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Effects of climate change on streamflow and nitrate pollution in an agricultural Mediterranean watershed in Northern Spain

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ABSTRACT

Predicting water quality and quantity response to climate change in a watershed is very difficult due to the complexity and uncertainties in estimating and understanding future hydrological conditions. However, hydrological models could simplify the processes and predict future impacts of agricultural activities. This study aimed to evaluate the applicability of the Soil Water Assessment Tool (SWAT) model for climate change prediction of streamflow and nitrate load in an agricultural Mediterranean watershed in northern Spain. The model was first evaluated for simulating streamflow and nitrate load under rainfed agricultural conditions in the Cidacos River watershed in Navarre, Spain. Then, climate change impact analysis on streamflow and nitrate load was conducted in the short-term (2011–2040), medium-term (2041–2070), and long-term (2071–2100) future projections relative to the historical baseline period (1971–2000) under the RCP4.5 and RCP8.5 CO_2 emission scenarios. The model evaluation showed a good model performance result during calibration (2000–2010) and validation (2011–2020) for streamflow (NSE = 0.82/0.83) and nitrate load (NSE = 0.71/0.68), indicating its suitability for adoption in the watershed. The climate change projection results showed a steady decline in streamflow and nitrate load for RCP4.5 and RCP8.5 in all projections, with the long-term projection scenario of RCP8.5 greatly affected. Autumn and winter saw a considerable drop in comparison to spring and summer. The decline in streamflow was attributed to the projected decrease in precipitation and increase in temperatures, while the nitrate load decline was consistent with the projected streamflow decline. Based on these projections, the long-term projection scenarios of RCP8.5 indicate dire situations requiring urgent policy changes and management interventions to minimize and mitigate the resulting climate change effects. Therefore, adapted agricultural management practices are needed to ensure sustainable water resource utilization and efficient nitrogen fertilizer application rates in the watershed to reduce pollution.

1. Introduction

Agriculture is one of the most important sectors of any regional or national economy globally. It is the primary source of livelihood and the backbone of most nations' economic systems, with more than 60% of the world population directly dependent on it (FAO, 2017). However, agricultural intensification puts great pressure on available water resources and the environment, potentially causing damage. These damages could range from soil erosion, which is much more common in agricultural environments than in other soil uses (García-Ruiz et al., 2015; Almagro et al., 2016; Boardman and Poesen, 2006), to water quality degradation caused by non-point source pollution (Chahor et al., 2014; Giménez et al., 2012; Merchán et al., 2018; Sutton et al., 2011). Studies conducted within the Navarre region of northern Spain have identified considerable nitrate and phosphate concentrations in streams in cereal crop areas, where the recommended thresholds are often exceeded, albeit with seasonal and annual variability (Casalí et al., 2008; Hernández-García et al., 2020; Merchán et al., 2019).

The Cidacos River watershed in the Navarre region has diverse land uses, with rainfed agriculture predominating. The watershed holds decades worth of nitrate concentration, discharge, and meteorological data collected by the Government of Navarre at various stations hence ideal for conducting investigations on agricultural activities' impact on the quality and quantity of water resources in the area. Some of the

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challenges associated with agricultural practices in the study area are nitrate pollution in surface waters, as evidenced by high nitrate concentration levels in the Cidacos River (Merchán et al., 2020), and anticipated climate change effects due to projected changes in temperature and precipitation affecting the cropping system (Funes et al., 2016; Trnka et al., 2011).

Climate change impacts on water resources can be quantified by using various Global or Regional Climate Models (GCMs or RCMs) and future radiative forcing scenarios, known as Representative Concentration Pathways (RCPs), to establish appropriate adaptation measures and policy interventions (Krysanova et al., 2017). The Mediterranean region, particularly Spain, is considered to be highly vulnerable to climate change effects due to its geographical location and the imbalance between the available water resources and the current demands (Vargas-Amelin and Pindado, 2014). Furthermore, most climate change model projections for Spain indicate an increase in temperature and a decrease in precipitation by the end of the 21st Century (Candela et al., 2012; Chirivella Osma et al., 2015; Estrela et al., 2012; Majone et al., 2012; Somot et al., 2008). Climate change is expected to affect all aspects of the environment, compounding agricultural effects on streamflow and nitrate exportation (Arora, 2019). For instance, a change in streamflow due to projected temperature and precipitation changes would result in extreme events such as droughts or floods. These events would, in turn, influence the nitrate dynamics in the watershed by changing the exported nitrate loads and concentration accumulated in the soils and water. Therefore, it is important to investigate the potential effects of climate change on streamflow and nitrate dynamics, as they would impair the current hydrological conditions and hinder the achievement of nitrate standards as stipulated by the European Water Framework Directive (European Communities, 2000).

Nitrate pollution is a global concern that affects water quality by making it unsafe for human consumption (WHO, 2017) and increases eutrophication (Sutton et al., 2011). Nitrate pollution contributors on a watershed could include agriculture, livestock, and aquaculture (Casalí et al., 2008; FAO, IWMI, 2018; Menció et al., 2016). Whilst some level of nitrate exportation is inevitable in agricultural areas, improved management practices could limit its effect on streams (Beaudoin et al., 2005; Boithias et al., 2014; Cameron et al., 2013). To address nitrate pollution challenge, the European Commission has established policy legislations such as the Nitrate Directive (ND, Directive 91/676/EEC) and the European Water Framework Directive (WFD, Directive 2000/60/EEC) to protect water bodies from agricultural nitrate pollution, with a nitrate concentration threshold of 50 mg L^{-1} for European rivers (European Communities, 1991). However, more research is needed to understand the spatial and temporal interactions of water quality variables and quantify the loads to assess their climate change impacts.

Mathematical models are fundamental tools for hydrological and environmental planning, with their greatest potential being the ability to generate scenarios in the face of voluntary or imposed changes in land use or management. The Soil Water Assessment Tool (SWAT) model is one of the best available tools for simulating the response of agricultural (or non-agricultural) watersheds to water quality. The SWAT model has been widely used by water resources experts to understand the characteristics of a watershed and predict its hydrological response to external (climate) and internal (water management, land management, etc.) drivers and their impacts. The SWAT model has been applied in the Mediterranean region and particularly Spain for streamflow analysis (Harraki et al., 2021; Jimeno-Sáez et al., 2018; Meaurio et al., 2015) and water quality assessment of nitrogen and nitrates (Epelde et al., 2015; Zabaleta et al., 2014; Zettam et al., 2020). The majority of nitrate studies conducted by the SWAT model focus on how to reduce nitrate pollution through land use changes (Ferrant et al., 2011; Wang et al., 2008), as well as regulating fertilizer application rates and other management practices like tillage (Boithias et al., 2014; Cerro et al., 2014; Ferrant et al., 2011; Liu et al., 2013). Despite several studies on streamflow and hydrological response to climate change, there has been very little research on the effects of climate change on water quality (Ficklin et al., 2010; Luz Rodríguez-Blanco et al., 2019; Martínková et al., 2011; Molina-Navarro et al., 2014). Moreover, there are no long-term climate change assessments of the effects of agricultural activities on streamflow and water quality in the Navarre region.

This study aimed to evaluate the SWAT model's applicability for climate change prediction of streamflow and nitrate load in a rainfed agricultural Mediterranean watershed in northern Spain. The model was first evaluated for its capacity to simulate streamflow and nitrate export under rainfed agricultural conditions in the upper reaches of the Cidacos River watershed and then used to assess the climate change impacts by comparing future projections to the historical baseline under two emission scenarios (RCP4.5 and RCP8.5). The findings from this study could provide valuable information on climate change adaptation and mitigation measures in the future and deepen the knowledge on nitrate exportation and pollution in the study area.

2. Materials and methods

2.1. Study area

The Cidacos River is one of the tributaries of the Aragón River, which is a tributary of the Ebro River. It is located approximately 15 km south of Pamplona, the capital of the Chartered Community of Navarre in Spain, at latitudes 42° 69' and 42° 34' North and longitudes 1° 72' and 1° 47' West. The Cidacos River drains a watershed area of 477 km^2 and runs north-south, with an approximate length of 44 km and width of 15 km in its widest section (Fig. 1). The watershed's headwater is somewhat mountainous in the north, with high altitudes of slightly over 1000 m above sea level, 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 the Aragón River. The watershed's climate is humid to dry, temperate, mild Mediterranean, with cold winters (monthly average: 4.7 °C to 5.4 °C in January) and warm summers (monthly average: 21.2 °C to 23.7 °C in August) that vary spatially from North to South. The annual average temperature ranges from 12.2 °C to 14.2 °C (north to south). The watershed receives annual precipitation from 800 mm in the north to 400 mm in the south, characterized by strong inter-annual variability and high summer aridity. The wettest months are April and May (Merchán et al., 2020).

Agriculture is the predominant land use in the watershed, accounting for 53% of the total area (Fig. 2a). Other major land uses in the watershed include forests (25%) and pasture and bushlands (17%). The remaining 5% comprises urban, residential areas, built-up land, bare land, and water bodies. Rainfed agriculture covers 176 km^2 (37% of the total area and 70% of cultivated land) and is primarily in the watershed's upper reaches until Olite town. Irrigated land, on the other hand, covers 77 km^2 (16% of the total area and 30% of the cultivated land) and is largely in the watershed's lower reaches. The main crops grown are rainfed winter cereals (wheat and barley) and vineyards (orchard). Other crops grown in small quantities include corn, tomatoes, and potatoes. The average annual fertilizer application rates range from 80 to 130 kg N ha^{-1} for winter cereals and 40–50 kg N ha^{-1} for vineyards (Oduor et al., 2023).

The most abundant soil textures in the watershed are loam and clay-loam, which are found in most agricultural areas, while loamy-sand and sandy-loam soils are found on eroded hillslopes. Red clay soils dominate the watershed with sandstone and mudstones. According to the FAO classification system (IUSS Working Group WRB, 2015), the watershed's predominant soil types are Haplic Calciols soils (51.6%), Fluvic Camisols soils (26.1%) which are mostly found along the river network path, and Alaric Regosols (18%). Haplic Phaeozem (1.7%), Calcic Castanea's (1.6%), Fluvic Phaeozem (0.4%), Eutric Fluvisols (0.3%), and Dystric Cambisols (0.2%) are among the other soils found in the watershed (Fig. 2b).

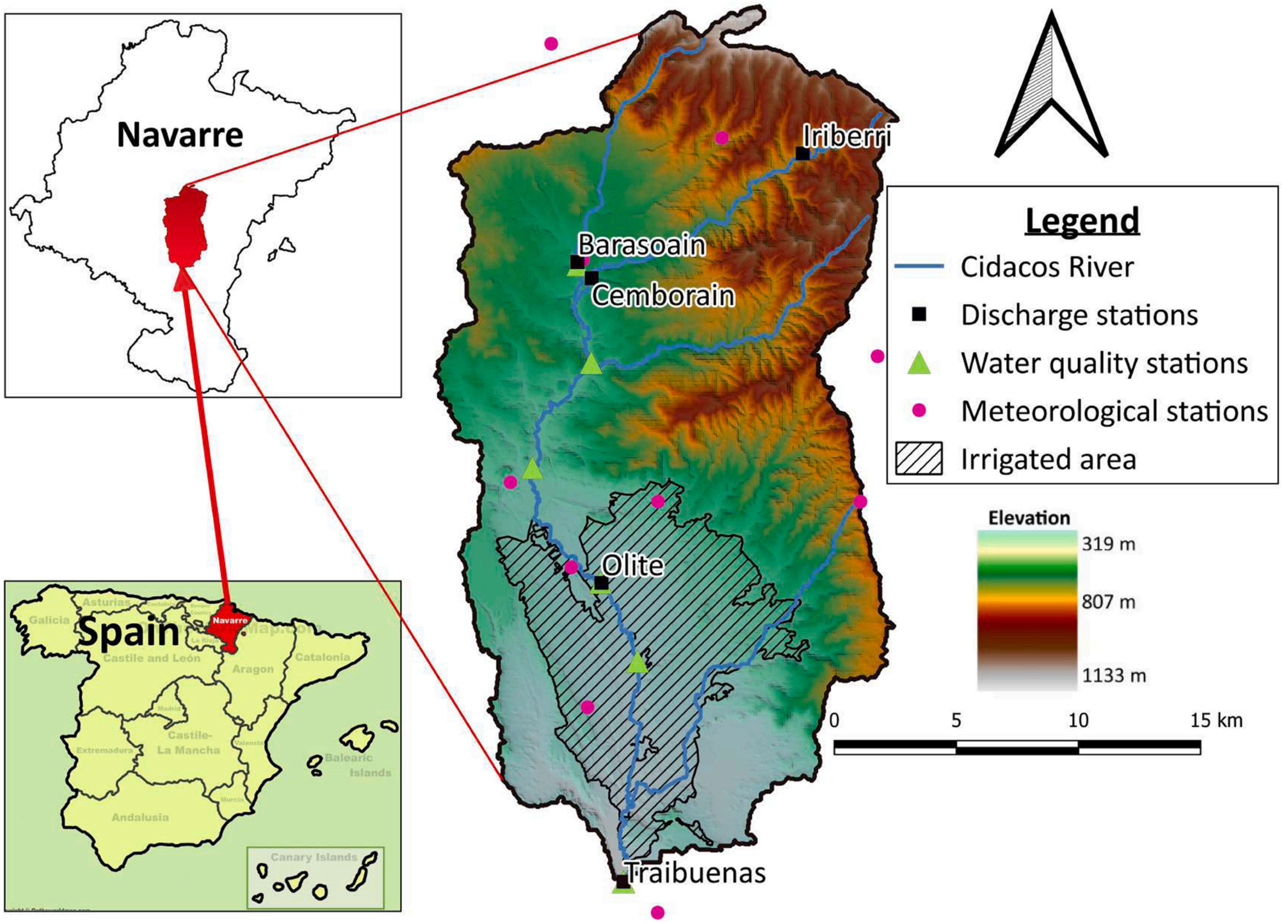

Fig. 1. The Cidacos River watershed location, elevation map, and measuring stations.

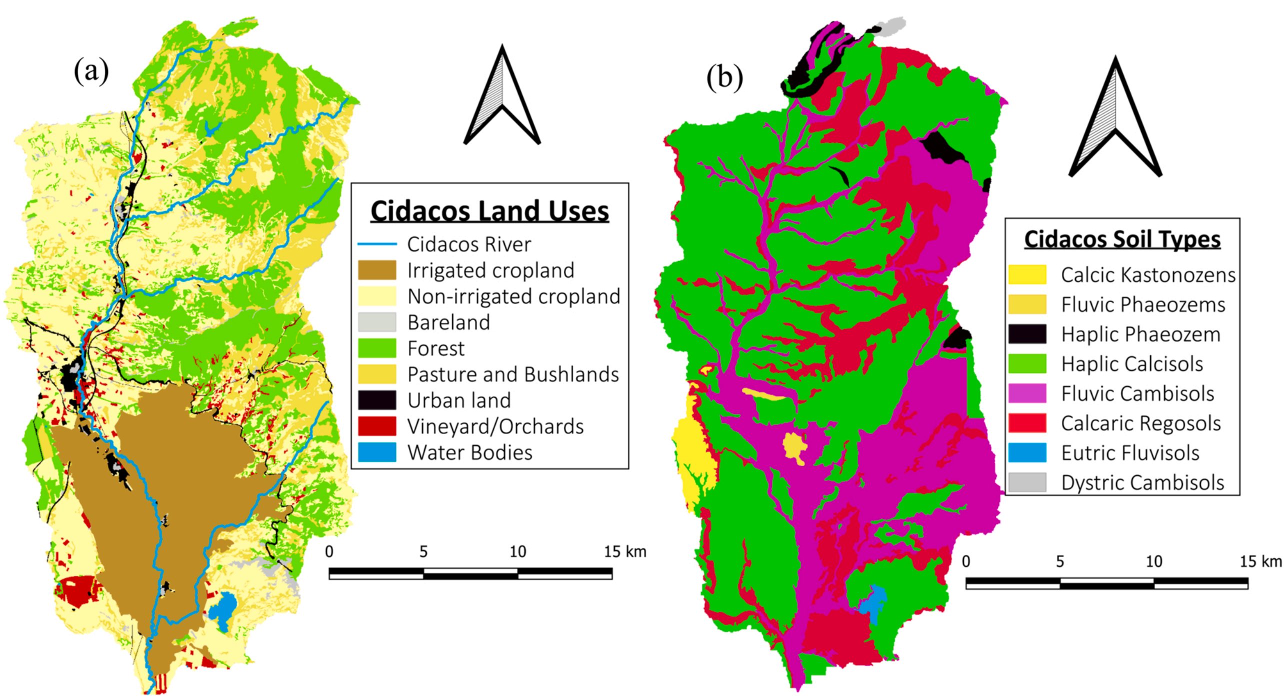

Fig. 2. The Cidacos River watershed (a) land use land cover map and (b) soil type map.

2.2. Description of the SWAT model

The SWAT model is an open-source software developed by the United States Department of Agriculture's Agricultural Research Service (USDA-ARS) to help water resource managers, policy experts, and decision-makers predict and quantify the impact of land use management on water and diffuse pollution in small and large watersheds with varying soil types, land use, and management practices (Neitsch et al., 2011). SWAT is a data-driven, semi-distributed, continuous timescale, physical and process-based hydrological model that simulates water, sediments, and agricultural chemicals or pollutant yields.

The SWAT simulation process divides the watershed's hydrology into land and routing phases. In the land phase, the water, sediment, and nutrient balances are calculated for each Hydrological Response Unit (HRU), whereas in the routing phase, the HRU outputs are aggregated and routed through the channel network to the outlet of the watershed (Arnold et al., 2012). The HRU consists of a unique homogeneous combination of similar land use, soil type, and topography characteristics (Neitsch et al., 2011). The water balance equation simulates the watershed's hydrological component (Arnold et al., 2012; Neitsch et al., 2011).

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 (Green and van Griensven, 2008). The movement and transformation of various forms of nitrogen within a watershed are introduced into the main channel through surface runoff and lateral subsurface flow and transported downstream with the flow (Arabi et al., 2008).

2.3. Data acquisition

The SWAT model requires geospatial and hydrometeorological input data variables. The data for this study were primarily obtained from the Government of Navarre agencies and websites, as shown in Table 1. The climate data were obtained at a daily time-step from 12 weather stations (both manual and automatic) located within and near the watershed from 1990 to 2020. The selected stations represented the spatial heterogeneity of the watershed's climate. The meteorological data included daily data of precipitation (mm), maximum and minimum daily temperatures (℃), solar radiation (MJ m^{-2} s^{-2}), wind speed (m s^{-1}), and relative humidity (%) data. The agricultural management information, which included planting and harvesting dates, the average annual fertilizer application rates, and the main crops cultivated in the watershed, were obtained from INTIA's technical team and extension advisors who conduct field interviews in consultation with key informants such as farmers within the watershed (see Table S1). The monthly observed streamflow data and nitrate loads from 2000 to 2020 were used for the model evaluation. The data used were obtained from the Olite gauging station since it was the only station in the watershed with consistent and extensive long-term data of observed discharge and nitrate concentration data, as it has been operational since 1988 and covers the watershed area under rainfed agriculture.

2.4. The SWAT model set-up and run

The model set-up was preceded by preparing and processing the necessary spatial datasets such as DEM, soil and land use grid maps, and discharge outlet points on the QGIS 3.18 interphase. The model was set up in the QSWAT3 1.1.1 interphase by performing watershed delineation, HRU creation, input editing, and running the SWAT model. The watershed was delineated using the DEM and the Cidacos River shapefile until the outlet at Traibuenas. Discretization was done using a minimum area threshold of 10 km^2 required to create streams, resulting in a watershed area of 477.02 km^2 with 23 sub-watersheds. A slope elevation band of 0–5%, 5–10%, and 10% and above was provided to the model. The watershed's overall elevation ranged from 315 m to 1150 m, with an average elevation of 560 m. By overlaying the LULC and soil grid maps and using a 5% threshold for land uses, soil type, and slope values, 1404 HRUs were generated. This threshold was chosen to eliminate minor land uses, soils, and slopes in each sub-watershed, facilitating model processing by improving its performance, speed, and efficiency. Using the SWAT editor, the weather data and agricultural management information were added to the model.

Table 1
The SWAT model input data requirement and their sources.

Dataset	Resolution	Source
Digital Elevation Model (DEM)	25 m, ETRS89 UTM Zone 30 N 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	1:25000	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 Navarra 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/) & INTIA (https://www.intiasa.es/)
Water Quality	Monthly (2000–2020)	Government of Navarre through Environmental Management of Navarre GAN-NIK (https://gan-nik.es/) & INTIA (https://www.intiasa.es/)
Agricultural Management	Annual	Consultation with the farmers & key stakeholders (INTIA)
Climate Change	Daily (1961–2100)	Platform on Adaptation to Climate Change in Spain (AdapteCCa) (https://escenarios.adaptecca.es/) & AEMET (http://www.aemet.es/en/serviciosclimaticos/cambio_climat/)

2.5. Sensitivity analysis, calibration, and validation

The SWAT Calibration and Uncertainty Procedures (SWATCUP) version 5.1.6, a standalone software, was used to perform sensitivity analysis, calibration, and validation of the model. The multi-site Sequential Uncertainty Fitting, version 2 (SUFI-2), a semi-automated inverse modeling routine procedure of SWATCUP, was used in this study. The model was run 500 times for each iteration, and the parameter sensitivity was determined by performing a global sensitivity analysis in which all parameters changed simultaneously. Multiple regression computations were used to identify the most sensitive parameters. The Latin hypercube-generated parameters are regressed against the objective function values in this system (Abbaspour, 2015). The t-test was used to determine the relative significance of each parameter.

The p-values and t-stat indices were used to assess the sensitivity of the parameters. The parameter was more sensitive when the p-value was lower, and vice versa. The best combination for obtaining the most sensitive parameter is a very small p-value and a large t-value (absolute).

Parameters that had p-values less than 0.05 were deemed highly sensitive. The parameter sensitivity was ranked using the t-stat index and the p-value to identify the most sensitive parameters that had the greatest impact on the model outputs (Arnold et al., 2012). Larger parameter uncertainties were initially assumed to ensure that most observed data fell within the 95% Prediction Uncertainty (95PPU) band (Abbaspour et al., 2018). 95PPU accounts for all the uncertainties within the model combined. The parameter ranges were adjusted after every iteration run during the calibration phase until most of the observed data were bracketed in the 95PPU band. The model was deemed satisfactory when more than 50% of the observed flow data were bracketed within the 95PPU.

The model was run from 1990 to 2020, with the first ten years (1990–1999) serving as the warm-up period to allow the model to reach an optimal state before reading the outputs. The model was then evaluated over the remaining period (2000–2020), which was divided into calibration (2000–2010) and validation (2011–2020) phases. The streamflow parameters were first satisfactorily calibrated and fixed before calibrating the nitrate parameters. The calibration parameters were chosen from the abundant existing literature on streamflow and nitrate calibration using the SWAT model in the Mediterranean region (Abbaspour, 2015; Abbaspour et al., 2018, 2015; Kamali et al., 2017; Kouchi et al., 2017; Rouholahnejad et al., 2014). To change the parameter values in SWAT, three methods (parameter qualifiers) are used: "R" which refers to a relative change of the specified parameter that increases or decreases the existing SWAT parameter value by multiplying it by (1 + fitted value) to obtain the new parameter value; "V" which refers to value change or replacement which means that the initial SWAT parameter value is to be directly replaced by the fitted value; and "A" which refers to addition and means that the fitted value is added to the initial SWAT parameter value. After the sensitivity analysis, the final streamflow and nitrate load calibration parameters were chosen.

The model results were presented graphically on the hydrograph plots for the simulated and observed values during the calibration and validation periods. The model's performance was evaluated using the commonly used statistical performance indicator techniques for hydrological modeling. Moriasi et al. (2007) have recommended the Nash-Sutcliffe efficiency (NSE), the Coefficient of Determination (R^2), the root mean square error to the standard deviation of observed data ratio (RSR), and the percent bias (PBIAS) as the most suitable quantitative statistical techniques for the SWAT model evaluation. The model performance was deemed satisfactory, provided the values of the NSE > 0.5, RSR ≤ 0.7, $R^2 > 0.5$, and PBIAS $\pm$ 25% for streamflow and PBIAS $\pm$ 55% for nitrate loads (Abbaspour et al., 2018).

2.6. Climate change scenario development

The climate change impact in the study area was analyzed using an ensemble of six Global Climate Models (ACCESS1–0, BCC-CSM1–1, CMCC-CM, GDFL-ESM2G, IPSL-CM5A-LR, and MPI-ESM-MR) from bias-corrected Coupled Model Intercomparison Project (CMIP) climate forcing data statistically downscaled on a 5 km grid for the Navarre region for historical and future data of RCP4.5 and RCP8.5 emission scenarios. This data was developed by the Spanish National Agency for Meteorology (AEMET) and was downloaded from the Platform on Adaptation to Climate Change in Spain (AdapteCCa) portal (AdapteCCa, 2021). These two projected radiative forcing scenarios represent the potential moderate (RCP4.5) and more aggressive (RCP8.5) climate change impact scenarios, with RCP4.5 assuming that greenhouse gas emissions will be gradually reduced in the coming years to achieve stability by 2100 and RCP8.5 assuming that greenhouse gas (GHG) emissions will continue to rise at current levels throughout the 21st century (IPCC 2014). Only the precipitation and temperature (maximum and minimum) datasets were used for the climate change simulation.

The calibrated SWAT model was used to simulate projected streamflow and nitrate load trends using climate change data (precipitation and temperature) as inputs while assuming all other variables to be constant. Crop heat units were used in the simulation to automatically assign agricultural management operations such as planting, harvesting, and fertilization periods. The simulation was run for each GCM from 1971 to 2000 for historical reference and 2011–2100 for the RCP4.5 and RCP8.5 future projection scenarios. In total, 18 simulations with six historical and 12 future projections were run (6 for each emission scenario). The future projection scenarios of streamflow and nitrate export were analyzed for three distinct periods categorized into short-term (2011–2040), medium-term (2041–2070), and long-term (2071–2100) by comparing each model to its historical period (1971–2000). Finally, the models' results were combined and averaged to obtain an ensemble for the climate change analysis.

3. Results and discussions

3.1. Model evaluation

3.1.1. Parameterization and sensitivity analysis

The most sensitive streamflow and nitrate load parameters, along with their calibrated values are shown in Table 2. The GW_DELAY parameter regulates the rate and duration of groundwater recharge. It estimates the time required for baseflow recharge. The ESCO parameter influences the watershed's evapotranspiration. The ESCO value was lower, indicating that most of the model's evaporative demands were extracted from deeper soil layers because the shallow layers could not meet them (Niraula et al., 2015; Qi et al., 2019). The CN2 is the curve number parameter typically affected by land use, soil permeability, and antecedent soil water conditions. The higher the CN2 value, the greater the surface runoff and the lower the infiltration and, thus, baseflow, and vice versa. The CDN parameter controls the amount of nitrate fertilizer lost due to denitrification. Denitrification losses are greater in areas with high moisture content than in areas with low moisture content. The ANION_EXCL parameter estimates the amount of nitrate exported by water. The FIXCO parameter regulates the amount of additional nitrogen provided to the plant to meet the legume demand when insufficient nitrate is in the root zone (Neitsch et al., 2011). The greater the FIXCO value, the more fixed the nitrogen demand, and vice versa.

Table 2
Selected sensitive streamflow and nitrate load parameters and their calibrated values.

Parameter	Description	Change Method[a]	Parameter Adjustment Values		
			Min. Value	Max. Value	Fitted Value
GW_DELAY.gw	Groundwater delay time (days)	V	20	80	53.54
ESCO.hru	Soil Evaporation compensation factor	R	-0.45	-0.28	-0.31
CN2.mgt	Initial SCS runoff CN number for moisture condition II	R	-0.20	0.20	-0.12
CDN.bsn	Denitrification exponential rate coefficient	V	0	1.62	0.04
ANION_EXCL.sol	Fraction of porosity (void space) from which anions are excluded	R	-0.15	0.53	-0.05
FIXCO.bsn	Nitrogen fixation coefficient	V	0.45	1.4	1.16

[a] The change method **R** multiplies the existing value with (1 +fitted value), indicating a relative change, whereas **V** replaces the existing value with the fitted value, indicating a value change

3.1.2. Streamflow calibration and validation

The maximum and minimum parameter values were used to account for the uncertainty for each parameter, with the fitted value providing the best simulation. The 95PPU was used to quantify model uncertainties, such as those related to parameters, input data, and structure. During the calibration and validation periods, the 95PPU results were represented by p-factor (0.56 and 0.65) and r-factor (0.70 and 0.67) values, respectively. These uncertainties may result in overestimation or underestimation by the model, often due to the model not fully capturing all the hydrologic components in the watershed because of the model's conceptual simplifications (Ficklin et al., 2013; Meaurio et al., 2015; Rostamian et al., 2008; Tolson and Shoemaker, 2007).

The model produced good results for streamflow prediction during calibration and validation, reproducing most of the observed discharge and its tendency over time (Fig. 3). The NSE values (0.82 and 0.83) and R^2 (0.83 and 0.84) during calibration and validation periods indicate a strong relationship between the observed and simulated values, indicating a 'good' fit. The negative PBIAS values (−8.7% and −5.6%) showed a slight but reasonable overestimation of the average flows by the model during the simulation periods. The RSR value of 0.42 was satisfactory because it was below the recommended threshold of less than 0.7, indicating a good model performance. The results of the four statistical performance indicators deemed the model to be 'very good' and capable of simulating monthly streamflow in the study area as per the Moriasi et al. (2007) recommendations. The validation period resulted in better model performance compared to the calibration period. This could be due to improved input data, such as precipitation and land use during the validation period. The validation period's input data was more accurate, such as precipitation with few to no missing gaps and using the most recent land use map from 2019. However, there were a few meteorological data inconsistencies before 2004, particularly for the automatic stations, as most were only operational after March 2004.

3.1.3. Nitrate load calibration and validation

The nitrate load parameters were calibrated after successfully calibrating and fixing the streamflow parameters. Comparisons between the observed and simulated monthly nitrate loads hydrographs (Fig. 4) indicated a good model performance. The uncertainties in nitrate load simulation were accounted for using the 95PPU represented by the p-factor (0.72 and 0.63) and r-factor (0.92 and 0.98) during calibration and validation periods, respectively. Some of the model weaknesses could have resulted from errors in the input data. These include estimations of missing precipitation data used to generate the discharge (Boithias et al., 2014); insufficient observed nitrate load data available since the concentration data were obtained from a highly scattered sampling frequency (in most cases collected only once per month at random dates and with several months having missing data) (Epelde et al., 2015); and information related to the agricultural management operations and practices such as fertilizer application or planting and harvesting dates (Zettam et al., 2020). The model simulation results were in good agreement with the observed data, indicating good accountability of the model's various agricultural inputs. The model's statistical performance was adequate, with acceptable NSE values (0.71 and 0.68) and R^2 values (0.72 and 0.79) during the calibration and validation periods, respectively. The PBIAS results show that the model underestimates the nitrate loads by − 9.2% and − 7% during the calibration and validation periods, respectively. These results are within the acceptable thresholds recommended by Moriasi et al. (2007), indicating a good model performance.

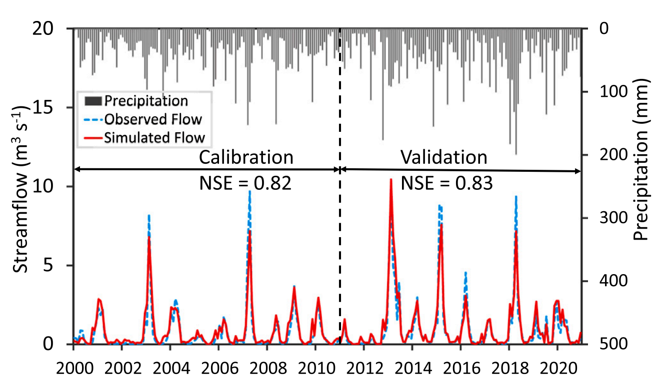

Fig. 3. Observed (dotted blue line) and simulated (solid red line) monthly discharge hydrographs and precipitation (grey bars) during the calibration (2000–2010) and validation (2011–2020) periods at the Olite gauging station in the Cidacos River.

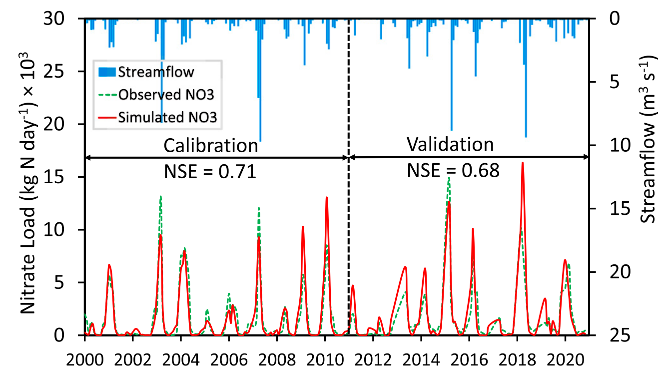

Fig. 4. Plot of observed (dotted green line) and simulated (solid red line) monthly nitrate load and measured streamflow (blue bars) during calibration and validation periods at the Olite gauging station in the Cidacos River.

The inter-annual and seasonal variability of nitrate load was very high throughout the simulation period. Loads were higher in wetter years than in dry years, and vice versa. Nitrate loads in the watershed increased from mid-autumn and peaked during winter, when precipitation was abundant, and thus streamflow, but gradually decreased from spring to summer, when precipitation was scarce. This could be attributed to increased streamflow and, to some extent, the nitrogen fertilizer application on agricultural fields because the planting season begins in October/November, increasing soil nitrogen levels, nitrate concentration, and, subsequently, nitrate loads in the watershed. Because nitrate load is a function of discharge used to transport it downstream, higher precipitation in the watershed during the winter and spring months will inevitably increase nitrate load exportation in the river. However, during the summer, when precipitation is scarce, resulting in limited streamflow, and there are almost no agricultural activities in the watershed's upper reaches, which rely primarily on rainfed farming, fewer nitrate loads are exported. These results are consistent with those obtained by Lam et al. (2009) when modeling agricultural catchments in Europe, where they reported that these patterns could be attributed to higher nitrogen concentrations in the winter due to nitrogen mobilization in the watershed and a lack of plant uptake, resulting in the accumulation of leachable nitrates and thus an increase in nitrogen concentration in streamflow during winter. Similar findings have also been reported by Donmez et al. (2020), who inferred that an increase in fertilization would lead to an increase in the amount of nitrate in the soil, which would be directly related to plant growth and agricultural production and management. According to Abbaspour et al. (2015), nitrate dynamics in agricultural watersheds are governed mainly by the fate and transportation of fertilizer in the soil, the rate of organic matter decomposition, and the prevailing climate.

3.2. Climate change impact analysis

3.2.1. Projected precipitation and temperature

Analysis of the future climate projection compared to the historical reference depicts a general decrease in precipitation and an increase in temperature. Fig. 5 illustrates the percent decrease in mean annual precipitation and increase in average temperature for all six climate models under RCP4.5 and RCP8.5 emission scenarios over the three projected periods relative to the historical period. The average decline in precipitation in the short-, medium-, and long-term projections were − 3.5%, − 6.9%, and − 7.6% for RCP4.5, and − 3.1%, − 7.9%, and − 14.8% for RCP8.5, respectively. The average temperature projections, on the other hand, increased progressively over the three periods, by 1.2 °C, 1.9 °C, and 2.2 °C for RCP4.5, and 1.4 °C, 2.0 °C, and 4.3 °C for RCP8.5. The projected precipitation data varied more among the selected models than the projected temperature, which was closely comparable across the models. These projections are similar to those reported by the European Environment Agency (EEA) (2017) for the Mediterranean region, indicating a significant increase in warming of 2 °C to 5°C from the 2050 s to the end of the 21st century, while the mean annual precipitation could decrease by − 5% to − 15%, and in the worst case scenario, up to − 25%, with an acceleration expected at the end of the century. Furthermore, winter and autumn project a higher decrease in precipitation than summer and spring, while summer temperature increases are greater than winter. Projected precipitation decline and temperature increase over the three time periods of 2040, 2070, and 2100 have also been reported for other studies within the Mediterranean region (Abd-Elmabod et al., 2020; Al-Mukhtar and Qasim, 2019; Fonseca and Santos, 2019). These projected climate changes are expected to alter the watershed's hydrological cycle by increasing the air temperature and, thus, evapotranspiration. A warmer atmosphere is expected to hold more water vapor, causing precipitation concentrations to rise, resulting in more frequent and intense extreme events (Abd-Elmabod et al., 2020; Navarra and Tubiana, 2013). However, greater losses in open surface waters and soils are also expected with the projected high evapotranspiration rates.

3.2.2. Effects of climate change on streamflow

The simulated climate change projections showed a declining effect on streamflow for both emission scenarios over all the projected periods analyzed with high interannual fluctuations (Fig. 6). The average annual streamflow decreased by − 11.5%, − 27.4%, and − 28.5% for RCP4.5 and − 8.5%, − 27.5%, and − 52.4% for RCP8.5 during the short-, medium-, and long-term future climate projections, respectively, compared to the historical reference period. This decline was mainly attributed to the projected decrease in precipitation and increasing temperatures for all the climate models used in the study (Fig. 5), leading to a rise in the watershed's evapotranspiration. Higher evapotranspiration rates and lower precipitation would result in declining discharge, unless there is a significant shift in the seasonal pattern with more precipitation occurring during colder seasons (Anand and Oinam, 2019; Molina-Navarro et al., 2016, 2014).

The long-term climate projection had the highest streamflow reductions for RCP4.5 (−28.5%) and RCP8.5 (−52.4%). The considerable streamflow reduction in RCP8.5 long-term projection compared to RCP4.5 was due to the continuous increase in temperature and decreasing precipitation caused by the lack of climate change mitigation measures to reduce the GHG emissions for this scenario. However, for RCP4.5, some mitigation measures to reduce GHG emissions are expected to be implemented gradually from the mid-term projection onwards. The RCP4.5 long-term projection showed greater extreme streamflow occurrences than the RCP8.5, which could be attributed to variations in precipitation and temperature intensity, timing, and frequency. This could potentially result in more frequent and severe floods and streamflow under RCP4.5, as well as drought and drier conditions under the RCP8.5 scenario, lowering the streamflow.

The projected long-term streamflow reductions were slightly higher in summer and autumn than in winter and spring for both emission scenarios (Fig. 7a). The long-term projection showed the greatest decrease, with a − 66.4% (RCP8.5) and − 42.0% (RCP4.5) reduction in streamflow during autumn. The declining patterns of the seasonal projections correspond to changes in precipitation and temperature. These findings are consistent with other studies in the Mediterranean climate that have found that annual streamflow in a watershed or on a regional scale is extremely sensitive to changes in precipitation, such that a slight decrease in precipitation in regions with high temperatures and consequently higher evapotranspiration rates would likely result in a significant reduction in runoff (Ficklin et al., 2013; Molina-Navarro et al., 2016, 2014). The results, especially the RCP8.5 long-term projection, have very strong implications for the water available in the river by the end of the century if the current global warming trends continue. A decline of more than 50% of the currently available streamflow would seriously affect the available water resources for the aquatic ecosystem, domestic consumption, and agricultural use.

3.2.3. Effects of climate change on nitrate load

The simulated future annual nitrate load decreased by − 21.7%, − 17.7%, and − 12.8% for RCP4.5 and − 20.5%, − 16.6%, and − 43.6%

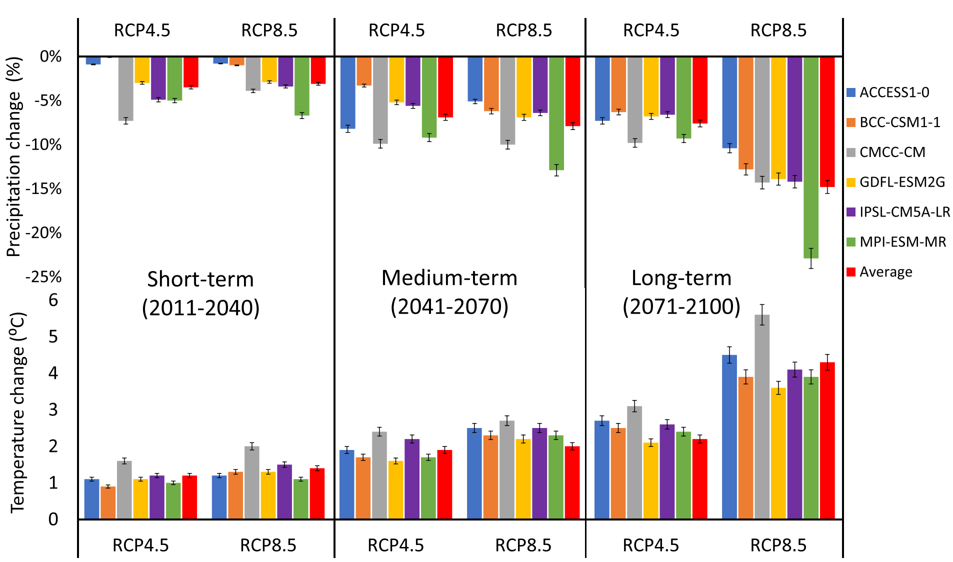

Fig. 5. Variation in average precipitation (%) and temperature changes (°C) for the six climate change models under RCP4.5 and RCP8.5 for short-, medium-, and long-term projections relative to historical reference.

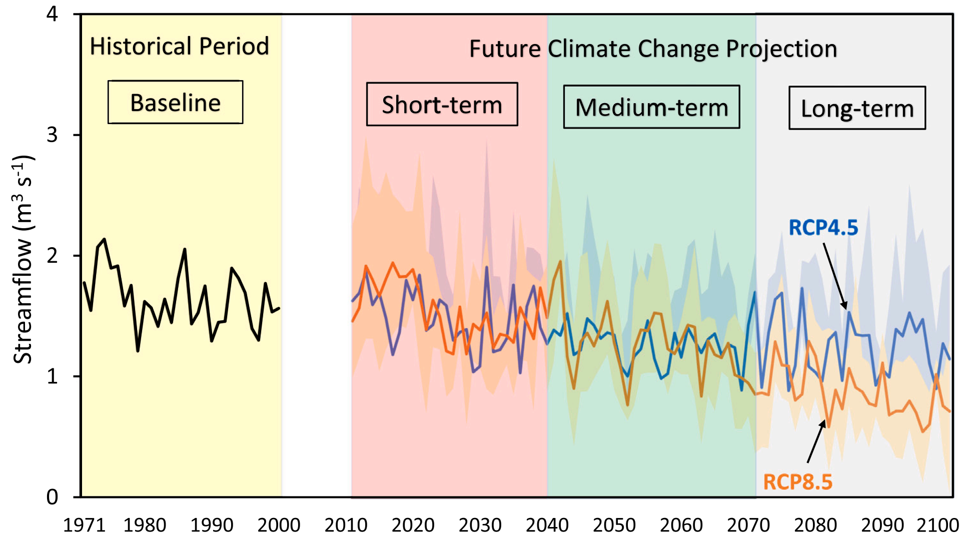

Fig. 6. Average annual streamflow evolution over historical (1971–2000), short-term (2011–2040), medium-term (2041–2070), and long-term (2071–2100) periods under RCP4.5 and RCP8.5 climate change projections.

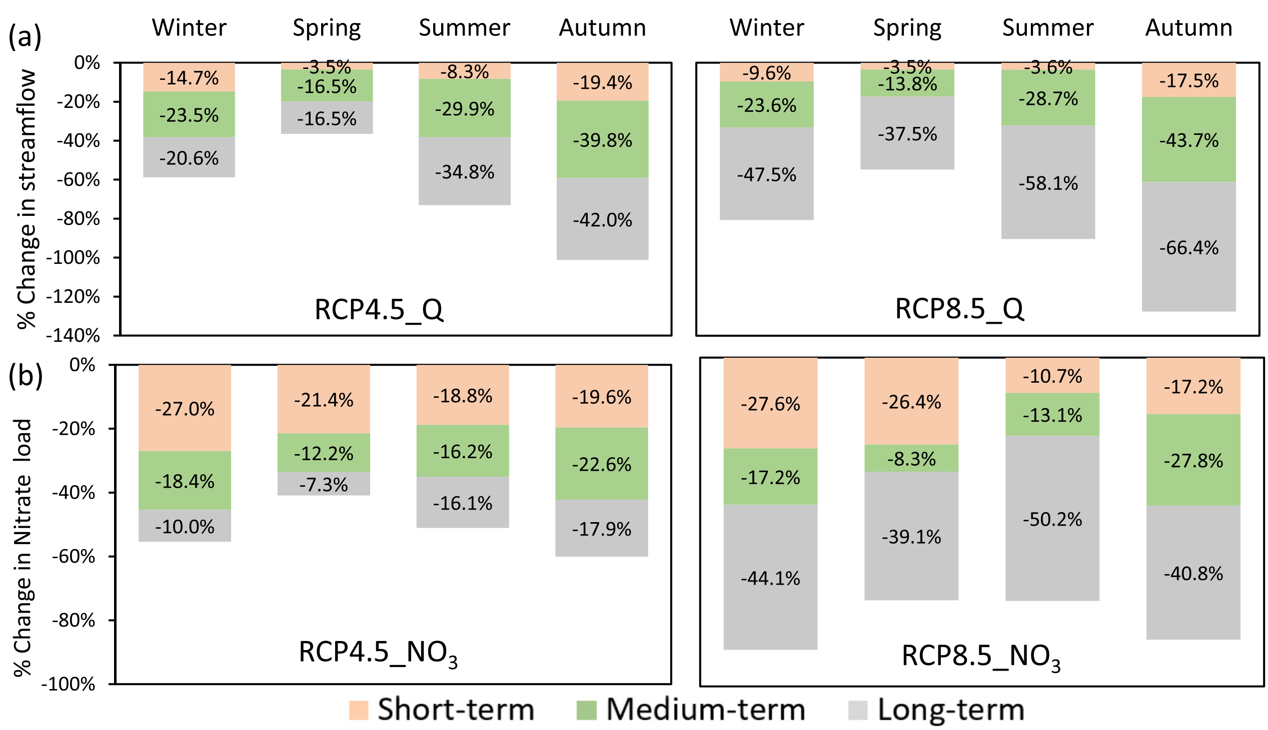

Fig. 7. Seasonal percent changes in projected future (a) streamflow and (b) nitrate load over the short-, medium-, and long-term periods under RCP4.5 and RCP8.5 emission scenarios relative to the historical reference.

for RCP 8.5 in the short-, medium-, and long- projections, respectively, compared to the historical reference period with very high interannual variability (Fig. 8). The short-term and medium-term nitrate load projections under RCP4.5 and RCP8.5 were quite similar. However, there was a considerable difference in the long-term projection, with RCP8.5 experiencing the greatest decline of − 43.6% compared to − 12.8 for RCP4.5. The decrease in projected nitrate load was primarily due to the reduction in projected streamflow. The statistical relationship between streamflow and nitrate load showed a good correlation (p-value <0.05) for RCP4.5 and RCP8.5 over the entire future projection period. This relationship indicates that streamflow, mainly driven by precipitation and temperature, would play an essential and critical role in determining the future nitrate export since it is the primary driving mechanism. However, the relationship between streamflow and nitrate load is not always linear, despite streamflow having the greatest influence. Other factors that could play a critical role in determining the sources, amounts, mobilization and transport pathways of nitrate in an agricultural watershed include precipitation amount and intensity, land cover type and practices, fertilization quantity and type, as well as cropping pattern and schedule (Boithias et al., 2014; Cameron et al., 2013; Parajuli and Risal, 2021). Additionally, the presence of buffer zones and riparian vegetation could help in minimizing nitrate mobilization and transport to surface waterways (Beaudoin et al., 2005). These results are consistent with other studies within the Mediterranean region

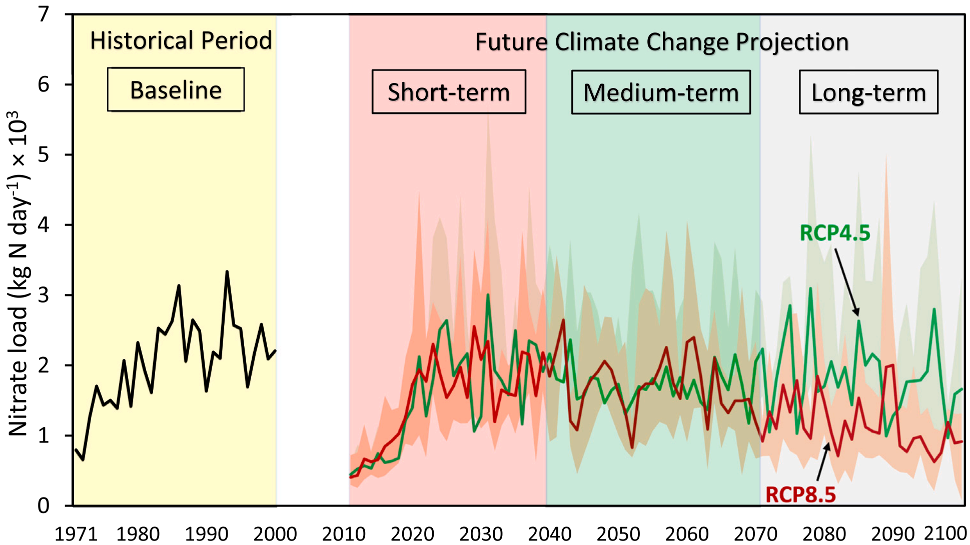

Fig. 8. Average annual nitrate load evolution over historical (1971–2000), short-term (2011–2040), medium-term (2041–2070), and long-term (2071–2100) periods under RCP4.5 and RCP8.5 climate change projections.

(Molina-Navarro et al., 2014) and the Iberian Peninsula (Carvalho-Santos et al., 2016) that have reported a decrease in nitrogen exportation due to the reduced streamflow. Mander et al. (2000) also found that reducing surface water runoff considerably reduced nitrate load exportation in cultivated areas.

The magnitude and frequency of nitrate load occurrence were greater in the long-term projection scenario of RCP4.5 than RCP8.5, similar to the streamflow results. This pattern could be due to the timing of agricultural management operations, such as fertilization, which could have coincided with heavy rainfall as well as more available water to transport the nitrates. The increase in frequency and severity of extreme precipitation events under RCP4.5 after the 2070 s in the Mediterranean region have been reported to result in increased streamflow and nitrate export (Almeida et al., 2022; Barredo et al., 2017; Giorgi and Lionello, 2008; Todaro et al., 2022).

Nitrate load decreased in all seasons and projected periods for both emission scenarios, with the greatest decrease occurring in summer for the RCP8.5 long-term projection (−50.2%) and autumn for the RCP4.5 medium-term projection (−22.6%) (Fig. 7b). This could be due to higher streamflow reduction and the aforementioned factors during the same period. However, the nitrate concentration is projected to rise by 4.1%, 34.8%, and 45.1% for RCP4.5% and 5.2%, 36.8%, 54.1% for RCP 8.5 in the short-, medium-, and long-term projections, respectively due to the faster streamflow decline than the nitrate exportation rate. This would result in the accumulation of more nitrates in the riverbed and soil, resulting in soil and groundwater pollution. Carvalho-Santos et al. (2016) observed an increasing trend in future nitrate concentration despite the declining nitrate loads attributed to declining streamflow. The projected increase in nitrate concentration at the end of the century will be of great concern as the current figures within the watershed already indicate a higher nitrate concentration. Findings by Merchán et al. (2020) have categorized the watershed as a "Nitrate Vulnerable Zone"; hence any increase in concentration is likely to exacerbate the problem further. Furthermore, increased nitrate concentration would increase eutrophication in the river, thus enhancing algae bloom and consequently degrading the water quality and resulting in higher water treatment costs (Tong et al., 2012). Table S2 in the supplementary materials summarizes all the climate change projection findings in terms of annual area-specific averages.

3.2.4. *Effect of climate change on agriculture*

Projected climate change is expected to heavily impact agricultural activities through changes in phenology and cropping cycle (Funes et al., 2016; Trnka et al., 2011), as well as higher water demands due to increased evapotranspiration (Saadi et al., 2015; Valverde et al., 2015). Consequently, crop yields are expected to decline, especially under the RCP8.5 long-term projection with no adaptation measures (Feyen et al., 2020), in addition to higher inter-annual variability and decreased production resilience (Zampieri et al., 2020). Reduced streamflow would greatly affect irrigation, particularly for corn and tomatoes grown in the study area, which relies heavily on irrigation. Extreme warming, as projected in the medium and long term, will shorten the growing seasons for most crops. According to Mougou et al. (2011), a temperature increase of 2.5 °C to 4 °C would shorten the growing period of wheat in the Mediterranean region by 16–30 days. Recurrent drought events could result in heavy agricultural losses of more than − 50% in irrigated areas and − 15% in rain-fed cereal production (Mougou et al., 2011). Other negative climate change impacts on agriculture in the region include the emergence of new and re-emerging crop pests and diseases, which would increase production losses, as well as increased wildfire incidences caused by the projected extreme weather. These factors would result in food insecurity and increased economic losses in the region and should thus be mitigated to limit the potential negative consequences.

3.2.5. *Climate change adaptation and mitigation measures*

Based on this study's findings, it's evident that climate change would negatively affect the agricultural areas in northern Spain and the Mediterranean region since water resources will be under great pressure and nitrate pollution of surface and groundwater will increase. As a result, robust agricultural policies, regulatory frameworks, and legislation aimed at climate change adaptation and mitigation would be required to minimize the potential negative impacts. The projected water scarcity and increased drought events would limit irrigation-based adaptation actions. However, agricultural management practices such as crop distribution, schedules, diversification, and rotation would be central to the adaptation strategy at the farm scale. Land use change by introducing drought-resistant crops could also improve climate change resilience. An effective adaptation and mitigation strategy would prioritize the following actions: (i) farming practices, which would include crop diversification, changing crop type and land use, and adjusting rotation patterns; (ii) water management practices, which would emphasize the need for technological development and innovation for crops and agricultural practices such as precision agriculture and modifying irrigation;

(iii) farm management practices that focus on diversifying income sources, such as the government establishing programs to ensure agricultural subsidies, the provision of insurance to farmers to stabilize their income, and agricultural financial assistance; (iv) agricultural management practices that focus on optimal nitrogen fertilization, the use of organic fertilizers, and soil health improvement. Nitrogen surplus in the soil can be reduced through efficient application according to the soil nitrogen availability and potential crop yield. Combining these adaptation and mitigation strategies would improve the potential for increasing or at least maintaining crop yield.

4. Conclusion

The effect of climate change on water resources and nitrate export is of major concern in the Mediterranean region and northern Spain due to its arid and semi-arid climatic conditions. This paper evaluated the adoption of the SWAT model for simulating current and future streamflow and nitrate loads under rainfed conditions in a Mediterranean agricultural area in northern Spain. The projected decline in precipitation and rise in temperature negatively impact both the streamflow and nitrate export on spatial and temporal scales. Reduced streamflow would reduce available water resources for agricultural and domestic use, resulting in lower agricultural yield, limited productivity, and conflicts over scarce water resources. Although the projected nitrate load would also decrease due to declining streamflow, the nitrate concentration levels are expected to rise due to the faster streamflow reduction rate than the nitrate load exportation rate, resulting in nitrate pollution by accumulation in the soil and riverbed as well as groundwater pollution through percolation.

This study's findings could help understand the scope of the climate change problem in northern Spain and develop appropriate adaptation and mitigation measures to help minimize the expected adverse effects. These measures could include more sustainable water resource management and better land management policies such as efficient nitrogen fertilization. These initiatives would be critical to adhere to the European Union's nitrate policies and legislation, as outlined in the Water Framework Directive and the Nitrate Directive. Despite this study's valuable findings, further research using a variety of climate models, ensembles, and other emission scenarios is needed to fully evaluate these impacts. More research could be done to help understand the scope and magnitude of the uncertainties by combining future climate, projected land uses, and population changes.

Declaration of Competing Interest

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

Data availability

Data will be made available on request.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agwat.2023.108378.

References

AdapteCCa, 2021, Platform on Adaptation to Climate Change in Spain [WWW Document]. Spanish Clim. Chang. Off. Biodivers. Found. URL ⟨https://escenarios.adaptecca.es/⟩ (accessed 11.2.21).

Cerro, I., Antigüedad, I., Srinivasan, R., Sauvage, S., Volk, M., Sanchez-Perez, J.M., 2014. Simulating land management options to reduce nitrate pollution in an agricultural watershed dominated by an alluvial aquifer. J. Environ. Qual. 43, 67–74. https://doi.org/10.2134/jeq2011.0393.

Chahor, Y., Casalí, J., Giménez, R., Bingner, R.L., Campo, M.A., Goñi, M., 2014. Evaluation of the AnnAGNPS model for predicting runoff and sediment yield in a small Mediterranean agricultural watershed in Navarre (Spain). Agric. Water Manag. 134, 24–37. https://doi.org/10.1016/j.agwat.2013.11.014.

Chirivella Osma, V., Capilla Romá, J.E., Pérez Martín, M.A., 2015. Modelling regional impacts of climate change on water resources: The Júcar Basin, Spain. Hydrol. Sci. J. 60, 30–49. https://doi.org/10.1080/02626667.2013.866711.

Donmez, C., Sari, O., Berberoglu, S., Cilek, A., Satir, O., Volk, M., 2020. Improving the applicability of the SWAT model to simulate flow and nitrate dynamics in a flat data-scarce agricultural region in the Mediterranean. Water 12, 1–24. https://doi.org/10.3390/w12123479.

Epelde, A.M., Cerro, I., Sánchez-Pérez, J.M., Sauvage, S., Srinivasan, R., Antigüedad, I., 2015. Application of the SWAT model to assess the impact of changes in agricultural management practices on water quality. Hydrol. Sci. J. 60, 825–843. https://doi.org/10.1080/02626667.2014.967692.

Estrela, T., Perez-Martin, M. a, Vargas, E., 2012. Impacts of climate change on water resources in Spain. Hydrol. Sci. J. -J. Des. Sci. Hydrol. 57, 1154–1167. https://doi.org/10.1080/02626667.2012.702213.

European Communities, 1991, Council Directive of 12 December 1991 concerning the protection of waters against pollution caused by nitrate from agricultural sources (91/676/EEC) (No. 91/676/EEC), Official Journal of the European Communities. European Communities.

European Communities, 2000, Directive 2000/60/EC of the European Parliament and the Council of 23 October 2000 establishing a framework for community action in the field of water policy (No. 2000/60/EC), Official Journal of the European Commission. European Communities.

Harraki, W.El, Ouazar, D., Bouziane, A., Harraki, I.El, 2021. Streamflow prediction upstream of a dam using SWAT and assessment of the impact of land use spatial resolution on model performance. Environ. Process. 8, 1165–1186. https://doi.org/10.1007/s40710-021-00532-0.

Hernández-García, I., Merchán, D., Aranguren, I., Casalí, J., Giménez, R., Campo-Bescós, M.A., Del Valle de Lersundi, J., 2020. Assessment of the main factors affecting the dynamics of nutrients in two rainfed cereal watersheds. Sci. Total Environ. 733, 1–48. https://doi.org/10.1016/j.scitotenv.2020.139177.

Intergovernmental Panel on Climate Change (IPCC), 2014. Climate change impacts, adaptation, and vulnerability. Part A: Global and sectoral aspects. Working Group II contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press,, United Kingdom and New York.

Abbaspour, K.C., 2015, SWAT-CUP: SWAT-Calibration and Uncertainty Programs (CUP) - A user manual, EAWAG Aquatic Research. EAWAG Swiss Federal Institute of Aquatic Science and Technology.

Abbaspour, K.C., Rouholahnejad, E., Vaghefi, S., Srinivasan, R., Yang, H., Kløve, B., 2015. A continental-scale hydrology and water quality model for Europe: Calibration and uncertainty of a high-resolution large-scale SWAT model. J. Hydrol. 524, 733–752. https://doi.org/10.1016/j.jhydrol.2015.03.027.

Abbaspour, K.C., Vaghefi, S.A., Srinivasan, R., 2018. A guideline for successful calibration and uncertainty analysis for soil and water assessment: A review of papers from the 2016 international SWAT conference. Water 10, 1–18. https://doi.org/10.3390/w10010006.

Abd-Elmabod, S.K., Muñoz-Rojas, M., Jordán, A., Anaya-Romero, M., Phillips, J.D., Laurence, J., Zhang, Z., Pereira, P., Fleskens, L., van der Ploeg, M., de la Rosa, D., 2020. Climate change impacts on agricultural suitability and yield reduction in a Mediterranean region. Geoderma 374, 1–14. https://doi.org/10.1016/j.geoderma.2020.114453.

Almagro, M., de Vente, J., Boix-Fayos, C., García-Franco, N., Melgares de Aguilar, J., González, D., Solé-Benet, A., Martínez-Mena, M., 2016. Sustainable land management practices as providers of several ecosystem services under rainfed Mediterranean agroecosystems. Mitig. Adapt. Strateg. Glob. Chang. 21 https://doi.org/10.1007/s11027-013-9535-2.

Al-Mukhtar, M., Qasim, M., 2019. Future predictions of precipitation and temperature in Iraq using the statistical downscaling model. Arab. J. Geosci. 12 https://doi.org/10.1007/s12517-018-4187-x.

Anand, V., Oinam, B., 2019. Future climate change impact on hydrological regime of river basin using SWAT model. Glob. J. Environ. Sci. Manag 5, 471–484. https://doi.org/10.22034/gjesm.2019.04.07.

Arabi, M., Frankenberger, J.R., Engel, B.A., Arnold, J.G., 2008. Representation of agricultural conservation practices with SWAT. Hydrol. Process. 22, 3042–3055. https://doi.org/10.1002/hyp.6890.

Casalí, J., Gastesi, R., Álvarez-Mozos, J., De Santisteban, L.M., Lersundi, J.D.V., de, Giménez, R., Larrañaga, A., Goñi, M., Agirre, U., Campo, M.A., López, J.J.,

Donézar, M., 2008. Runoff, erosion, and water quality of agricultural watersheds in central Navarre (Spain). Agric. Water Manag. 95, 1111–1128. https://doi.org/10.1016/j.agwat.2008.06.013.
FAO, 2017. The Future of Food and Agriculture - Trends and Challenges. FAO,, Rome.
Arnold, J.G., Kiniry, J.R., Srinivasan, R., Williams, J.R., Haney, E.B., Neitsch, S.L., 2012, Input/Output documentation Soil & Water Assessment Tool, version 2012. Texas Water Resources Insitute, TR-439, Texas.
Arora, N.K., 2019. Impact of climate change on agriculture production and its sustainable solutions. Environ. Sustain. 2, 95–96. https://doi.org/10.1007/s42398-019-00078-w.
Barredo, J.I., Caudullo, J.I., Mauri, G., 2017. Mediterranean habitat loss under RCP4.5 and RCP8.5 climate change projections, European Commision; JRC Technical Reports. Luxembourg. https://doi.org/10.2760/622174.
Beaudoin, N., Saad, J.K., Van Laethem, C., Machet, J.M., Maucorps, J., Mary, B., 2005. Nitrate leaching in intensive agriculture in Northern France: Effect of farming practices, soils, and crop rotations. Agric. Ecosyst. Environ. 111, 292–310. https://doi.org/10.1016/j.agee.2005.06.006.
Boardman, J., Poesen, J., 2006. Soil erosion in Europe. John Wiley Sons, Ltd. Br. Libr. https://doi.org/10.1002/0470859202.
Boithias, L., Srinivasan, R., Sauvage, S., Macary, F., Sánchez-Pérez, J.M., 2014. Daily nitrate losses: Implication on long-term river quality in an intensive agricultural catchment of Southwestern France. J. Environ. Qual. 43, 1–10. https://doi.org/10.2134/jeq2011.0367.
Cameron, K.C., Di, H.J., Moir, J.L., 2013. Nitrogen losses from the soil/plant system: A review. Ann. Appl. Biol. https://doi.org/10.1111/aab.12014.
Candela, L., Tamoh, K., Olivares, G., Gomez, M., 2012. Modelling impacts of climate change on water resources in ungauged and data-scarce watersheds. Application to the Siurana catchment (NE Spain). Sci. Total Environ. 440, 253–260. https://doi.org/10.1016/j.scitotenv.2012.06.062.
Carvalho-Santos, C., Nunes, J.P., Monteiro, A.T., Hein, L., Honrado, J.P., 2016. Assessing the effects of land cover and future climate conditions on the provision of hydrological services in a medium-sized watershed of Portugal. Hydrol. Process. 30, 720–738. https://doi.org/10.1002/hyp.10621.
European Environment Agency (EEA), 2017, Climate Change, impacts and vulnerability in Europe 2016. An indicator-based report. EEA Report No1/2017. Copenhagen. https://doi.org/10.2800/534806.
FAO, IWMI, 2018, More people, more food, worse water? A global review on water pollution from agriculture. FAO and IWMI, Rome and Colombo.
Ferrant, S., Oehler, F., Durand, P., Ruiz, L., Salmon-Monviola, J., Justes, E., Dugast, P., Probst, A., Probst, J.L., Sanchez-Perez, J.M., 2011. Understanding nitrogen transfer dynamics in a small agricultural catchment: Comparison of a distributed (TNT2) and a semi distributed (SWAT) modeling approaches. J. Hydrol. 406, 1–15. https://doi.org/10.1016/j.jhydrol.2011.05.026.
Feyen, L., Ciascar, J., Gosling, S., Ibarreta, D., Soria, A., Dosio, A., Naumann, G., Russo, S., Formetta, G., Forzieri, G., Girardello, M., Spinoni, J., Mentaschi, L., Bisselink, B., Bernhard, J., Gelati, E., Adamovic, M., Guenther, S., de Roo, A., Cammalleri, C., Dottori, F., Bianchi, A., Alfieri, L., Vousdoukas, M., Mongelli, I., Hinkel, J., Ward, P., Costa, H., de Rigo, D., Libertà, G., Houston Durrant, T., San-Miguel-Ayanz, J., Barredo, J., Mauri, A., Caudullo, G., Ceccherini, G., Mauri, A., Beck, P., Cescatti, A., Hristov, J., Toreti, A., Pérez Domínguez, I., Dentener, F., Fellmann, T., C., Elleby, Ceglar, A., Fumagalli, D., Niemeyer, S., Cerrani, I., Panarello, L., Bratu, M., Després, J., Adamovic, M., Szewczyk, W., Matei, A., Mulholland, E., Olariaga, M., 2020. Climate change impacts and adaptation in Europe: JRC Science for policy report. JRC PESETA IV final report. Publ. Off. Eur. Union. Luxemb. https://doi.org/10.2760/171121.
Ficklin, D.L., Luo, Y., Luedeling, E., Gatzke, S.E., Zhang, M., 2010. Sensitivity of agricultural runoff loads to rising levels of CO_2 and climate change in the San Joaquin Valley watershed of California. Environ. Pollut. 158, 223–234. https://doi.org/10.1016/j.envpol.2009.07.016.
Ficklin, D.L., Stewart, I.T., Maurer, E.P., 2013. Effects of projected climate change on the hydrology in the Mono Lake Basin, California. Clim. Change 116, 1–21. https://doi.org/10.1007/s10584-012-0566-6.
Fonseca, A.R., Santos, J.A., 2019. Predicting hydrologic flows under climate change: The Tâmega Basin as an analog for the Mediterranean region. Sci. Total Environ. 668, 1013–1024. https://doi.org/10.1016/j.scitotenv.2019.01.435.
Funes, I., Aranda, X., Biel, C., Carbó, J., Camps, F., Molina, A.J., de Herralde, F., Grau, B., Savé, R., 2016. Future climate change impacts on apple flowering date in a Mediterranean subbasin. Agric. Water Manag. 164, 19–27. https://doi.org/10.1016/j.agwat.2015.06.013.
Giménez, R., Casalí, J., Grande, I., Díez, J., Campo-Bescós, M.A., Álvarez-Mozos, J., Goñi, M., 2012. Factors controlling sediment export in a small agricultural watershed in Navarre (Spain). Agric. Water Manag. 110, 1–8. https://doi.org/10.1016/j.agwat.2012.03.007.
Giorgi, F., Lionello, P., 2008. Climate change projections for the Mediterranean region. Glob. Planet. Change 63, 90–104. https://doi.org/10.1016/J.GLOPLACHA.2007.09.005.
Green, C.H., van Griensven, A., 2008. Autocalibration in hydrologic modeling: Using SWAT2005 in small-scale watersheds. Environ. Model. Softw. 23, 422–434. https://doi.org/10.1016/j.envsoft.2007.06.002.
IUSS Working Group WRB, 2015, World reference base for soil resources 2014: International soil classification system for naming soils and creating legends for soil maps, update 2015, World Soil Resources Reports No. 106. Food and Agriculture Organization (FAO) of the United Nations, Rome, Italy. https://doi.org/10.1017/S0014479706394902.
Jimeno-Sáez, P., Senent-Aparicio, J., Pérez-Sánchez, J., Pulido-Velazquez, D., 2018. A comparison of SWAT and ANN Models for daily runoff simulation in different climatic zones of Peninsular Spain. Water 10, 1–19. https://doi.org/10.3390/w10020192.
Kamali, B., Abbaspour, K.C., Yang, H., 2017. Assessing the uncertainty of multiple input datasets in the prediction of water resource components. Water 9, 1–16. https://doi.org/10.3390/w9090709.
Kouchi, D.H., Esmaili, K., Faridhosseini, A., Sanaeinejad, S.H., Khalili, D., Abbaspour, K. C., 2017. Sensitivity of calibrated parameters and water resource estimates on different objective functions and optimization algorithms. Water 9. https://doi.org/10.3390/w9060384.
Krysanova, V., Vetter, T., Eisner, S., Huang, S., Pechlivanidis, I., Strauch, M., Gelfan, A., Kumar, R., Aich, V., Arheimer, B., Chamorro, A., Van Griensven, A., Kundu, D., Lobanova, A., Mishra, V., Plötner, S., Reinhardt, J., Seidou, O., Wang, X., Wortmann, M., Zeng, X., Hattermann, F.F., 2017. Intercomparison of regional-scale hydrological models and climate change impacts projected for 12 large river basins worldwide - a synthesis. Environ. Res. Lett. 12, 1–12. https://doi.org/10.1088/1748-9326/aa8359.
Lam, Q.D., Schmalz, B., Fohrer, N., 2009. Ecohydrological modelling of water discharge and nitrate loads in a mesoscale lowland catchment, Germany. Adv. Geosci. 21, 45–55. https://doi.org/10.5194/adgeo-21-49-2009.
Liu, R., Zhang, P., Wang, X., Chen, Y., Shen, Z., 2013. Assessment of effects of best management practices on agricultural non-point source pollution in Xiangxi River watershed. Agric. Water Manag. 117, 9–18. https://doi.org/10.1016/j.agwat.2012.10.018.
Luz Rodríguez-Blanco, M., Mercedes Taboada-Castro, M., Arias, R., Teresa Taboada-Castro, M., 2019, Assessing the expected impact of climate change on nitrate load in a small Atlantic agro-forested catchment, in: Climate Change and Global Warming. pp. 11–24. https://doi.org/10.5772/intechopen.80709.
Majone, B., Bovolo, C.I., Bellin, A., Blenkinsop, S., Fowler, H.J., 2012. Modeling the impacts of future climate change on water resources for the Gallego River basin (Spain). Water Resour. Res. 48, 1–18. https://doi.org/10.1029/2011WR010985.
Mander, Ü., Kull, A., Kuusemets, V., Tamm, T., 2000. Nutrient runoff dynamics in a rural catchment: Influence of land-use changes, climatic fluctuations and ecotechnological measures. Ecol. Eng. 14, 405–417. https://doi.org/10.1016/S0925-8574(99)00064-6.
Martínková, M., Hesse, C., Krysanova, V., Vetter, T., Hanel, M., 2011. Potential impact of climate change on nitrate load from the Jizera catchment (Czech Republic). Phys. Chem. Earth 36, 673–683. https://doi.org/10.1016/j.pce.2011.08.013.
Meaurio, M., Zabaleta, A., Uriarte, J.A., Srinivasan, R., Antigüedad, I., 2015. Evaluation of SWAT model's performance to simulate streamflow spatial origin. The case of a small forested watershed. J. Hydrol. 525, 326–334. https://doi.org/10.1016/j.jhydrol.2015.03.050.
Menció, A., Mas-Pla, J., Otero, N., Regàs, O., Boy-Roura, M., Puig, R., Bach, J., Domènech, C., Zamorano, M., Brusi, D., Folch, A., 2016. Nitrate pollution of groundwater; all right., but nothing else? Sci. Total Environ. 539, 241–251. https://doi.org/10.1016/j.scitotenv.2015.08.151.
Merchán, D., Casalí, J., Del Valle de Lersundi, J., Campo-Bescós, M.A., Giménez, R., Preciado, B., Lafarga, A., 2018. Runoff, nutrients, sediment and salt yields in an irrigated watershed in southern Navarre (Spain). Agric. Water Manag. 195, 120–132. https://doi.org/10.1016/j.agwat.2017.10.004.
Merchán, D., Luquin, E., Hernández-García, I., Campo-Bescós, M.A., Giménez, R., Casalí, J., Del Valle de Lersundi, J., 2019. Dissolved solids and suspended sediment dynamics from five small agricultural watersheds in Navarre, Spain: A 10-year study. Catena 173, 114–130. https://doi.org/10.1016/j.catena.2018.10.013.
Merchán, D., Sanz, L., Alfaro, A., Pérez, I., Goñi, M., Solsona, F., Hernández-García, I., Pérez, C., Casalí, J., 2020. Irrigation implementation promotes increases in salinity and nitrate concentration in the lower reaches of the Cidacos River (Navarre, Spain. Sci. Total Environ. 706, 1–11. https://doi.org/10.1016/j.scitotenv.2019.135701.
Molina-Navarro, E., Trolle, D., Martínez-Pérez, S., Sastre-Merlín, A., Jeppesen, E., 2014. Hydrological and water quality impact assessment of a Mediterranean limno-reservoir under climate change and land use management scenarios. J. Hydrol. 509, 354–366. https://doi.org/10.1016/j.jhydrol.2013.11.053.
Molina-Navarro, E., Hallack-Alegría, M., Martínez-Pérez, S., Ramírez-Hernández, J., Mungaray-Moctezuma, A., Sastre-Merlín, A., 2016. Hydrological modeling and climate change impacts in an agricultural semiarid region. Case study: Guadalupe River basin. Mex. Agric. Water Manag. 175, 29–42. https://doi.org/10.1016/j.agwat.2015.10.029.
Moriasi, D.N., Arnold, J.G., Van Liew, M.W., Bingner, R.L., Harmel, R.D., Veith, T.L., 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. Trans. ASABE 50, 885–900. https://doi.org/10.13031/2013.23153.
Mougou, R., Mansour, M., Iglesias, A., Chebbi, R.Z., Battaglini, A., 2011. Climate change and agricultural vulnerability: A case study of rain-fed wheat in Kairouan, Central Tunisia. Reg. Environ. Chang. 11, 137–142. https://doi.org/10.1007/s10113-010-0179-4.
Navarra, A., Tubiana, L., 2013. Regional assessment of climate change in the Mediterranean. Volume 3: Case studies. Advances in Global Change Research, 1st ed.,. Springer,, Dordrecht. https://doi.org/10.1007/978-94-007-5769-1.
Neitsch, S.L., Arnold, J.G., Kiniry, J.R., Williams, J.R., 2011. Soil & Water Assessment Tool theoretical documentation, version 2009, Texas Water Resources Institute. Texas.
Niraula, R., Meixner, T., Norman, L.M., 2015. Determining the importance of model calibration for forecasting absolute/relative changes in streamflow from LULC and climate changes. J. Hydrol. 522, 439–451. https://doi.org/10.1016/j.jhydrol.2015.01.007.
Oduor, B.O., Campo-Bescós, M.Á., Lana-Renault, N., Echarri, A.A., Casalí, J., 2023. Evaluation of the impact of changing from rainfed to irrigated agriculture in a

Mediterranean watershed in Spain. Agriculture 13, 1–18. https://doi.org/10.3390/agriculture13010106.
Parajuli, P.B., Risal, A., 2021. Evaluation of climate change on streamflow, sediment, and nutrient load at watershed scale. Climate 9, 1–14. https://doi.org/10.3390/cli9110165.
Qi, J., Wang, Q., Zhang, X., 2019. On the use of NLDAS2 weather data for hydrologic modeling in the Upper Mississippi River basin. Water 11, 1–18. https://doi.org/10.3390/w11050960.
Rostamian, R., Jaleh, A., Afyuni, M., Mousavi, S.F., Heidarpour, M., Jalalian, A., Abbaspour, K.C., 2008. Application of a SWAT model for estimating runoff and sediment in two mountainous basins in central Iran. Hydrol. Sci. J. 53, 977–988. https://doi.org/10.1623/hysj.53.5.977.
Rouholahnejad, E., Abbaspour, K.C., Srinivasan, R., Bacu, V., Lehmann, A., 2014. Water resources of the Black Sea basin at high spatial and temporal resolution. Water Resour. Res. 50, 5866–5885. https://doi.org/10.1002/2013WR014132.
Saadi, S., Todorovic, M., Tanasijevic, L., Pereira, L.S., Pizzigalli, C., Lionello, P., 2015. Climate change and Mediterranean agriculture: Impacts on winter wheat and tomato crop evapotranspiration, irrigation requirements and yield. Agric. Water Manag. 147, 1–13. https://doi.org/10.1016/j.agwat.2014.05.008.
Somot, S., Sevault, F., Déqué, M., Crépon, M., 2008. 21st century climate change scenario for the Mediterranean using a coupled atmosphere–ocean regional climate model. Glob. Planet. Change 63, 112–126. https://doi.org/10.1016/j.gloplacha.2007.10.003.
Sutton, M.A., Howard, C.M., Erisman, J.W., 2011. The European Nitrogen Assessment - Sources, Effects and Policy Perspectives. Cambridge University Press,. https://doi.org/10.1017/CBO9780511976988.
Todaro, V., D'Oria, M., Secci, D., Zanini, A., Tanda, M.G., 2022. Climate change over the Mediterranean region: Local temperature and precipitation variations at five pilot sites. Water (Switz.) 14, 1–24. https://doi.org/10.3390/w14162499.
Tolson, B.A., Shoemaker, C.A., 2007. Cannonsville Reservoir watershed SWAT2000 model development, calibration and validation. J. Hydrol. 337, 68–86. https://doi.org/10.1016/j.jhydrol.2007.01.017.
Tong, S.T.Y., Sun, Y., Ranatunga, T., He, J., Yang, Y.J., 2012. Predicting plausible impacts of sets of climate land land use change scenarios on water resources. Appl. Geogr. 32, 477–489. https://doi.org/10.1016/j.apgeog.2011.06.014.
Trnka, M., Olesen, J.E., Kersebaum, K.C., Skjelvåg, A.O., Eitzinger, J., Seguin, B., Peltonen-Sainio, P., Rötter, R., Iglesias, A., Orlandini, S., Dubrovský, M., Hlavinka, P., Balek, J., Eckersten, H., Cloppet, E., Calanca, P., Gobin, A., Vučetić, V., Nejedlik, P., Kumar, S., Lalic, B., Mestre, A., Rossi, F., Kozyra, J., Alexandrov, V., Semerádová, D., Žalud, Z., 2011. Agroclimatic conditions in Europe under climate change. Glob. Chang. Biol. 17, 2298–2318. https://doi.org/10.1111/j.1365-2486.2011.02396.x.
Valverde, P., Serralheiro, R., de Carvalho, M., Maia, R., Oliveira, B., Ramos, V., 2015. Climate change impacts on irrigated agriculture in the Guadiana river basin (Portugal). Agric. Water Manag. 152, 17–30. https://doi.org/10.1016/j.agwat.2014.12.012.
Vargas-Amelin, E., Pindado, P., 2014. The challenge of climate change in Spain: Water resources, agriculture and land. J. Hydrol. 518, 243–249. https://doi.org/10.1016/j.jhydrol.2013.11.035.
Wang, S., Kang, S., Zhang, L., Li, F., 2008. Modelling hydrological response to different land-use and climate change scenarios in the Zamu River basin of Northwest China. Hydrol. Process. 22, 2502–2510. https://doi.org/10.1002/hyp.6846.
WHO, 2017. Guidelines for Drinking-water Quality: Fourth Edition Incorporating the first Addendum. World Health Organization,, Geneva.
Zabaleta, A., Meaurio, M., Ruiz, E., Antigüedad, I., 2014. Simulation climate change impact on runoff and sediment yield in a small watershed in the Basque Country, Northern Spain. J. Environ. Qual. 43, 235–245. https://doi.org/10.2134/jeq2012.0209.
Zampieri, M., Toreti, A., Ceglar, A., Naumann, G., Turco, M., Tebaldi, C., 2020. Climate resilience of the top ten wheat producers in the Mediterranean and the Middle East. Reg. Environ. Chang. 20, 1–9. https://doi.org/10.1007/s10113-020-01622-9.
Zettam, A., Taleb, A., Sauvage, S., Boithias, L., Belaidi, N., Sanchez-Perez, J.M., 2020. 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. 192, 1–17. https://doi.org/10.1007/s10661-020-08482-0.