Economic valuation of the natural service of nitrate regulation provided by rivers including dilution effects: application to a semiarid region, the Ebro basin (Spain)

Javier Tapia[a], Jorge Bielsa[a*], Yolanda Martínez[a], Sabine Sauvage [b], Roxelane Cakir [b], Mélanie Raimonet [b], Magali Gerino[b], José Miguel Sánchez-Pérez[b]

[a] Department of Economic Analysis, AgriFood Institute of Aragon (IA2), University of Zaragoza, Spain.

[b] ECOLAB, Université de Toulouse, CNRS, INPT, UPS, Toulouse, France.

* Corresponding author: jbielsa@unizar.es. Postal address: Facultad Economía y Empresa, Gran Vía 2, 50005 Zaragoza

Abstract

Nitrogen pollution is one of the main problems affecting the environment and human activity. This is especially true in semiarid regions where the quantity-quality relationship is a key constraining factor. In this paper, we propose and apply a method to quantify the economic value of the environmental service a river provides as a nitrate purifier/diluent. The main aim is to provide water management decisions with a solid and easily replicable method. Specifically, this study proposes a method to assess two processes of the water quality regulation service, namely, nitrate removal and dilution, through avoided decontamination costs. The proposed method is applied in the Ebro river basin as a role model. This application allows us to define the concepts in an operational manner. Since water availability forecasts are a key factor, we explore and analyze our results in accordance with several climate-change scenarios. The results show that the economic value of the removal service at the watershed scale amounts to about €92.3 million per year. Estimates of the annual economic value of the dilution service in the whole basin range more largely from €21.8 to €111.7 million, depending on the climate change scenario considered. To our knowledge, this is the first time that the dilution and removal services provided by rivers are jointly assessed.

Keywords: water management; nitrate decontamination; water quality regulation service; removal service; dilution service; semiarid region.

Highlights

- When it comes to nitrogen pollution, water quantity-quality relationship is a key factor for water quality regulation services in semiarid regions.
- The article proposes and applies a method to quantify the economic value at a watershed scale of the environmental service provided by rivers as nitrate purifiers/diluents.
- In addition to the already analyzed service of nitrogen removal, the paper defines and proposes a calculation method for measuring the dilution service provided by water flows.
- The method requires a large amount of georeferenced data and is water-management oriented.
- The total value of nitrate regulation services provided by the Ebro river basin is found to be between €92.7 and €134.8 million per year.

1. Introduction

Nitrogen pollution is one of the main problems affecting the environment and human activity (Tilman et al., 2001). This is especially true in semiarid regions where the quantity-quality relationship is a key constraining factor. Although general ideas on the importance of considering quality aspects as an inherent part of water quantity management are not lacking, we do need concepts with practical application to achieve a successful environmental management (Groffman et al., 2006). With this goal in mind, we propose and apply a method to quantify the economic value of the environmental service a river provides as a nitrate purifier and diluent. This specific goal belongs to the broad field of economic valuation of environmental services, which started with the work of Costanza et al. (1997). A very useful review of the state of the art a decade afterwards can be found in Haines-Young and Potschin (2009).

Nevertheless, for the specific purpose at hand, there is also extensive literature on negative externalities of nitrate entering rivers due to human activity. Using contingent valuation techniques, which were extremely popular at the time, Loomis et al. (2000) suggested measuring the willingness to pay for the recovery of a river's dilution capacity. Along different lines, Ribaudo et al. (2005) posited an emission permit scheme mirroring the one successful in reducing nitrous oxide emissions by coal power plants, which were responsible for acid rain. In the case studied by Ribaudo et al. (2005), urban residents were supposed to pay farmers to reduce to a minimum their use of fertilizers. The end goal was to curb the rampant eutrophication process that was taking place in the Mississippi Delta.

Both the 'Coasian' market approach by Ribaudo et al. (2005) and more centralized or state-dependent management and valuation are required to tackle the non-point source pollution of nitrate. Approaches from the industrial ecology field are not lacking. For instance, Watanabe and Ortega (2011) addressed the problem in terms of the energy involved both in the contamination and decontamination processes. According to these authors, the nitrogen cycle, whether in water or air, can be reduced to an energy-loss process. In this vein, reversing these losses necessarily involves energy provision and subsequent monetary expenditure. In this case, the value of pollution is derived from the costs we are forced to bear to return to the previous status.

Other indirect methods in the literature involve environmental services that focus on valuing wetlands' ability to remove nitrogen. By devoting a portion of land to absorb the nitrogen present in the water, the avoided costs minus the costs associated with these new wetlands might approximate the value of the environmental service provided. Based on this approach, Jenkins et al. (2010) aimed to calculate the value of wetlands used for nitrogen mitigation in the Mississippi River basin. La Notte et al. (2012) and Grossman (2012) took a similar approach for the Mediterranean region and the Elbe River, respectively. Although these papers differ significantly, they share a common ground: service values are calculated under the assumption that the basins contain enough water and land to substantially reduce the nitrogen load. In other words, the purification relies on the relative abundance of water and suitable land to become a nitrogen-reducing wetland.

Nevertheless, the water volume in rivers in southwestern Europe is lower than in the Elbe or the Mississippi. These southern Europe basins simply do not have enough water to afford new wetlands around the main rivers. In fact, such low streamflow is precisely the reason why the concentrations of some pollutants are so high and increasing. Consequently, although these contributions provide relevant and up-to-date information, we cannot follow the same scheme to analyze nitrogen pollution in the semiarid regions of southern Europe. That is why we need to add the value of the ability to dilute and not only the removal capacity of rivers.

From a wider and more general point of view, Liu et al. (2012) suggested measuring the nitrogen footprint in every major river on earth. Their calculations are based on very broad standards as they aim to provide a picture of the problem worldwide. Although this paper proves highly useful in developing a global perspective of the major figures, this approach is overly aggregate when it comes to investigate the problem of nitrification in some specific areas.

Furthermore, instead of focusing on economic valuation, their approach addresses water resource measurement, that is, the volume of water 'involved' in diluting the nitrogen discharged into watercourses. Nonetheless, this idea does fit in with our study because water scarcity is, as we mentioned earlier, one of the main features of our basins. The ratio between the total mass of pollutant and the total available water points to a dilution capacity indicator in semiarid regions.

This concept is not new. In the early 1970s, Falkenmark and Lindh (1974) introduced the idea of the water required for pollutant dilution. Bielsa et al. (1998) termed this concept 'degradative consumption'. Finally, it gained in popularity under the better-known term of 'grey water' in the precise definitions provided by Hoekstra and Chapagain (2008) and Hoekstra et al. (2011). More recently, Zeng et al (2013) and Liu et al (2016) have also addressed the quantity/quality relationship in order to assess water scarcity. They have proposed new indicators able to integrate both aspects and act as a more accurate measure of water availability in watersheds.

The growing problem of nitrate concentration stems from both the amount of nitrogen discharged into the rivers, the dilution capacity of these rivers and the river ability to transform N forms into N^2. High concentrations can be diluted by higher stream flows in rivers, while decreasing stream flows due to climate change can increase pollutant concentrations. In the end, what is at stake is the capacity of the environment—in this case the rivers—to act as a diluent for the pollutant resulting from human activity. That is why the quantity-quality relationship is so important in semiarid regions, where there is not enough water to feed wetlands.

Against this background, our methodological approach consists in calculating the value of environmental services through avoided costs, i.e. the hypothetical cost to replace the service that the ecosystem now provides at no cost. Papers cited above (Liu et al. 2010 and Grossman 2012) rely on the same approach but with a major difference: they assign the avoided costs as a positive value of the wetlands. According to La Notte et al. (2012), we suggest considering those avoided costs when they decontaminate and dilute the nitrogen load as a value of the environmental service provided by stream flows. We follow the work by Martinez and Albiac (2004), who assess environmental damage through depuration costs arising from agricultural activities.

This method requires a high amount of georeferenced data to prove useful in water-management decisions. We perform these calculations using the Ebro river basin as a role model. As suggested by Lowe et al (2018), our proposal aims to contribute to the literature on the valuation of environmental services provided by rivers by defining a method based on avoided costs. This study chose to focus on nitrogen-related environmental services instead of other services such as sediment removal, heavy metal or pesticides because it is more accurate to work with solute components to consider the dilution process. Nitrate compounds are the worldwide common proxy of water quality. About water quality regulation service assessment, so far we have been working on the Nitrate retention regarding only the biological process of denitrification at the scale of watershed or river reaches. Nitrate removal is one way to quantify water quality regulation service but others approaches to quantify these services are possible (Shamshirband et al., 2019) based on other indicators like water-sediment regulation (Xia et al. 2016), chlorophyll a regulation (Neal et al. 2006), bacteria regulation

(Mokondoko et al. 2016). Nitrate removal quantification was done in most of the papers through SWAT modelling that also allows quantifying the inputs of N from water runoffs. Concerning the modelling approach with SWAT model, previous studies worked at modelling hydrology and water quality at the scale of the South of Europe (Cakir et al. 2020b), including Ebro. A methodology to evaluate the nitrate removal was developed over the Garonne basin (Cakir et al. 2020a) only based on biological removal. Then we transferred in this paper this methodology to the Ebro basin based on the modelling approach at the scale of the South of Europe published in (Cakir et al. 2020b) which was never led before. Other hydrological models have been applied to the Ebro River but do not model nitrate retention. Romero et al. (2016) valuated N retention in soil with the Nani model, based on statistical relationships. Our study worked on the in-stream retention using a process-based model (including biological and physical processes). The novelty with respect to previous works steams in its economic approach.

Specifically, we propose to value economically two types of nitrogen-related environmental services: removal and dilution. Both biological removal and dilution processes are two different processes that are involved in the water quality regulation service: biological removal and physical dilution. Both biological removal and dilution processes are two different processes that participate in the same service of water quality regulation service for nitrate since they are both acting to reduce in stream N concentrations. The economic valuations of these two processes are different which led this study to separate the biological removal and dilution process. To our knowledge, this is the first time that the dilution and removal services provided by rivers are jointly assessed. The applied example allows us to define the concepts in an operational way, sheds light on practical knowledge of environmental processes related to nitrate and incorporates the economic valuation that pursues sustainable river management. Since water availability forecasts are a key factor, we explore and analyse our results using different climate-change scenarios.

The structure of the paper is as follows. We start with the precise definition of the proposed valuation method and the dataset we need to calculate it. We then present and discuss the results of these calculations in the specific case of the Ebro River basin to highlight the policy implications we can draw from them. The conclusions section closes the paper with a brief summary of the concepts and results we can draw from our work.

2. Method and data

Below we outline our specific proposal for quantifying the value of the ecosystem services a river provides as a nitrate decontaminant. We begin with the methodological approach and then we present the dataset we used. We selected the Ebro River basin (85.534 km²) in northeastern Spain as a pilot area for the study in a context of semiarid region, so the climate change forecasts deserve attention. This river has some advantages as a case study: on the one hand we have the necessary data over a sufficiently long period of years, with a high and homogeneous periodicity; on the other, it is a river that supports an important industrial, agricultural and livestock human activity. In its basin there are municipalities of various sizes, which we found interesting in order to illustrate the differences in cost. The Ebro River has been the subject of numerous management plans in recent decades that include water transfers to other basins, plans to extend the irrigated area and also new regulation works (dams).

2.1. Method

We consider that the nitrogen-related services provided by rivers fall into two main types:

i) Nitrogen removal service (RS), which is the natural depuration of free-flowing water as a result of the bio-physical activity of river ecosystems.

ii) Dilution service (DS), which occurs when the river acts as a dissolvent/diluent for the nitrate discharged into the watercourse using a specific water flow.

The RS value is obtained by combining available data on removal rates in each stretch of the river (e.g. kg per linear meter of stream) with geographical data on the river length. These data were provided by a hydro-agro-environmental modelling approach calibrated on data measured, including the point and non-point sources of nitrate (Cakir et al. 2020b) and the bio-physical processes involved in the nitrate removal in the rivers reaches (Cakir et al. 2020a). The DS value is demonstrated when a temporary (drought) or permanent (global warming) event reduces water flows and, as a result, increases nitrate concentrations. The total value of the environmental service of water decontamination is the sum of both types of services (RS and DS).

Our approach involves valuing RS and DS using avoided costs, i.e. the costs the river saves by keeping nitrate concentrations below a specific level, whether by removal or dilution. This definition allows us to quantify, compare and analyze how these environmental services performed by the river evolve. We will assess these avoided costs taking as a reference the average treatment costs incurred by the treatment plants in the region.

In order to estimate the current avoided costs due to the DS performed by rivers, we must consider a benchmark for the stream flows which we can compare with. Specifically, we take as a reference value the resulting stream flows after different scenarios of climate change. In addition, global warming also alters the RS value, since reduced water flows affect biological activity. Our geographical unit of reference is the hydrological subsystem (HS), which is a division of the basin into smaller units, which are the management units used in Spain by the water agencies. This division makes it possible to combine both global and specific views. The decision to use the HS instead of other possible options (entire basins, sub-basins, water bodies, municipalities, etc) is that it is the basic unit of management used in Spain by the water agencies, as is the case in most of the important basins in Europe. Our proposal aims to provide a useful tool for the real management of water, so we adapt as much as possible to the current conditions.

Below we explain our proposal for both services economic valuation in a more detailed manner.

2.1.1. Economic valuation of the nitrogen removal service

The economic valuation of the RS consists in calculating the costs that would be involved in treating the mass of nitrogen the river ecosystem removes on a natural basis. In other words, using the avoided treatment costs, we value the natural depuration service the river performs. This involves multiplying the river's natural nitrogen depuration rate (in kg per linear meter) by the length of the river system in linear meters in every specific HS. Then, we multiply the result by the cost per unit of mass ($€\cdot kgN^{-1}$) specific for each subsystem in accordance with its equivalent population in terms of treatment costs, as detailed below. Consequently, we obtain a different value in each HS, which can be represented in a map of the RS value.

The different value for each HS has two origins: firstly, every subsystem's removal capacity is different and, secondly, each HS involves different treatment costs depending on its specific population features. The removal capacity also depends on the river status, which includes the streamflow. For this reason, streamflow losses due to global warming change the river's removal capacity and thus reduce the value of the service in the future.

The following dataset is needed to estimate the value of this service:

- Nitrogen mass removed per year by the river, measured in units of total nitrogen removal per linear kilometer of system ($kgN \cdot m^{-1} \cdot year^{-1}$).
- Treatment costs attributed to the plants operating in the subsystem, population in the municipalities and data on specific livestock activities.
- A river network shapefile including the length (i.e. Geographical Information System (GIS) layer), which can be obtained automatically in the geographical model based on digital elevation model.

Concerning the calculation of every HS treatment costs; we follow Grossman (2012) to assign different cost depending on the population size of the HS. Grossman's work details different costs according to the connected person equivalents based on actual data showing relevant question regarding the cost of treatment: the existence of economies of scale (the larger the population the lower the unit costs). Since the average unit costs of each HS, including both fixed and variable costs, critically depend on the population, we need to assign a measure that considers population dispersion or concentration in each one. For this reason, we assign a cost to every HS based on an average weighted for the population size of the municipalities that form it and for livestock activities in the area.

2.1.2. Economic valuation of the dilution service

As mentioned above, quantifying the DS essentially consists in measuring the streamflow capacity to decrease the nitrogen load. This benefit is considered as efficient only when nitrogen concentration is not exceeding the maximum concentration levels allowed. This capacity obviously depends on the quantity of water flowing at each point. In other words, we are measuring the river's capacity to dilute the nitrogen load.

Hoekstra and Chapagain (2008) proposed to express grey water as the volume of 'natural' water that would be needed to dilute the extra pollutant that is currently being discharged into the river. It expresses the DS in general terms and in the quantity of water units. This approach is very useful for comparing different basins or assessing how pollution issues evolve over time.

By contrast, our proposal aims to obtain monetary and locally specific measures. Consequently, we only consider the part of the DS the river currently performs using its streamflow. Any streamflow reduction will mean a loss in this service and, therefore, a monetary cost in terms of subsequent treatment costs. Thus, we need a reference point to value the service currently offered in terms of the future treatment costs that are now 'hidden' by the present streamflow. To sum up, the DS can always be valued on a monetary basis by means of the costs saved by the dilution capacity.

Next, we outline the specific calculations we have to perform. We use a database of the concentrations (*c*) in mg of total N per liter and the streamflow (*q*) in cubic meters per second at a number of gauging points in the basin. This allows us to draw a map of how the mass of nitrogen evolves across the basin. For the sake of simplicity, we represent each basin with three subsystems: the main river (1) with two gauging points, and two tributaries with only one gauging point each: one on the left bank (2) and the other on the right bank (3).

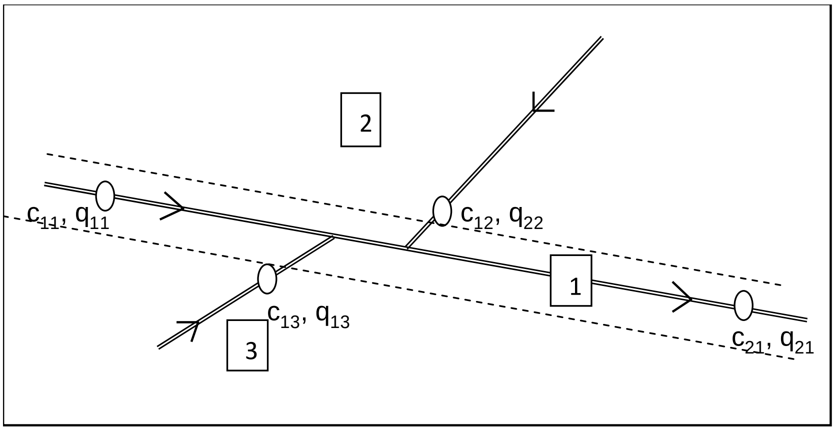


Figure 1. *Simplified data representation of a basin with three sub-systems and four gauging points;* c_{ij} *is the concentration (*$mg \cdot L^{-1}$*) of nitrogen in the point i of the HS number j;* q_{ij} *is the same for the discharge (*$m^3 \cdot s^{-1}$*).*

The nitrogen mass accumulated flowing at every point is Q_{ij} (in $mg \cdot s^{-1}$):

$$Q_{ij} = c_{ij} q_{ij} \quad [1]$$

Once we have this map of nitrogen masses, we have to distinguish between the natural loads, i.e. the nitrogen that would be there anyway, and the nitrogen produced by human activity. This split needs a reference for natural concentrations (c^{nat}), which can be observed at the head of the rivers, where there is almost no human activity. In fact, this only changes the reference or the lower bound we start from.

$$Q_{ij}^{nat} = c^{nat} q_{ij} \quad [2]$$

We also need to establish the 'acceptable' or admitted concentrations (cmax), i.e. the levels of mass per volume of water that trigger the water treatment process in the basin. This threshold logically entails an arbitrary judgment, but it is essential if we are to value the service in monetary terms. Furthermore, as Joseffson (2018) points out, a specification of the generic concept of ‘water damage’ is required to have an operative measure of this typing point for actual water management.

The first calculation of grey water volume (GW in L·s-1) can be directly drawn from the series of data we have. This value, accumulated up to a specific point (i , j), would be the volume of ‘natural’ water (cnat in mg·L-1) required to dilute the extra mass of nitrogen discharged, which results in acceptable water, i.e. water with concentration (cmax):

$$GW_{ij} = \frac{Q_{ij} - Q^{nat}}{c^{\max} - c^{nat}} \qquad [3]$$

In other words, GW is an aggregate measure of dilution needs in absolute terms and can be expressed in physical units of water. The higher the standard of acceptable water (the lower cmax), the higher the volume of water committed to compensate for this non-point source pollution. This indicator allows us to assess the evolution of pollution or to compare the situation of different basins using the ratio between GW and total water resources. If the ratio exceeds one, the basin no longer has dilution capacity.

Nevertheless, our proposal aims to go further and calculate an economic value for this spontaneous dilution the river performs using its streamflow. Taking a reference point with concentration and flow c, q respectively, that is, a mass of nitrogen Q = c q, and assuming that this mass of nitrogen is constant (exogenous), we can express the concentration in terms of the streamflow as follows:

$c = \frac{Q}{q}$, given a total nitrogen mass Q, we can represent a curve which inversely relates *c* and *q*, as shown in Figure 2.

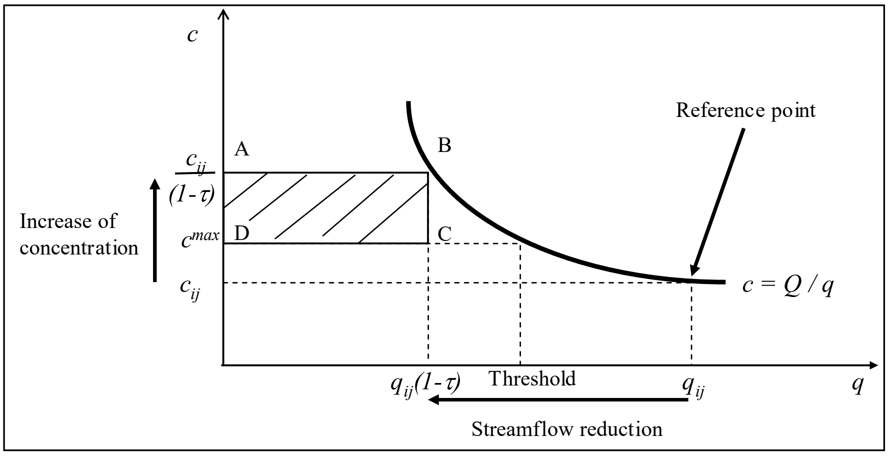


Figure 2. *Nitrogen mass to depurate if* $c_{ij} < c^{\max}$

We suppose initially that, with a reference point, $c_{ij} < c^{max}$ it happens a reduction of streamflow at a rate τ that leads us to a non-allowed concentration.

The pollutant quantity in excess to remain at the c^{max} level, that is to say, the treatment needs resulting from the streamflow reduction at that rate τ, is drawn on Figure 2 by the area ABCD, that is, $\left(\frac{c_{ij}}{(1-\tau)} - c^{\max}\right) q_{ij}(1-\tau)$. We have to artificially remove that amount of nitrogen from water. We can calculate the total costs D_{ij} (euros) associated with the depuration of this pollutant amount:

$$D_{ij} = d_{ij}\left(\frac{c_{ij}}{(1-\tau)} - c^{\max}\right) q_{ij}(1-\tau); \qquad [4]$$

where d_{ij} is the unitary cost per unit of mass (in *euros.mg* $^{-1}$).

If the initial concentration is already higher than $c^{\max}$, the reduction of streamflow leads to additional needs of decontamination as represented in Figure 3.

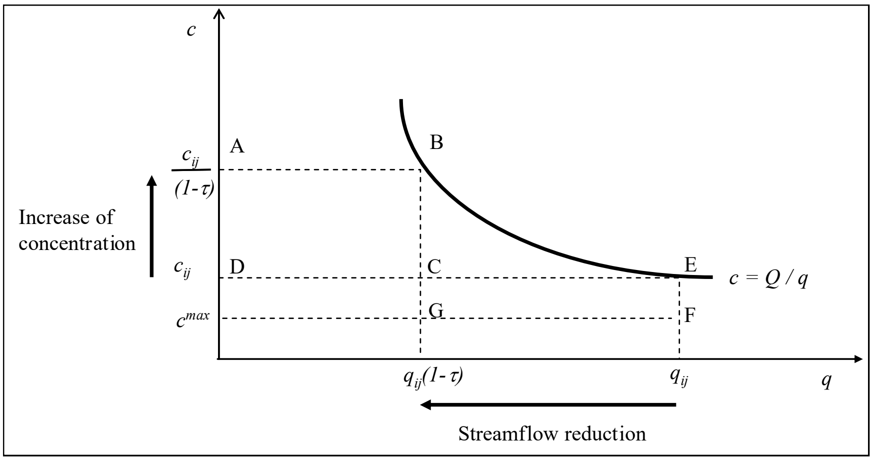


Figure 3. *Nitrogen mass to depurate if* $c_{ij} > c^{\max}$.

We have to calculate the net amount of nitrogen we should now remove as the difference between the areas ABCD (treatment needs due to higher concentration) minus CEFG (pollutant content in water loss previously treated), which is $c^{\max} q_{ij}\tau$. The treatment costs associated with this streamflow reduction can be calculated the same way as above multiplying that mass by the unitary costs per unit of mass d_{ij}:

$$D_{ij} = d_{ij} c^{\max} q_{ij}\tau \qquad [5]$$

In a third possible case, that is, when $\frac{c_{ij}}{(1-\tau)} < c^{\max}$, the expression [4] will be negative and the avoided costs will, logically, be zero.

This methodological approach considers only the actual avoided costs with respect to a legal reference. If we wanted to consider the costs caused by all the nitrogen present in the water, the calculations would be easier and the amount much higher.

That is precisely one more difference with the grey water approach. All the nitrogen mass that exceeds the river's natural load generates dilution needs and, therefore, entails a positive grey water footprint of pollution. By contrast, our method assumes that the costs are zero until the concentration reaches a given allowed threshold and that they increase due to additional loads.

Finally, the DS value rises as the total mass of nitrogen (Q) increases. Graphically, that means that the equal-mass curve moves away from the origin of the graph. Consequently, all the areas under the curve, i.e. the nitrogen to be treated, are larger.

2.2. Hydrological and Treatment Cost Data

2.2.1. Georeferenced hydrological information

The Ebro River basin is composed of 20 HS, which are the units used to calculate the value of the environmental services. Initially, we had 1,241 potential gauging points for nitrogen concentrations, of which only 115 have measurements on 12 months of the year for a long enough span (Confederación Hidrográfica del Ebro (CHE), 2018). With the aim of gaining an adequate picture of the basin, we choose 45 points distributed as shown in Figure 4. These points contain the information that we assign to every HS.

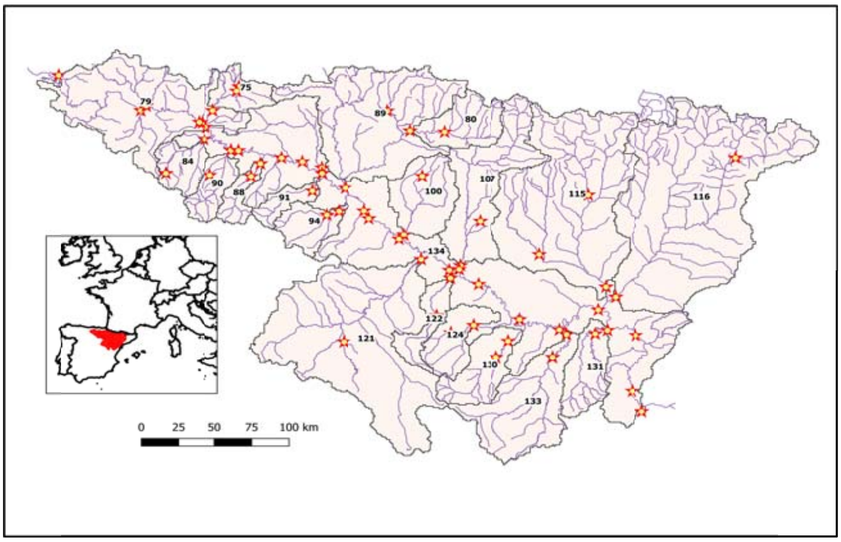


***Figure 4:** Gauging points and hydrological subsystems selected in this study.*

The dataset needed for our study comes from three different sources: hydrological information on surface streamflows, nitrogen concentrations and natural nitrogen removal rates at the same points by modelling approach (CHE 2018, Cakir et al. 2020b) and, finally, treatment costs associated with nitrogen removal (own calculations based on Grossman, 2012).

The streamflow and nitrogen concentrations were simulated by a calibrated and validated Soil and Water Assessment Tool (SWAT) model (Arnold et al., 1998) taking into account human pressures stemming from crop management (fertilization and irrigation), dam management, and municipalities' effluents (with a population equivalent capacity of over 50,000) (Cakir et al., 2020b). SWAT model is able to simulate in-stream retention using a process-based model (including biological and physical processes) whereas other models, such as Nani model (Romero et al., 2016), will only simulate N retention in soil.

SWAT is a public hydro-agro-environmental model that is widely known (Fu et al., 2019; Gassman et al., 2007) and applied all around the world from small to large scale (Abbaspour et al., 2015). Based on the river network and the topography, the watershed is divided into sub-basins at each river junction between the tributaries and the main river (Fig. 1). The model is based on hydrologic response units (HRU aggregated at sub-basin level). In each sub-basin, an HRU is characterized by a soil, land use and slope combination. The hydrology and nitrogen components are calculated at HRUs level and then computed at the sub-basin level. For the sake of clarity, we call each of these geographical basin subdivisions a hydrological subsystem (HS).

The model was set up and calibrated at monthly time scale based on different data set (irrigation, dams, Waste Water Treatment Plants (WWTP), discharge and nitrogen concentration in different stations). The procedure of calibration and validation is detailed in Cakir et al. (2020b). SWAT documentation available online explained the theory and details of SWAT modeling components (Neitsch et al., 2011). SWAT simulates the fluxes of nitrogen on surface water and the natural removal of nitrogen at the level of each reach associated to each HS (Cakir et al. 2020a) at a monthly basis and can predict the impact of climatic changes on the fluxes and removal processes.

As a result of the SWAT model, information is available at each outlet of the Ebro River's tributaries (Fig. 4). The following subsystem outputs in the Ebro watershed is extracted using a monthly time step and computed annually from 1980 to 2010: (1) streamflow in $m^3 \cdot s^{-1}$; (2) nitrogen concentration in $kgN \cdot year^{-1}$; and (3) nitrogen removal, estimated by the in-out nitrogen load difference divided by the length of the reach in $kgN \cdot m^{-1} \cdot year^{-1}$. Detailed and consistent data are available for this 20-year series.

We can calculate both the evolution and spatial distribution of the nitrogen load across the Ebro River basin when we know streamflow and concentration. This raw calculation represents the ex-post result once the river has performed its RS. Meanwhile, these simulated nitrogen concentrations are the result of a certain streamflow at every point which dilutes the mass of nitrogen that has reached this specific point for whatever reason (either natural or anthropogenic).

2.2.2. Climate scenarios

The set of forecasts by Estrela et al. (2012) considers two different scenarios. For the Ebro River basin, where the study area is located, the forecasts for a moderate climate-change scenario indicate that available water will decrease by 11% and 14% over the time horizons 2041–2070 (medium term) and 2071–2100 (long term), respectively. Forecasts in a more pessimistic scenario estimate a reduction in water resources of 14% and 28% in the medium and long term, respectively. Vautard et al. (2014), with more updated information, consider a likely scenario of a precipitation drop of around 20% in southeastern Europe. Thus, we can consider this figure as a moderate guess for the lower bound of a likely general reduction in water resources. We will take this central scenario as a reference in our calculations.

However, there are also more spatially precise and specific water availability forecasts in our zone. We will use them as a sensitivity analysis of the results depending on water resources reduction. For the recent period, the MESAN (the Mesoscale Analysis System) re-analyses of air temperature and precipitation at a daily scale (Landelius et al., 2016) were used as SWAT model inputs from 1980 to 2010. The choice of MESAN was based on the following criteria: (i) the validity of the reanalysis compared to other available meteorological datasets, deemed satisfactory for France (Raimonet et al., 2017) and Spain (unpublished data), (ii) a gridded dataset at a daily time step and covering Europe, as the aim of the methodology developed in this paper is to be applied at the European scale, and (iii) a meteorological dataset that has been used to correct the bias of EURO-CORDEX (0.11°; Jacob et al., 2020) climate model outputs for the 21st century. For the future period, this study used six climate projections from the EURO-CORDEX project (Table 1) obtained from different global and regional climate models and the RCP (Representative Concentration Pathway) 8.5 extreme scenario (+8.5 W/m^2 in 2100 compared to preindustrial values) developed by IPCC (Intergovernmental Panel on Climate Change). These projections have been corrected for bias based on MESAN re-analysis (see Raimonet et al., (2018) for more details).

Table 1. *List of EURO.CORDEX climate model outputs used in this study. Details about GCM and RCM models can be found on the EURO-CORDEX website (http://www.euro-cordex.net/).*

RCP(1)	GCM	RCM	Institute	Code
	IPSL-CM5A-MR	IPSL-INERIS-WRF331F	IPLS(2)	IPSL85
	CNRM-CM6	CLMcom-CCLM4-8-17	CNRM(3)	CNRM85
	MPI-ESM-LR	CLMcom-CCLM4-8-17	MPI-M(4)	MPI85_CLM
8.5	MPI-ESM-LR	MPI-CSC-REMO2009	MPI-M(4)	MPI85_MPI
	EC-EARTH	CLMcom-CCLM4-8-17	ICHEC(5)	ICHEC85_CLM
	EC-EARTH	KNMI-RACMO22E	ICHEC(5)	ICHEC85_KNMI

(1) Representative Concentration Pathway
(2) Institut Pierre-Simon Laplace
(3) Centre national de recherches météorologiques
(4) Max Planck Institute for Meteorology
(5) The Irish Centre for High-End Computing

2.2.3. Treatment plant costs

To assign treatment costs, we first need to consider the maximum concentration allowed at the plant outlet or the specific concentration that triggers the need for depuration. This limit is set in Spain by a Royal Decree (11/1995) in response to the requirements by the Nitrate

Directive 91/271/CEE in a range from 2.25 to 3.38 mg total $N\cdot L^{-1}$ where TN (total nitrogen) includes all nitrogen forms.

According to the Ebro River Basin Authority (Confederación Hidrográfica del Ebro, CHE), moderate quality is defined between 2 and 5.6 $mgTN\cdot L^{-1}$. To be conservative in the results, we set the higher limit (5.6 $mgTN\cdot L^{-1}$) as either the threshold for starting the plant activity or the concentration that the water has to achieve when it leaves the treatment plant. This is the value of the above mentioned c^{max}.

Concerning costs, the Spanish National Statistical Institute (INE) reports a cost range from €0.76 to €0.79 per m^3 for treated wastewater in general. According to Martínez and Albiac (2004), the specific cost of denitrification amounts to around €0.076 per m^3. This means that removing 1 kg of nitrogen, assuming a stream with a concentration of 11 mg $TN\cdot L^{-1}$ on entering the treatment plant, would result in an expense of 6.9€. However, the use of a single cost would limit the accuracy in estimating the value of the service when population and livestock activities determine treatment's unit cost.

Alternatively, as mentioned above, Grossman (2012) obtains an average total cost from a large sample of treatment plants (more than 2,000). Their data include fixed and variable denitrification costs. For this reason, the average cost per unit of treated mass decreases with the size of the plant. Grossman (2012) establishes six categories depending on the plant size of the plant (Table 2) which in turn depends on the population equivalent The distinction between costs according to population makes it possible to obtain a more realistic assessment of the treatment costs in each HS when the population is unevenly distributed, as is the case in our study basin. For this reason, we are going to use Grossman' data costs in accordance with the population and livestock activities and we also incorporate a measure that considers dispersion or concentration in each HS.

Table 2. *Treatment costs and plant size.*

Population equivalent	*Treatment costs (€/kgTN)*
<1000	14
1000 - 5000	9
5000 - 10000	6
10000 - 50000	3
50000 - 100000	2.5
> 100000	2

Source: Grossman (2012)

Calculations were made using GAMS (General Algebraic Modeling System, Brooke et al., 1998) which is a modelling system generally used in ecological and economics fields particularly suitable to model environmental economics problems. The model was solved with the CONOPT2 algorithm.

3. Results and Discussion

Table 3 shows the municipalities where the stations are located and the initial values of nitrogen concentrations and water flows by subsystem. These values will be considered our starting point for the calculation of nitrogen-related river services. There are 12 HS that exceed the 5.6 mg $N \cdot L^{-1}$ limit above which it is necessary to assume treatment costs , including five stations with a concentration higher than the 11.3 $mg \cdot L^{-1}$ limit set by the EU Nitrate Directive. Only eight HS have a better water quality than the minimum established.

We present the results related to the RS and DS separately to facilitate discussion. First, we present the estimates related to the RS, since the value of this service is common across all climate-change scenarios considered. Then, we present the magnitudes related to the DS under the different climate-change scenarios that have been defined. Finally, we discuss the calculations obtained considering the total value of the service (RS+DS).

Table 3. *Initial annual average values of nitrogen concentration and water flows by subsystem.*

HS	Annual average nitrogen concentration ($mgN \cdot L^{-1}$)	Annual average water flow ($Hm^3 \cdot year^{-1}$)
75	**12.99**	42.93
79	**9.77**	786.83
80	3.28	248.09
84	**15.43**	41.86
88	3.96	50.51
89	3.70	957.29
90	4.78	106.49
91	4.48	12.66
94	**8.23**	6.86
100	**31.50**	85.75
107	5.60	111.60
115	2.42	723.42
116	5.53	753.55
121	**15.47**	40.39
122	**19.52**	28.80
124	**8.24**	1.41
130	**10.47**	6.22
131	**6.93**	17.34
133	**7.03**	14.07
134	**9.76**	3,491.08

3.1. Nitrogen Removal Service (RS)

Table 4 includes the estimated mean nitrogen natural removal coefficients by subsystem during the period studied, the length of the watercourses, the corresponding total annual nitrogen removed and the final RS value. The removal rates modeled by SWAT had been validated on the Garonne river nitrate removal by comparing with in situ measurement of removal rates (See Cakir et al. 2020a). According to Billen et al. (2018) and Wolheim et al. (2008), nitrogen removal in hydrosystem eliminates between 10 and 50% of the diffuse sources, the same studies estimated the leaching rate of croplands areas ranging from 11 to 46 kgN/ha/yr. If we applied this rate to our study cas, we found an average total annual nitrogen removed over the entire basin equal to 16,583 tons whereas our simulation estimated an average of 18,449 tons. The simulations are in the range of nitrogen removal rate of previous studies. Our estimates show that subsystems 115, 116 and 134 have the highest natural removal capacity within the basin, with values >2 kg·m-1. These subsystems correspond to the sub-basins of the Cinca and Segre rivers and the main course of the Ebro River, respectively, where there is more population pressure and where the majority of the agricultural and livestock activities in the basin take place (mainly pig production). Subsystems 79 and 89 also achieve nitrogen removal levels over 1.3 tons per year.

Considering the initial nitrogen concentrations in these subsystems (Table 3), we can conclude that the high natural removal capacity (>1 kg·m-1) plays a key role in keeping nitrogen concentrations below 11.3 mg·N.L-1, which is the limit established by the EU Nitrate Directive as a "good status" for all waters. For example, subsystem 116 would have a nitrogen concentration of 12.38 mg·N.L-1 instead of the current concentration of 5.53 mg·N.L-1 if the rivers did not remove the quantity of nitrogen that they currently do.

Table 4. *Natural nitrogen removal coefficients, length of the reach, total annual nitrogen removed and RS value by subsystem.*

HS	Natural removal coefficients ($kg{\cdot}m^{-1}$)	Length of watercourses (m)	Total annual nitrogen removed (kg)	Unit treatment cost ($€{\cdot}kg^{-1}$)	RS value (€)
75	0.2	202,600	40,520	2.77	112,240
79	1.4	984,226	1,377,916	3.40	4,684,914
80	0.3	373,260	111,978	6.33	708,821
84	0.3	224,866	67,460	9.51	641,545
88	0.2	86,635	17,327	8.56	148,319

89	1.5	1,004,314	1,506,471	4.64	6,990,025
90	0.3	260,816	78,245	9.52	744,892
91	0.2	97,185	19,437	5.66	110,013
94	0.3	162,806	48,842	6.45	315,031
100	0.6	276,685	166,011	5.48	909,740
107	0.7	584,248	408,974	7.65	3,128,651
115	2.0	1,452,006	2,904,013	4.71	13,677,901
116	2.7	1,973,227	5,327,715	5.99	31,913,000
121	0.9	946,584	851,926	7.21	6,142,386
122	0.1	135,560	13,556	6.36	86,216
124	0.3	178,263	53,479	12.41	663,674
130	0.4	270,052	108,021	11.17	1,206,595
131	0.3	308,956	92,687	9.62	891,649
133	0.6	545,011	327,007	7.99	2,612,786
134	2.8	1,759,830	4,927,524	3.57	17,591,261

The RS values shown in Table 4 vary greatly between subsystems, given that the value of the RS depends on three components: retention coefficient, length of water courses and unit treatment cost.. On the one hand, the high values of subsystems 116, 115, 134, 79 and 89 are related to a coefficient value greater than 1 and to the length of watercourses. All these subsystems have a unit cost of treatment lower than €6·kg-1 (i.e. more than 5,000 inhabitants). On the other hand, the high value in subsystem 121 is due to the length of the watercourses, despite having a removal coefficient <1 kg·m-1, and also to the higher unit treatment cost, since the population is smaller. In contrast, subsystem 75, located in the Ebro headwaters, where the population density is high, has a low RS value due to its low removal coefficient, short length and low treatment costs (see Table 4). According to our calculations, the amount of nitrogen retained by the river totals around 18,352 tons per year. As a result, the value of the service considering the treatment costs in each subsystem amounts to about €92.3 million per year.

3.2. Dilution service (DS)

According to our method for calculating the DS, we must consider the increase in the nitrogen concentration resulting from the water flow reduction as a reference for the current economic value of this service. Therefore, for the stations where the initial concentration is less than 5.6 mgN·L-1, the DS value will be noticeable only when the water flow reduction causes a concentration above 5.6 mg·N.L-1; in such a case, it will be necessary to incur treatment costs. In contrast, when a station currently has a concentration above 5.6 mg·N.L-1, we must calculate the costs that will be involved in returning to the initial nitrogen concentration level.

We first focus on the scenario of a 20% reduction in water flow and then we present the results under the other climate-change forecasts explained in the previous section. Table 5 shows the final annual average concentrations of nitrogen and water flows in each subsystem after a 20% reduction. In this scenario, five subsystems (numbers 80, 88, 89, 91 and 115) maintain a final concentration lower than 5.6 mg·L-1, which means that incurring treatment costs is not necessary and the DS value is zero. In the other subsystems, treatment costs either appear or increase with respect to the initial costs, since the final concentrations are above 5.6 mg·N.L-1.

Although a flow reduction in the same proportion has been simulated, the differences in the DS value between subsystems are remarkable, with a minimum of €28,700 (subsystem 124) and a maximum of €24.3 million (subsystem 134). The differences recorded in the DS values are related to the differences between HS in population (which implies different unit treatment costs) and in the initial water flow (see Table 3).

The highest DS values, associated with higher avoided costs, are found in the subsystems with the highest initial flows (subsystems 134, 79 and 116). By contrast, subsystems with the lowest DS values correspond to those with the lowest initial water flows, despite having higher unit treatment costs (subsystems 124, 94 and 130). According to this result, we can state that the DS is not homogeneous in the whole basin for two reasons: firstly, due to the existence of economies of scale in the unit treatment cost; and, secondly, because the flow loss in the subsystems with a higher initial flow will generate more costs than in those with an initially low flow, since our methodology values the absolute dilution capacity of the water (not the relative one). In Table 5, we find that the highest DS values are obtained in HS whose initial water flow is greater. This indicates that in these HS the current high water flows avoid incurring a high treatment cost. This result is critical for understanding that the existence of high water flows in certain areas of the river is providing remarkable DS by avoiding treatment costs. Quantifying this DS has important consequences for actual river management, as it influences how the impacts associated with any project that reduces water flows in the basin are estimated. The final DS value of this scenario in the whole basin exceeds €42.2 million per year.

Table 5. *Final nitrogen concentration, water flow and DS value for a reduction of 20% in water flow by subsystem.*

HS	Nitrogen concentration ($mg \cdot L^{-1}$)	Water flow (Hm^3)	Nitrogen to be diluted (kg)	Unit treatment cost (€·kg^{-1})	DS value (€)
75	16.24	34.34	111,514.1	2.77	308,894
79	12.21	629.46	1,535,255.3	3.40	5,219,868
80	4.10	198.47	0	6.33	0
84	19.28	33.49	129,233.3	9.51	1,229,009
88	4.95	40.41	0	8.56	0
89	4.63	765.83	0	4.64	0
90	5.98	85.19	32,207.2	9.52	306,613
91	5.60	10.13	0	5.66	0
94	10.28	5.49	11,299.7	6.45	72,883
100	39.37	68.60	540,334.7	5.48	2,961,034
107	7.00	89.28	124,739.6	7.65	954,258
115	3.03	578.74	0	4.71	0
116	6.91	602.84	789,242.4	5.99	4,727,562
121	19.34	32.31	124,982.8	7.21	901,126
122	24.40	23.04	112,418.7	6.36	714,983
124	10.30	1.13	2,318.9	12.41	28,778
130	13.09	4.98	13,033.6	11.17	145,585
131	8.66	13.87	24,039.9	9.62	231,264
133	8.79	11.26	19,775.3	7.99	158,005
134	12.20	2,792.86	6,809,276.2	3.57	24,309,116

Table 6 contains the estimates for water flows in the other climate-change scenarios considered, the variations (in percentage) with respect to the initial situation (Table 3) and the final nitrogen concentrations. These data enable us to identify the subsystems in which flows are expected to decrease, therefore increasing treatment costs. The results show that estimates vary significantly and neither the amount nor the sign of the variation between the different scenarios considered coincide. For example, the IPSL85 scenario predicts a loss of 4% of water flows for subsystem 79, while the ICHEC85_CLM predicts a loss of 27%, and the ICHEC85_KNMI scenario anticipates an increase of 18% in the same subsystem. The average flow fluctuation is -13% for IPSL85, +13% for CNRM_85, -11% for MPI85_CLM, +9% for MPI85_MPI, -2% for ICHEC85_CLM and +17% for ICHEC85_KNMI. Thus, we can consider these calculations as a kind of sensitivity analysis.

Although the estimates anticipate flow increases in some of the subsystems, the calculation of the current value of the DS requires a flow reduction. In other words, to estimate the treatment costs that the current river flow is avoiding, we need to know what the nitrogen concentration would be if the water was not diluting part of it. Thus, in the scenarios and/or subsystems with an increase in flow, we cannot assign any avoided costs. Therefore, according to our proposal, it is the final nitrogen concentration in each scenario when streamflow decreases that is relevant for the calculation of the economic value of the dilution service.

The information contained in Table 6 shows that in 64 of the 120 estimates, a flow decrease is expected and, consequently, an increase in the final nitrogen concentration. Therefore, treatment costs that are currently being avoided by a higher flow will have to be incurred. Specifically, subsystems 90, 91, 107 and 116, which initially presented concentrations lower than 5.6 mg·N.L-1, exceed this threshold in some climate-change scenarios. In contrast, subsystems 80, 88, 89 and 115 are still below 5.6 mg·N.L-1 after the flow decreases, and, consequently, have no DS value. For all other subsystems, as already explained, the corresponding DS value will be assigned by calculating the difference in concentrations.

Table 7 shows the final calculations of the DS value in all scenarios considered. The zero value in this table means that the expected flow reduction will not result in an increase in treatment costs (there is no avoided costs because nitrogen concentration is <5.6 mg·N.L-1). In contrast, when a value does not appear in this table, it means a flow increase is expected and, therefore, the value of the current DS cannot be calculated (the avoided cost cannot be observed). Predicted values vary between €112,000 and €21.8 million depending on the scenario, with ICHEC85_CLM and IPSL85 giving a higher total DS value for the basin. Compared to those calculated in the -20% scenario, all values are significantly lower. However, in all scenarios considered, subsystems 80, 88, 89 and 115 give a DS value of zero, since in these subsystems the initial N concentration is very low (<4 mg·L-1), so there is no need for treatment in any scenario. Subsystem 134 achieves the highest value in all flow reduction scenarios, followed by subsystems 116 and 79.

Table 6. *Estimated annual average water flows (Hm^3) and nitrogen concentration ($mg \cdot L^{-1}$) under climate-change scenarios.*

HS	IPSL85		CNRM85		MPI85_CLM		MPI85_MPI		ICHEC85_CLM		ICHEC85_KNMI	
	Water flow (Hm^3)	**Nitrogen ($mg \cdot L^{-1}$)**	**Water flow (Hm^3)**	**Nitrogen ($mg \cdot L^{-1}$)**	**Water flow (Hm^3)**	**Nitrogen ($mg \cdot L^{-1}$)**	**Water flow (Hm^3)**	**Nitrogen ($mg \cdot L^{-1}$)**	**Water flow (Hm^3)**	**Nitrogen ($mg \cdot L^{-1}$)**	**Water flow (hm^3)**	**Nitrogen ($mg \cdot L^{-1}$)**
75	47.74 (11%)	11.68	42.46 (-1%)	13.13	41.34 (-4%)	13.49	39.82 (-7%)	14.00	39.64 (-8%)	14.07	51.38 (20%)	10.85
79	752.59 (-4%)	10.21	669.34 (-15%)	11.48	760.00 (-3%)	10.11	811.41 (3%)	9.47	576.44 (-27%)	13.33	926.57 (18%)	8.29
80	218.03 (-12%)	3.73	231.86 (-7%)	3.51	223.67 (-10%)	3.64	251.03 (1%)	3.24	218.10 (-12%)	3.73	230.61 -7%	3.53
84	34.47 (-18%)	18.74	36.58 (-13%)	17.66	37.30 (-11%)	17.32	42.27 (1%)	15.28	31.70 (-24%)	20.37	52.75 (26%)	12.24
88	38.35 (-24%)	5.22	41.24 (-18%)	4.85	40.63 (-20%)	4.92	45.89 (-9%)	4.36	36.81 (-27%)	5.44	54.60 (8%)	3.67
89	968.85 (1%)	3.66	849.33 (-11%)	4.17	866.54 (-9%)	4.09	893.00 (-7%)	3.97	1007.68 (5%)	3.52	1005.24 (5%)	3.53
90	80.43 (-24%)	6.33	86.87 (-18%)	5.86	87.15 (-18%)	5.84	103.63 (-3%)	4.91	78.54 (-26%)	6.48	115.58 (9%)	4.41
91	8.58 (-32%)	6.61	11.12 (-12%)	5.10	9.99 (-21%)	5.67	13.88 (10%)	4.08	9.59 (-24%)	5.91	14.34 (13%)	3.95
94	5.01 (-27%)	11.27	6.27 (-9%)	9.01	5.85 (-15%)	9.65	7.43 (8%)	7.59	5.19 (-24%)	10.87	7.86 (14%)	7.19
100	81.59 (-5%)	33.11	131.29 (53%)	20.57	81.78 (-5%)	33.03	106.57 (24%)	25.35	114.63 (34%)	23.56	106.36 (24%)	25.40
107	105.72 (-5%)	5.91	123.52 (11%)	5.06	141.80 (27%)	4.41	149.59 (34%)	4.18	121.73 (9%)	5.13	108.80 (-3%)	5.74
115	790.74 (9%)	2.22	667.83 (-8%)	2.63	660.89 (-9%)	2.65	892.88 (23%)	1.96	910.24 (26%)	1.93	961.33 (33%)	1.82
116	704.34 (-7%)	5.91	921.27 (22%)	4.52	754.97 (0%)	5.52	786.10 (4%)	5.30	635.30 (-16%)	6.56	686.76 (-9%)	6.07

121	36.76 (-9%)	17.00	101.76 (152%)	6.14	34.51 (-15%)	18.11	49.00 (21%)	12.75	69.72 (73%)	8.96	75.01 (86%)	8.33
122	27.52 (-4%)	20.43	39.58 (37%)	14.21	24.19 (-16%)	23.24	29.61 (3%)	18.99	39.60 (38%)	14.20	36.69 (27%)	15.33
124	1.09 (-23%)	10.65	1.87 (33%)	6.20	1.14 (-19%)	10.21	1.81 (29%)	6.39	1.62 (15%)	7.15	1.61 (15%)	7.18
130	4.42 (-29%)	14.75	7.16 (15%)	9.10	4.83 (-22%)	13.48	7.34 (18%)	8.88	6.91 (11%)	9.43	6.79 (9%)	9.59
131	17.01 (-2%)	7.07	24.55 (42%)	4.89	12.60 (-27%)	9.54	18.24 (5%)	6.59	14.77 (-15%)	8.14	23.19 (34%)	5.18
133	7.39 (-47%)	13.39	15.10 (7%)	6.55	11.01 (-22%)	8.98	17.16 (22%)	5.76	9.28 (-34%)	10.66	15.62 (11%)	6.33
134	3070.26 (-12%)	11.10	3552.99 (2%)	9.59	3246.40 (-7%)	10.49	3646.64 (4%)	9.34	3248.57 (-7%)	10.49	3992.00 (14%)	8.53

Table 7. *Value of dilution services under different climate-change scenarios (€ per year) by subsystem.* [*]

HS	Dilution service					
	Scenario IPSL85	Scenario CNRM85	Scenario MPI85_CLM	Scenario MPI85_MPI	Scenario ICHEC85_CLM	Scenario ICHEC85_KNMI
75	-	16,716	57,003	111,720	118,368	-
79	1,135,763	3,897,180	889,899	-	6,978,508	-
80	0	0	0	-	0	0
84	1,085,611	775,983	670,139	-	1,491,739	-
88	0	0	0	0	0	-
89	-	0	0	0	-	-
90	560,558	217,275	202,273	0	661,561	-
91	48,934	0	4,075	-	16,856	-
94	98,380	31,631	53,658	-	88,723	-
100	719,317	-	686,301	-	-	-
107	250,291	-	-	-	-	118,074
115	-	0	0	-	-	-
116	1,324,010	-	-	-	3,638,952	1,913,649
121	404,741	-	655,653	-	-	-
122	159,282	-	572,088	-	-	-
124	32,561	-	27,828	-	-	-
130	210,978	-	162,453	-	-	-
131	22,129	-	316,253	-	171,359	-
133	375,209	-	171,799	-	269,002	-
134	14,651,209	-	8,518,840	-	8,443,099	-
Total	**20,990,431**	**4,938,785**	**12,988,262**	**111,720**	**21,878,167**	**2,031,723**

[*] The zero value means that the expected flow reduction will not result in an increase in treatment costs; when a value does not appear, it means a flow increase is expected.

3.3. Total value of water services

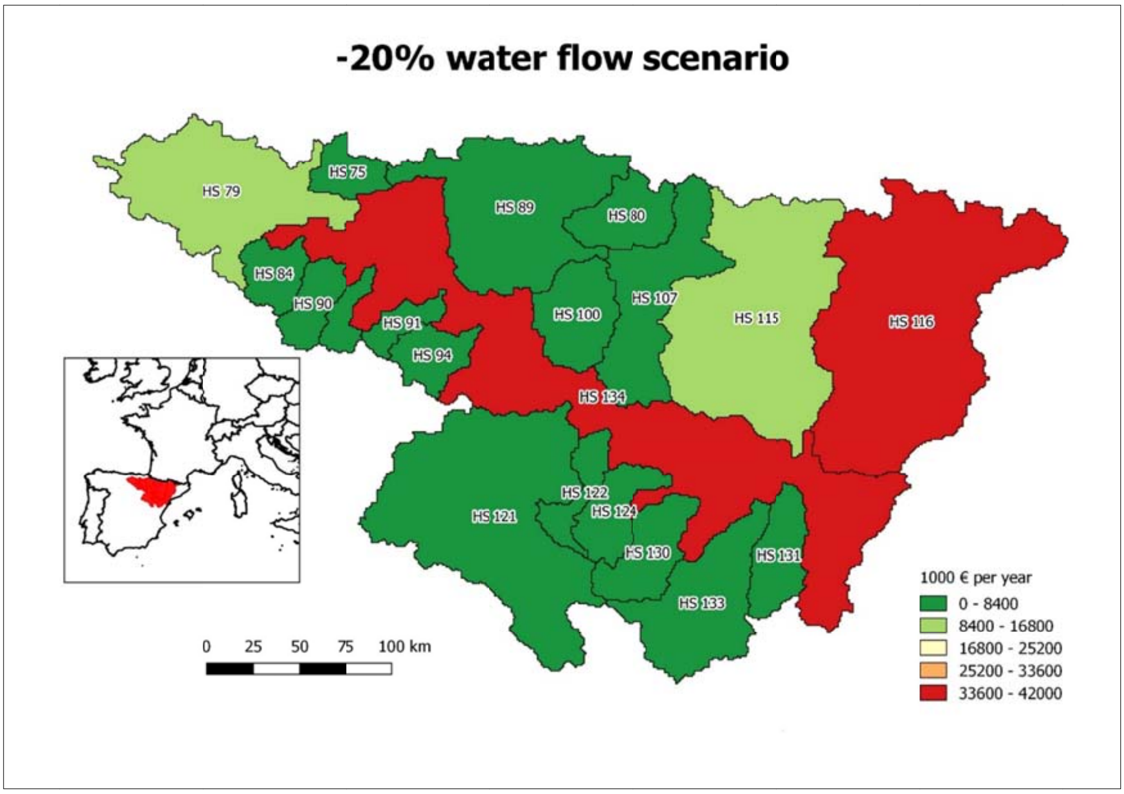
Figure 5 shows the sum of the values of the RS and DS in all scenarios considered for the whole water basin. The results indicate a total value between €92.7 and €134.8 million. In spite of the differences that depend on the varying climate-change forecasts, if we consider an identical probability of occurrence for all scenarios, then the expected value of the service amounts to €107.6 million, the majority of which (€92.3 million) corresponds to the RS. It is worth noting here that we have used a very conservative definition of the DS value. Since we consider only the dilution the river performs so that the water merely reaches the threshold, the result is very sensitive to that limit. Any change in this concentration benchmark would in turn change the total value of this service.

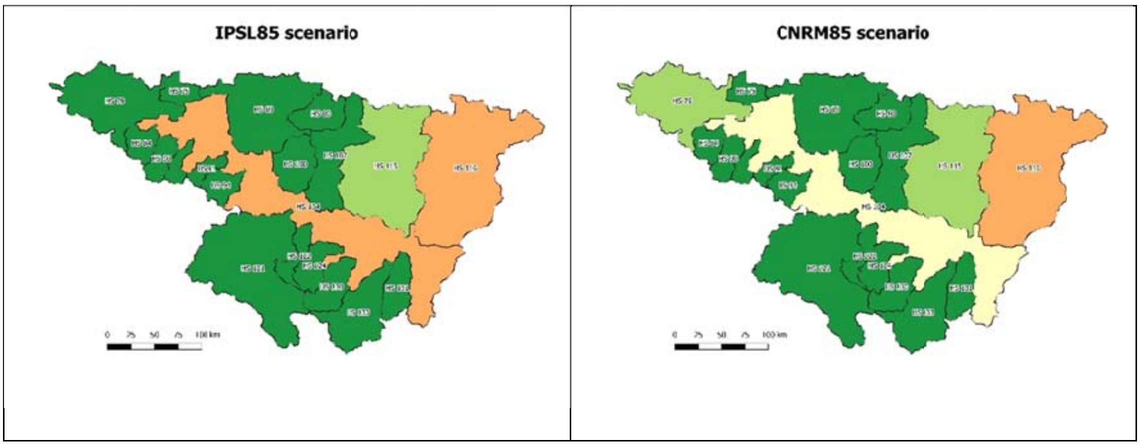
Considering the subsystem values, we can conclude that the highest water service values are attained in subsystems 134, 116 and 79 (Ebro main course, Segre River basin, and Ebro headwaters, respectively), which have the highest nitrogen removal and dilution values of the whole basin. Subsystem 115 (Cinca River basin) has a high total value only due to the RS, since, in this case, the DS is zero in all scenarios. Headwater subsystems 91, 75 and 94, however, have the lowest total values, which coincides with lower retention capacity (<0.4 kg·m-1) and low dilution values.

Of course, the figures calculated here should be viewed with caution due to the inherent limitations of the approach we have taken. Given that the avoided cost considered corresponds only to the excess of nitrogen diluted by the water that remains as a result of the different climate-change scenarios, the DS is being underestimated. In addition, the DS is undervalued in subsystems where the limit of 5.6 $mg \cdot L^{-1}$ is not exceeded.

An overall assessment of the results obtained in this study allows us to affirm that circulating water has an important economic value linked to the biological activity of the fluvial ecosystem and as a nitrate diluent that must be considered in the decision-making process of institutions involved in water management. This is especially relevant in those projects that involve water withdrawal, such as the increase in irrigated surface area, reforestation plans or even transfers to other basins based on the argument that some subsystems in the basin have abundant water flows. Our calculations show that the water flow and nitrate transformation by biogeochemistry in these subsystems avoid high treatment costs. This must be economically valued for a proper consideration of all river management that may favor or impair these services. However, the economic values linked to nitrogen-related environmental services are only a part of the total value of these services. The removal of other compounds (sediment, pesticides, heavy metal...), which participate in water regulation services, are not considered in the present study.

Despite the limitations of the proposed approach, the suggested method provides a consistent and easily replicable tool for economically valuing nitrogen-related water services, which is one of the main causes of concern related to non-point pollution in rivers worldwide and, especially, in Europe.





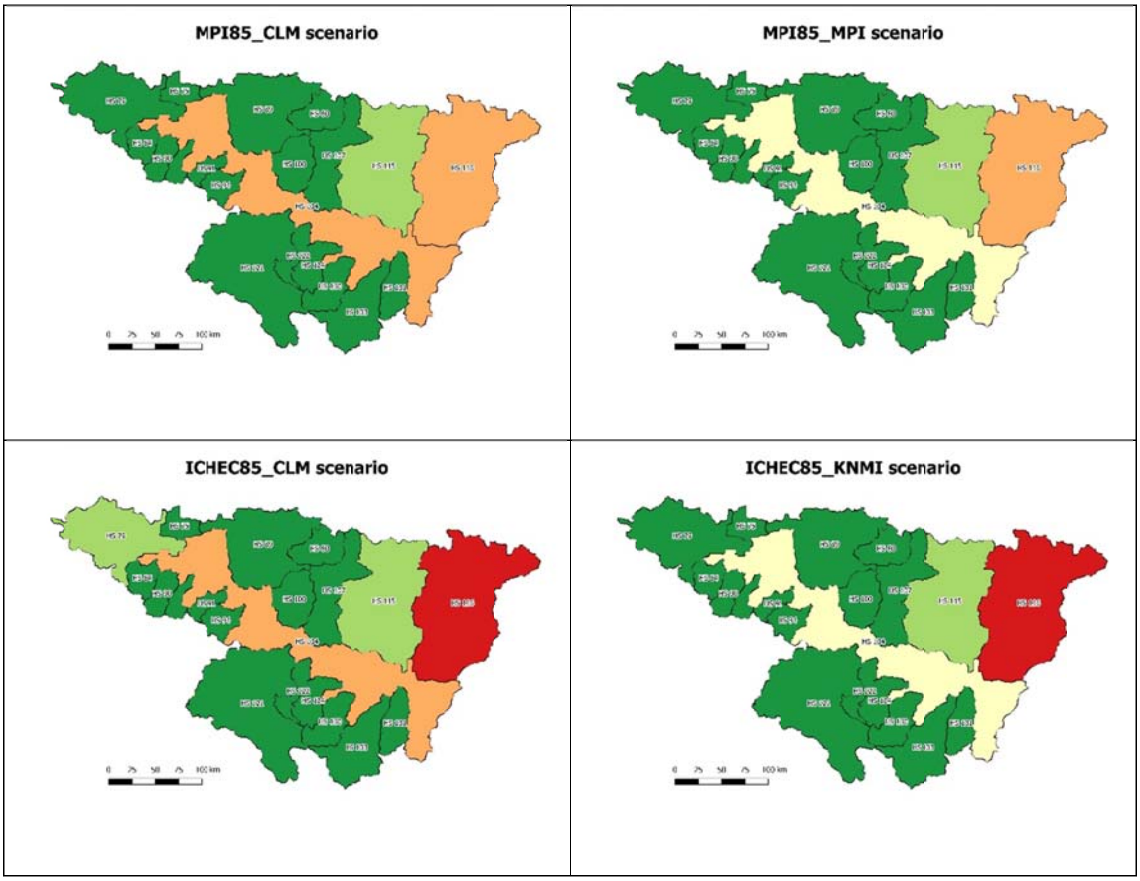


Figure 5. *Total value of the service (RS+DS) under different climate change scenarios (10^3 € per year) by subsystem.*

4. Conclusions

The first novelty of the study is to assess the water quality regulation service by taking into account both the physical dilution and the biological N retention which was never led before. The second novelty of this paper is to estimate an economic value to these functions and to sum them up in order to get an estimation of benefits that come from natural processes. The economic valuation was run through avoided decontamination costs. Other attempts of economic valuations of this service exist in the literature but are rare.

The third novelty of this paper is to test the influence of climate changes on the economic value that is only possible with the help of a modelling approach that includes physical and biological processes. Indeed, climate change is more influencing water discharges than biological processes, and only this coupled approach (dilution plus biological removal) allows the demonstration of climate change influence on this natural service of water quality regulation. This is the first try to our knowledge to show the influence of climate changes on the economic values of this service.

Moreover, policymakers face the challenge of designing policies that consider environmental and economic perspectives in a comprehensive and easy-to-apply way. This requires the implementation of management tools that incorporate the economic valuation of river ecosystem services. The main objective of this paper is to provide such a tool for a specific environmental problem that is particularly acute in southwestern Europe, namely, nitrate pollution combined with growing water scarcity. This quantity-quality relationship is expressed and calculated using the environmental service we call the dilution service (DS). Additionally, the good status of rivers also performs a service in terms of the removed nitrogen, which is already known as the removal service (RS). Both services clearly enhance the natural system capacity for tackling the (growing) mass of nitrogen discharged into water flows, that is, they contribute significantly to the water regulation service.

Specifically, these services avoid the treatment costs we would otherwise have to incur. Our proposal, therefore, is based on the avoided costs method. It combines estimated data on natural removal from geographical-information tools, available nitrogen concentration measures and streamflow data from some reference gauging points. Since a specific reference status is necessary to calculate the DS, several climate-change-scenario forecasts have been considered in the study. All the calculations and procedures have been developed under a geographical-information system (output from SWAT) and calculated by way of GAMS. Both data and codes would be available on request.

The results show that the ecosystem services linked to biological activity in river ecosystems, i.e., what we call the Removal Service (RS), has a high value. For its part, the Dilution Service (DS), which depends critically on the quality standard and on the assumption made on future stream flows, provides a wide range of results. Any further increase in the nitrogen load or any additional decrease in water flows would raise the value of this service currently rendered by rivers. Once we have established these reference points, the method we propose in this paper allows us to quantify the quantity-quality relationship, which is so important in semiarid regions as the one studied in this paper.

To the best of our knowledge, this study is the first methodological proposal and empirical application in the literature that jointly assess these two basic nitrogen-related environmental services at the scale of the watershed. The values of the avoided cost obtained in this study provide a good argument in favor of the conservation of both the natural river bed and its discharge. The accuracy of our results relies on the quality of the data. Provided that we could count with geographically breakdown and detailed data, we could export this method to other chemical pollutants (pesticides, heavy metals) and territories. A natural extension is the Phosphorus pollution, which has similar origin and dynamics. With respect to territories, this approach is initially more suitable to semiarid regions, as pointed out above, but global warming may increase the number of watersheds in which quantity shortfalls end up leading to quality problems as well. As other monetary valuations, our method could well be integrated in a wider assessment of environmental services valuation projects.

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