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

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11 ***Abstract***

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

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

29 ***Highlights***

- 30 • When it comes to nitrogen pollution, water quantity-quality relationship is a key factor for
31 water quality regulation services in semiarid regions.
32 • The article proposes and applies a method to quantify the economic value at a watershed
33 scale of the environmental service provided by rivers as nitrate purifiers/diluents.
34 • In addition to the already analyzed service of nitrogen removal, the paper defines and
35 proposes a calculation method for measuring the dilution service provided by water flows.
36 • The method requires a large amount of georeferenced data and is water-management
37 oriented.
38 • The total value of nitrate regulation services provided by the Ebro river basin is found to be
39 between €92.7 and €134.8 million per year.
40

41 **1. Introduction**

42 Nitrogen pollution is one of the main problems affecting the environment and human
43 activity (Tilman et al., 2001). This is especially true in semiarid regions where the quantity-
44 quality relationship is a key constraining factor. Although general ideas on the importance of
45 considering quality aspects as an inherent part of water quantity management are not lacking,
46 we do need concepts with practical application to achieve a successful environmental
47 management (Groffman et al., 2006). With this goal in mind, we propose and apply a method
48 to quantify the economic value of the environmental service a river provides as a nitrate
49 purifier and diluent. This specific goal belongs to the broad field of economic valuation of
50 environmental services, which started with the work of Costanza et al. (1997). A very useful
51 review of the state of the art a decade afterwards can be found in Haines-Young and Potschin
52 (2009).

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

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

71 Other indirect methods in the literature involve environmental services that focus on
72 valuing wetlands' ability to remove nitrogen. By devoting a portion of land to absorb the
73 nitrogen present in the water, the avoided costs minus the costs associated with these new

74 wetlands might approximate the value of the environmental service provided. Based on this
75 approach, Jenkins et al. (2010) aimed to calculate the value of wetlands used for nitrogen
76 mitigation in the Mississippi River basin. La Notte et al. (2012) and Grossman (2012) took a
77 similar approach for the Mediterranean region and the Elbe River, respectively. Although
78 these papers differ significantly, they share a common ground: service values are calculated
79 under the assumption that the basins contain enough water and land to substantially reduce the
80 nitrogen load. In other words, the purification relies on the relative abundance of water and
81 suitable land to become a nitrogen-reducing wetland.

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

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

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

102 This concept is not new. In the early 1970s, Falkenmark and Lindh (1974) introduced
103 the idea of the water required for pollutant dilution. Bielsa et al. (1998) termed this concept
104 ‘degradative consumption’. Finally, it gained in popularity under the better-known term of
105 ‘grey water’ in the precise definitions provided by Hoekstra and Chapagain (2008) and
106 Hoekstra et al. (2011). More recently, Zeng et al (2013) and Liu et al (2016) have also

107 addressed the quantity/quality relationship in order to assess water scarcity. They have
108 proposed new indicators able to integrate both aspects and act as a more accurate measure of
109 water availability in watersheds.

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

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

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

140 (Mokondoko et al. 2016). Nitrate removal quantification was done in most of the papers
141 through SWAT modelling that also allows quantifying the inputs of N from water runoffs.
142 Concerning the modelling approach with SWAT model, previous studies worked at modelling
143 hydrology and water quality at the scale of the South of Europe (Cakir et al. 2020b), including
144 Ebro. A methodology to evaluate the nitrate removal was developed over the Garonne basin
145 (Cakir et al. 2020a) only based on biological removal. Then we transferred in this paper this
146 methodology to the Ebro basin based on the modelling approach at the scale of the South of
147 Europe published in (Cakir et al. 2020b) which was never led before. Other hydrological
148 models have been applied to the Ebro River but do not model nitrate retention. Romero et al.
149 (2016) valuated N retention in soil with the Nani model, based on statistical relationships. Our
150 study worked on the in-stream retention using a process-based model (including biological
151 and physical processes). The novelty with respect to previous works steams in its economic
152 approach.

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

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

171
172

173 **2. Method and data**

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

185 **2.1. Method**

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

- 187 i) Nitrogen removal service (RS), which is the natural depuration of free-flowing
188 water as a result of the bio-physical activity of river ecosystems.
- 189 ii) Dilution service (DS), which occurs when the river acts as a dissolvent/diluent for
190 the nitrate discharged into the watercourse using a specific water flow.

191 The RS value is obtained by combining available data on removal rates in each stretch
192 of the river (e.g. kg per linear meter of stream) with geographical data on the river length.
193 These data were provided by a hydro-agro-environmental modelling approach calibrated on
194 data measured, including the point and non-point sources of nitrate (Cakir et al. 2020b) and
195 the bio-physical processes involved in the nitrate removal in the rivers reaches (Cakir et al.
196 2020a). The DS value is demonstrated when a temporary (drought) or permanent (global
197 warming) event reduces water flows and, as a result, increases nitrate concentrations. The total
198 value of the environmental service of water decontamination is the sum of both types of
199 services (RS and DS).

200 Our approach involves valuing RS and DS using avoided costs, i.e. the costs the river
201 saves by keeping nitrate concentrations below a specific level, whether by removal or dilution.
202 This definition allows us to quantify, compare and analyze how these environmental services

203 performed by the river evolve. We will assess these avoided costs taking as a reference the
204 average treatment costs incurred by the treatment plants in the region.

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

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

219

220 *2.1.1. Economic valuation of the nitrogen removal service*

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

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

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

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

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

252

253 *2.1.2. Economic valuation of the dilution service*

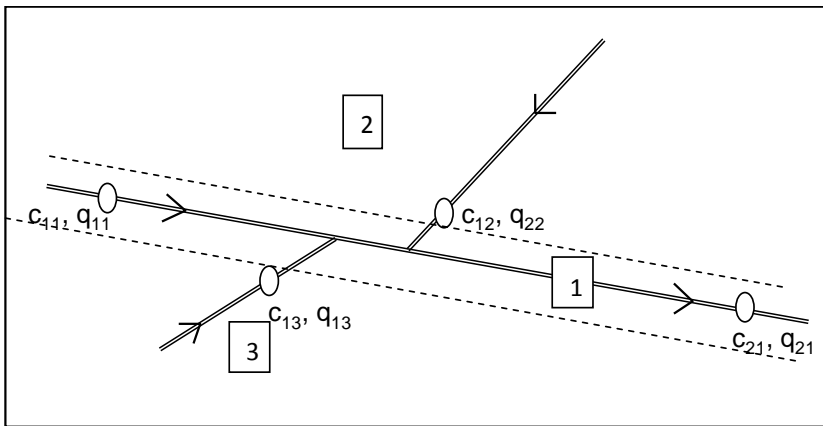
254 As mentioned above, quantifying the DS essentially consists in measuring the
255 streamflow capacity to decrease the nitrogen load. This benefit is considered as efficient only
256 when nitrogen concentration is not exceeding the maximum concentration levels allowed.
257 This capacity obviously depends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

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

264 By contrast, our proposal aims to obtain monetary and locally specific measures.
265 Consequently, we only consider the part of the DS the river currently performs using its
266 streamflow. Any streamflow reduction will mean a loss in this service and, therefore, a
267 monetary cost in terms of subsequent treatment costs. Thus, we need a reference point to value
268 the service currently offered in terms of the future treatment costs that are now 'hidden' by the

269 present streamflow. To sum up, the DS can always be valued on a monetary basis by means of
 270 the costs saved by the dilution capacity.

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



277

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

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

$$282 \quad Q_{ij} = c_{ij} q_{ij} \quad [1]$$

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

$$288 \quad Q_{ij}^{\text{nat}} = c^{\text{nat}} q_{ij} \quad [2]$$

289 We also need to establish the ‘acceptable’ or admitted concentrations (c_{max}), i.e. the
 290 levels of mass per volume of water that trigger the water treatment process in the basin. This
 291 threshold logically entails an arbitrary judgment, but it is essential if we are to value the
 292 service in monetary terms. Furthermore, as Joseffson (2018) points out, a specification of the

293 generic concept of ‘water damage’ is required to have an operative measure of this typing
 294 point for actual water management.

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

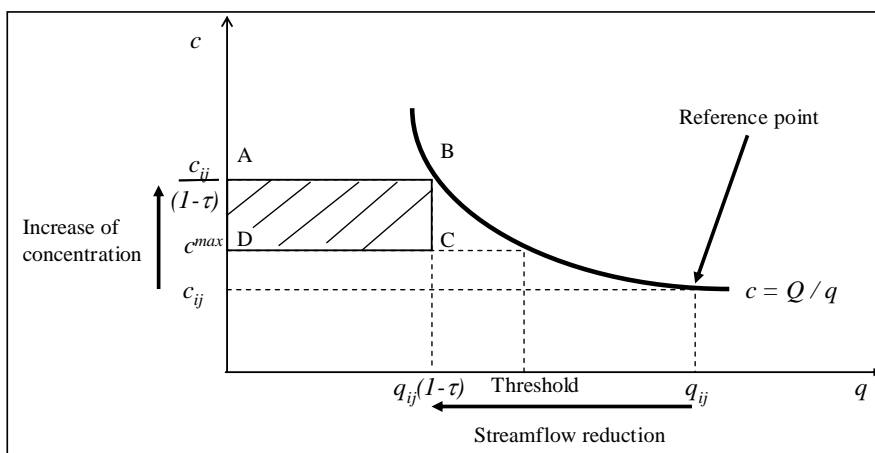
$$299 \quad GW_{ij} = \frac{Q_{ij} - Q^{nat}}{c^{max} - c^{nat}} \quad [3]$$

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

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

311 $c = \frac{Q}{q}$, given a total nitrogen mass Q, we can represent a curve which inversely relates c and

312 q, as shown in Figure 2.



313

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

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

317 The pollutant quantity in excess to remain at the c^{max} level, that is to say, the treatment
 318 needs resulting from the streamflow reduction at that rate τ , is drawn on Figure 2 by the area

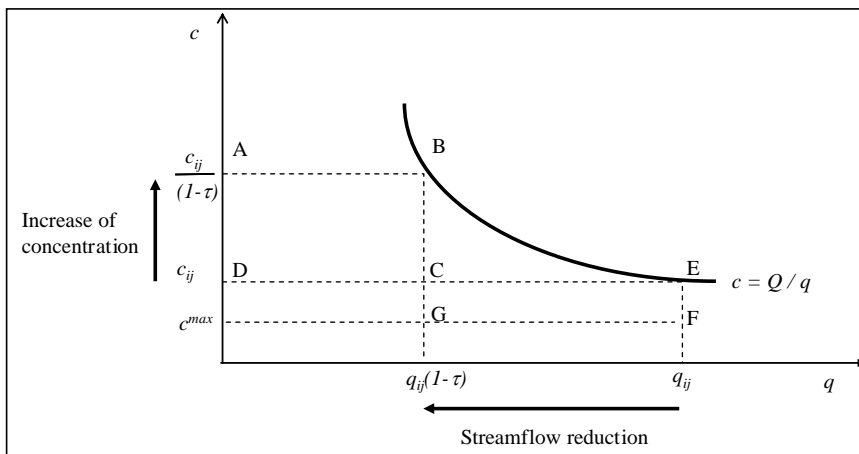
319 ABCD, that is, $\left(\frac{c_{ij}}{(1-\tau)} - c^{max} \right) q_{ij}(1-\tau)$. We have to artificially remove that amount of

320 nitrogen from water. We can calculate the total costs D_{ij} (euros) associated with the depuration
 321 of this pollutant amount:

$$322 \quad D_{ij} = d_{ij} \left(\frac{c_{ij}}{(1-\tau)} - c^{max} \right) q_{ij}(1-\tau); \quad [4]$$

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

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



326

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

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

$$333 \quad D_{ij} = d_{ij} c^{max} q_{ij} \tau \quad [5]$$

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

335 and the avoided costs will, logically, be zero.

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

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

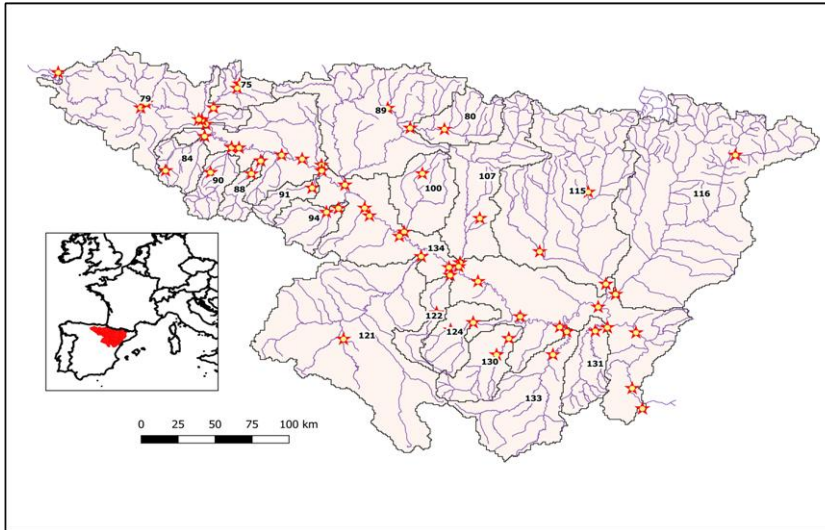
344 Finally, the DS value rises as the total mass of nitrogen (Q) increases. Graphically, that
345 means that the equal-mass curve moves away from the origin of the graph. Consequently, all
346 the areas under the curve, i.e. the nitrogen to be treated, are larger.

347

348 ***2.2. Hydrological and Treatment Cost Data***

349 *2.2.1. Georeferenced hydrological information*

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



356

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

358 The dataset needed for our study comes from three different sources: hydrological
 359 information on surface streamflows, nitrogen concentrations and natural nitrogen removal
 360 rates at the same points by modelling approach (CHE 2018, Cakir et al. 2020b) and, finally,
 361 treatment costs associated with nitrogen removal (own calculations based on Grossman,
 362 2012).

363 The streamflow and nitrogen concentrations were simulated by a calibrated and
 364 validated Soil and Water Assessment Tool (SWAT) model (Arnold et al., 1998) taking into
 365 account human pressures stemming from crop management (fertilization and irrigation), dam
 366 management, and municipalities' effluents (with a population equivalent capacity of over
 367 50,000) (Cakir et al., 2020b). SWAT model is able to simulate in-stream retention using a
 368 process-based model (including biological and physical processes) whereas other models,
 369 such as Nani model (Romero et al., 2016), will only simulate N retention in soil.

370 SWAT is a public hydro-agro-environmental model that is widely known (Fu et al.,
 371 2019; Gassman et al., 2007) and applied all around the world from small to large scale
 372 (Abbaspour et al., 2015). Based on the river network and the topography, the watershed is
 373 divided into sub-basins at each river junction between the tributaries and the main river (Fig.
 374 1). The model is based on hydrologic response units (HRU aggregated at sub-basin level). In
 375 each sub-basin, an HRU is characterized by a soil, land use and slope combination. The
 376 hydrology and nitrogen components are calculated at HRUs level and then computed at the
 377 sub-basin level. For the sake of clarity, we call each of these geographical basin subdivisions a
 378 hydrological subsystem (HS).

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

387 As a result of the SWAT model, information is available at each outlet of the Ebro
388 River's tributaries (Fig. 4). The following subsystem outputs in the Ebro watershed is
389 extracted using a monthly time step and computed annually from 1980 to 2010: (1)
390 streamflow in $\text{m}^3 \cdot \text{s}^{-1}$; (2) nitrogen concentration in $\text{kgN} \cdot \text{year}^{-1}$; and (3) nitrogen removal,
391 estimated by the in-out nitrogen load difference divided by the length of the reach in $\text{kgN} \cdot \text{m}^{-1}$
392 $\cdot \text{year}^{-1}$. Detailed and consistent data are available for this 20-year series.

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

399

400 2.2.2. *Climate scenarios*

401 The set of forecasts by Estrela et al. (2012) considers two different scenarios. For the
402 Ebro River basin, where the study area is located, the forecasts for a moderate climate-change
403 scenario indicate that available water will decrease by 11% and 14% over the time horizons
404 2041–2070 (medium term) and 2071–2100 (long term), respectively. Forecasts in a more
405 pessimistic scenario estimate a reduction in water resources of 14% and 28% in the medium
406 and long term, respectively. Vautard et al. (2014), with more updated information, consider a
407 likely scenario of a precipitation drop of around 20% in southeastern Europe. Thus, we can
408 consider this figure as a moderate guess for the lower bound of a likely general reduction in
409 water resources. We will take this central scenario as a reference in our calculations.

410 However, there are also more spatially precise and specific water availability forecasts
411 in our zone. We will use them as a sensitivity analysis of the results depending on water

412 resources reduction. For the recent period, the MESAN (the Mesoscale Analysis System) re-
 413 analyses of air temperature and precipitation at a daily scale (Landelius et al., 2016) were used
 414 as SWAT model inputs from 1980 to 2010. The choice of MESAN was based on the
 415 following criteria: (i) the validity of the reanalysis compared to other available meteorological
 416 datasets, deemed satisfactory for France (Raimonet et al., 2017) and Spain (unpublished data),
 417 (ii) a gridded dataset at a daily time step and covering Europe, as the aim of the methodology
 418 developed in this paper is to be applied at the European scale, and (iii) a meteorological
 419 dataset that has been used to correct the bias of EURO-CORDEX (0.11°; Jacob et al., 2020)
 420 climate model outputs for the 21st century. For the future period, this study used six climate
 421 projections from the EURO-CORDEX project (Table 1) obtained from different global and
 422 regional climate models and the RCP (Representative Concentration Pathway) 8.5 extreme
 423 scenario (+8.5 W/m² in 2100 compared to preindustrial values) developed by IPCC
 424 (Intergovernmental Panel on Climate Change). These projections have been corrected for bias
 425 based on MESAN re-analysis (see Raimonet et al., (2018) for more details).

426

427 **Table 1.** List of EURO-CORDEX climate model outputs used in this study.
 428 Details about GCM and RCM models can be found on the EURO-CORDEX website
 429 (<http://www.euro-cordex.net/>).
 430

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

- 431 (1) Representative Concentration Pathway
 432 (2) Institut Pierre-Simon Laplace
 433 (3) Centre national de recherches météorologiques
 434 (4) Max Planck Institute for Meteorology
 435 (5) The Irish Centre for High-End Computing
 436

437 2.2.3. Treatment plant costs

438 To assign treatment costs, we first need to consider the maximum concentration allowed
 439 at the plant outlet or the specific concentration that triggers the need for depuration. This limit

440 is set in Spain by a Royal Decree (11/1995) in response to the requirements by the Nitrate
 441 Directive 91/271/CEE in a range from 2.25 to 3.38 mg total N·L⁻¹ where TN (total nitrogen)
 442 includes all nitrogen forms.

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

448 Concerning costs, the Spanish National Statistical Institute (INE) reports a cost range
 449 from €0.76 to €0.79 per m³ for treated wastewater in general. According to Martínez and
 450 Albiac (2004), the specific cost of denitrification amounts to around €0.076 per m³. This
 451 means that removing 1 kg of nitrogen, assuming a stream with a concentration of 11 mg
 452 TN·L⁻¹ on entering the treatment plant, would result in an expense of 6.9€. However, the use
 453 of a single cost would limit the accuracy in estimating the value of the service when
 454 population and livestock activities determine treatment's unit cost.

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

465 **Table 2.** Treatment costs and plant size.

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

466 Source: Grossman (2012)

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

471

472 **3. Results and Discussion**

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

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

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

HS	Annual average nitrogen concentration ($\text{mgN}\cdot\text{L}^{-1}$)	Annual average water flow ($\text{Hm}^3\cdot\text{year}^{-1}$)
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

486 *3.1. Nitrogen Removal Service (RS)*

487 Table 4 includes the estimated mean nitrogen natural removal coefficients by subsystem
488 during the period studied, the length of the watercourses, the corresponding total annual
489 nitrogen removed and the final RS value. The removal rates modeled by SWAT had been
490 validated on the Garonne river nitrate removal by comparing with in situ measurement of
491 removal rates (See Cakir et al. 2020a). According to Billen et al. (2018) and Wolheim et al.
492 (2008), nitrogen removal in hydrosystem eliminates between 10 and 50% of the diffuse
493 sources, the same studies estimated the leaching rate of croplands areas ranging from 11 to 46
494 kgN/ha/yr. If we applied this rate to our study cas, we found an average total annual nitrogen
495 removed over the entire basin equal to 16,583 tons whereas our simulation estimated an
496 average of 18,449 tons. The simulations are in the range of nitrogen removal rate of previous
497 studies. Our estimates show that subsystems 115, 116 and 134 have the highest natural
498 removal capacity within the basin, with values >2 kg·m⁻¹. These subsystems correspond to
499 the sub-basins of the Cinca and Segre rivers and the main course of the Ebro River,
500 respectively, where there is more population pressure and where the majority of the
501 agricultural and livestock activities in the basin take place (mainly pig production).
502 Subsystems 79 and 89 also achieve nitrogen removal levels over 1.3 tons per year.

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

509
510 **Table 4.** *Natural nitrogen removal coefficients, length of the reach, total annual nitrogen*
511 *removed and RS value by subsystem.*

HS	Natural removal coefficients (kg·m⁻¹)	Length of watercourses (m)	Total annual nitrogen removed (kg)	Unit treatment cost (€·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

512

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

526

527 3.2. Dilution service (DS)

528 According to our method for calculating the DS, we must consider the increase in the
529 nitrogen concentration resulting from the water flow reduction as a reference for the current
530 economic value of this service. Therefore, for the stations where the initial concentration is
531 less than 5.6 mgN·L⁻¹, the DS value will be noticeable only when the water flow reduction

532 causes a concentration above 5.6 mg·N.L-1; in such a case, it will be necessary to incur
533 treatment costs. In contrast, when a station currently has a concentration above 5.6 mg·N.L-1,
534 we must calculate the costs that will be involved in returning to the initial nitrogen
535 concentration level.

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

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

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

565

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

HS	Nitrogen concentration (mg·L⁻¹)	Water flow (Hm³)	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

568

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

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

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

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

609 **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.

614 3.3. Total value of water services

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

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

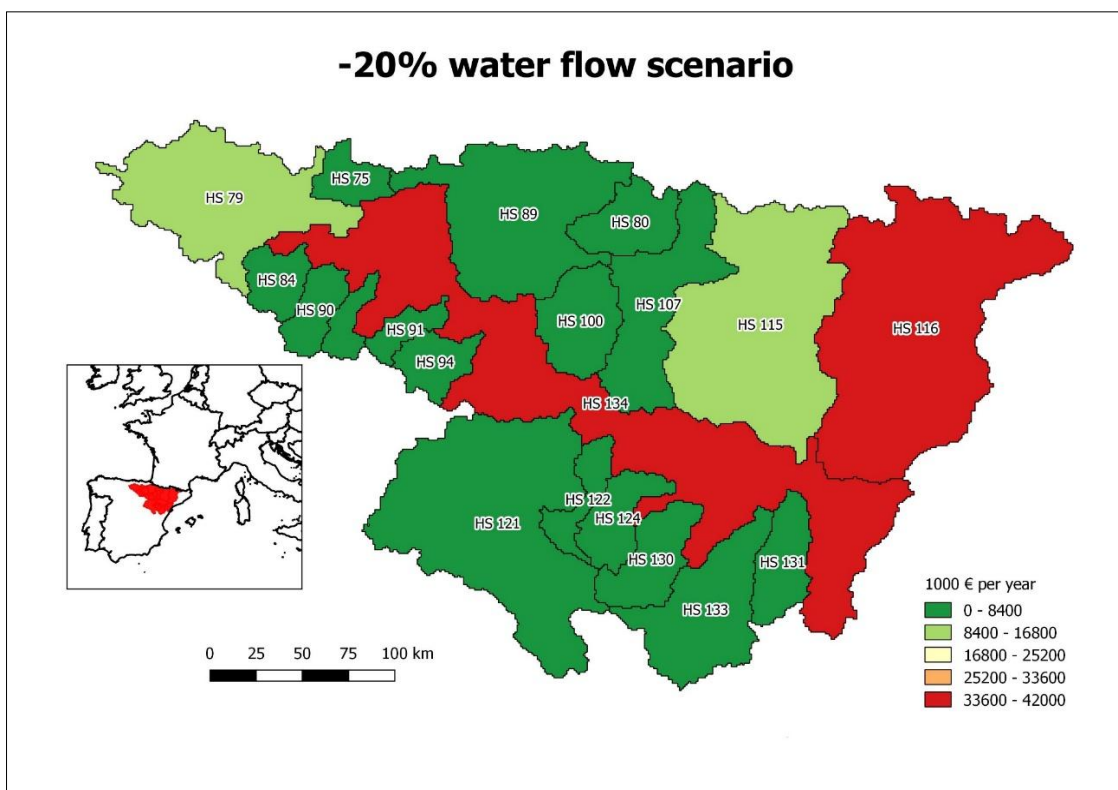
632 Of course, the figures calculated here should be viewed with caution due to the
633 inherent limitations of the approach we have taken. Given that the avoided cost
634 considered corresponds only to the excess of nitrogen diluted by the water that remains
635 as a result of the different climate-change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

638 An overall assessment of the results obtained in this study allows us to affirm that
639 circulating water has an important economic value linked to the biological activity of
640 the fluvial ecosystem and as a nitrate diluent that must be considered in the decision-
641 making process of institutions involved in water management. This is especially
642 relevant in those projects that involve water withdrawal, such as the increase in irrigated
643 surface area, reforestation plans or even transfers to other basins based on the argument
644 that some subsystems in the basin have abundant water flows. Our calculations show
645 that the water flow and nitrate transformation by biogeochemistry in these subsystems
646 avoid high treatment costs. This must be economically valued for a proper consideration

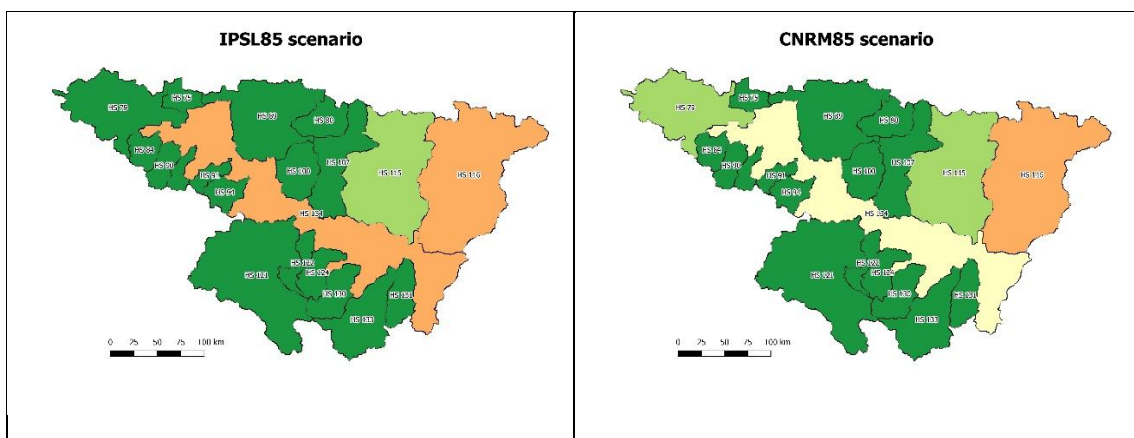
647 of all river management that may favor or impair these services. However, the economic
648 values linked to nitrogen-related environmental services are only a part of the total
649 value of these services. The removal of other compounds (sediment, pesticides, heavy
650 metal...), which participate in water regulation services, are not considered in the
651 present study.

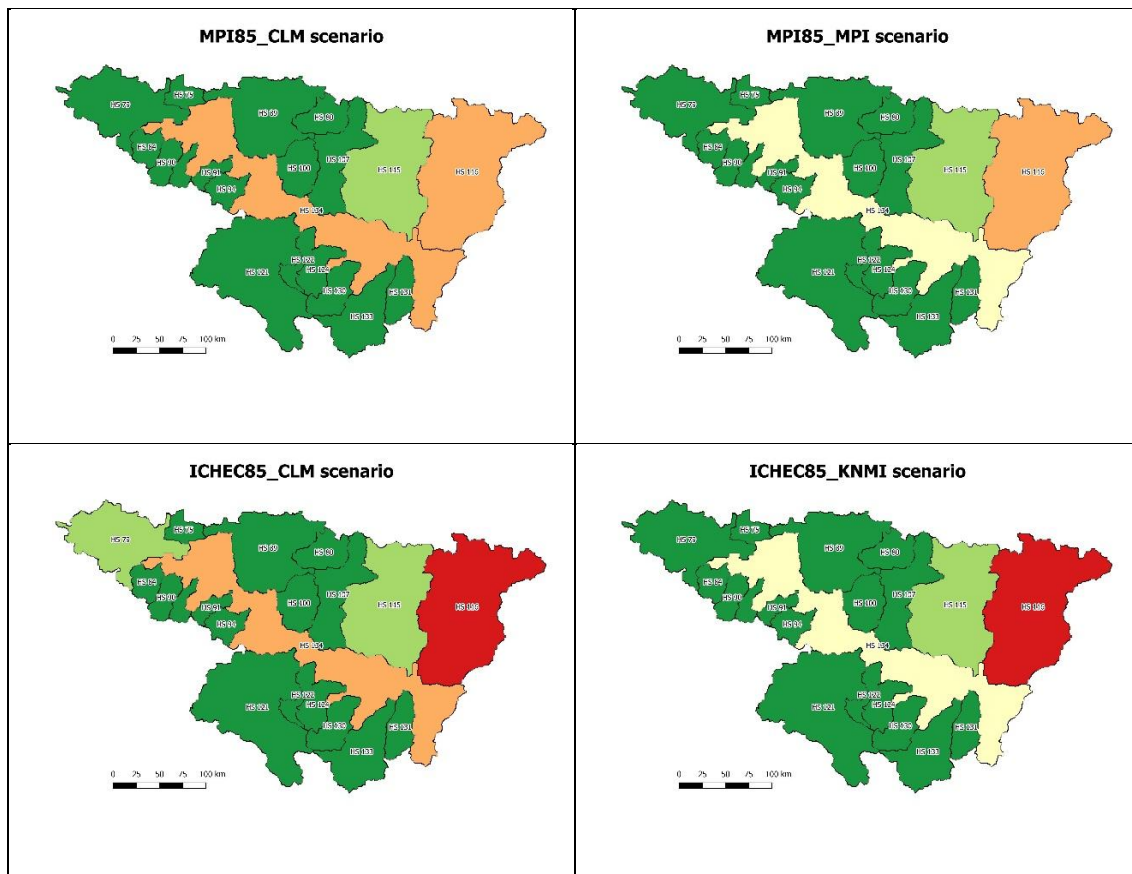
652 Despite the limitations of the proposed approach, the suggested method provides a
653 consistent and easily replicable tool for economically valuing nitrogen-related water
654 services, which is one of the main causes of concern related to non-point pollution in
655 rivers worldwide and, especially,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

656



657





658 **Figure 5.** Total value of the service (RS+DS) under different climate change scenarios
 659 (10^3 € per year) by subsystem.

660 **4. Conclusions**

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

668 The third novelty of this paper is to test the influence of climate changes on the
 669 economic value that is only possible with the help of a modelling approach that includes
 670 physical and biological processes. Indeed, climate change is more influencing water
 671 discharges than biological processes, and only this coupled approach (dilution plus
 672 biological removal) allows the demonstration of climate change influence on this

673 natural service of water quality regulation. This is the first try to our knowledge to show
674 the influence of climate changes on the economic values of this service.

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

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

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

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

718

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724 75927-R

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