT model

Citation: Oduor, B.O.; Campo-Bescós, M.Á.; Lana-Renault, N.; Echarri, A.A.; Casalí, J. Evaluation of the Impact of Changing from Rainfed to Irrigated Agriculture in a Mediterranean Watershed in Spain. *Agriculture* **2023**, *13*, 106. https://doi.org/10.3390/agriculture13010106

Academic Editor: Yanqun Zhang

Received: 22 November 2022
Revised: 22 December 2022
Accepted: 27 December 2022
Published: 30 December 2022

1. Introduction

Agricultural intensification and increased demand for high-value food production because of market liberalization and population growth have put a lot of pressure on the available water resources and the environment. Agriculture is the largest global freshwater consumer, accounting for more than 70% of the global freshwater resources withdrawals [1,2] and nearly 90% consumptive water use [3]. Irrigation accounts for more than 70% of the agricultural water demand [4]. In the 50 years from 1965 to 2015, the global area under irrigation farming more than doubled [5]. The need for more agricultural production, combined with the effects of climate change, pollution, population growth, and water conflicts, is expected to drive up the demand for irrigation even further [1]. On average, irrigated agricultural productivity per unit of land is more than double that of rainfed cultivation, resulting in increased production intensity and crop diversification [1].

In Spain, irrigation, which covers only 16% of the agricultural land, contributes more than 50% of total agricultural output, six times more than rainfed areas [6]. Over the ten years between 2003 and 2013, the irrigated area in Europe increased by 13.4%. In Spain, the increase was approximately 16% between 2007 and 2017 [6]. Most of the irrigable areas in Europe are mainly found within the Mediterranean region, with Italy and Spain having the largest share of irrigated agricultural lands [7].

Spain's agricultural activities have relied on rainfed and irrigated cultivation, with more focus on irrigation in recent years. According to the Heinrich Böll Foundation [8], rainfed agricultural lands in Spain's dry Mediterranean areas have decreased by 23% over the last 30 years, mainly due to low productivity and inadequate support from the Common Agricultural Policy (CAP). However, the Spanish Ministry of Agriculture reported an increase in irrigated acreage of more than 400,000 ha (which accounted for 16.2% of the irrigated land) over the past decade as of 2018 [6]. Navarre, located in Northern Spain and mostly in the Mediterranean region, has experienced a relatively rapid expansion of irrigated lands. The irrigated area in Navarre increased by around 25% between 2000 and 2020, with more pressurized irrigation systems installed in recent years [9]. Irrigation expansion in the Navarre region was accelerated by establishing the "Canal de Navarra" project to convert 59,160 ha into irrigation. Approximately 40% (22,363 ha) of the proposed land has been converted into irrigation within the project's first phase, with the remainder being transformed in the second phase [10]. The lower reaches of our study area, the Cidacos River watershed, is part of the area converted from rainfed cultivation to irrigation under the project's first phase, with approximately 7700 ha of its total cultivated area converted to irrigation.

Previous studies have shown that the conversion from rainfed to irrigated agriculture affects water quality by increasing its salinity [11,12] and nitrate pollution [13–15]. Nitrate pollution leads to eutrophication, which endangers water quality for human consumption and the environment [16,17]. Although factors such as cultivation, livestock farming, and aquaculture may contribute to an increase in nitrate pollution in agricultural areas [5,18,19], the introduction of irrigation through agricultural intensification would result in higher nitrate loads and yields in such areas. In Spain, for example, flood irrigated areas have reported nitrate yield values exceeding 100 N kg ha^{-1} yr^{-1} [20,21]; pressurized irrigation systems have reported values ranging from 20 to 70 N kg ha^{-1} yr^{-1} [14,22–25]; and rainfed agricultural areas tend to report lower nitrate levels with values ranging from 16 to 37 N kg ha^{-1} yr^{-1} [14,18,25]. Other European countries, such as Sweden and Estonia, have also recorded lower nitrate yields in rainfed areas, ranging from 6 to 32 N kg ha^{-1} yr^{-1} and 10 to 40 N kg ha^{-1} yr^{-1}, respectively [26,27]. Irrigation is generally implemented in arid and semi-arid environments where the nitrate load under rainfed agriculture is usually lower. Hence, a change from rainfed to irrigated agriculture in these areas is likely to increase the nitrate load export from a watershed; thus, estimating their quantities is essential to determine the potential impacts. The introduction of irrigation also affects the hydrology of the irrigated areas. This includes the surface and groundwater by increasing the flows and recharging the groundwater aquifer, particularly when irrigation water is obtained outside the watershed [4].

There has been limited research comparing the hydrological behavior and quality of return flows in agricultural areas before and after irrigation implementation. However, such information is important because shifting from rainfed to irrigated agriculture can change the water regime and increase the concentrations and exports of agrochemicals, which are very harmful to the environment. This study expands on the baseline study by Merchán et al. [14], which found an increase in the salt and nitrate concentrations in the Cidacos River's lower reaches, where irrigation has been implemented for the past decade. However, no information was provided about the irrigation's impact on streamflow and nitrate exportation. This was primarily due to the lack of observed streamflow data before the irrigation period (streamflow measurement in the irrigated section began in June 2017), making it impossible to understand the streamflow patterns before this period and calculate the nitrate export. This analysis is essential in the current context of the increasing scarcity of water resources and growing concern about the contamination of aquifers and surface waters with nitrates and other substances, mainly from agriculture. Furthermore, even when there are relatively long data series of the behavior before and after irrigation, the comparison is not entirely accurate because the response of the rainfed period is not compared to that of the same period and climatic conditions in irrigation. The latter can only be possible by simulating the rainfed scenario and comparing it with the same period after irrigation.

Therefore, this study aimed to use the Soil Water Assessment Tool (SWAT) model to simulate and understand the behavior of the Cidacos River in the irrigated area from mid-2017 to 2020 before irrigation implementation and then compare those simulated results (rainfed condition) with the measured values (post-irrigation). The findings from this study contribute critical information to the implementation of the European Communities' Nitrate Directive (ND, Directive 91/676/EEC) and the Water Framework Directive (WFD, Directive 2000/60/EC), both of which are concerned with protecting water bodies against nitrate pollution from agricultural areas [28,29].

2. Materials and Methods

2.1. Study Area Description

The Cidacos River is a tributary of the Aragón River, which is one of the tributaries of the Ebro River. The study area is located between latitudes 42°69′ and 42°34′ North and longitudes 1°72′ and 1°47′ West, about 15 km south from the city of Pamplona, the capital of the Chartered Community of Navarre in Spain. The Cidacos River flows from north to south for about 44 km and drains a 477 km^2 watershed area (Figure 1a). The watershed's headwater is somewhat mountainous, with high altitudes of slightly more than 1100 m above sea level in the north, but then crosses down to slightly uneven to low terrain of approximately 300 m above sea level in the south at the river's mouth in Traibuenas where it joins with the Aragón River. The watershed's climate is humid to dry temperate, mild Mediterranean, with cold winters (average daily temperatures of 4.7 °C to 5.4 °C in January) and high levels of summer aridity (average daily temperatures of 21.2 °C to 23.7 °C in August) that varies spatially from North to South. The annual average temperatures range from 12.2 °C to 14.2 °C (North to South). The watershed receives annual precipitation ranging from 800 mm in the north to 400 mm in the south, and it's characterized by a strong inter-annual and seasonal irregularity. The wettest months are April and May, while the driest are July and August [14]. The annual evapotranspiration rate is approximately 1150 mm yr^{-1}, with nearly 76% occurring between April and September.

Agriculture is the most dominant land use (Figure 1b) in the watershed, covering over 60% of the total area. The other land uses in the watershed include forests (20%), pasture, and bushlands (15%), and the remaining 5% is made up of urban land, bare land, and water bodies. Rainfed agriculture (which covers approximately 260 km^2) is practiced upstream of the watershed until Olite station, whereas irrigated agriculture (which covers about 77 km^2) is practiced primarily downstream of the watershed, stretching from Olite town up to the river's mouth in Traibuenas. Before implementing the "Canal de Navarra" irrigation project, less than 5 km^2 of the watershed was irrigated (traditional flood irrigation). The irrigated area obtains its water from the Navarre canal, which flows from the Itoiz Reservoir on the Irati River, located about 70 km north of the study area. The conversion from rainfed cultivation to irrigation within the study area has been gradual, with most of the changes occurring between 2009 and 2012. By 2013, approximately 90% of the current irrigated area had been converted from rainfed to irrigation. The main irrigation method used in the study area is pressurized (sprinkler) (71%) with buried fixed sprinkler systems. Drip irrigation (20.3%) and pivot/canon irrigation (8.5%) are two other types of irrigation used in the area [30].

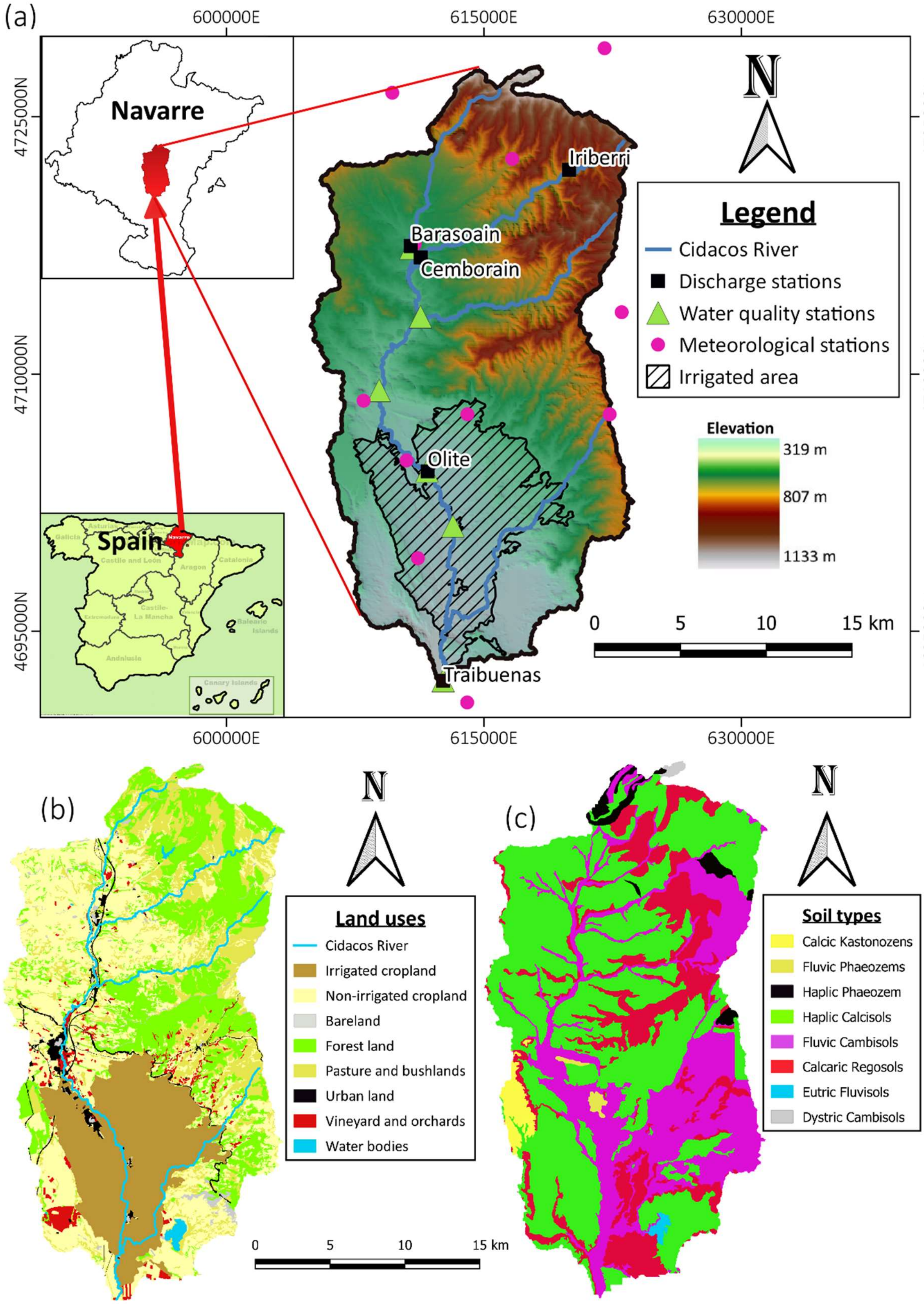

Figure 1. The Cidacos River watershed (**a**) location, elevation, and stations; (**b**) land use map, and (**c**) soil map.

Winter cereals (mainly wheat and barley) and vineyards (orchards) are the main crops grown in the rainfed cultivated area. Corn, tomatoes, and potatoes are among the crops grown in the irrigated area. The annual average fertilization rates in the rainfed area range from 80 to 130 kg N ha^{-1} for winter cereals and 40 to 50 kg N ha^{-1} for vineyards. Crop diversity is greater in the irrigated area, with crops such as corn, peas, grasses, and vegetables. Consequently, fertilizer applications have increased to meet the increased production expectations. The annual average fertilization rates in the irrigated region are 260 kg N ha^{-1} for corn and 120 kg N ha^{-1} for tomatoes and potatoes. Table 1 shows the annual fertilization rates and cropping cycle for each crop in the study area. Details about the area of cultivated crops, irrigation water consumption, and crop yield are shown in the supplementary material (Table S1).

Table 1. Crop types, cropping cycle, and fertilization schedules in the Cidacos River watershed.

Type of Crop	Cropping Cycle	Tillage Date	Fertilization Dates	Annual Fertilization (N kg yr^{-1} ha^{-1})	Type of Fertilizer Applied *
Wheat	1 Nov–1 Jul	01 Oct	1 Oct	40	9-23-30
			01 Jan	60	Urea + Ammonium sulfate
			01 Mar	100	Urea
Winter barley	1 Nov–1 Jul	01 Oct	01 Oct	40	9-23-30
			01 Jan	60	Urea + Ammonium sulfate
			01 Mar	100	Urea
Corn	1 May–1 Nov	01 Apr	15 Apr	40	9-23-30
			15 Jun	260	Urea
Tomato	10 May–15 Sept	01 Apr	15 Apr	60	9-23-30
			15 Jun	120	8-4-10
Potato	1 May–15 Sept	01 Apr	15 Apr	60	9-23-30
			15 Jun	120	NAC 27%

* 9-23-30 contains 9% nitrogen (N), 23% phosphorous (P), and 30% potassium (K) (typically used for plants with high P and K requirements); Urea + ammonium sulfate fertilizer has a nitrogen content of 38% and a sulfate content of 18.75%; urea has a nitrogen content of 46%; 8-4-10 is composed of 8% nitrogen, 4% phosphorous, and 10% potassium (commonly used as a top dressing in crops that require an additional boost of N and K, and in soils with minor P deficiencies); NAC 27% is composed of calcium ammonium nitrate, which contains 27% nitrogen.

The watershed is mainly covered by red mudstone and sandstone clays. The most common soil textures in the watershed are loam and clay-loam soils, which are found in most agricultural areas, whereas loamy-sand and sandy-loam soils are commonly found on eroded hillslopes. The soil types in the watershed were classified using the FAO classification system (Figure 1c). The watershed's predominant soils are Haplic Calcisols (51.6%), Fluvic Cambisols (26.1%), which are mostly found along the river network path, and Calcaric Regosols (18%). Other soils found in the watershed are Haplic Phaeozem (1.7%), Calcic Kastanozems (1.6%), Fluvic Phaeozem (0.4%), Eutric Fluvisols (0.3%), and Dystric Cambisols (0.2%).

2.2. Data Acquisition and Processing

Most of the data used in this study were obtained from the Government of Navarre agencies and websites, as shown in Table 2. The spatial data were projected to the ETRS89 UTM Zone 30N coordinate reference system. The geospatial dataset used for the study included a 25 m resolution Digital Elevation Map (DEM) (Figure 1a), the 2019 Land Use Land Cover (LULC) map (Figure 1b), and the soil type map (1:25,000 scale) (Figure 1c). The climate data were obtained at a daily timestep from 25 meteorological stations located within and around the study area from 1990 to 2020. The high number of weather stations were used to adequately represent the spatial variability of the region's climate and to improve the model's accuracy. The meteorological data included precipitation, maximum and minimum daily temperatures, solar radiation, wind speed, and relative humidity.

Monitoring water quality, specifically nitrate concentration levels, has been ongoing in the watershed since 2000 at various locations (Barasoain, Pueyo, Tafalla, Olite, Beire, and Traibuenas). The nitrate concentration data are collected through a highly scattered sampling frequency once per month on random dates, with some months skipped. The monthly streamflow and nitrate data at the Olite gauging station from 2000 to 2020 were used in the model evaluation. However, streamflow data at the outlet in Traibuenas has only been measured since June 2017. Hence, the observed nitrate load at Traibuenas could only be calculated from mid-2017 to 2020.

Table 2. The SWAT model input data requirement and their sources.

Dataset	Resolution	Source
Digital Elevation Model (DEM)	25 m, ETRS89 UTM Zone 30N projection	Government of Navarre, Spatial Data Infrastructure of Navarre (IDENA), Digital Elevation Model data (https://sitna.navarra.es/geoportal/geop_sitna/geoportal.aspx)
Land Use Map	25 m, 2019 LULC map	Government of Navarre, Spatial Data Infrastructure of Navarre (IDENA), Land Use/Cover data (https://sitna.navarra.es/geoportal/geop_sitna/geoportal.aspx)
Soil Type Map	1:25,000	Government of Navarre, Navarre Spatial Data Infrastructure (IDENA), Soil type data (https://sitna.navarra.es/geoportal/geop_sitna/geoportal.aspx)
Meteorological Data	Daily (1990–2020)	Government of Navarre, Meteorology, and climatology of Navarre website (http://meteo.navarra.es/estaciones/mapadeestaciones.cfm)
Streamflow	Daily (2000–2020)	Government of Navarre, Water in Navarre website (http://www.navarra.es/home_es/Temas/Medio+Ambiente/Agua/Documentacion/DatosHistoricos/) and INTIA (https://www.intiasa.es/)
Water Quality (Nitrate)	Monthly (2000–2020)	Government of Navarre through, Environmental Management of Navarre GAN-NIK (https://gan-nik.es/) and INTIA (https://www.intiasa.es/)
Agricultural Management	Annual	Consultation with the farmers and key stakeholders (INTIA)
Irrigation Data	Monthly (2017–2020)	INTIA reports (https://www.intiasa.es/) and Aguacanal (https://www.aguacanal.es/en/)

The Olite gauging station covers the watershed area under rainfed agriculture, and thus, it was used for the model calibration and validation, whereas the Traibuenas gauging station covers the watershed area under irrigated agriculture. The streamflow, irrigation, and nitrate concentration data at the watershed's outlet in Traibuenas were obtained from the Navarre Institute of Agri-food Technologies and Infrastructures (INTIA). INTIA is the government agency operating and managing all irrigated areas in Navarre. Field interviews and consultation with key informants who are farmers in the watershed were used to obtain agricultural management information (Table 1). The contribution of nitrate pollution from point sources in the study area is negligible, accounting for only about 1.5% of the N loads [14], thus, it was not considered in the modeling.

2.3. The SWAT Model Description

The SWAT model is a freely available open-source software developed by the United States Department of Agriculture's Agricultural Research Service (USDA-ARS). The model assists water resources managers, policy experts, and decision makers in predicting and quantifying land use management's impact on water and diffuse pollution in small and large watersheds with different soil types, land use, and management practices [31]. SWAT is a data-driven, semi-distributed, continuous timescale, physical and process-based hydrological model that simulates water flow, sediments, agricultural chemicals, and pollutant yields.

The SWAT model simulation process divides the watershed into multiple sub-basins, which are sub-divided further into smaller unique homogeneous combinations of similar land use, soil type, and topography characteristics known as the Hydrological Response Units (HRUs) [31]. The model uses HRUs to describe the watershed's spatial heterogeneity and to represent its basic computational unit. The water balance equation is used to simulate the hydrological components of the watershed [31,32]. The Soil Conservation Service (SCS) curve number method [33] is used to estimate surface runoff, while the Muskingum routing method [34] is used to simulate flow routing into the channels. The FAO's Penman-Monteith method is used to calculate the potential evapotranspiration.

Nitrate and nitrogen transportation and transformation are simulated at the HRU level through denitrification, nitrification, mineralization, plant uptake, decay, fertilization, and volatilization processes. The organic and mineral nitrogen cycles are simulated in SWAT by dividing the nutrients in the soil into organic and inorganic parts and component pools, which can increase or decrease depending on the transformation and additions or losses occurring within each pool [35]. The movement and transformation of various forms of nitrogen within a watershed are introduced into the main channel via surface runoff and lateral subsurface flow and transported downstream with the flow [36].

2.4. The SWAT Model Set-Up, Calibration, and Validation

The SWAT model was set up by delineating the watershed, creating HRUs, editing inputs, and running the model. First, the watershed was delineated using a minimum area threshold of 10 km^2 required to create streams, resulting in a 477.02 km^2 watershed area with 23 sub-watersheds. After overlaying the land use and soil type maps with the slope bands, the sub-watersheds were further subdivided into 1404 HRUs. A 5% threshold for land use, soil type, and slope was used to eliminate smaller HRUs that did not meet this threshold, thus improving the model performance. The SWAT editor was then updated with meteorological data and agricultural management information (fertilizer application). Finally, the model was run at daily timesteps with simulations from 1990 to 2020, with the first 10 years (1990–1999) serving as the model warm-up period and the remaining 21 years (2000–2020) for the model evaluation, divided into calibration (2000–2010) and validation (2011–2020). The simulation outputs were extracted at a monthly timestep.

After the initial model run, the outputs were transferred to the SWAT Calibration and Uncertainty Programs (SWATCUP), where parameterization, sensitivity analysis, calibration, and validation were performed using the multi-site Sequential Uncertainty Fitting, version 2 (SUFI-2) [37]. The sensitivity of the parameters was established through the global sensitivity analysis after initially running the model 500 times. The most sensitive streamflow and nitrate load parameters were identified and used for the model calibration and validation. The parameter ranges were adjusted after each calibration iteration until most of the observed data were bracketed within the 95 Percent Prediction Uncertainty (95PPU) band [38]. The streamflow parameters were first calibrated and then set constant before calibrating the nitrate load parameters.

The model was calibrated and validated using monthly streamflow and nitrate load observations at the Olite gauging station. The statistical model evaluation techniques recommended by Moriasi et al. (2007) were used to compare the observed and simulated results. The four statistical indicators used to evaluate the model in this study were the Coefficient of Determination (R^2), Nash–Sutcliffe Efficiency (NSE), Root Mean Square Error (RMSE) to Observations Standard Deviation Ratio (RSR), and Percent Bias (PBIAS). The model was considered calibrated satisfactorily when the values of $R^2 > 0.5$, NSE > 0.5, RSR ≤ 0.7, and PBIAS $\pm$ 25% for streamflow and PBIAS $\pm$ 55% for nitrate loads [38–41]. After evaluating the model performance satisfactorily, the simulations at the watershed's outlet in Traibuenas under rainfed conditions were extracted from mid-2017 to 2020. These simulated results (rainfed) were compared to the measured observation (irrigated) at Traibuenas to determine the impact of irrigation implementation. The Wilcoxon–Mann–Whitney Rank-Sum Test [42] was used to test the statistical significance of the medians for the different periods.

2.5. Irrigation Impact Assessment

The impact of the change from rainfed to irrigated agriculture for the watershed was assessed at the outlet in Traibuenas using an irrigation impact index that was established by calculating the ratio of the change (in streamflow, nitrate load, and nitrate concentration) in the post-irrigation (observed/irrigated) and pre-irrigation (simulated/rainfed) for the downstream and upstream sections located at Traibuenas and Olite, respectively to the area converted to irrigation as follows:

$$\mathrm{III_i} = \frac{\Delta \mathrm{Post}_{(\mathrm{ds-us})\mathrm{i}} - \Delta \mathrm{Pre}_{(\mathrm{ds-us})\mathrm{i}}}{\Delta \mathrm{IA}} \tag{1}$$

where $\mathrm{III_i}$ represents the irrigation impact indices for streamflow ($m^3\ ha^{-1}$), nitrate load (kg ha^{-1}), and nitrate concentration (mg L^{-1} ha^{-1}) for the period considered; $\Delta \mathrm{Post}_{(\mathrm{ds-us})\mathrm{i}}$ represents the change in the post-irrigation values between downstream (Traibuenas) and upstream (Olite) sections for each of the variables; $\Delta \mathrm{Pre}_{(\mathrm{ds-us})\mathrm{i}}$ represents the change in the pre-irrigation values between downstream (Traibuenas) and upstream (Olite) sections for each variable; ΔIA is the change in the irrigated area in hectares.

These indices helped calculate the annual rate of change per unit area, which could be used to estimate similar changes in the watershed as well as compare different watersheds. The variation was computed as the percentage of the average annual change for each variable at the watershed outlet.

3. Results and Discussion

3.1. Model Evaluation

3.1.1. Sensitivity Analysis

The most influential parameters for the model's calibration and validation were identified using a global sensitivity analysis. The curve number, soil evaporation factor, and groundwater delay time were the most sensitive streamflow calibration parameters, while the denitrification factor and nitrate percolation coefficient were the most sensitive nitrate calibration parameters. Table 3 shows the ranges of selected sensitive parameters during streamflow and nitrate load calibration. The curve number is an important parameter for the watershed's hydrology because it directly influences the surface runoff and infiltration rate. Since the initial model underestimated the baseflow and overestimated runoff, the default curve number parameter values in each HRU were reduced by 12%, resulting in slightly reduced surface runoff and increased infiltration. The evaporation factor was sensitive because agricultural areas in the Mediterranean regions have high evapotranspiration rates. The model's evaporation generation capacity increased by lowering the default parameter value, thus appropriately representing the watershed's evaporative demand [43]. Similar sensitivity analysis findings have been obtained by other researchers in the Mediterranean catchments [43–46]. The greatest influence on nitrate load was the amount of fertilizer lost to denitrification. Denitrification losses are higher in areas with high moisture content than in dry regions; thus, its parameter value was set very low due to the watershed's Mediterranean climatic conditions. The nitrate percolation parameter governs how much nitrate is removed by surface runoff relative to the amount percolated. Typically, the default value ranges from 0.01 to 1; a lower value closer to 0 means that all of the nitrate is percolated and not in the surface runoff, while a percolation coefficient of 1 indicates that the surface runoff has the same nitrate content as percolation [32]. Because of the high nitrate concentration levels in the groundwater measurements, which indicate a high watershed nitrate percolation rate, this parameter was set relatively low in the model.

Table 3. The most sensitive parameters during streamflow and nitrate load calibration.

Parameter	Description	Change Method *	Parameter Adjustment Values		
			Min. Value	Max. Value	Fitted Value
CN2.mgt	Initial SCS runoff CN number for moisture condition II	R	−0.2	0.20	−0.12
ESCO.hru	Soil Evaporation compensation factor	R	−0.40	−0.28	−0.31
GW_DELAY.gw	Groundwater delays (days)	V	20	80	53.54
CDN.bsn	Denitrification exponential rate coefficient	V	0	1.62	0.04
NPERCO	Nitrate Percolation coefficient	V	0.01	1	0.17

* R is a relative change method that multiplies the existing value with (1 + fitted value), whereas V replaces the existing value with the fitted value.

3.1.2. Calibration and Validation

The model simulation results at monthly timestep during the calibration and validation of streamflow and nitrate loads are shown in Figures 2 and 3. The magnitude and temporal dynamics of the discharge and nitrate load peaks and lows were captured satisfactorily, as evidenced by their hydrographs. Furthermore, the statistical evaluation results for streamflow and nitrate load were very good during the calibration and validation, respectively, as detailed in Table 4. Given the numerous uncertainties associated with modeling, these findings demonstrated good accountability of the various model input data and agricultural management practices used.

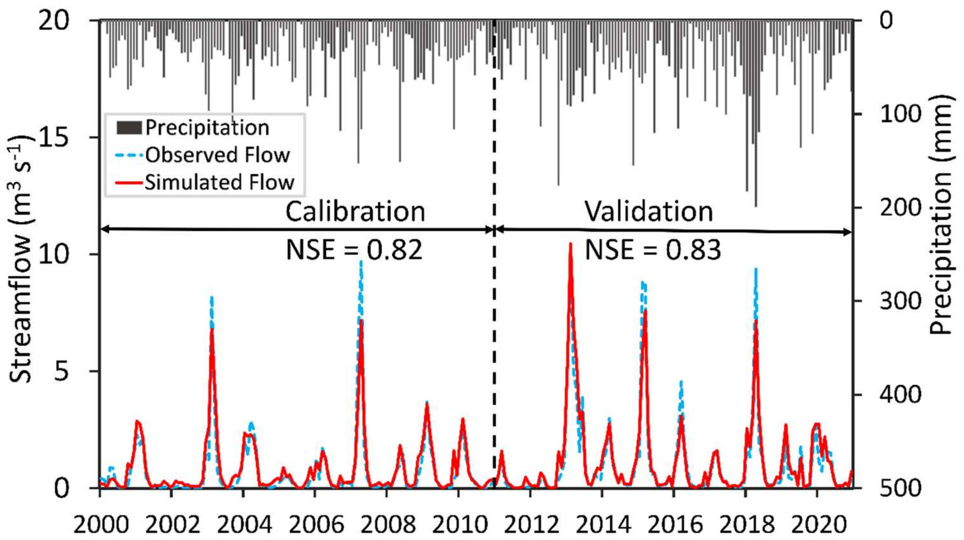

Figure 2. Observed and simulated average monthly streamflow and precipitation at the Olite gauging station from 2000 to 2020.

The model uncertainties were accounted for by 95PPU, represented by the r-factor and p-factor. The r-factor represents the ratio of the average distance between the 95PPU band by the standard deviation of the observed data whereas the p-factor represents the percentage of observed data bracketed within the 95PPU band [38]. More than 60% of the streamflow and nitrate load data for the entire simulation period were bracketed within the 95PPU band. Therefore, these results were acceptable as they fell within the normal range suggested by Abbaspour et al. [38]. The negative PBIAS values indicated an overall slight overestimation of the streamflow and nitrate load, although the values were within the acceptable range. Uncertainties in the model output may be caused by uncertainties in the input, measured data, or the model itself. As reported by other similar studies [47–51], these uncertainties could range from missing precipitation data, reclassification of the soil and land use maps, and insufficient measured data used for calibration.

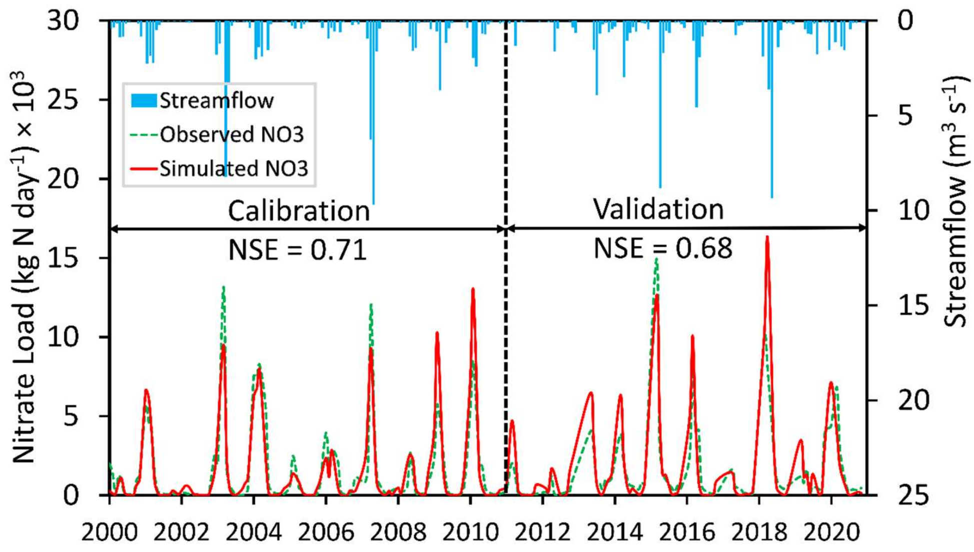

Figure 3. Observed and simulated average monthly nitrate load and streamflow at the Olite gauging station from 2000 to 2020.

Table 4. The SWAT model performance metrics during the calibration and validation of streamflow and nitrate loads.

Performance Indicator	Streamflow		Nitrate Load	
	Calibration	Validation	Calibration	Validation
p-factor	0.56	0.65	0.72	0.63
r-factor	0.70	0.67	0.92	0.98
NSE	0.82	0.83	0.71	0.68
R^2	0.83	0.84	0.72	0.79
PBIAS	−8.7%	−5.6%	−9.2%	−7%
RSR	0.42	0.42	0.54	0.56

The streamflow and nitrate export distribution patterns over the simulation period (2000–2020) were characterized by interannual and seasonal variability (Figures 2 and 3). Wetter years with more precipitation had higher streamflow and, consequently, nitrate export, and vice versa. The seasonal distribution followed a similar pattern, with higher streamflow and nitrate export in winter and spring than in summer and autumn. Summer had the lowest streamflow due to high evapotranspiration and little or no precipitation, limiting cultivation to almost zero. As a result, the nitrate inputs (fertilizer) were reduced, as were the loads. Nitrate dynamics in agricultural watersheds are primarily governed by the fate and transportation of fertilizer in the soil, organic matter decomposition, and the prevailing climate [40,52].

3.2. Irrigation Dynamics in the Watershed

The conversion of agricultural land from rainfed to irrigation in the study area began in late 2006, with nearly 70% of the changes occurring between 2009 and 2012. By 2020, at least 16% of the watershed had been converted into irrigated land. This study evaluated the conversion from rainfed to irrigation using the available data (mid-2017 to 2020). The seasonal irrigation patterns show that in winter, irrigation is minimal (only 1%), while in the summer, irrigation water applications are high (57%) (Figure 4). Irrigation was mostly carried out during periods of low precipitation, especially from July to September (Figure 4a). Similar seasonal irrigation patterns have been observed in other semi-arid irrigated watersheds within the Ebro basin [24,53–55] and around the world [56].

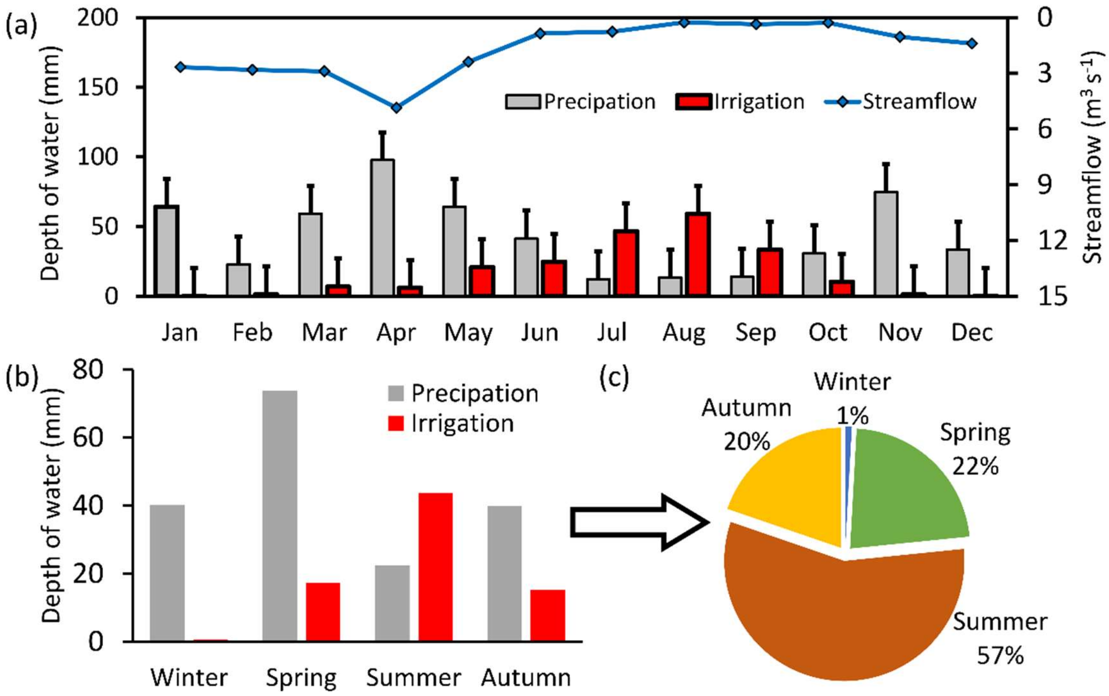

Figure 4. (**a**) Monthly average precipitation, irrigation, and streamflow distribution at the watershed outlet in Traibuenas from mid-2017 to 2020, (**b**) seasonal precipitation and irrigation distribution pattern, and (**c**) the percentage of irrigation water applied each season.

According to the most recent INTIA reports [30,57,58], the average water inflow into the irrigated section of the watershed was 51% precipitation, 31% river inflow from the Olite gauging station, and 17% irrigation water from the Navarre canal. In 2020, evapotranspiration accounted for 33.5% of output, groundwater storage accounted for 3.5%, and the outflow at the Traibuenas gauging station accounted for 42%. The irrigation performance efficiency in the study area was relatively high (84.6%), indicating a well-managed irrigation system. However, there is a spatial variation, with some irrigated plots having lower efficiencies than others, which are compensated for by the higher ones. The irrigation efficiency value was slightly higher than the figures reported by other researchers in the Ebro basin, such as 76% [53] and 72% [59] in watersheds with predominantly sprinkler irrigation.

3.3. Observed Nitrate Concentration Dynamics

Nitrate concentrations at the watershed outlet in Traibuenas have been monitored since 2000. The average annual nitrate concentration distribution pattern from 2000 to 2020 is depicted in Figure 5. During the pre-irrigation period (2000–2008), the average nitrate concentration at the Traibuenas gauging station was 27.72 mg L^{-1} (median value of 27.49 mg L^{-1}; interquartile range (IQR): 34.77 mg L^{-1} to 23.77 mg L^{-1}) with maximum and minimum concentrations of 57.20 mg L^{-1} and 4 mg L^{-1}, respectively. The monthly median values ranged from 45.64 mg L^{-1} in January to 11.33 mg L^{-1} in September. For the pre-irrigation period, 80 nitrate concentration samples were analyzed, with only 6.3% (5 samples) exceeding the 50 mg L^{-1} threshold recommended by Nitrate Directives and 45% (36 samples) falling below the 25 mg L^{-1} for unaffected waters. The nitrate concentration varied between the years, with the lowest value recorded in 2002 due to a severe drought, resulting in limited cultivation, and the highest in 2007, due to abundant precipitation and, thus, increased cultivation. The seasonal cycles were not consistent for all the years during the pre-irrigation period, with high nitrate concentrations in winter and spring and low concentrations in summer and autumn. The temporal fluctuation in nitrate concentration before the irrigation implementation was related to the precipitation distribution pattern for each season in a specific year. These findings are consistent with those made by Orellana-Macías et al. [60] in a study of nitrate vulnerable zones evolution in the north-east of Spain and Hernández-García et al. [61] in a small rainfed experimental watershed in Navarre, where they both observed an increase in nitrate concentration during the years with more precipitation compared to years with less due to increased cultivation and subsequent crop fertilization. During the transition period (2009–2012), the nitrate concentration significantly declined because this was when the irrigation infrastructure was being constructed; thus, most of the agricultural land was left uncultivated except for the upstream area which was unaffected by this development.

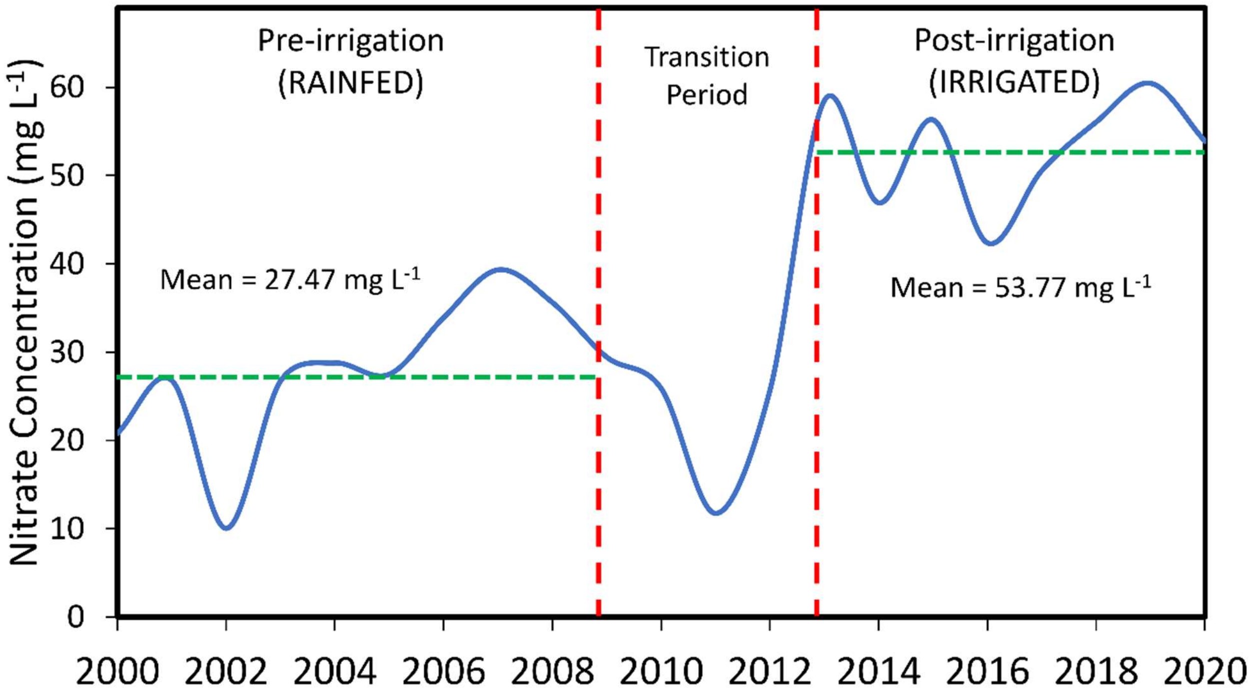

Figure 5. The average annual nitrate concentration distribution pattern at the watershed outlet in Traibuenas before and after irrigation implementation from 2000 to 2020.

During the post-irrigation period (2013–2020), 71 nitrate concentration samples were analyzed. The nitrate concentration in the post-irrigation period was twice as high as in the pre-irrigation period (Figure 5), with a mean of 53.14 mg L^{-1} (median value of 54.99 mg L^{-1}; IQR: 57.95 mg L^{-1} to 47.83 mg L^{-1}) and maximum and minimum concentrations of 97.10 mg L^{-1} and 16.04 mg $L^{-1,}$ respectively. The monthly median concentration values ranged from 73.70 mg L^{-1} in September to 31.9 mg L^{-1} in March. Hernández-García et al. [61] obtained similar results when comparing the nitrate concentration levels in rainfed and irrigated experimental watersheds in Navarre, with the findings indicating a threefold higher nitrate concentration in the irrigated watershed than in the rainfed one. The collected samples exceeded the recommended Nitrate Directive threshold of 50 mg L^{-1} in 56.3% (40 samples), indicating that the river was contaminated with nitrate, while only 2.8% (2 samples) fell below the 25 mg L^{-1} level (unaffected waters). The nitrate concentration was significantly higher ($p < 0.01$) during the post-irrigation period than during the pre-irrigation period from May to December. Following the irrigation implementation, the seasonal cycle of nitrate concentrations was substantially altered; the peak concentration shifted from January–February to August–September, and the lowest concentration shifted from September–October to March–April.

The nitrate concentration patterns in the watershed may be related to the cropping practices before and after irrigation. Before irrigation, the main crops grown in the watershed were mostly rainfed winter cereals (wheat and barley), which required less nitrate fertilization. However, following the implementation of irrigation, high-value crops such as tomatoes and corn, which require more nitrogen fertilization, were introduced into the watershed, increasing nitrate concentration levels, particularly during the summer and autumn. Lower nitrate concentration and export levels from rainfed cultivated areas in Navarre have also been reported by other studies [18,61,62]. During the pre-irrigation period, cultivation was mostly carried out during the winter and spring when there was enough precipitation. However, this decreased during the summer and autumn when productivity was low due to lack of precipitation, resulting in lower nitrate concentration. Hernández-García et al. [61] obtained similar seasonal patterns during rainfed conditions in their studies of small experimental watersheds within Navarre with similar characteristics to the Cidacos River watershed. The post-irrigation phase, however, sees year-round cultivation with irrigation supporting farming during the summer and autumn, a period when productivity was previously low.

3.4. Variations in Streamflow and Nitrate (Load and Concentration) due to Irrigation

The irrigation impact index and the average annual variation after irrigation implementation showed a positive response in streamflow, nitrate load, and nitrate concentration. The annual irrigation impact index per unit irrigated area (Equation (1)) shows that irrigation increased the streamflow (952.33 m^3 ha^{-1}, +18.8%), nitrate load (68.17 kg ha^{-1}, +62.3%), and nitrate concentration (0.89 mg L^{-1} ha^{-1}, +79%) at the watershed outlet (Figure 6). These findings are comparable to those obtained by Merchán et al. [63] who reported an increase in streamflow, nitrate load, and nitrate concentration by 23%, 27%, and 8%, respectively in the Lerma catchment within the Ebro basin in Spain after irrigation implementation. However, the variation in exported nitrate load and concentration was slightly higher in this study than in Merchán et al. [63] because the Lerma catchment had higher nitrogen concentration levels before irrigation implementation than the Cidacos River watershed due to different fertilization management practices. Similar annual nitrate exportation rates after irrigation implementation have been reported in Monegros within the Ebro basin at 49 kg ha^{-1} [23] and in La Violada irrigation district in northeast Spain at 66 kg ha^{-1} [20]. Likewise, the reported increases in nitrate concentration values after irrigation of 0.7 to 0.8 mg L^{-1} ha^{-1} in the middle Ebro River basin [64] and 0.91 mg L^{-1} ha^{-1} in the Arba River basin [65] are in close agreement with our findings.

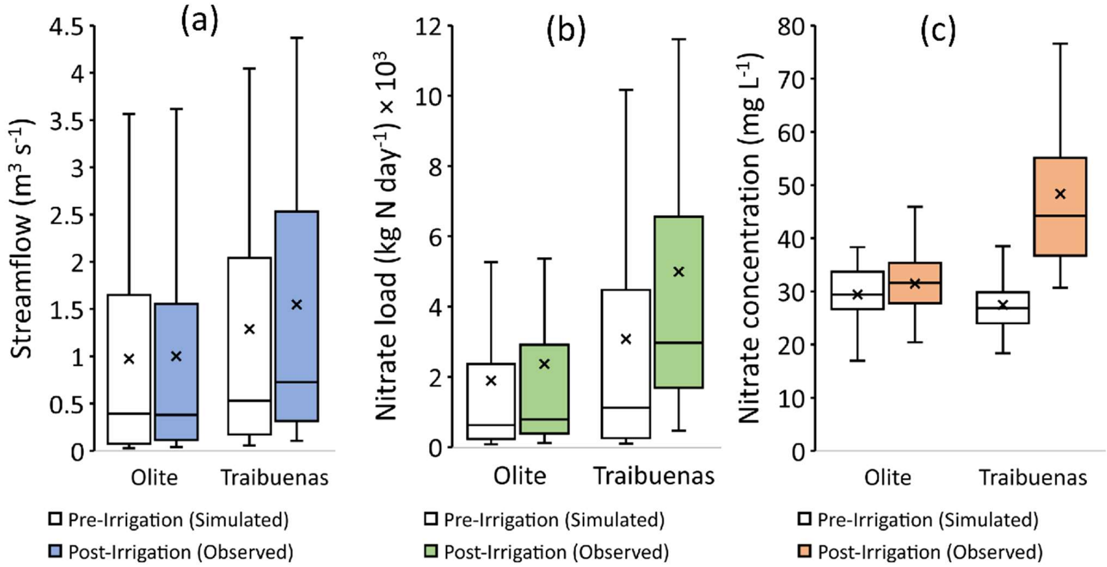

Figure 6. Comparison of average annual changes in (**a**) streamflow, (**b**) nitrate load, and (**c**) nitrate concentration before and after irrigation at Olite and Traibuenas stations from mid-2017 to 2020.

The increased streamflow in the post-irrigation period was consistent with the addition of irrigation water from outside the watershed via the Navarre canal. The irrigation impact on streamflow was more pronounced in the summer and autumn compared to winter and spring, resulting in changes in the watershed's hydrological behavior (Figure 7). More research into the effects of these changes on flora and fauna is needed in the future to understand their impacts.

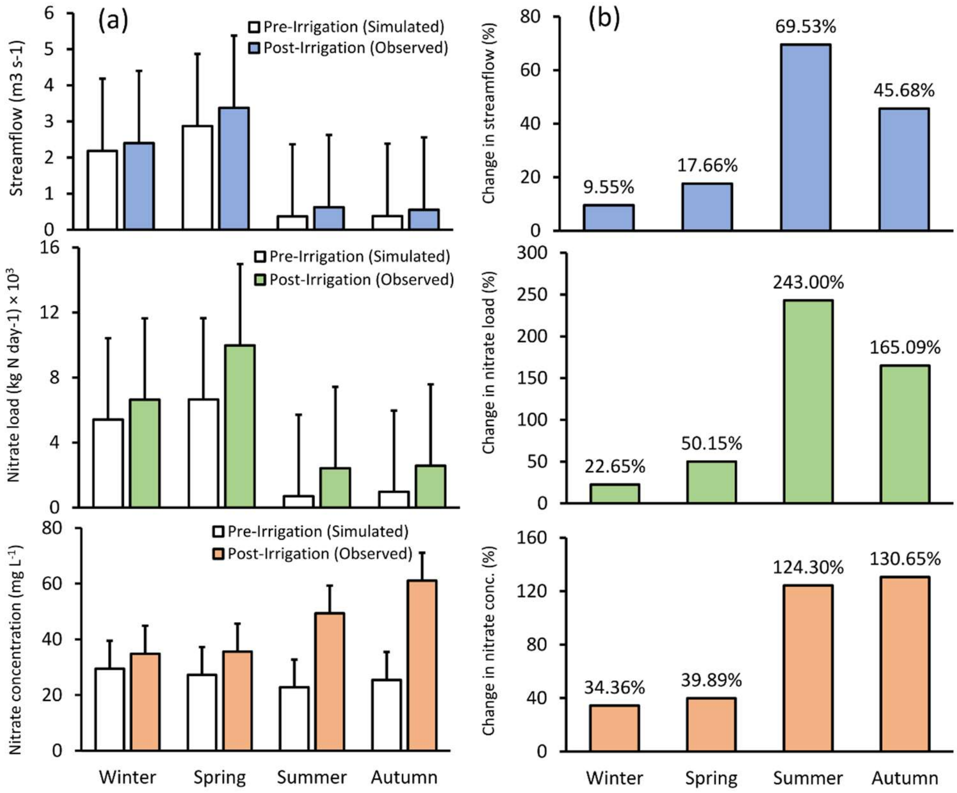

Figure 7. Seasonal comparison of (**a**) pre-irrigation and post-irrigation results and (**b**) the percentage changes after irrigation implementation for streamflow, nitrate load, and concentration at the Traibuenas gauging stations from mid-2017 to 2020.

The changes in nitrate (load and concentration) in the post-irrigation period were attributed to increased nitrogen fertilizer application resulting from cultivating high-value crops (with high nitrogen fertilizer demand) to boost productivity due to irrigation. Furthermore, the introduction of irrigation has resulted in a shift in cropping cycles because crops can now receive water throughout the year, with rainfall primarily supporting agriculture in the winter and spring and irrigation in the summer and autumn. The concentration and exported nitrate were comparatively higher in the summer and autumn (from May to October) due to nitrate mobilization resulting from irrigation, increased fertilizer application during that period, and low streamflow despite the irrigation water contributions. The highest nitrate concentration in the post-irrigation period was observed from August to October, which could be influenced by the top-dressing fertilization [24,63,64].

The increase in exported nitrate load during the summer was very high (243%) due to increases in both streamflow (70%) and nitrate concentration (124%) in the same period (Figure 7). The exported nitrate load is directly influenced by streamflow and nitrate concentration, whereby a slight increase in streamflow produces a greater change in the exported nitrate load than a slight increase in the nitrate concentration. This effect of increased flow on the exported nitrate loads has been reported in other studies in irrigated areas [20,63]. Given the importance of nitrate exportation, some studies [24,63,64,66] have proposed its adoption in agricultural management decisions for nitrogen impact assessment rather than relying solely on the Nitrate Directive's nitrate concentration thresholds.

4. Conclusions

This paper examined the impact of changing from rainfed to irrigated agriculture on streamflow, nitrate load, and nitrate concentration in a Mediterranean watershed in northern Spain by simulating the rainfed conditions using the SWAT model and compared them to the current post-irrigation period. The results indicate a significant increase in the annual streamflow, nitrate load, and nitrate concentration at the watershed outlet in the post-irrigation period. Higher irrigation impact was observed during summer and autumn when irrigation was at its peak than in winter and spring. The increase in streamflow was explained by additional water coming from irrigation, whereas the increase in nitrate export and concentration was attributed to increased fertilization from the cultivation of high nitrogen consuming crops. The implementation of irrigation and subsequent agricultural intensification resulted in changing cropping patterns and doubling of nitrate concentrations at the outlet, exceeding the Nitrate Directive thresholds recommended by the European Commission. Therefore, nitrate minimization practices such as efficient nitrogen fertilizer application and the creation of nitrogen buffer zones along the river's riparian zone should be considered to control nitrate exportation and pollution from cultivated lands into the river. Despite this study's valuable and significant findings, more data are needed to further analyze and assess the impact of irrigation especially during summer and autumn, which was modified following irrigation. The methodology and findings from this study can be applied to other areas with similar conditions, allowing a more comprehensive assessment of the effect of changing from rainfed to irrigated agriculture on streamflow and nitrate pollution. These findings could assist farmers, water experts, and policy/decision makers in improving water resources management at the watershed level and be useful in guiding the development of new irrigation systems, thereby improving sustainable agriculture.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/agriculture13010106/s1, Table S1: The area of cultivated crops, irrigation water consumption, and crop yield.

Author Contributions: B.O.O.: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Resources, Software, Validation, Visualization, Writing—original draft, Writing—review and editing. M.Á.C.-B.: Conceptualization, Project administration, Resources, Funding acquisition, Supervision, Writing—review and editing. N.L.-R.: Supervision, Funding acquisition, Writing—review and editing. A.A.E.: Data acquisition. J.C.: Conceptualization, Project administration, Resources, Funding acquisition, Supervision, Writing—review and editing. All authors have read and agreed to the published version of the manuscript.

Funding: This study was supported by funding from the European Union's H2020 research and innovation program under Marie Sklodowska-Curie grant agreement no. 801586, and from the Ministerio de Economía y Competitividad (Government of Spain) via Research Project CGL2015-64284-C2-1-R and PID2020-112908RB-I00 funded by MCIN/AEI/10.13039/501100011033/FEDER "Una manera de hacer Europa".

Institutional Review Board Statement: Not applicable.

Acknowledgments: The authors are grateful to INTIA for the information on irrigation and all the Government of Navarre agencies that supported the data acquisition for this research.

Conflicts of Interest: The authors declare that they have no known competing financial interests or personal relationships that could have influenced the work reported in this paper.

References

1. World Bank Water in Agriculture. Available online: https://www.worldbank.org/en/topic/water-in-agriculture#1 (accessed on 2 March 2022).
2. FAO. *Water for Sustainable Food and Agriculture*; Food and Agriculture Organization of the United Nations: Rome, Italy, 2017.
3. Siebert, S.; Burke, J.; Faures, J.M.; Frenken, K.; Hoogeveen, J.; Döll, P.; Portmann, F.T. Groundwater Use for Irrigation—A Global Inventory. *Hydrol. Earth Syst. Sci.* **2010**, *14*, 1863–1880. [CrossRef]
4. Zeng, R.; Cai, X. Analyzing Streamflow Changes: Irrigation-Enhanced Interaction between Aquifer and Streamflow in the Republican River Basin. *Hydrol. Earth Syst. Sci.* **2014**, *18*, 493–502. [CrossRef]
5. FAO; IWMI. *More People, More Food, Worse Water? A Global Review on Water Pollution from Agriculture*; Mateo-Sagasta, J., Zadeh, S.M., Turral, H., Eds.; FAO; Rome, Italy, IWMI: Colombo, Sri Lanka, 2018; ISBN 9789251307298.
6. MAPA Gestión Sostenible de Regadíos. Available online: https://www.mapa.gob.es/es/desarrollo-rural/temas/gestion-sostenible-regadios/ (accessed on 24 May 2021).
7. European Parliamentary Research Service (EPRS). *Irrigation in EU Agriculture*; EPRS: Brussels, Belgium, 2019.
8. Heinrich Böll Foundation. *Agriculture Atlas: Facts and Figures on EU Farming Policy*, 1st ed.; The European Commission's LIFE Programme: Brussels, Belgium, 2019.
9. DDRMAAL Estadísticas Agrícolas. Negociado de Estadística. Available online: http://www.navarra.es/home_es/Temas/Ambito+rural/Indicadores/agricultura.htm (accessed on 26 August 2021).
10. Government of Navarre Description of the Navarre Canal. Available online: https://www.canasa.es/proyecto/descripcion-canal-de-navarra (accessed on 10 March 2022).
11. Duncan, R.A.; Bethune, M.G.; Thayalakumaran, T.; Christen, E.W.; McMahon, T.A. Management of Salt Mobilisation in the Irrigated Landscape—A Review of Selected Irrigation Regions. *J. Hydrol.* **2008**, *351*, 238–252. [CrossRef]
12. Pulido-Bosch, A.; Rigol-Sanchez, J.P.; Vallejos, A.; Andreu, J.M.; Ceron, J.C.; Molina-Sanchez, L.; Sola, F. Impacts of Agricultural Irrigation on Groundwater Salinity. *Environ. Earth Sci.* **2018**, *77*, 197. [CrossRef]
13. Muñoz-Carpena, R.; Ritter, A.; Socorro, A.R.; Pérez, N. Nitrogen Evolution and Fate in a Canary Islands (Spain) Sprinkler Fertigated Banana Plot. *Agric. Water Manag.* **2002**, *52*, 93–117. [CrossRef]
14. Merchán, D.; Sanz, L.; Alfaro, A.; Pérez, I.; Goñi, M.; Solsona, F.; Hernández-García, I.; Pérez, C.; Casalí, J. Irrigation Implementation Promotes Increases in Salinity and Nitrate Concentration in the Lower Reaches of the Cidacos River (Navarre, Spain). *Sci. Total Environ.* **2020**, *706*, 135701. [CrossRef]
15. Stamatis, G.; Parpodis, K.; Filintas, A.; Zagana, E. Groundwater Quality, Nitrate Pollution and Irrigation Environmental Management in the Neogene Sediments of an Agricultural Region in Central Thessaly (Greece). *Environ. Earth Sci.* **2011**, *64*, 1081–1105. [CrossRef]
16. WHO. *Guidelines for Drinking-Water Quality: Fourth Edition Incorporating the First Addendum*; World Health Organization: Geneva, Switzerland, 2017; ISBN 9789241549950.
17. Sutton, M.A.; Howard, C.M.; Erisman, J.W. *The European Nitrogen Assessment—Sources, Effects and Policy Perspectives*; Cambridge University Press: Cambridge, UK, 2011.
18. Casalí, J.; Gastesi, R.; Álvarez-Mozos, J.; De Santisteban, L.M.; de Lersundi, J.D.V.; Giménez, R.; Larrañaga, A.; Goñi, M.; Agirre, U.; Campo, M.A.; et al. Runoff, Erosion, and Water Quality of Agricultural Watersheds in Central Navarre (Spain). *Agric. Water Manag.* **2008**, *95*, 1111–1128. [CrossRef]
19. Menció, A.; Mas-Pla, J.; Otero, N.; Regàs, O.; Boy-Roura, M.; Puig, R.; Bach, J.; Domènech, C.; Zamorano, M.; Brusi, D.; et al. Nitrate Pollution of Groundwater; All Right..., but Nothing Else? *Sci. Total Environ.* **2016**, *539*, 241–251. [CrossRef]
20. Barros, R.; Isidoro, D.; Aragüés, R. Irrigation Management, Nitrogen Fertilization, and Nitrogen Losses in the Return Flows of La Violada Irrigation District (Spain). *Agric. Ecosyst. Environ.* **2012**, *155*, 161–171. [CrossRef]
21. García-Garizábal, I.; Causapé, J.; Abrahao, R. Nitrate Contamination and Its Relationship with Flood Irrigation Management. *J. Hydrol.* **2012**, *442–443*, 15–22. [CrossRef]
22. Andrés, R.; Cuchí, J.A. Salt and Nitrate Exports from the Sprinkler-Irrigated Malfarás Creek Watershed (Ebro River Valley, Spain) during 2010. *Environ. Earth Sci.* **2014**, *72*, 2667–2682. [CrossRef]
23. Cavero, J.; Beltrán, A.; Aragüés, R. Nitrate Exported in Drainage Waters of Two Sprinkler-Irrigated Watersheds. *J. Environ. Qual.* **2003**, *32*, 916–926. [CrossRef] [PubMed]
24. Merchán, D.; Causapé, J.; Abrahão, R.; García-Garizábal, I. Assessment of a Newly Implemented Irrigated Area (Lerma Basin, Spain) over a 10-Year Period. II: Salts and Nitrate Exported. *Agric. Water Manag.* **2015**, *158*, 288–296. [CrossRef]
25. Merchán, D.; Casalí, J.; Del Valle de Lersundi, J.; Campo-Bescós, M.A.; Giménez, R.; Preciado, B.; Lafarga, A. Runoff, Nutrients, Sediment and Salt Yields in an Irrigated Watershed in Southern Navarre (Spain). *Agric. Water Manag.* **2018**, *195*, 120–132. [CrossRef]
26. Iital, A.; Klõga, M.; Pihlak, M.; Pachel, K.; Zahharov, A.; Loigu, E. Nitrogen Content and Trends in Agricultural Catchments in Estonia. *Agric. Ecosyst. Environ.* **2014**, *198*, 44–53. [CrossRef]
27. Kyllmar, K.; Forsberg, L.S.; Andersson, S.; Mårtensson, K. Small Agricultural Monitoring Catchments in Sweden Representing Environmental Impact. *Agric. Ecosyst. Environ.* **2014**, *198*, 25–35. [CrossRef]
28. European Communities. *Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 Establishing a Framework for Community Action in the Field of Water Policy*; European Communities: Brussels, Belgium, 2000.

29. European Communities. *Council Directive of 12 December 1991 Concerning the Protection of Waters against Pollution Caused by Nitrate from Agricultural Sources (91/676/EEC)*; European Communities: Brussels, Belgium, 1991.
30. INTIA. *Informe Del Seguimiento de La Zona Regable Del Canal de Navarra Durante La Campaña 2020: Balance Hídrico, Calidad Del Riego y Contaminación Por Sales y Nitratos*; INTIA: Pamplona, Spain, 2021.
31. Neitsch, S.L.; Arnold, J.G.; Kiniry, J.R.; Williams, J.R. *Soil & Water Assessment Tool Theoretical Documentation Version 2009*; Texas A&M University System: College Station, TX, USA, 2011.
32. Arnold, J.G.; Kiniry, J.R.; Srinivasan, R.; Williams, J.R.; Haney, E.B.; Neitsch, S.L. *Input/Output Documentation Soil & Water Assessment Tool, Version 2012*; Texas Water Resources Insitute, TR-439: College Station, TX, USA, 2012.
33. USDA Soil Conservation Service (SCS). *National Engineering Handbook, Section 4: Hydrology*; USDA Soil Conservation Service (SCS): Washington, DC, USA, 1972.
34. Cunge, J.A. On the Subject of a Flood Propagation Computation Method (Muskingum Method). *J. Hydraul. Res.* **1969**, *7*, 205–230. [CrossRef]
35. Green, C.H.; van Griensven, A. Autocalibration in Hydrologic Modeling: Using SWAT2005 in Small-Scale Watersheds. *Environ. Model. Softw.* **2008**, *23*, 422–434. [CrossRef]
36. Arabi, M.; Frankenberger, J.R.; Engel, B.A.; Arnold, J.G. Representation of Agricultural Conservation Practices with SWAT. *Hydrol. Process.* **2008**, *22*, 3042–3055. [CrossRef]
37. Abbaspour, K.C. *SWAT-CUP: SWAT-Calibration and Uncertainty Programs (CUP)—A User Manual*; EAWAG Aquatic Research: Dübendorf, Switzerland, 2015; ISBN 9780975840047.
38. Abbaspour, K.C.; Vaghefi, S.A.; Srinivasan, R. A Guideline for Successful Calibration and Uncertainty Analysis for Soil and Water Assessment: A Review of Papers from the 2016 International SWAT Conference. *Water* **2018**, *10*, 6. [CrossRef]
39. Moriasi, D.N.; Arnold, J.G.; Van Liew, M.W.; Bingner, R.L.; Harmel, R.D.; Veith, T.L. Model Evaluation Guidelines for Systematic Quantification of Accuracy in Watershed Simulations. *Trans. ASABE* **2007**, *50*, 885–900. [CrossRef]
40. Abbaspour, K.C.; Rouholahnejad, E.; Vaghefi, S.; Srinivasan, R.; Yang, H.; Kløve, B. A Continental-Scale Hydrology and Water Quality Model for Europe: Calibration and Uncertainty of a High-Resolution Large-Scale SWAT Model. *J. Hydrol.* **2015**, *524*, 733–752. [CrossRef]
41. Santhi, C.; Muttiah, R.S.; Arnold, J.G.; Srinivasan, R. A GIS-Based Regional Planning Tool for Irrigation Demand Assessment and Savings Using SWAT. *Trans. Am. Soc. Agric. Eng.* **2005**, *48*, 137–147. [CrossRef]
42. Helsel, D.R.; Hirsch, R.M.; Ryberg, K.R.; Archfield, S.A.; Gilroy, E.J. Statistical Methods in Water Resources: U.S. Geological Survey Techniques and Methods, Book 4, Chapter A3. In *Book 4, Hydrologic Analysis and Interpretation*; U.S Department of the Interior, Ed.; US Geological Survey: Reston, VA, USA, 2020; p. 458.
43. Niraula, R.; Meixner, T.; Norman, L.M. Determining the Importance of Model Calibration for Forecasting Absolute/Relative Changes in Streamflow from LULC and Climate Changes. *J. Hydrol.* **2015**, *522*, 439–451. [CrossRef]
44. Molina-Navarro, E.; Hallack-Alegría, M.; Martínez-Pérez, S.; Ramírez-Hernández, J.; Mungaray-Moctezuma, A.; Sastre-Merlín, A. Hydrological Modeling and Climate Change Impacts in an Agricultural Semiarid Region. Case Study: Guadalupe River Basin, Mexico. *Agric. Water Manag.* **2016**, *175*, 29–42. [CrossRef]
45. Molina-Navarro, E.; Trolle, D.; Martínez-Pérez, S.; Sastre-Merlín, A.; Jeppesen, E. Hydrological and Water Quality Impact Assessment of a Mediterranean Limno-Reservoir Under Climate Change and Land Use Management Scenarios. *J. Hydrol.* **2014**, *509*, 354–366. [CrossRef]
46. Ficklin, D.L.; Luo, Y.; Zhang, M. Watershed Modelling of Hydrology and Water Quality in the Sacramento River Watershed, California. *Hydrol. Process.* **2012**, *27*, 236–250. [CrossRef]
47. Epelde, A.M.; Cerro, I.; Sánchez-Pérez, J.M.; Sauvage, S.; Srinivasan, R.; Antigüedad, I. Application of the SWAT Model to Assess the Impact of Changes in Agricultural Management Practices on Water Quality. *Hydrol. Sci. J.* **2015**, *60*, 825–843. [CrossRef]
48. Meaurio, M.; Zabaleta, A.; Uriarte, J.A.; Srinivasan, R.; Antigüedad, I. Evaluation of SWAT Model's Performance to Simulate Streamflow Spatial Origin. The Case of a Small Forested Watershed. *J. Hydrol.* **2015**, *525*, 326–334. [CrossRef]
49. Rostamian, R.; Jaleh, A.; Afyuni, M.; Mousavi, S.F.; Heidarpour, M.; Jalalian, A.; Abbaspour, K.C. Application of a SWAT Model for Estimating Runoff and Sediment in Two Mountainous Basins in Central Iran. *Hydrol. Sci. J.* **2008**, *53*, 977–988. [CrossRef]
50. Tolson, B.A.; Shoemaker, C.A. Cannonsville Reservoir Watershed SWAT2000 Model Development, Calibration and Validation. *J. Hydrol.* **2007**, *337*, 68–86. [CrossRef]
51. Zettam, A.; Taleb, A.; Sauvage, S.; Boithias, L.; Belaidi, N.; Sanchez-Perez, J.M. Applications of a SWAT Model to Evaluate the Contribution of the Tafna Catchment (North-West Africa) to the Nitrate Load Entering the Mediterranean Sea. *Environ. Monit. Assess.* **2020**, *192*, 510. [CrossRef] [PubMed]
52. Boithias, L.; Srinivasan, R.; Sauvage, S.; Macary, F.; Sánchez-Pérez, J.M. Daily Nitrate Losses: Implication on Long-Term River Quality in an Intensive Agricultural Catchment of Southwestern France. *J. Environ. Qual.* **2014**, *43*, 46–54. [CrossRef]
53. Andrés, R.; Cuchí, J.A. Analysis of Sprinkler Irrigation Management in the LASESA District, Monegros (Spain). *Agric. Water Manag.* **2014**, *131*, 95–107. [CrossRef]
54. García-Garizábal, I.; Causapé, J.; Merchán, D. Evaluation of Alternatives for Flood Irrigation and Water Usage in Spain Under Mediterranean Climate. *CATENA* **2017**, *155*, 127–134. [CrossRef]
55. García-Garizábal, I.; Causapé, J.; Abrahao, R. Application of the Irrigation Land Environmental Evaluation Tool for Flood Irrigation Management and Evaluation of Water Use. *CATENA* **2011**, *87*, 260–267. [CrossRef]

56. Scott, J.; Rosen, M.R.; Saito, L.; Decker, D.L. The Influence of Irrigation Water on the Hydrology and Lake Water Budgets of Two Small Arid-Climate Lakes in Khorezm, Uzbekistan. *J. Hydrol.* **2011**, *410*, 114–125. [CrossRef]
57. INTIA. *Informe Corregido Del Seguimiento de La Zona Regable Del Canal de Navarra Durante La Campaña 2018: Balance Hídrico, Calidad Del Riego y Contaminación Por Sales y Nitratos*; INTIA: Pamplona, Spain, 2019.
58. INTIA. *Informe Del Seguimiento de La Zona Regable Del Canal de Navarra Durante La Campaña 2019: Balance Hídrico, Calidad Del Riego y Contaminación Por Sales y Nitratos*; INTIA: Pamplona, Spain, 2020.
59. Skhiri, A.; Dechmi, F. Impact of Sprinkler Irrigation Management on the Del Reguero River (Spain). I: Water Balance and Irrigation Performance. *Agric. Water Manag.* **2012**, *103*, 120–129. [CrossRef]
60. Orellana-Macías, J.M.; Merchán, D.; Causapé, J. Evolution and Assessment of a Nitrate Vulnerable Zone over 20 Years: Gallocanta Groundwater Body (Spain). *Hydrogeol. J.* **2020**, *28*, 2207–2221. [CrossRef]
61. Hernández-García, I.; Merchán, D.; Aranguren, I.; Casalí, J.; Giménez, R.; Campo-Bescós, M.A.; Del Valle de Lersundi, J. Assessment of the Main Factors Affecting the Dynamics of Nutrients in Two Rainfed Cereal Watersheds. *Sci. Total Environ.* **2020**, *733*, 139177. [CrossRef] [PubMed]
62. Lassaletta, L.; García-Gómez, H.; Gimeno, B.S.; Rovira, J.V. Headwater Streams: Neglected Ecosystems in the EU Water Framework Directive. Implications for Nitrogen Pollution Control. *Environ. Sci. Policy* **2010**, *13*, 423–433. [CrossRef]
63. Merchán, D.; Causapé, J.; Abrahão, R. Impact of Irrigation Implementation on Hydrology and Water Quality in a Small Agricultural Basin in Spain. *Hydrol. Sci. J.* **2013**, *58*, 1400–1413. [CrossRef]
64. Causapé, J.; Quílez, D.; Aragüés, R. Assessment of Irrigation and Environmental Quality at the Hydrological Basin Level II. Salt and Nitrate Loads in Irrigation Return Flows. *Agric. Water Manag.* **2004**, *70*, 211–228. [CrossRef]
65. CHE (Confederación Hidrográfica del Ebro) Control de Los Retornos de Las Actividades Agrarias de La Cuenca Del Ebro: Evaluación de Tendencias En La Calidad Del Agua, Control Experimental de Los Retornos y Propuesta de Red de Control (in Spanish). Available online: www.chebro.es (accessed on 19 August 2022).
66. Arauzo, M.; Valladolid, M.; Martínez-Bastida, J.J. Spatio-Temporal Dynamics of Nitrogen in River-Alluvial Aquifer Systems Affected by Diffuse Pollution from Agricultural Sources: Implications for the Implementation of the Nitrate Directive. *J. Hydrol.* **2011**, *411*, 155–168. [CrossRef]