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Application of the SWAT model to assess the impact of changes in agricultural management practices on water quality

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Abstract An excessive use of nitrogen in agricultural regions leads to nitrate pollution of surface and groundwater systems. The Alegria River watershed (Basque Country, northern Spain) is an agricultural area dominated by a Quaternary shallow aquifer that has suffered nitrate-related problems since the 1990s. Our objective was to use the SWAT hydrological water quality model for long-term backward simulation (1990–2011) considering main changes in management practices to determine their impact on water quality. Hydrology, crop yield, nitrogen losses and soil nitrogen budgets were simulated satisfactorily. Nitrogen budgets indicated that annual N inputs exceed outputs (which consider main N loss pathways), resulting in mean N surpluses of 114 and 65 kg ha^{-1} year^{-1} in the periods 1990–1999 and 2000–2011, respectively. In the long-term, trends in N surplus generally follow those of fertilization input, which directly affect groundwater nitrate concentration. The characteristics of the aquifer and non-point source pollution have enabled us to properly simulate the historical trends in N concentration in the Vitoria-Gasteiz aquifer.

Key words SWAT; soil N budget; nitrogen losses; agricultural practices; long-term simulation; nitrates

Application du modèle SWAT à l'évaluation de l'impact des modificationss des pratiques agricoles sur la qualité de l'eau

Resumé Une utilisation excessive de l'azote dans les régions agricoles conduit à la pollution par les nitrates des eaux superficielles et souterraines. Le bassin versant de la rivière Alegria (Pays Basque, Nord de l'Espagne) est une zone agricole dont l'aquifère quaternaire peu profond souffre depuis les années 1990 de ce type de problème. Notre objectif a été d'utiliser le modèle hydrologique de qualité de l'eau SWAT pour simuler l'historique de cette pollution (1990–2011), en prenant en compte les principaux changements de pratiques culturales, afin de déterminer l'impact de ces modifications sur la qualité de l'eau. L'hydrologie, les rendements agricoles, les pertes d'azote et les bilans d'azote du sol ont été simulés de manière satisfaisante. Les bilans d'azote ont indiqué que les apports annuels de N dépassaient les sorties (qui sont considérées comme les principales sources de pertes). Le résultat est un excédent moyen de 114 kg ha^{-1} an^{-1} pour la période 1990–1999 et de 65 pour la période 2000–2011. Les tendances à long terme des excédents d'azote sont en accord avec les apports de fertilisants, qui affectent directement la concentration en nitrates des eaux souterraines. A partir des caractéristiques de l'aquifère et des apports diffus de pollution d'origine agricole, il a donc été possible de simuler correctement l'évolution à long terme des teneurs en azote de l'aquifère Vitoria-Gasteiz.

Mots clefs SWAT ; bilan de N du sol ; pertes d'azote ; pratiques agricoles ; simulation à long terme ; nitrates

1 INTRODUCTION

In regions with intense agricultural management, surface water and groundwater are usually affected by anthropogenic pollution resulting from the use of high doses of pesticides and fertilizers and inadequate irrigation techniques. Although in such regions nitrate leaching seems to be an inevitable process, an improvement in management practices leading to a higher N fertilizer use efficiency is thought to reduce the potential for nitrate contamination of groundwater (Bijay-Singh and Sekhon 1995). Nowadays, particular attention is being focused on groundwater quality, especially in regions where it is the main source of drinking water. Current regulations, such as the European Water Framework Directive (EC 2000), recognize and attempt to address this problem. In fact, there are several hazards related to nitrate pollution of waters, from health hazards linked to consumption of nitrate-bearing water, to the proliferation of toxic algae and hypoxia (Exner *et al.* 2010).

The environmental impact of agricultural practices depends on many different factors, such as crop type, hydrometeorological conditions (climatology and hydrogeology), crop management practices and soil characteristics (Jégo *et al.* 2008). Several authors have demonstrated the effect of different land covers on the hydrology of watersheds (Pikounis *et al.* 2003), a factor that is also directly linked to the nutrient transport within a watershed, especially within the root zone. In lowland watersheds where the water table is quite shallow, groundwater transport plays a key role in the transport of pollutants from the soils into the water system (Wriedt and Rode 2006) and, particularly during flood events, streams are at risk of contamination due to the close connection between them and the aquifer (Cerro *et al.* 2014a).

The Alegria River watershed (Basque Country, northern Spain) is a lowland area with extensive agricultural land use. It is dominated by a Quaternary aquifer with a quite shallow groundwater table, making this watershed especially vulnerable to groundwater pollution (Schmalz *et al.* 2007, Lam *et al.* 2011). Almost the whole study area was designated as a Vulnerable Zone for nitrate pollution from agricultural sources at the end of the 1990s, in line with the European Directive (EC 1991). With the aim of reducing the pollution, various measures were taken, including the establishment of a Code of Good Practices and changes in the origin of irrigation water. These measures led to a decrease in groundwater N concentration and, in turn, a decrease in the N concentration in the river. It should also be noted, though, that groundwater concentration has a slow response to fertilizer applied (Exner *et al.* 2010) and, hence, the decrease in N input to the system was appreciated some time after regulations were introduced. Nowadays, the limit value for nitrates (50 mg L^{-1} according to the European Directive) is exceeded at some points in the aquifer but rarely in the river (Cerro 2013).

Due to the multiple processes involved in the dynamics of pesticides and fertilizers, modelling is considered extremely valuable as it can help to quantify the pollution, making balances at the scale of the watershed and guiding decisions to improve management (Jégo *et al.* 2008, Cerro *et al.* 2014a). In this study, the SWAT model (Soil and Water Assessment Tool) was used, being considered one of the most useful models for long-term simulation in predominantly agricultural watersheds (Borah and Bera 2003), and robust in predicting nutrient losses at the watershed scale (Gassman *et al.* 2007, Ferrant *et al.* 2011, Cerro *et al.* 2014a).

There are many studies on nitrate pollution of agricultural watersheds that have been carried out with the SWAT model. They are usually focused on pollution mitigation scenarios by changes in land use (Wang *et al.* 2008, Ferrant *et al.* 2013), fertilization doses (Ferrant *et al.* 2013, Liu *et al.* 2013, Boithias *et al.* 2014, Cerro *et al.* 2014b) and other management practices, such as the burial of straw, tillage intensity or fertilization dosage (Ferrant *et al.* 2013, Liu *et al.* 2013, Cerro *et al.* 2014b) by applying a previously calibrated and validated model. Although some studies consider varying conditions on agricultural management obtained from interpolation (Bracmort *et al.* 2006) or from tools like Land Use Update and Soil Assessment (Koch *et al.* 2012), most of the studies analyse the effect of alternative scenarios implemented all at once. In any case, calibration and validation of the model are usually performed before the implementation of the scenarios of interest. In this study, however, we simulate varying management practices since the year 1986. Although we calibrate and validate the model for a period in which there is plenty of available field data, we also carry out a validation of other two previous periods, which we consider necessary as the results obtained during them is significant for the conclusion of this work.

The main objectives of this study were to: (a) test how well the SWAT model connects nutrient surpluses and groundwater pollution with the river, (b) perform a long-term (1990–2011) simulation taking into account various changes in agricultural practices to see how nutrient dynamic was affected over the study period; and (c) assess how well the model simulates the nitrate concentration within the aquifer. The fact that the study period was when important agricultural management changes were registered helps us to identify the aspects which most influence water quality. These aspects should be carefully regarded in the future when new agricultural management legislation is being implemented.

2 MATERIALS AND METHODS

2.1 Study area

The Alegria River watershed (Fig. 1) is a sub-basin (115 km^2) of the Ebro River basin and it is located in the Basque Country (northern Spain). The underlying materials in the lowland are Quaternary fluvial and alluvial deposits (a Quaternary aquifer) lying over impermeable marls. The average thickness of the Quaternary formation is 5 m. The elevation of the watershed ranges from 506 to 1098 m a.s.l., with a mean of 613 m a.s.l. The average annual precipitation is 650 mm and seasonal variability is significant. Autumn is the rainiest season, registering around 30% of the annual precipitation. High temperature variability is observed on both annual and daily scales (daily means ranging from 0 to 25ºC, with changes within a day by as much as 20ºC). River discharge was measured at the outlet of the study area. The average discharge in Alegria River for the period 2009–2011 was 0.32 m^3 s^{-1}, the annual mean water outflow being 11 hm^3. Observed minimum and maximum discharges were 0.02 and 9.55 m^3 s^{-1}, respectively. The water of the upper part of the watershed is routed to a reservoir outside the watershed through the Alegria channel (Fig. 1). This study is focused on the area downstream of the channel (53 km^2) as the channel is believed to route all the surface water from the upper part. The alluvial land is characterized by high clay content soils, resulting in a high water retention capacity, a property that makes them suitable for cultivation (Cerro 2013). Approximately 70% of the studied area is in agricultural use, the remaining part being covered by forest and pastures (Table 1). The main crops in the area are rainfed grains (wheat, oats and barley) and irrigated sugar beet and potato.

Table 1 Land-use classification in the Alegria River watershed.

Land-use type	Area (km^2)	Area (%)
Pasture	3.5	6.6
Deciduous forest	9.2	17.2
Agricultural land	37.1	69.4
Water	0.4	0.8
Urban	3.2	6
	53.4	*100*

2.2 SWAT model description

The SWAT model is a semi-distributed basin-scale model developed by the United States Department of Agriculture (USDA). In this study we used the ArcSWAT interface (ArcSWAT Version 2009.93.7 b)

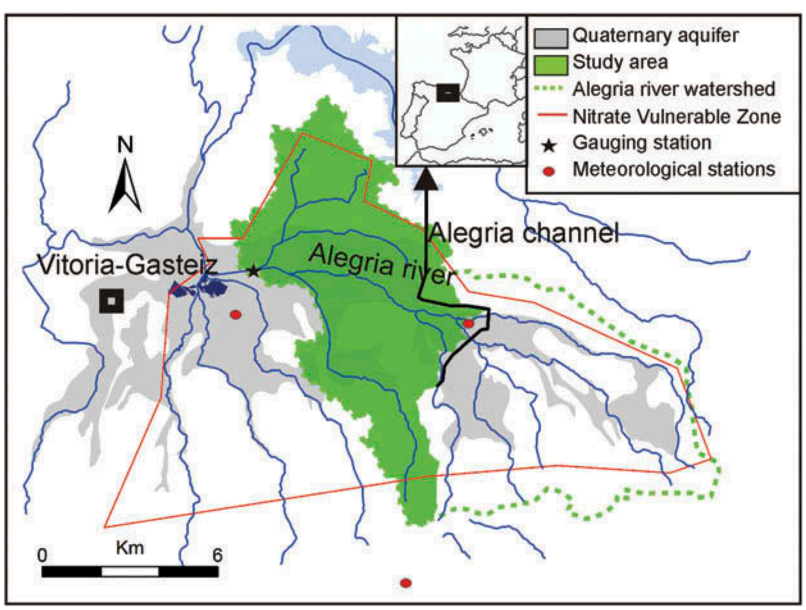


Fig. 1 Location of the study area and positions of gauging and meteorological stations in the Alegria River watershed.

for ArcGIS (Winchell *et al.* 2007). The SWAT model can predict long-term impacts of land use and agricultural management on water (Arnold *et al.* 1998). It is process-based, operates at daily and monthly time steps, and it includes hydrological, chemical, ecological and management practices modules.

The SWAT model divides the watershed into sub-basins connected by a stream network, and further delineates each sub-basin into hydrologic response units (HRUs), which consist of unique combinations of land cover, slope and soil type. SWAT simulates each HRU separately and calculates daily water balances. It is assumed that there is no interaction between HRUs (Srinivasan *et al.* 2010). The model considers multiple hydrological processes occurring in the soil: infiltration, evapotranspiration, percolation into a deeper aquifer and water losses by runoff, as well as lateral and groundwater flow. It also simulates N transport and transformation at HRU scale considering the basic processes of denitrification, volatilization and plant uptake. SWAT distinguishes five different pools for mineral and organic N. Channel water routing can be calculated with different variations of the kinematic wave model. SWAT uses Manning's equation to define the rate and velocity of flow. The water balance components within the reach segment and, in turn, the outflow to the next reach are calculated considering all the losses (via evaporation, transmission, return flow from bank storage and diversions). It is also possible to model nutrient transformations within the channel network. More detailed information about the SWAT model can be found in Neitsch *et al.* (2011).

The SWAT model has been widely calibrated and validated through the comparison of observed and simulated streamflow data, as well as nutrient and sediment loads at watershed outlets. However, publications related to the vegetation growth module are not so abundant (Nair *et al.* 2011), despite the fact that its proper calibration is considered an essential factor for good performance of the model. The importance of the correct simulation of vegetation growth lies in its influence on water and nutrient balances of a system, especially for agricultural watersheds. The SWAT model's crop growth module is based on the Erosion/Productivity Impact Calculator model (EPIC; Williams *et al.* 1989).

2.3 Input data for SWAT

The main inputs for the SWAT model are meteorological data, elevation, soil and land-use maps. Meteorological data used were obtained from the Basque Meteorology Agency (Euskalmet) and consist of precipitation, temperature, wind, solar radiation and humidity from three meteorological stations: two (Alegria-C056 and Arkaute-C001) located within the watershed at elevations below 550 m, and the third (Kapildui-C047) at a site about 2 km to the south at an elevation of 1173 m (Fig. 1). We believe that it is useful to consider meteorological information from this station located outside of the watershed due to the differences in its data (overall, higher precipitation and lower temperatures) compared to those from the other stations. The digital elevation model (LIDAR 2008, 5 × 5 m) was obtained from the Geoeuskadi website (www.geo.euskadi.net). For each soil represented in the soil map (Iñiguez *et al.* 1980), an average texture classification was obtained from the data developed by the Corporation of the Basque Government for the Rural and Marine Environment (HAZI) (Table 2). The most dominant soil types in the area are vertisol, cambisol and rendzina, and their general texture has been classified as a combination of loam, silty loam and clay loam, respectively. Aquifer material is characterized by a higher sand content and higher saturated conductivity (Table 2). The bulk density (BD), available water content (AWC) and saturated conductivity (Ksat) values were obtained from the Soil Water Characteristics Program developed by the USDA (Saxton and Rawls 2009). From the data provided by the Basque Institute for Agricultural Research and Developmet (Neiker-Tecnalia), organic carbon content (OC) was estimated to be 0.5% for all soil classes. According to the input data, the watershed was discretized into 66

Table 2 Soil classes and main characteristics.

Soil type	Layer depth (mm)	Clay (%)	BD (g/cm^3)	AWC (vol %)	Ksat (mm/h)	OC (%)
ROccl	1000	29	1.35	0.17	7.25	0.5
ROCCrc	2000	29	1.35	0.17	7.24	0.5
CCcvrc	2000	33	1.35	0.16	5.66	0.5
VCV (1)	1000	25	1.41	0.15	9.84	0.5
VCV (2)	4000	14	1.41	0.17	21.36	0.5
Ccrorc	2000	25	1.38	0.17	9.37	0.5
CCvc	2000	23	1.43	0.14	11.93	0.5
CV	1000	16	1.45	0.13	23.36	0.5
Lulro	500	30	1.39	0.15	6.61	0.5

ROccl: Ochric Rendzina; ROCCrc: Ochric Rendzina with Calcic Cambisol; CCcvrc, Ccrorc, CCcv: Calcic Cambisols; VCV: Vertisol with Vertic Cambisol (1 and 2, top and bottom layers, respectively); CCvc: Calcic Cambisol with Vertic Cambisol, CV: Vertic Cambisol; and, Lulro: Ortic Luvisol.

sub-basins and divided into 590 HRUs, created from the combination of three slope classes (0–5, 5–15 and >15%), eight soil types and 17 land uses. The land-use, soil and slope thresholds used for the HRU delineation was 3 ha for all the cases.

In the Alegria River watershed, the majority of the land was devoted to non-irrigated agriculture until the 1950s, but irrigated agriculture had become the dominant practice by the 1990s. Arrate *et al.* (1997) documented that, during this decade, the area occupied by non-irrigated crops was just 15% of that occupied by irrigated crops. However, by the end of the 1990s, the trend had reversed, with the non-irrigated cropland being more abundant than the irrigated land, a pattern that has remained until the present day.

To simulate this variability, areas of irrigated and non-irrigated crops were changed progressively. For the first years of the simulation, irrigated crops were set to occupy most of the arable lands, and the pattern changed through the mid-1990s until the year 2000, since when the distribution has been constant, with non-irrigated crops across the larger part of the arable land. In order to simplify the implementation of the model, from the 17 different land uses, the two covering the smallest areas were gathered into a single category of 'other dominant crops'. Then, regular crop rotations and specific management practices for each crop category (Table 3) were established, based on land cover data acquired from Neiker-Tecnalia. Crop rotation and management data are essential aspects of this modelling for obtaining accurate estimations of water and crop yields (Srinivasan *et al.* 2010). In this study, the following crop rotations were considered:

- Rotation between non-irrigated crops (wheat/barley)
- Rotation between irrigated and non-irrigated crops (barley/sugar beet/wheat, barley/potato/wheat and sugar beet/wheat).

Table 3 Crop cycles and management practices.

Crop	Cycle	Tillage	Fertilization
Wheat	2 Nov.–31 Jul.	25 Oct.	25 Jan. (15-15-15) 5 Apr. (Nac 27%)
Winter barley	2 Nov.–31 Jul.	25 Oct.	5 Feb. (15-15-15) 5 Apr. (Nac 27%)
Spring barley	5 Feb.–31 Jul.	1 Feb.	25 Feb. (15-15-15) 5 Apr. (Nac 27%)
Sugar beet	15 Feb.–1 Nov.	10 Feb.	10 Feb. (8-15-15) 5 May (Nac 27%)
Potato	20 Apr.–20 Oct.	15 Apr.	15 Mar. (7-10-20) 20 May (Nac 27%)

Nac: Nitric ammoniacal fertilizer.

To interpret the results, the study period was divided into two periods (1990–1999 and 2000–2011) representing different agricultural practices. Specifically, the cut-off year 2000 was established in line with the implementation of the Code of Good Practices, which led to a drop in fertilizer use and a change in the origin of the irrigation water. From the data provided by Neiker-Tecnalia, annual fertilizer input was estimated to be approximately 960 and 680 t year^{-1} for the first and second periods of the simulation, respectively.

Figure 2 represents the simulated amounts of top and dressing fertilizer applications. Note the decrease in fertilization over the periods, reflecting the real data. The secondary *x*- and *y*-axes represent the estimated nitrate concentration for irrigation water up to the year 2000. In previous decades, pumping water for irrigation was a common practice, a process that led to recirculation of the nitrate from the aquifer. After the implementation of the Code of Good Practices, however, farmers started taking the water from ponds not connected to the aquifer. As the SWAT model does not consider NO_3 concentration in irrigation water, the corresponding N load was calculated for 10 mm of water (each irrigation dose)

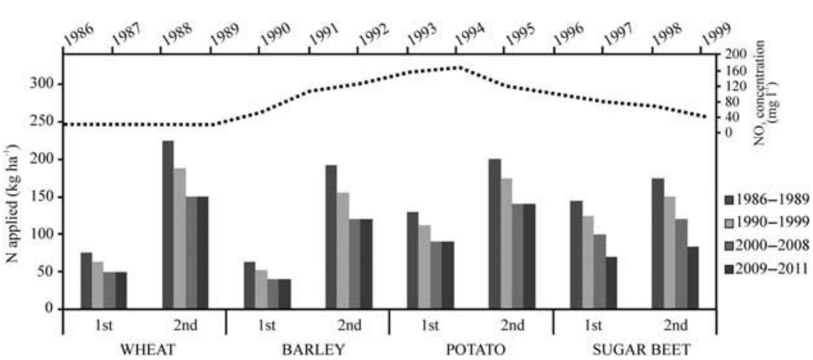


Fig. 2 Evolution of top (first) and dressing (second) fertilization (kg N ha^{-1}) during the simulated period. Nitrate concentration considered in irrigation water is shown on the secondary *y*-axis.

based on the annual groundwater nitrate measurements. The calculated value was applied simultaneously with each irrigation dose as mineral N fertilizer. This annual groundwater concentration dataset was taken from the Groundwater Control Network website of the Water Agency of the Basque Country (www.telur.es/redbas).

Irrigated crops (sugar beet and potato) receive, respectively, about 100 and 120 mm of water a year. So, depending on the annual NO_3 concentration of the groundwater, N added with irrigation water is equivalent to fertilization doses ranging between 4.5 and 37.8 kg ha^{-1} year^{-1} for sugar beet areas, and between 5.4 and 45.4 kg ha^{-1} year^{-1} for potato fields. Due to the large extent of irrigated crops areas, especially in the first years of the simulation, a detailed irrigation schedule had to be established in order to avoid all the HRUs receiving irrigation at the same time.

2.4 Model evaluation

The performance of the SWAT model was evaluated using the following statistical indices: percent bias (PBIAS), coefficient of determination (R^2), root mean square error (RMSE), RMSE-observations standard deviation ratio (RSR) and Nash-Sutcliffe efficiency (NSE) (Moriasi *et al.* 2007). The PBIAS measures the average tendency of simulated data to be larger or smaller than the observed counterparts. Positive and negative values indicate model underestimation and overestimation bias respectively. The performance is better for small magnitudes of PBIAS. The R^2 represents the proportion of total variance in the observed data. Values range from 0 to 1, where 1 is the best performance. The NSE coefficient indicates how well the plot of observed *vs* simulated values fits the 1:1 line. Its value ranges from –∞ to 1, with NSE = 1 being the optimal value. Lastly, RSR represents the ratio of RMSE and standard deviation of observed data. Its value ranges from 0 to a large positive value, with 0 being the optimal value.

To evaluate the daily and monthly results of this study, the following criteria were used: NSE satisfactory at >0.5; PBIAS satisfactory at below ±25% for streamflow and below ±70% for nitrogen; RSR satisfactory at <0.7 (Moriasi *et al.* 2007); and R^2 satisfactory at >0.5 (Green *et al.* 2006).

2.5 Data used for calibration and validation

The whole simulation was performed with a daily time step; the first 4 years (1986–1989) were excluded from the results since they were used as a warm-up period. The calibration “by stages” detailed by Nair *et al.* (2011), leaving nutrient load calibration as last step, has been used in many publications (e.g. Behera and Panda 2006, Ferrant *et al.* 2013) and seemed the most appropriate for our purposes. First, a sensitivity analysis was carried out in order to identify calibration parameters. Then, by manual calibration, the parameters presented in Table 4 were adjusted. Streamflow was the first variable to be

Table 4 Main variables used for sensitivity and calibration in SWAT.

Parameter	Input file	Description	Range	Calibrated value
EPCO	.bsn	Plant uptake compensation factor	0.01–1	1
ESCO	.bsn	Soil evaporation compensation factor	0.01–1	0.9
SURLAG	.bsn	Surface runoff lag coefficient	1–24	5
CDN	.bsn	Denitrification exponential rate coefficient	0–3	0.01
CMN	.bsn	Rate coefficient for mineralization of the humus active organic nutrients	0.0001–0.003	0.002
RSDCO	.bsn	Rate coefficient for mineralization of the residue fresh organic nutrients	0–0.1	0.01
NPERCO	.bsn	Nitrogen percolation coefficient	0.01–1	0.8
SDNCO	.bsn	Denitrification threshold water content	0–1	0.95
ALPHA_BF	.gw	Baseflow alpha factor	0–1	0.35
GW_DELAY	.gw	Groundwtaer delay	0–500	0.5
GW_REVAP	.gw	Groundwater revap coefficient	0.02–0.2	0.02
REVAPMN	.gw	Threshold depth of water in the shallow aquifer for revap to occur	0–8000	2000
GWQMN	.gw	Threshold depth of water in the shallow aquifer required for return flow to occur	0–5000	800
HLIFE_NGW	.gw	Half-life of the nitrate in shallow aquifer	0–5000	2500
CN2	.mgt	Curve number	0–100	±10% of initial values

calibrated, although it was necessary to re-calibrate it once crop-related parameters had been well adjusted. After calibrating streamflow and crop yield, N cycle-related parameters were adjusted. Initial values for the following parameters were established using data provided by Neiker-Tecnalia: initial nitrate concentration in the shallow aquifer, SHALLST_N = 15 mg L^{-1}; initial nitrate concentration in the soil layer, SOL_NO3 = 5 mg kg^{-1}; and initial organic nitrogen concentration in the soil layer, SOL_ORGN = 700 mg kg^{-1}. For calibration and validation of the model the following field data was used:

2.5.1 Streamflow River discharge was calibrated at the outlet of the watershed using field measured water levels, which were converted to discharge using a rating curve. This work was performed within the framework of the Aguaflash Project (Interreg-SUDOE) by Cerro (2013). All available discharge data were used, from which the period 21 October 2009–31 December 2010 was used for calibration and 1 January 2011–31 December 2011 for validation. The most sensitive parameters to the water balance and the assigned values are shown in Table 4.

2.5.2 Crop yield Many studies have demonstrated the difficulties of capturing annual variations in crop yield well with the SWAT model (and the EPIC crop growth sub-model on which SWAT is based) (Huang *et al.* 2006, Srinivasan *et al.* 2010). For this reason, prediction of yield for individual years is not recommended (Moulin *et al.* 1993), and researchers use long-term average data to compare simulated and measured crop response variability. In this study, the crop yield evaluation focused on the comparison of simulated data with mean values from recent years (2002–2011). The studied crops were wheat, barley, oats, potato and sugar beet. Given the agricultural similarities between barley and oats, we decided to simulate them as a single group, which was renamed as barley (so, from now on, barley data will be a mean value for these two crops). Real crop yield data from the last 10 years were obtained from the Agricultural Department of the Basque Government. The SWAT model estimates dry crop yield and, hence, in order to perform comparisons with observed data (given in wet weight), simulated values (tonnes per hectare) were converted from dry to wet weights. For that conversion, the following moisture contents were considered: wheat and barley, 10%; potato, 75%; and sugar beet, 80% (Kenneth and Hellevang 1995, Scanlon 2005, NDSU 2006). The harvest index of sugar beet was adjusted in the crop database, owing to a rather low estimated yield. We assumed a value of 1.1 instead of the default value of 0.95. It should be kept in mind that simulated yield corresponds to the potential value and so higher yields than observed values were expected in the simulation.

The evapotranspiration process was calculated by the Hargreaves PET method.

2.5.3 Nitrogen In the context of different research studies, N-NO_3 concentration has been measured in the Alegria River in three different periods: 1990–1994, 2001–2005 and 2009–2011. Arrate (1994) made an exhaustive hydrogeological study of the Quaternary aquifer of Vitoria-Gasteiz. He gathered the water quality data from wells and the river for spatial and temporal analyses. Martinez (2008) focused her research on the hydrochemical temporal evolution of surface water and groundwater. Cerro (2013) studied the impact of floods on water quality, for which she monitored physicochemical parameters in the Alegria River over a period of two years.

Commonly, in the field, what is measured is nitrate concentration (mg L^{-1}); hence, in order to process SWAT-simulated nitrogen export, it was essential to convert the field-measured concentrations into continuous series of N-NO_3 load (t d^{-1}). To do so, a continuous streamflow dataset is required. However, given that in the first two periods mentioned above there were no field-measured streamflow data, it was decided to use the simulated streamflow values in all the cases, once values during both calibration and validation had shown a good fit. Therefore, simulated streamflow and scattered field-measured N-NO_3 concentrations were used to obtain continuous load data series. This was performed with LOADEST, a FORTRAN program commonly used to estimate constituent load in streams and rivers (Runkel *et al.* 2004). There are several uncertainties in this process: first, the uncertainty inherent in the discharge values used, these being simulated; and, second, the representation of daily N-NO_3 load based on one or two measurements per month for some cases, especially during 1990–1994 (Arrate 1994) and 2001–2005 (Martinez 2008). The N-NO_3 load was calibrated for the period 28 October 2009–27 October 2010 and validated for three different periods: Val-1: 27 November 1990–29 June 1994;

Val-2: 2 January 2001–22 December 2005; and Val-3: 28 October 2010–17 June 2011.

Nitrogen mass balance was calculated by equation (1) and results were expressed in terms of: (a) unit of mass per time (t N year^{-1}), calculated as seasonal mean values for the periods 1990–1999 and 2000–2011; (b) unit of mass per time and cultivated area (kg ha^{-1} year^{-1}), which represents annual mean values for each study period and only considering agricultural land cover (necessary data for validation of N fluxes); and (c) unit of mass per time and total area (kg ha^{-1} year^{-1}), which represents annual mean values for each period (necessary data for visualization of N fluxes at the sub-basin scale):

$$\Delta N = (N_{\mathrm{fert}} + N_{\mathrm{atm}} + N_{\mathrm{min}}) - \left(N_{\mathrm{denit}} + N_{\mathrm{up}} + N_{\mathrm{losses}}\right) \quad (1)$$

where N_{fert} is N in fertilizers and in irrigation water; N_{atm} is atmospheric N deposition; N_{min} is mineralized N; N_{denit} is N from denitrification; N_{up} is crop uptake N; and N_{losses} is N exported by streams.

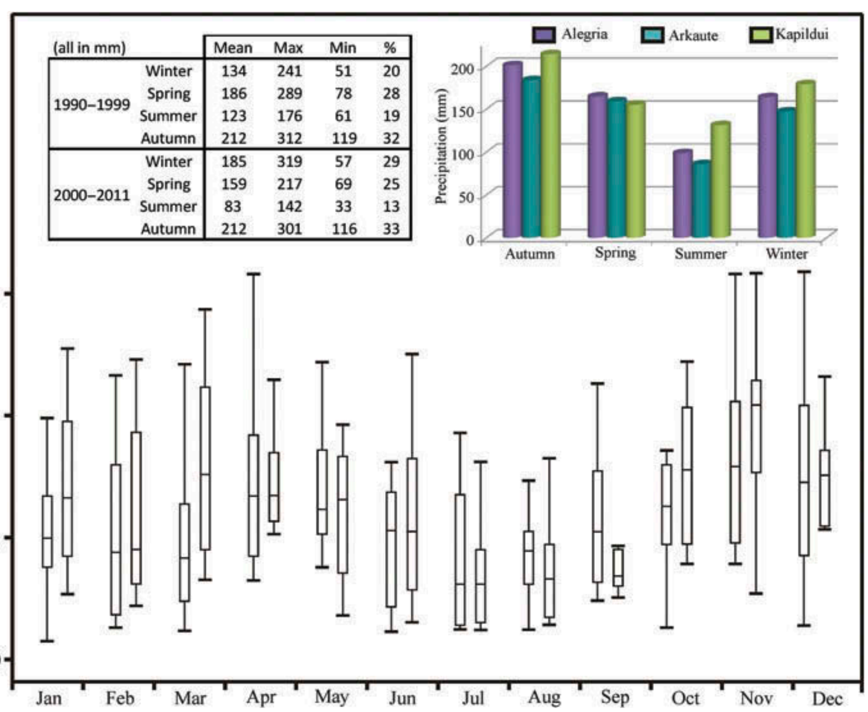
The N fluxes on arable land (N wash-off, N uptake by plants, N denitrification and N mineralization) were indirectly validated by the comparison of simulated data with data obtained by Sanchezperez *et al.* (2003) and Jégo *et al.* (2008) in studies carried out in the same study area. In addition, the literature was consulted to contrast the results with reasonable ranges in agricultural lands (Krysanova and Haberlandt 2002, Burkart *et al.* 2005, Castaldelli *et al.* 2013, Ferrant *et al.* 2013, Boithias *et al.* 2014). To simulate atmospheric deposition a rainfall nitrogen concentration value was estimated (1 mg L^{-1}) so that annual deposition was similar to the data obtained by Sanchezperez *et al.* (2003) in the same area. In order to fit N fluxes and transport within the watershed, the most sensitive parameters were adjusted (Table 4).

3 RESULTS AND DISCUSSION

3.1 Hydrology

For the period 1990–2011, the model interpolated 660 mm mean annual precipitation, of which 68% was lost by evapotranspiration. The estimated values seemed to be within acceptable ranges according to available data for the study area (Cerro 2013).

Precipitation data of the periods 1990–1999 and 2000–2011 (Fig. 3) suggest drier summers and wetter



(all in mm)		Mean	Max	Min	%
1990–1999	Winter	134	241	51	20
	Spring	186	289	78	28
	Summer	123	176	61	19
	Autumn	212	312	119	32
2000–2011	Winter	185	319	57	29
	Spring	159	217	69	25
	Summer	83	142	33	13
	Autumn	212	301	116	33

Fig. 3 Paired box plots reporting average monthly rain data from three meteorological stations and corresponding to the periods 1990–1999 and 2000–2011. Median, 25th and 75th percentiles, and minimum and maximum values are shown. The table provides statistical information (mean, maximum, minimum and percentage) about seasonal precipitation and the graph represents the seasonal precipitation average in each of the meteorological stations considered in the study.

winters during the second period. In the upper part of Fig. 3 the seasonal mean precipitation values for each meteorological station considered in the study are shown. In general, it can be noted that Kapildui is the station in which higher precipitation is registered and Arkaute is the one with the lower precipitation. The lengths of the box plots in Fig. 3 indicate greater variability in the rainfall during the second period in most months.

Mean daily discharge for the calibration period was 0.39 $m^3 s^{-1}$ in the observed data and 0.43 $m^3 s^{-1}$ in the simulated data. For the validation period, the observed and simulated mean discharges were 0.22 and 0.21 $m^3 s^{-1}$, respectively. Daily discharges were satisfactorily simulated in the calibration period and showed good agreement during the validation period (Fig. 4; Table 5). Although baseflow was simulated quite well, most of the streamflow peaks were underestimated. It is important to note that the highest flow peak (January 2010) was related to snowmelt. Multiple iterations were completed, changing parameters linked to snow, in order to obtain a better fit of the hydrograph; however, for this event this process was not very successful. Another point to be highlighted is the uncertainty associated with the observed streamflow data. As noted by Cerro *et al.* (2014a), there are some uncertainties in the rating curve for very high flows. In addition, we should remember that there is uncertainty related to the interpolation of rainfall data, an aspect that could have influenced the simulation of high-flow values.

Based on the statistical indices, flow simulations are, in the worst case, satisfactory. In general, the statistical indices improve from daily to monthly analysis as long-term means have relatively smaller errors than short-term values (Winter 1981) (Table 5). Specifically, for the daily analysis, the NSE and R^2 values for the model were 0.59 and 0.59, respectively, for the calibration period and 0.72 and 0.73 for validation period, while for the monthly analysis, NSE and R^2 were 0.85 and 0.88, respectively, for the calibration period and 0.95 and 0.95 for the validation period. In all cases, PBIAS was at worst case 11%, and also RSR indicated satisfactory results.

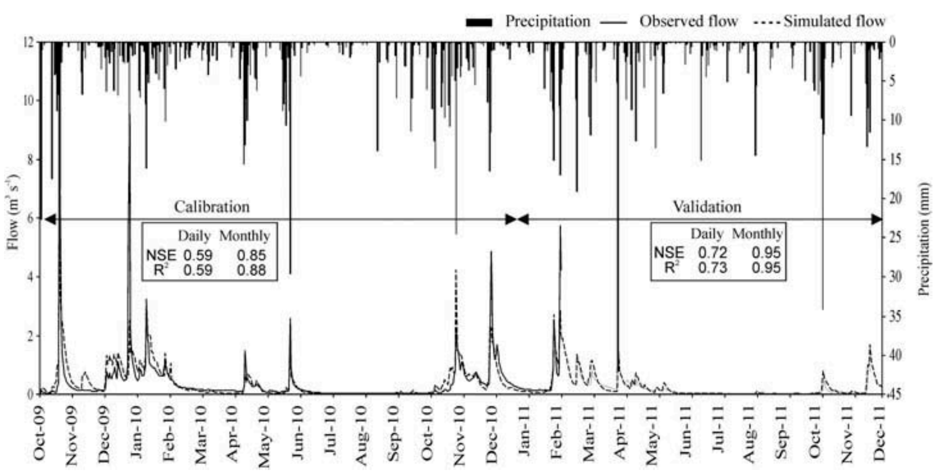


Fig. 4 Observed and simulated daily discharge at the Alegria watershed outlet ($m^3 d^{-1}$). Daily and monthly statistical values are provided for calibration and validation periods.

Table 5 Statistical summary for streamflow and nitrates.

	Streamflow				Nitrate							
	Daily		Monthly		Daily				Monthly			
	Cal.	Val.	Cal.	Val.	Cal.	Val-1	Val-2	Val-3	Cal.	Val-1	Val-2	Val-3
NSE	0.59	0.72	0.85	0.95	0.77	0.74	0.73	0.75	0.92	0.94	0.94	0.67
R^2	0.59	0.73	0.88	0.95	0.86	0.76	0.74	0.83	0.99	0.96	0.95	0.91
PBIAS (%)	−10	6	−11	4	−7	7	−12	29	−7	7	12	28
RSR	0.64	0.53	0.39	0.24	0.48	0.51	0.52	0.50	0.29	0.25	0.25	0.57

Simulated annual water yield calculated from the recorded precipitation grouped into periods 1990–1999 and 2000–2011 was quantified as 12 and 13 hm^3. Field measurements indicate that, during the streamflow calibration period (15 months) 15 hm^3 was exported from the watershed through the main channel, while during the validation period (12 months) it was 7 hm^3. Simulation results show similar values, 16 and 8 hm^3 being the exported water yield for calibration and validation periods, respectively.

The streamflow components indicate that 85% is contributed by groundwater, 3% by lateral flow and 12% by surface runoff. This distribution of components corresponds well to the special characteristics of lowland areas, as reported by Lam *et al.* (2011).

3.2 Crop yield

Although the annual variability of crop yield was not well reproduced, simulated crop yields were well adjusted to the observed values in the longer term (10 years). Observed and simulated mean values for the period 2002–2011 were, respectively: 5.4 and 5.2 t ha^{-1} $year^{-1}$ for wheat; 4.8 and 7.0 t ha^{-1} $year^{-1}$ for barley; 34.7 and 40.6 t ha^{-1} $year^{-1}$ for potato; and 80.9 and 84.7 t ha^{-1} $year^{-1}$ for sugar beet. In principle, higher values were expected for the simulated yield, as they represent the potential crop yield not influenced by any disease. Further, differences between the observed and simulated yields on the annual scale could be explained by discrepancies between the observed and simulated crop stresses: with the simplification of adopting fixed planting and harvesting dates (throughout the simulation period), we have not taken into account modifications in dates that could have been made by farmers due to the meteorological conditions in a given year. The differences in crop cycles from year to year are likely to imply different levels of stress to that simulated, which in turn will have an impact on simulated crop yield.

Figure 5 illustrates the simulated annual dry yield for each crop, and also the mean annual precipitation (mm) and solar radiation (M J m^2 d^{-1}). Although a slight positive relationship can be perceived between precipitation and solar radiation with crop yield, it is difficult to obtain good regression coefficients on the annual scale.

3.3 Nutrients

3.3.1 Soil nitrogen mass balance In the nitrogen mass balance, a positive ΔN value means that nitrogen is being retained in the system and negative ΔN that nitrogen inputs are lower than outputs during a given period, considering the outputs as the N losses by all the possible pathways. The nitrogen mass balance results were processed to obtain representative differences between the periods 1990–1999 and 2000–2011 (Tables 6 and 7; Fig. 6).

In the Fig. 6 the input, output and surplus differences between the two periods can be distinguished: the lower N input and output during the period 2000–2011 led to a lower N surplus over the study area. These maps not only show the differences between

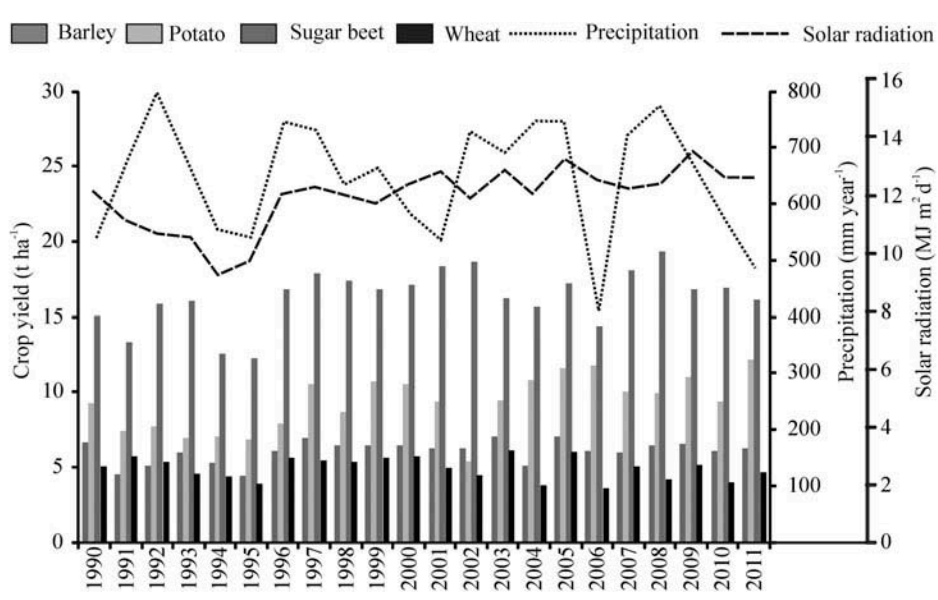


Fig. 5 Simulated annual crop yield (t ha^{-1} $year^{-1}$), mean precipitation (mm $year^{-1}$) and mean solar radiation (MJ m^2 d^{-1}).

Table 6 Soil system N budget for cultivated lands in the Alegria River watershed (annual average values and standard deviation are given). Values are expressed per hectare of cultivated land (kg N ha^{-1} year^{-1}).

	1990–1999		2000–2011	
	kg N ha^{-1} year^{-1}	% of total	kg N ha^{-1} year^{-1}	% of total
Input				
Fertilization	260 (27)	*61*	185 (14)	*54*
Atmospheric deposition	7 (1)	*2*	6 (1)	*2*
Mineralization	159 (11)	*37*	149 (10)	*44*
∑Input	**426 (34)**	***100***	**340 (21)**	***100***
Output				
Plant uptake	206 (20)	*66*	211 (18)	*77*
Denitrification	64 (17)	*21*	30 (7)	*11*
Stream losses	40 (12)	*13*	34 (12)	*12*
∑Output	**312 (31)**	***100***	**275 (27)**	***100***
∑Input – ∑output	*114 (41)*		*65 (19)*	

the periods, but also allow one to identify the zones that contribute most to the N pollution of the watershed.

The simulation results of total N input to the system show an input of 1668 ± 124 t year^{-1} in 1990–1999 and 1364 ± 76 t year^{-1} in 2000–2011 (Table 7). In Table 6, inputs and outputs are expressed in percentages. The main N input is from fertilization (54–61% of the total) followed by that from mineralization process both of humus and active pools (37–44%), and lastly, from atmospheric deposition (2%).

Total N output was estimated to be 1231 ± 119 and 1091 ± 103 for the periods 1990–1999 and 2000–2011, respectively (Table 7). The main N output is due to plant uptake (66–77%), followed by stream losses (12–13%) and denitrification (11–21%) (Table 6). Each of the processes shows quite large variability within a given year, mainly due to the distribution of fertilization and precipitation. In order to quantify the differences over the year, N balance inputs and outputs were averaged for each season for the periods 1990–1999 and 2000–2011 (Table 7). It should be noted, though, that in Table 7 total inputs and outputs are represented, and these are affected not only by meteorology and crop growth stage, but also by the surface area occupied by each crop during the corresponding period. Regarding the N inputs, it should be noted that the highest values are registered during spring and are related to fertilization. Mineralization is the second process that contributes notably to N inputs, the maximum value being registered in autumn, coinciding with the harvesting days. In contrast, plant uptake is the process which most contributes to the N output, with maximum values during spring in both periods. However, whereas in the period

Table 7 Soil system N budget for the Alegria River watershed (annual seasonal average values and standard deviation are given). Values are expressed as mass net values (t N year^{-1}).

	1990–1999				2000–2011			
	High fertilization doses + irrigation water taken from the aquifer				Low fertilization doses + irrigation water taken from the aquifer			
	Winter	Spring	Summer	Autumn	Winter	Spring	Summer	Autumn
Input								
Fertilization	343 (33)	580 (86)	38 (21)	0 (0)	208 (13)	474 (39)	0 (0)	0 (0)
Atmospheric deposition	7 (3)	10 (4)	7 (2)	11 (3)	10 (4)	8 (2)	4 (2)	11 (3)
Mineralization	182 (17)	115 (18)	143 (22)	233 (21)	154 (7)	92 (4)	181 (11)	221 (22)
∑Input (Seasonal)	**532 (43)**	**704 (94)**	**187 (18)**	**245 (22)**	**372 (20)**	**575 (42)**	**185 (11)**	**232 (23)**
∑Input (Total)	**1668 (124)**				**1364 (76)**			
Output								
Plant uptake	136 (59)	510 (59)	141 (57)	25 (7)	234 (47)	518 (60)	48 (10)	35 (14)
Denitrification	69 (20)	71 (26)	49 (12)	58 (11)	34(8)	31 (7)	23 (6)	31 (7)
Stream losses	61 (40)	37 (41)	8 (7)	65 (40)	67 (37)	18 (15)	3 (2)	49 (29)
∑Output (Seasonal)	**266 (76)**	**619 (70)**	**198 (61)**	**148 (45)**	**335 (45)**	**567 (63)**	**74 (14)**	**115 (41)**
∑Output (Total)	***1231 (119)***				***1091 (103)***			
∑Input – ∑output (Seasonal)	*266 (114)*	*86 (114)*	*−11 (65)*	*96 (39)*	*37 (36)*	*118 (28)*	*111 (17)*	*118 (28)*
∑Input – ∑output (Total)	*437 (156)*				*273 (72)*			

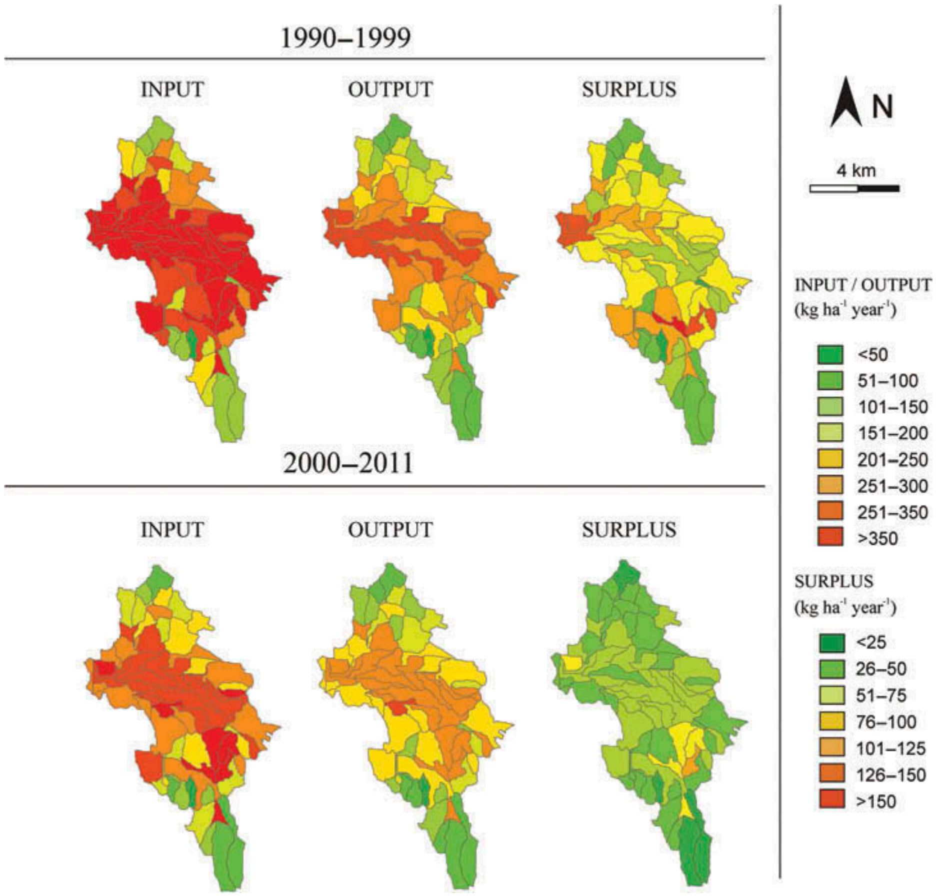


Fig. 6 Spatial distribution of simulated nitrogen input, output and surplus (kg N ha^{-1} $year^{-1}$) for the periods 1990–1999 and 2000–2011.

1990–1999 N uptake is quite significant during the winter and summer, in the period 2000–2011, the values are quite low during the summer. This may be due, in part, to the change of crop type and, in turn, the crop cycle from one period to the other. Denitrification decreases from first to the second period, but remains quite similar over the seasons except for summer, where meteorological conditions are less favourable for this process to occur. In both the periods 1990–1999 and 2000–2011, stream losses are the highest during autumn and winter, a fact directly linked to the precipitation registered during these seasons.

In order to validate simulated N fluxes, only the results obtained from agricultural lands were considered. In this way, an indirect validation of the N balance was performed using regional and general ranges documented in the literature. The results obtained suggest a total N surplus equivalent to area weighted mean values of 114 and 65 kg ha^{-1} $year^{-1}$ for 1990–1999 and 2000–2011, respectively (Table 6). The relationship between outputs and inputs is estimated to be 73% in the first period and 81% in the second. These values seem to be within acceptable ranges. In fact, in a study carried out in the same area during the years 1993–1994, this relationship was estimated to be 87% on a cultivated area and 26% on non-cultivated land (Sanchezperez *et al.* 2003). As would be expected, the average value obtained in our study is much closer to that previously obtained for the cultivated area. This is attributable to the fact that our values were based on larger temporal and spatial scales, comprising the processes at the watershed scale, including those occurring in the intervals between cultivation periods.

The mineralization fluxes remained similar over time: 159 and 149 kg ha^{-1} $year^{-1}$ for the 1990–1999 and 2000–2011 periods, respectively. These values are higher than those reported by Jégo *et al.* (2008) in the same study area, which range from 55 to 69 kg ha^{-1} $year^{-1}$, but are within the range for general arable lands reported by Krysanova and Haberlandt (2002), ranging from 40 to 180 kg ha^{-1} $year^{-1}$. In the literature, N uptake ranges are reported for several different crops; grains have the lowest uptake (100–200 kg ha^{-1} year^{-1}), while it is somewhat higher for root crops—potato: 142–233 kg ha^{-1} year^{-1} and sugar beet: 200–250 kg ha^{-1} year^{-1}, according to Krysanova and Haberlandt (2002) and Tyler *et al.* (1983), respectively. In our study, annual mean values were calculated (Table 6): 206 and 211 kg ha^{-1} year^{-1} for the periods 1990–1999 and 2000–2011, respectively; these values seem to be within reasonable ranges. In any case, it is interesting to separate N uptake fluxes for each crop. The simulated results show 177, 199, 205 and 269 kg ha^{-1} year^{-1} for potato, barley, wheat and sugar beet, respectively.

The higher plant uptake for 2000–2011 compared to 1990–1999, together with the fact that the fertilization rate was markedly higher during the first period, provides evidence for the over-fertilization that has been occurring in recent decades in the watershed.

Regarding denitrification fluxes in arable lands, it can be noted that, whereas the value obtained for 1990–1999 (64 kg ha^{-1} year^{-1}) is slightly higher than the ranges reported for general arable lands (20–60 kg ha^{-1} year^{-1} according to Krysanova and Haberlandt 2002), the value obtained for 2000–2011 (30 kg ha^{-1} year^{-1}) is within the expected range.

Total N outputs were quantified and the relationship with regard to the precipitation regime was obtained. In fact, meteorology is thought to be an important factor in determining the system N balance. On a seasonal scale, we did not find a strong association between the precipitation regime and total N output from the system. In contrast, on the annual scale, precipitation and N outputs are well correlated (Fig. 7). Indeed, this relationship demonstrates a difference in the effect of precipitation over the two periods: a given annual precipitation is associated with a considerably higher N output in the period 1990–1999 than in 2000–2011.

3.3.2 Nitrogen losses and nitrate concentration in surface waters Using calibrated parameter data, the SWAT model successfully predicted N-NO_3 load at the outlet of the Alegria River (Fig. 8). With a daily time step, the model performance was very good during the calibration period (NSE: 0.77) and good during the validation periods (NSE: 0.74, 0.73 and 0.75).

The main differences between the observed and simulated data can be seen in high peaks (Fig. 8). These differences might be associated with the following uncertainties: (a) streamflow calibration, which directly affects nutrient simulation; (b) representativeness of the data, especially for the periods 1990–1994 and 2001–2005, when field

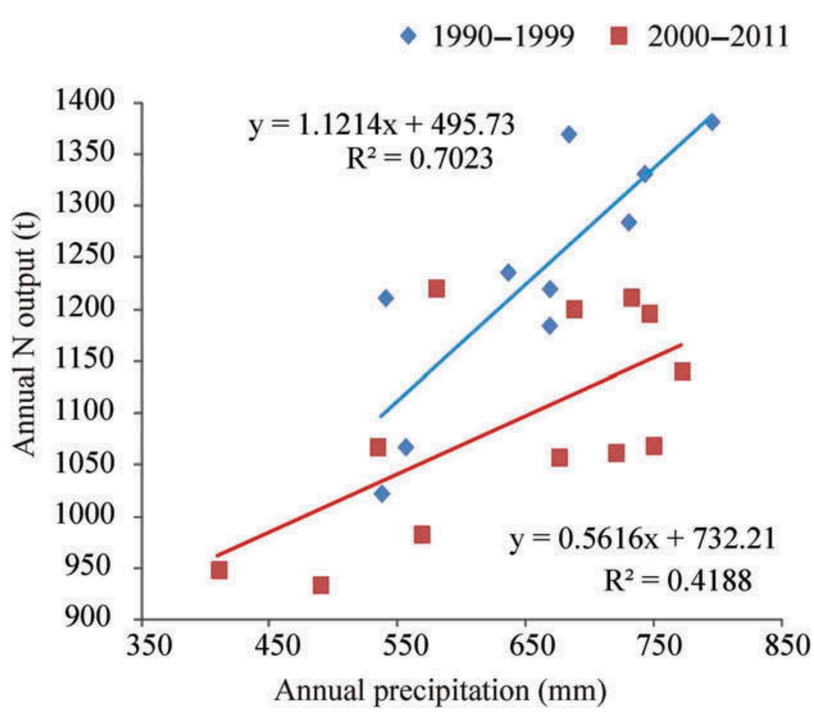


Fig. 7 Relationship between annual precipitation and annual N output for the periods 1990–1999 and 2000–2011.

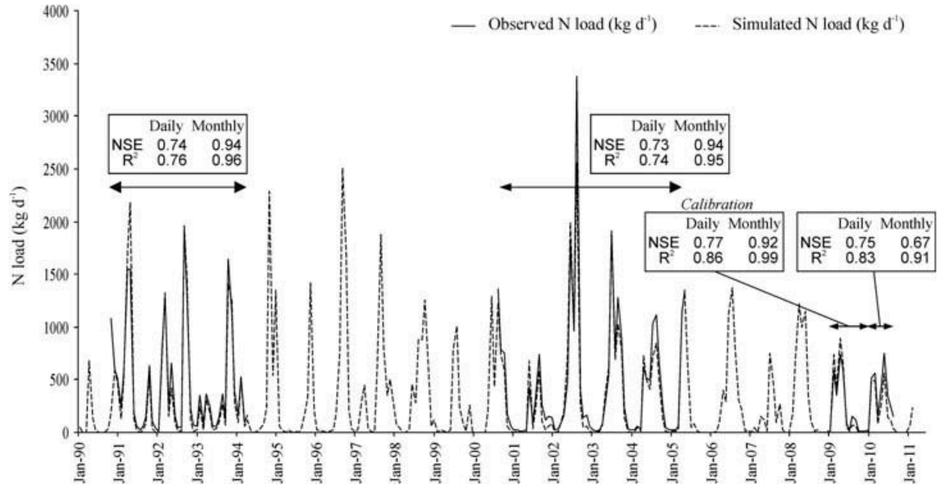


Fig. 8 Observed and simulated monthly average nitrogen load (kg N d^{-1}) at the Alegria watershed outlet. Daily and monthly statistical values are provided for calibration and validation periods.

measurements used for daily estimates were scarce; and (c) input related to management practices, which have been assumed to be the same over time in the simulation.

The high nitrate peaks coincide with flood events, reflecting the fact that they are pushed through the system in rainy periods. During the drought season, nitrate tends to accumulate in the aquifer, behaving as nitrate storage, until higher water flows flush them out (Cerro *et al.* 2014a). Indeed, the annual exported amount of nitrate is largely controlled by hydrological conditions (simulated water yield and nitrogen export in the period 1990–2011 having a value of R^2 = 0.74).

There is a pronounced decrease in exported N-NO_3 over the years. While in 1990–1999 the mean simulated annual value was 132 t, in 2000–2011 it fell to 110 t. Although a lower precipitation regime could explain lower export during a given year, we think that for longer periods only a reduction in fertilizer use and change in irrigation water origin could explain this reduction in the exported nitrogen.

The general trend of nitrate concentration (mg L^{-1}) in Alegria River was adequately simulated (Fig. 9). It should be noted that, whereas simulated NO_3 monthly averages were calculated from daily data, the observed NO_3 averages were obtained from data with a highly-scattered sampling frequency (in many cases one or two samples per month in the periods 1990–1994 and 2000–2005). A more accurate sampling strategy between 2009 and 2011 is reflected in less variation between the observed and simulated monthly averages. Especially in the last period, it can be seen that during the low water level periods the difference between the observed

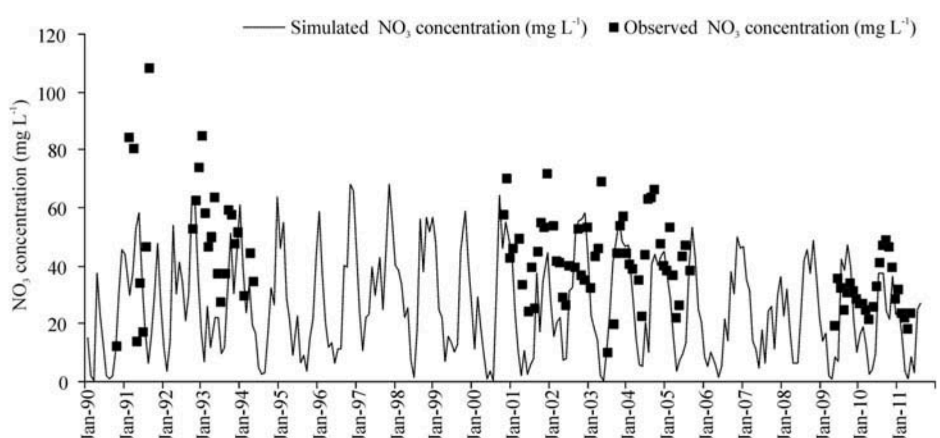


Fig. 9 Observed and simulated monthly average nitrate concentration (mg NO_3 L^{-1}) at the Alegria watershed outlet.

and simulated nitrate concentration is higher. This pattern has already been studied by several authors (Arrate 1994, Martinez 2008) and has been linked to the N uptake by plants in river channels.

3.3.3 Nitrate concentration in groundwater The assumption of no interaction between HRUs and the lack of a full groundwater balance (SWAT only simulates nutrients in a shallow aquifer) may mean that groundwater contributions to streamflow are not representative of the nitrate concentration in the aquifer. However, in the present study, the fact that groundwater is the main streamflow component in an area where the aquifer is quite shallow, there is no regional discharge and there is diffuse pollution over the 85% of the aquifer, led us to think about the possibility of simulating groundwater nitrate concentration through the groundwater contribution to streamflow.

Following this approach, we calculated the mean NO_3 concentration in the groundwater contribution to streamflow and obtained a similar trend to that described by Sanchezperez *et al.* (2003), which was representative of the groundwater NO_3 concentration. Figure 10 represents the mean annual NO_3 concentration for all the HRUs in the watershed and also the concentration obtained from cultivated HRUs. As can be observed, the mean watershed nitrate concentration increases over the first years of the simulation, reaching maximum values during the 1990s decade and decreasing over the following years, a pattern that has already been reported by several authors (Arrate *et al.* 1997, Sanchezperez *et al.* 2003, Martinez 2008). The trend was directly linked to high fertilizer doses applied during the 1990s, as well as to the recirculation of the water used for irrigation. Further, there is also considerable difference between the trend in global nitrate concentration (all HRUs of the watershed) and the trend obtained from cultivated HRUs. The latter shows an annual concentration value approx. 10–20 mg L^{-1} higher than the annual value obtained from HRUs of the entire watershed.

3.3.4 N dynamics The environmental impact of the agricultural practices is controlled by the crop type, hydrometeorological conditions, crop management practices and soil characteristics (Jégo *et al.* 2008). In Fig. 11, it can be observed that both N export (t $year^{-1}$) through streams and nitrate concentration in the river (mg L^{-1}) are highly dependent on the precipitation regime (R^2 = 0.75). In the same figure, the different pathways for discharge flow are presented. It can be seen that groundwater flow is the dominant pathway, accounting for 85% of the total discharge, on average. It should be noted that the low value of lateral flow is related to a shallow

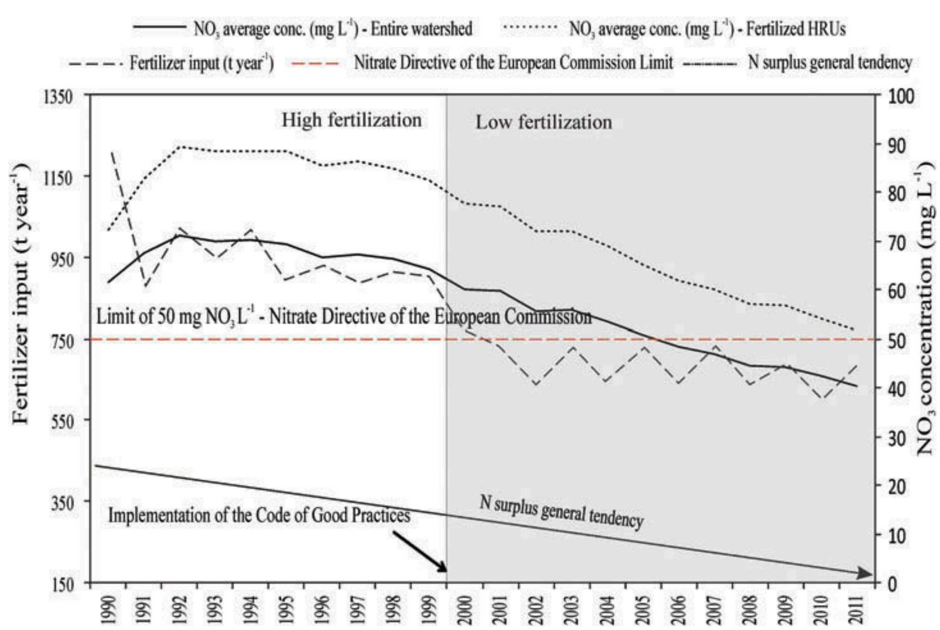


Fig. 10 Simulated annual evolution of groundwater NO_3 concentration (mg NO_3 L^{-1}) (considering the HRUs of the entire watershed and only the agricultural HRUs) and annual fertilizer input (t $year^{-1}$). High and low fertilization periods (1990–1999 and 2000–2011, respectively) are highlighted. The dashed (red) line indicates the nitrate limit set by the European Commission and the arrow the simulated trend of annual N surplus.

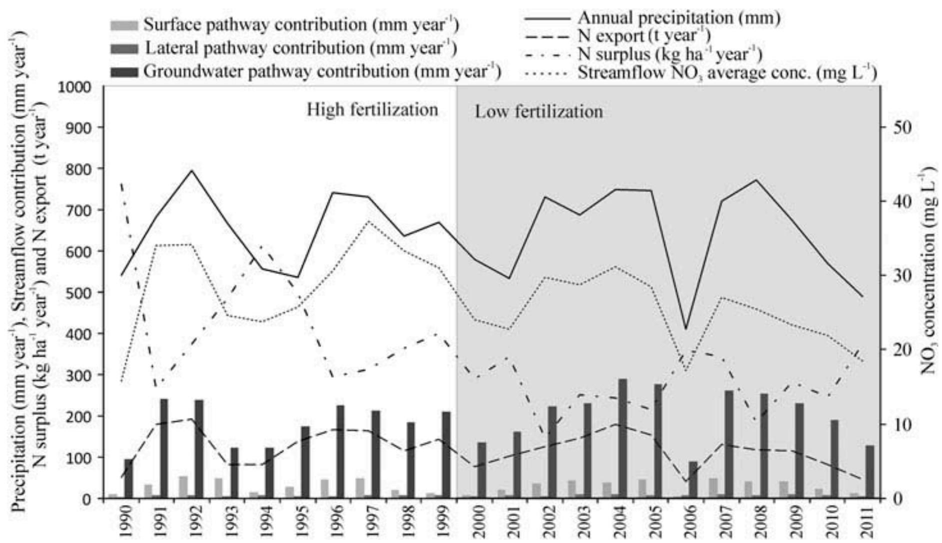


Fig. 11 Simulated annual streamflow contribution by surface, lateral and groundwater pathways. High and low fertilization periods (1990–1999 and 2000–2011, respectively) are highlighted. Annual precipitation (mm year^{-1}), N surplus (kg N ha^{-1} year^{-1}), N export (t N year^{-1}) and streamflow NO_3 concentration (mg NO_3 L^{-1}) are also shown.

groundwater table and the small surface contribution due to the flat topography (Lam *et al.* 2010). Table 8 summarizes the correlation between each of the water contribution pathways and annual precipitation, N export and nitrate concentration in the river. In each case, 22 values were considered, and both Pearson correlation and bilateral significance are shown. Annual precipitation has high correlation coefficients with surface and groundwater contributions: $R^2 = 0.841$ ($p < 0.01$) and 0.847 ($p < 0.01$), respectively, the groundwater contribution being the factor most strongly correlated with N export: $R^2 = 0.841$ ($p < 0.01$). Figure 10 shows annual groundwater NO_3 concentration. It can be appreciated that it does not follow the annual variability seen in the river concentration, showing in contrast a stronger response to fertilizer input than to meteorological conditions (Table 8). Figure 10 illustrates the lag time required to decrease nitrate concentration in the aquifer after environmental measures have been

Table 8 Correlation matrix of simulated variables obtained by SWAT model.

	Prec.	SW	LW	GW	Fert.	Δ*N*	N export	GW conc.	River conc.
Prec.	1								
SW	0.841**	1							
	0.000								
LW	0.308	0.424*	1						
	0.163	0.049							
GW	0.847**	0.723**	0.563**	1					
	0.000	0.000	0.006						
Fert.	0.003	−0.040	−0.499*	−0.334	1				
	0.990	0.859	0.018	0.128					
Δ*N*	−0.485*	−0.408	−0.472*	−0.675**	0.782**	1			
	0.022	0.060	0.027	0.001	0.000				
N export.	0.807**	0.720**	0.379	0.841**	0.069	−0.423*	1		
	0.000	0.000	0.082	0.000	0.761	0.050			
GW conc.	0.142	0.106	−0.297	−0.142	0.780**	0.433*	0.369	1	
	0.527	0.640	0.180	0.529	0.000	0.044	0.092		
River conc.	0.760**	0.610**	0.298	0.681**	0.128	−0.418	0.883**	0.492*	1
	0.000	0.003	0.178	0.000	0.572	0.053	0.000	0.020	

** Correlation significant at $p = 0.01$ level, * significant at $p = 0.05$ level.
Prec.: precipitation, SW: surface water contribution, LW: lateral contribution, GW: groundwater contribution, Fert.: fertilization, Δ*N*: N surplus, GW conc.: groundwater concentration; River conc.: river concentration.

taken. We estimate that a 25% decrease in fertilizer use led to a decrease of around 30% in both aquifer concentration and nitrogen export through river discharge. It took approximately a decade from the date the implementation of the Code of Good Practices became compulsory, in 2000, until the nitrate concentration in the aquifer reached the target set in the code. The relationship between nitrogen export and precipitation regime can be explained by the nutrient flushing process. Usually, nutrients are flushed out of the landscape during hydrologically active periods, especially during flood events, while in drier ones they are retained in the aquifer (Oeurng *et al.* 2010, Cerro *et al.* 2014a). This would explain the higher nitrate concentration in the river when precipitation is high. The flushing process may have different origins, such as, for instance, the common surface runoff occurring after saturation of the nitrogen-enriched top layers (Bhat *et al.* 2007, Oeurng *et al.* 2010, Zhang *et al.* 2010). However, in the Alegria River watershed, this explanation does not apply, as the groundwater is the main source for the nitrates in the streams. In this watershed, the larger proportions of both water and nutrients are first stored in the aquifer and are then released into the streams. This behaviour was analysed by Cerro *et al.* (2014b), who carried out an exhaustive study of the nitrate transport during floods in the watershed. Their simulation results estimated that 97% of the total N export came through the aquifer to the main stream.

Though the regression coefficient was not high (R^2 = 0.17; Table 8), N surplus follows an opposite tendency to nitrogen export or nitrate concentration in streams (Fig. 11). This can be explained by the strong correlation of N surplus with groundwater contribution and annual fertilizer input, which, in turn, affect the nitrogen export and nitrate concentration in streams, among other factors.

To sum up, whereas the groundwater concentration shows a direct response to fertilizer input (R^2 = 0.78; $p < 0.01$), the streamflow concentration is influenced mainly by exported N (a relationship that shows R^2 = 0.883; $p < 0.01$). Exported nitrogen is, in turn, highly correlated to the annual precipitation (R^2 = 0.807; $p < 0.01$).

4 CONCLUSIONS

It this study, the SWAT model was used to simulate (for the period 1990–2011) discharge and N-NO_3 load in the Alegria River outlet, and crop yield and nitrogen fluxes in arable lands. Streamflow simulation showed satisfactory and good agreement with the observed data for the calibration and validation periods (NSE and R^2 at the daily time step of, respectively, 0.59 and 0.59 for the calibration period and 0.72 and 0.83 for the validation period). Although in the long term (10 years) crop yield was well simulated on average, the model did not give a good fit of the annual variation in crop yield.

In addition, the N fluxes were indirectly validated by comparing the simulated values with general ranges for arable land available in the literature. The results obtained in this study showed that, in arable land, N inputs in the watershed exceeded outputs by 114 and 65 kg ha^{-1} $year^{-1}$ in the periods 1990–1999 and 2000–2011, respectively. These results could be linked to hydrological conditions, as well as to the agricultural management in each period. The higher plant N uptake in 2000–2011 compared to 1990–1999, where the fertilization rate was known to be higher during the first period, underlines the over-fertilization that has occurred in the watershed in recent decades. The nitrate load in the river outlet showed very good performance, with NSE and R^2 at a daily time step of, respectively, 0.77 and 0.86 in the calibration period, and in the ranges 0.73–0.75 and 0.76–0.83, respectively, in the validation periods.

Hydrometeorological conditions, crop type and management, and soil characteristics are the factors that most influence the environmental impact of agricultural practices. Our results have shown the slow response of nitrate concentration in the aquifer to the decrease in fertilizer doses. Specifically, it seems that it took a decade to reach the target of the Code of Good Practices adopted in 2000.

We have deduced that in the long term, N surplus generally follows the trend in fertilization input, which directly affects the groundwater nitrate concentration. However, streamflow nitrate concentration is governed not only by groundwater nitrate concentration but also by surface water contribution. Due to the aquifer characteristics, it has been possible to reproduce historical trends in aquifer nitrate concentration with the SWAT hydrological model, an achievement that has not been reported before for any other watershed. We also note that SWAT can be used in similar hydrogeological and agricultural settings to obtain annual groundwater concentration tendency. Moreover, it is remarkable that the nitrate groundwater concentration in the aquifer is within the target of the European Nitrate Directive with the actual agricultural management.

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