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A dynamic hydro-economic model to assess the effectiveness and economic benefits and costs of wetland restoration and creation

Noureddine Bouzidi [a,b,*], C. Dionisio Pérez-Blanco [b,c]

[a] *Department of Economics and Economic History, Universidad de Salamanca, C/ Francisco Tomás y Valiente s/n, 37007 Salamanca, Spain*
[b] *IMDEA Water Institute, Av. Punto Com, 2, Alcala de Henares, Madrid 28805, Spain*
[c] *Euro-Mediterranean Center on Climate Change, Via della Libertà, 12, 30121 Venezia VE, Italy*

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ABSTRACT

This paper studies the environmental and socioeconomic performance and sustainability of wetland creation/restoration in agricultural watersheds under nonstationary climatic conditions. To this end, we develop a dynamic hydroeconomic modeling framework that integrates the Soil and Water Assessment Tool Plus (SWAT+), Positive Mathematical Programming (PMP), and economic valuation via the benefit transfer method to calculate the effectiveness (wetland area), benefits (economic value of wetland expansion) and costs (foregone agricultural profit) of wetland creation/restoration. Methods are illustrated with an application to the Flumen Watershed wetland restoration and construction project in NE Spain. Results highlight significant tradeoffs between environmental (wetland) and economic (agriculture) water uses, which aggravate over time and are particularly relevant under more severe climate change scenarios. Despite the growing costs and decreasing benefits of wetland creation/restoration due to reduced water availability and wetland surface under climate change, our results show that total wetland benefits over the series offset total wetland costs under all simulations and scenarios. Only during the last years of the series and for the most pessimistic climate change scenarios and combinations of models, costs start exceeding benefits. These results suggest a positive and robust performance of the wetland restoration and creation project in the Flumen Watershed.

1. Introduction

The threat of growing water scarcity, driven by increasing irrigation water demand and aggravated by climate change-induced decreasing supplies (WWF, 2022), demands immediate and multifaceted solutions that address environmental degradation and related socioeconomic challenges (IPBES, 2019). While conventional grey engineering and technologies will play an important role in addressing water scarcity, a distinct and promising paradigm shift is underway building on nature-based solutions (NBS) (Albert et al., 2021; Souliotis and Voulvoulis, 2022). In the context of water scarcity, NBS have been defined as “multi-functional measures that aim to protect water resources and address water-related challenges by restoring or maintaining ecosystems as well as natural features and characteristics of water bodies using natural means and processes” (European Commission, 2014). A key challenge in the design of NBS under growing water scarcity and competition for the resource relates to the consideration of potential socioeconomic-environmental trade-offs in water use and how these may affect NBS sustainability over time (IPBES, 2019), including under uncertainty (Van Zanten et al., 2023).

A prime example of socioeconomic-environmental trade-offs in NBS is the tension between irrigated agriculture and wetlands in agricultural watersheds (Pérez-Blanco and Sapino, 2022). While increasing irrigation water allocation is necessary for food production (Grafton et al., 2018; Perry, 2011), it often comes at the cost of reduced water availability for environmental uses and wetland area loss (Hambäck et al., 2023), which in turns reduces biodiversity and the valuable services these ecosystems offer (Stroud, 2022). Conversely, prioritizing water allocation to wetlands may constrain agricultural water supply, posing potential threats to food production and the economy (Acreman et al., 2011). Understanding this trade-off is critical for ensuring the sustainability of food production and the long-term health of wetlands and the ecosystem services they deliver (Hambäck et al., 2023; Pérez-Blanco and Sapino, 2022). From a biophysical perspective, successfully managing the tradeoff calls for an in-depth understanding of the hydrological connectivity within the watershed, including factors like

* Corresponding author.
E-mail address: nori_bouzidi@usal.es (N. Bouzidi).

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evapotranspiration rates, water infiltration, and groundwater recharge in different land-use scenarios, as well as the contribution of these water flows to the provision of food and other ecosystem services (Barbier, 2011; Matos and Roebeling, 2022). From a socioeconomic perspective, managing and understanding the trade-off necessitates information on the economic value of agricultural production against the economic value derived from the ecosystem services provided by wetlands (including *inter alia* water purification, flood control, recreation, and the long-term economic consequences of wetland degradation), as well as mapping and understanding the implications for the stakeholders involved, including farmers, environmentalists, local communities, and policymakers (Pérez-Blanco and Sapino, 2022; Van Zanten et al., 2023). Moreover, this trade-off analysis and management needs to be implemented in a context of socioeconomic and environmental change, which drives growing uncertainty and calls for the design of robust governance frameworks with the capacity to enhance welfare under most plausible futures (Cohen-Shacham et al., 2016).

Despite the significant and highly integrated environmental and socioeconomic aspects that characterize NBS such as wetlands and their management, their study is typically focused either on understanding biophysical or socioeconomic aspects, or a part thereof, and rarely account for uncertainties such as climate change—which are critical in the sustainable management of environmental assets over the long run. The (limited) studies integrating socioeconomic and hydrological aspects into hydroeconomic models of wetland creation/restoration typically address the costs for economic water users (often farmers), but exclude an assessment of the wider benefits of wetlands (González-López et al., 2023a,b; Asbjornsen et al., 2015; Van Zanten et al., 2023). Moreover, assessments of wetland restoration/creation policies, particularly those using socioeconomic modeling, are often stationary (i.e., the statistical properties of the data series such as mean and variance do not change over time) (González-López et al., 2023a,b) and rely on point predictions that provide a single estimate without reflecting the variability and uncertainty present in models and real-world conditions, which can lead to overconfidence in model outcomes and underestimation of the uncertainties associated with water management decisions (Reichert and Mieleitner, 2009).

This study aims to address gaps in socioecological integration and uncertainty quantification in NBS performance assessments by evaluating biophysical aspects of wetland restoration/creation and the related socioeconomic costs and benefits for main economic water users (irrigators) and the wider society, as well as the relevant socioeconomic-environment tradeoffs, under nonstationary climate change conditions and accounting for key uncertainties in modeling. Biophysical aspects are modeled relying on the Soil and Water Assessment Tool Plus (SWAT+) model (Bieger et al., 2017), which is coupled with a microeconomic Positive Mathematical Programming (PMP) model of irrigators' behavior to assess economic users' responses to alternative water availability scenarios and thus quantify the costs of wetland restoration/creation (de Frahan et al., 2007a). The benefits from wetland restoration/creation are obtained by multiplying the change in the surface of wetlands modeled in SWAT+ under alternative management and climatic scenarios by their per hectare benefits as obtained via a benefit transfer approach (Smith, 2018). The modeling approach works under a dynamic setup and incorporates uncertainty quantification via scenario-based approaches (climate change scenarios), sensitivity analysis (SWAT+) and multi-model ensemble techniques (benefit transfer). Methods are illustrated with an application to the Flumen watershed in NE Spain, an agricultural watershed that witnessed significant wetland creation and restoration activities over 2011–2014, and whose socio-ecological performance and tradeoffs under climate change have not been assessed yet.

By integrating costs and benefits with hydrological aspects, incorporating temporal and spatial dynamics, and quantifying key sources of uncertainty, our approach can provide a more comprehensive assessment of the socioeconomic and environmental performance of wetland creation/restoration policies within agricultural watersheds and under conditions of uncertainty—an information that is instrumental towards identifying robust policies that achieve environmental and socioeconomic sustainability under most plausible futures. In the context of this research, sustainability is defined as the capacity of wetland creation/restoration programs to maintain their environmental and socioeconomic benefits over time and under conditions of uncertainty, and is assessed by evaluating whether the total economic benefits of wetland restoration outweigh the associated costs (foregone agricultural profits) while ensuring wetland sustainability under alternative nonstationary climatic scenarios (RCP 2.6, 4.5, and 8.5).

2. Case study: flumen watershed

The Flumen River watershed, covering an area of 1400 km^2, is located within the Ebro River Basin in northeastern Spain. Its only permanent tributary is the Isuela River. The northern region features medium mountains composed of conglomerates and limestone, while the central and southern areas consist of plains predominantly used for intensive irrigated agriculture (Sorando et al., 2019). The region also includes low mountain ranges that support forests and pastures (Macary et al., 2011). Most of the watershed is characterized by gently sloping terrain, with gradients less than 5 % (Comín et al., 2014). Elevations range from 230 m above sea level in the southernmost part to 1694 m in the northern mountainous area. The climate shifts from semi-humid in the north to semi-arid in the central and southern sections. Although the average annual rainfall is 413 mm, there is considerable interannual variation of 32.7 %, with frequent heavy summer storms (Sorando et al., 2019). The average annual temperature is 14.5 °C, peaking at 26 °C in July and dropping to 5 °C in January, with high evapotranspiration rates of 800–1000 mm per year (Comín et al., 2014; Darwiche-Criado et al., 2015).

Water from the Cinca and Gállego Rivers is channeled via two canals running parallel to the Flumen River in the northeast and northwest of the watershed. These canals, originating in the Pyrenees, provide irrigation to the central and southern regions through a network of cemented or piped channels. This area is part of Europe's largest irrigation system, *Riegos del Alto Aragón*, which also features channels to manage excess water and agricultural runoff (Sánchez-Chóliz and Sarasa, 2013). The higher altitude areas of the watershed, situated above the canal-fed zones, are used for dry farming, mainly barley and wheat, and contain natural and reforested pine woods and pastures (Darwiche-Criado et al., 2017).

Throughout its course, the flow of the Flumen River is monitored at several gauging stations, revealing the predominant land uses within each sector of the watershed. Near Quicena, the water flow demonstrates minimal human impact, as it originates from densely forested regions. As the river approaches Barbues and merges with the Isuela River, it traverses Huesca, a major urban center with a population of 53,000. By the time it reaches Albalatillo, the river's flow increases significantly, showing substantial seasonal variations influenced heavily by agricultural activities (Sorando et al., 2019). Agricultural irrigation demands a significant volume of water, totaling 800 Hm^3 annually, supported by infrastructure including three dams: Montearagón (43.18 hm^3), Santa María De Belsué (13 hm^3), and Cienfuens (1 hm^3) (Chebro, 2016). Additionally, two canals divert water from the Cinca and Gállego Rivers, flowing northeast and northwest of the Flumen Watershed and running parallel to the Flumen River through neighboring watersheds (Chebro, 2016). Return flows from agricultural water uses total approximately 700 tons N $year^{-1}$ and are producing nontrivial qualitative impacts on surface water bodies, which have been labeled as "poor" in the basin plan (Chebro, 2016). Nitrate infiltration rates of 100–250 $kgNha^{-1}year^{-1}$ in irrigated sub-watersheds and lateral flow rates of up to 1400–2000 $kgNha^{-1}year^{-1}$ have been observed in non-irrigated watersheds (Sorando et al., 2019).

The LIFE CREAMAgua project, conducted from 2011 to 2014, aimed to address water quality concerns in rivers affected by agricultural runoff including the Flumen Watershed (Comín et al., 2014; Darwiche-Criado et al., 2017; Sánchez-Chóliz and Sarasa, 2013). By implementing ecological practices, the project sought to demonstrate the potential for improving water quality and increasing biodiversity through wetland restoration and creation and the rehabilitation of riverbanks on a basin-wide scale (Darwiche-Criado et al., 2017). It also intended to highlight the role of local governance via NBS in promoting environmental assets in sparsely populated rural areas.

The project focused on restoring and constructing 16 wetlands, covering a total of 78 ha of permanently flooded land, 60 ha of temporarily flooded land, and 400 ha of buffer lands set aside for conservation. These wetlands were strategically placed throughout the basin, including 11 in-stream and 5 off-stream sites, as well as 19 riverbank initiatives along the Flumen River, spanning 70 ha (Comín et al., 2014; Darwiche-Criado et al., 2015). Restoration efforts began in 2011 and concluded by April 2013, with expenses ranging from €2,500 to €4,500 per hectare for wetlands and €1,100 to €2,100 per hectare for riverbanks (EUROPEAN COMMISSION, 2014). Monitoring outcomes indicated a substantial enhancement in water quality, with a reduction of nitrate levels of 80–95 % in 45 % of the wetlands and 10–50 % in another 45 %, and 10 % of the wetlands showing no reduction. In-stream wetlands were particularly effective in improving water quality as they retained more pollutants (Darwiche-Criado et al., 2015). Notably, the construction and restoration of wetlands within an agricultural watershed altered the watershed's hydrodynamics and constrained agricultural water supply, leading to a trade-off with irrigation that emerges during drought events, and will be aggravated into the future due to climate change—thus creating an added cost for wetland construction and restoration projects (Stroud, 2022).

The Flumen River project lacks a comprehensive socioecological performance assessment that quantifies its environmental and economic impacts and feasibility, the relevant tradeoffs, and the returns it offers to society over time—a gap we address in this paper. To this end we analyze the impacts of alternative climate change and management scenarios on the provision (effectiveness) and value of ecosystem services (benefits), as well as on agricultural practices and economic outcomes over the irrigated areas that rely on water from the Flumen Watershed (costs). Irrigated areas in the Flumen Watershed include 2,488.12 ha with registered water uses (Chebro, 2023) and a total water demand of 9,251,054 cubic meters, distributed over 34 municipalities (the agents in the PMP model) (Fig. 1) (Chebro, 2023; Gobierno de Aragón, 2023). Relevant crops include Alfalfa (21.51 %), Almond tree (2.34 %), Rice (0.90 %), Oats (1.09 %), Barley (42.76 %), Onion (0.41 %), Sunflower (0.75 %), Dried pea (3.72 %), Corn (14.97 %), Apple tree (0.13 %), Olive grove (0.22 %), Wheat (9.52 %), and Vetch (1.68 %), with a gross margin ranging from 351.48 EUR/ha/year (Wheat) to 10,000.55 EUR/ha/year (Onion) and averaging 1,283 EUR/ha/year.

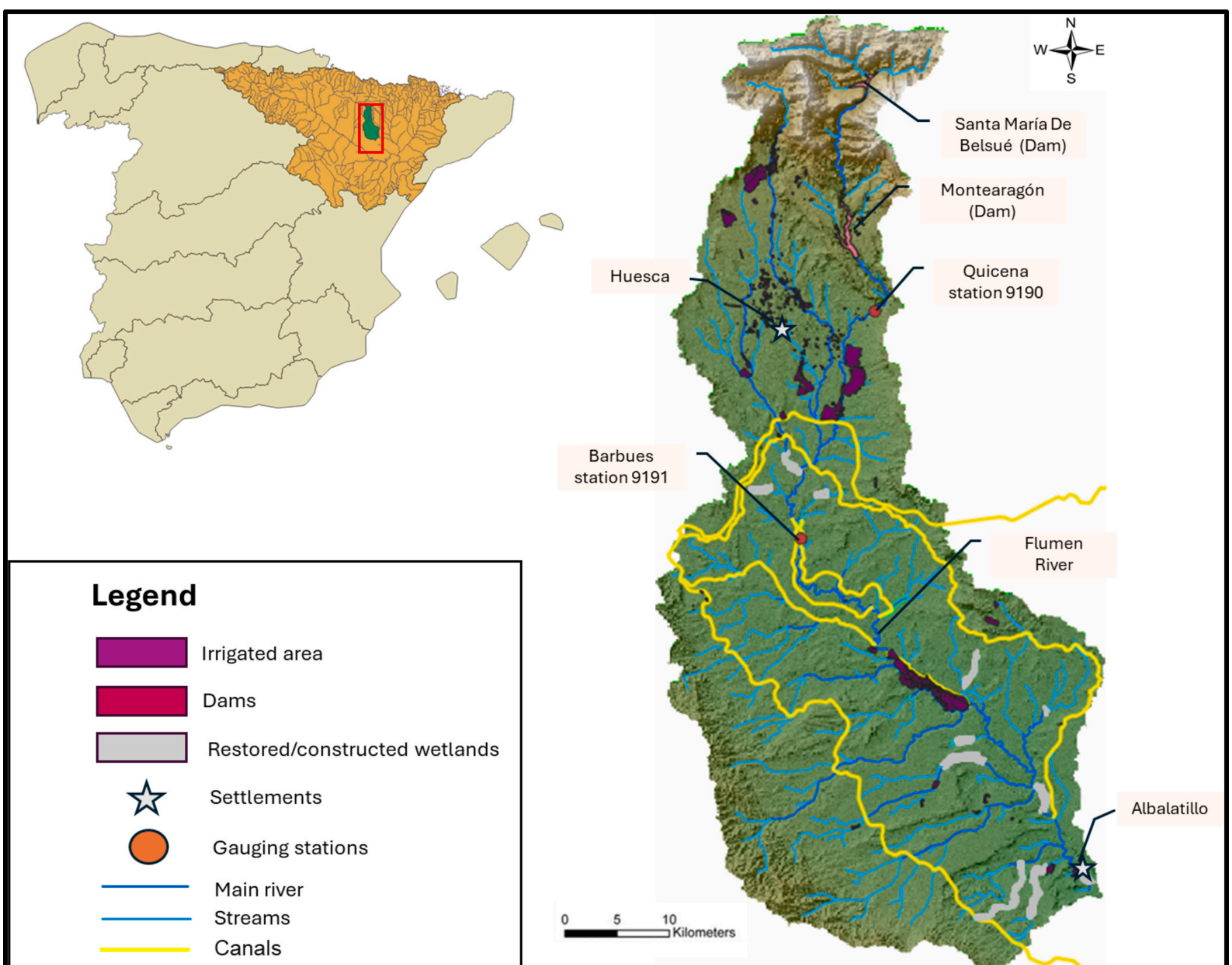

Fig. 1. Flumen watershed, registered irrigated area and the location of restored and constructed wetlands (CEDEX, 2023).

3. Methods and data

The present study presents a dynamic and integrated hydroeconomic modeling framework (Fig. 2) that evaluates the biophysical aspects of wetland restoration/creation using SWAT+ (Section 3.1), the related socioeconomic costs for irrigators using a PMP model (Section 3.2), and the wider socioeconomic benefits provided by wetlands using a benefit transfer approach (Section 3.3), while accounting for key sources uncertainty. The coupling between the water (SWAT+) and human (PMP and benefit transfer) system models is implemented through two protocols: one bidirectional protocol that interconnects the PMP and SWAT+ models, and one sequential protocol that works from the benefit transfer model to the SWAT+ model (Section 3.4). The resultant modeling framework is used to assess the environmental and socioeconomic performance of wetland restoration/creation under alternative management and climatic scenarios (described in Section 3.5). Choice of SWAT+, PMP and benefit transfer methods responds to both the intrinsic advantages of each model and their widespread use, flexibility and generalizability. The intrinsic advantages of each model include the more realistic representation of irrigators' behavior provided by positive PMP models as compared to normative models, the ability of SWAT+ to assess the impact of nonstationary climatic conditions on water availability and wetland surface areas at the watershed scale, and the capacity of benefit transfer methods to account for uncertainty. Moreover, SWAT+, PMP and benefit transfer are arguably one of the most (if not the most) widely used approaches for hydrologic modeling, water user modeling, and economic valuation, respectively, which supports relevance and the potential adoption of the proposed tool by a large community of modelers. Finally, it should be noted that the coupling framework is flexible and generalizable, and can be applied to alternative hydrologic (e.g., HEC-HMS, MODFLOW), socioeconomic (e.g., Linear Programming, Positive Multi-Attribute Utility Programming, or alternative PMP models) and economic valuation (e.g., ad-hoc valuation methods such as contingent valuation) methods. This is further discussed in Section 5.

3.1. *Eco-hydrological model (SWAT+)*

The SWAT+ model is an advanced watershed simulation tool used for predicting the impact of land management practices on water, sediment, and agricultural chemical yields in large complex watersheds with varying soils, land use, and management conditions over long periods (Janjić and Tadić, 2023). SWAT+ is a semi-distributed model that considers the basin to be a mosaic of smaller spatially defined units, or sub-basins, which can be further divided into smaller Hydrological Response Units (HRUs). HRUs are defined as "lumped areas within a sub-basin that are comprised of unique land cover, soil, slope and management combination, which, together, comprise the main data inputs to the SWAT model" (Neitsch et al., 2011). This section presents the data inputs used in SWAT+ (Section 3.1.1), the sensitivity analysis conducted with the SWAT+ Toolbox to inform robust model calibration (Section 3.1.2), and the model calibration and validation (Section 3.1.3).

3.1.1. *Data*

To implement the SWAT+ (Soil and Water Assessment Tool Plus) model for the Flumen Watershed, comprehensive climatic and hydrological data were collected to ensure accurate and reliable simulation results. The climatic data necessary for the SWAT+ model, including precipitation and temperature records, were obtained from the Spain Weather Datasets for SWAT (SWAT Global Data, 2023). This dataset spans from 1951 to 2019, providing a robust historical climate profile essential for long-term hydrological modeling. The soil map for the Flumen watershed was created using data from the FAO (Food and Agriculture Organization) database (FAO, 2023). This database provides detailed soil properties, including soil texture, depth, and chemical composition, which are critical for modeling soil–water interactions, infiltration rates, and nutrient cycling within the SWAT+ framework. The slope was obtained from the Digital Elevation model provided by (USGS, 2023). The agricultural land and water use are generated in the microeconomic PMP model and integrated into the SWAT+ model leveraging the PMP-SWAT+ bidirectional protocol (see Section 3.4), while the location of wetlands was determined leveraging the comprehensive study conducted by the LIFE CREAMAgua project (CREAMAgua, 2011). Finally, other land use data was obtained from (ICEARAGON, 2023). The complete database used to populate the hydroeconomic modeling framework, including the SWAT+ model, is available in Annex 1.

3.1.2. *Sensitivity analysis*

The sensitivity analysis was conducted using the SWAT+ Toolbox

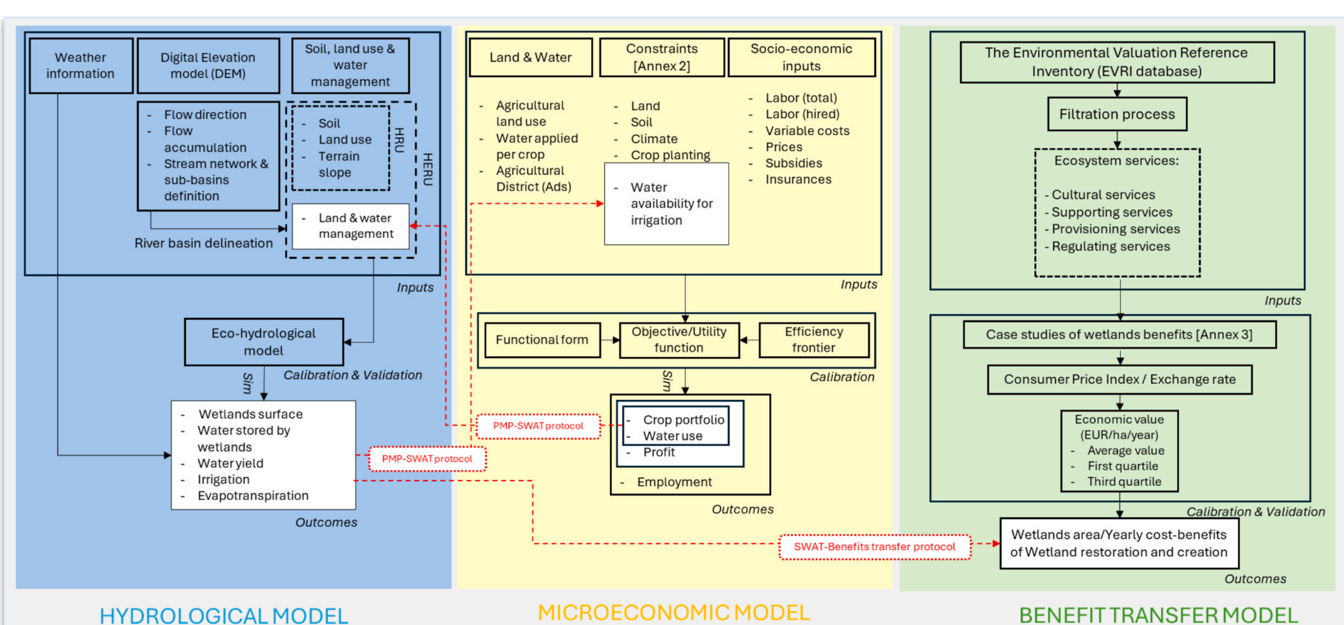

Fig. 2. Conceptual design of the hydroeconomic modeling framework.

and employing the Sobol index method (Nossent et al., 2011). This step was crucial for identifying the most influential parameters affecting the model outputs, which in turn are fine-tuned during the model calibration in Section 3.1.3. The Sobol index method is a global sensitivity analysis technique that quantifies the contribution of each input parameter to the output variability (Zhang et al., 2013). By systematically varying the parameters within their respective ranges and observing the resulting changes in model performance, we were able to determine which parameters had the most significant impact on the simulation results. The Sobol index method involves the following steps:

- *Parameter Sampling*: Generate many parameters sets using a sampling strategy that covers the entire parameter space.
- *Model Simulation*: Run the SWAT+ model for each parameter set to produce output data.
- *Variance Decomposition*: Analyze the output data to decompose the total variance into contributions from individual parameters and their interactions.
- *Index Calculation*: Calculate the Sobol indices, which indicate the sensitivity of the model output to each parameter.

The sensitive parameters identified through the sensitivity analysis include Cn3_swf, Cn2, epco, esco, k, awc, slope, Lat_ttime, Lat_len, chw, d50, chn, and Chd.

3.1.3. Calibration

The calibration of the SWAT+ model was carried out using the Dynamically Dimensioned Search (DDS) algorithm within the SWAT+ Toolbox. DDS is a robust optimization technique designed to efficiently search the parameter space and find the optimal set of parameters that best match the observed data. The iterative nature of DDS allows for dynamic adjustment of search dimensions, leading to improved convergence and calibration accuracy (Tolson and Shoemaker, 2007). The objective function in this context is typically defined as the discrepancy between the observed and simulated river flows, quantified using metrics Nash-Sutcliffe Efficiency (NSE) (Tolson and Shoemaker, 2007). During this step, the parameters identified as sensitive were fine-tuned to ensure that the model accurately replicated the observed hydrological processes of the Flumen watershed. An eight-year period, from 1993 to 2000, was used for the calibration of daily river flow in the Flumen watershed. Two observation points were utilized to compare the simulated river flow against observed data that were collected from stations 9190 and 9191 located in the municipalities of Quicena and Barbués (Fig. 1). This period provided a sufficiently long timeframe to capture a wide range of hydrological conditions, including variations in precipitation, evapotranspiration, and other climatic factors. The calibration process involved adjusting the model parameters to minimize the discrepancies between the simulated and observed river flows. Following the calibration, the model was validated using a six-year period from 2002 to 2007, again utilizing the same two observation points for daily river flow. The validation process involved running the model with the calibrated parameters and comparing the simulated outputs with the observed data for the validation period. This step allowed us to assess the model's performance and ensure that it could reliably predict river flow under different conditions. The calibration and validation outcomes, which are presented in Table 1, demonstrated the model's robustness and its ability to accurately simulate the hydrological dynamics of the Flumen watershed. The NSE values, with most above the acceptable threshold of 0.5, indicate good predictive power, particularly for Station 9190 (NSE = 0.611 during calibration and 0.480 during validation). RMSE values decrease during validation, showing improved accuracy, while PBIAS values remain within the acceptable ± 25 % range, indicating no significant over- or underestimation (D N Moriasi et al., 2007).

Table 1
Output of the calibration and validation of SWAT+ for Flumen watershed with daily discharge data.

Calibration of daily discharge (1993–2000)				
Stations	**Functions**			
	NSE[a]	**MSE**[b]	**RMSE**[c]	**PBias**[d]
9190	0.611	1.311	1.145	−10.168
9191	0.467	5.379	2.319	9,979
Validation of daily discharge (2002–2007)				
Stations	**Functions**			
	NSE	**MSE**	**RMSE**	**PBias**
9190	0.480	1.005	1.002	−20.454
9191	0.484	3.241	1.800	13.407

[a] The Nash-Sutcliffe efficiency (NSE) is a metric to assess the performance of a model relative to observed data. Developed by Nash and Sutcliffe in 1970, NSE is a normalized statistic that compares the residual variance of the model to the variance of the measured data.

[b] Mean Squared Error (MSE) is a statistical metric that quantifies the average squared difference between observed and predicted values. It indicates model accuracy, with lower values reflecting better performance.

[c] Root Mean Squared Error (RMSE) is a metric that measures the square root of the average squared differences between observed and predicted values. It provides an indication of model prediction accuracy, expressed in the same units as the variable being modeled. Lower RMSE values signify better model performance.

[d] Percent Bias (PBIAS) measures the average percentage difference between observed and simulated values, indicating whether the model tends to overestimate (negative PBIAS) or underestimate (positive PBIAS) the observations.

3.2. Costs model (PMP)

PMP is a technique used in agricultural economics to calibrate an objective/utility function (see Equation (3) that can perfectly replicate observed data (i.e., the calibration error is zero) (Mérel and Howitt, 2014). Different PMP models have been developed over time (Pérez-Blanco and Sapino, 2022), including the original Howitt (1995) model adopted in this paper. In PMP models the decision variable is the crop portfolio x, which is defined as a combination of water and land use and management practices (each of which can be treated as an individual crop) that conditions the expected total gross margin – the relevant attribute in the model that drives utility. The model constraints ensure that land use and resource allocations (water, labor, others) are within feasible limits, accounting for water availability, land constraints, and other technical coefficients such as crop rotation coefficients (see Eq. (4) for a mathematical statement of the domain, and Eq. (6) for the water availability constraint). A detailed description of the model constraints that conform the domain is available in Annex ll.

3.2.1. Data

To implement the PMP Howitt model, we collected socio-economic data of the Flumen watershed from alternative sources. Land use data was collected using open data on cultivated areas from The General Technical Secretariat of the Department of Agriculture, Livestock and Food, of the Government of Aragon (Gobierno de Aragón, 2023). The land use included 26 crops between irrigated and rainfed crops across 34 municipalities, which are the agents in the PMP model (i.e., 34 representative agents that grouped irrigators by municipality were identified and calibrated). key inputs for each crop, including costs, prices, yields and labor were gathered from the database on Farm Cost and Income Studies (ECREA, 2023), while the amount of water required per crop for irrigation was determined though the Agroclimatic Information System for Irrigation (SIAR, 2023) A summary of the data used to populate the PMP model is presented in Table 2, while the complete database is available in Annex I.

Table 2
Economic inputs of the PMP model of Flumen watershed.

Inputs	Description	Historical data	Source
Cost	The production cost for each crop	15 years (2006–2020)	Farm Cost and Income Studies (ECREA)
Surface	The area of land allocated to each crop	1 year (2020)	Common Agricultural Policy – PAC
Prices	Market prices for each crop	15 years (2006–2020)	Farm Cost and Income Studies (ECREA)
Yield	The output per unit area for each crop	15 years (2006–2020)	Farm Cost and Income Studies (ECREA)
Water Requirement	The amount of water needed for irrigation for each crop.	1 year (2020)	Agroclimatic Information System for Irrigation (SIAR)
Labor	The labor requirements (both total and hired) for each crop.	15 years (2006–2020)	Farm Cost and Income Studies (ECREA)
Subsidies	The coupled subsidies for each crop.	15 years (2006–2020)	Farm Cost and Income Studies (ECREA)

3.2.2. Calibration

The PMP calibration method involves a three-stage process designed to replicate observed base-year data (de Frahan et al., 2007b; Howitt, 1995; Mérel and Howitt, 2014). In the first stage, we formulate and solve a linear programming (LP) model. The primary goal of this stage is to maximize the profits associated with agricultural production, while adhering to various resource constraints such as land, labor, and water availability. The utility function is expressed as:

$$U = \max \sum_{c} \pi_c x_c - \sum_{i} \lambda_i b_i \tag{1}$$

$$\text{s.t.: } Ax \leq b \tag{2}$$

where πc represents the profit per unit of activity c, xc is the level of activity c, λi denotes the shadow price of resource i, bi is the availability of resource i, A is the matrix of technical coefficients, and b is the vector of resource availabilities.

In the second stage of the Positive Mathematical Programming (PMP) model, the cost parameters are estimated using the dual values (shadow prices) obtained from the Linear Programming (LP) model above. These dual values reflect the marginal value of each constraint, representing the opportunity cost of resources and the implicit costs associated with the agricultural activities. To capture the non-linear behavior of costs with respect to changes in activity levels, a quadratic cost function is adopted. This quadratic form allows for a more accurate representation of the increasing marginal costs that typically occur as production expands. The general form of the quadratic cost function is:

$$C(x) = \frac{1}{2} x^T Q_x + C^T x \tag{3}$$

where Q is a symmetric positive definite matrix representing cost parameters, C is a vector of linear cost coefficients, x is the vector of activity levels, and T represents data availability for the subset of crops being modeled.

The third and last stage involves integrating the quadratic cost function in Eq. (3) into the model in Eq (4), thereby transforming the utility function into a nonlinear programming (NLP) problem. This step allows the model to perfectly reflect the observed behavior of irrigators by accounting for nonlinearities in cost structures and resource allocation decisions. The objective function for the NLP model is designed to maximize the total expected gross margin obtained as the difference between the total revenue generated by crop activities and the total costs incurred, as follows:

$$U = \max \sum_{c} \pi_c x_c - \frac{1}{2} x^T Q_x - c^T x \tag{4}$$

$$\text{s.t.: } Ax \leq b \tag{5}$$

Using the PMP utility function in Eq. (6) we can explore the crop portfolio responses (and the related water use and gross margin outcomes) of irrigators to alternative shocks, including water availability shocks, which are a particularly relevant constraint in our application to wetland creation/restoration and can be mathematically stated as follows:

$$\sum_{c \in CT} (\text{watreq}(c) \cdot X(c)) \leq \text{watlim} \tag{6}$$

where watreq(c) represents the water requirements for crop c, X(c) is the crop portfolio, and watlim is the water availability constraint. The water availability constraint is generated in the SWAT+ model and integrated into the PMP model leveraging the PMP-SWAT+ bidirectional protocol (see Section 3.4). By simulating the crop portfolio responses of irrigators to alternative water availability constraints, we can estimate the variations in the gross margin, reflecting the profitability of agricultural activities at each level of water availability, which allows us to estimate the foregone gross margin (or costs) of limited water availability due to wetland expansion. In turn, the new crop portfolio is a relevant force to the SWAT+ model that conditions water dynamics in the following year.

3.3. Benefit transfer model

Various methods have been used to estimate the economic benefits produced by wetlands, which are typically divided between stated preference methods and revealed preference methods. Stated preference methods, including Contingent Valuation and Choice Experiments, ask individuals to voice their willingness to pay for specific services, and capture both use (values based on actual use of the asset) and non-use values (not associated with actual use, or even an option to use the asset) (Costanza et al., 1998) . On the other hand, revealed Preference methods such as Hedonic Pricing and the Travel Cost Method, *inter alia*, can derive the use value of the asset from observed behavior and expenses linked to ecosystem services (Hanley, 2013). Both stated and revealed preferences valuation methods quantify benefits using ad-hoc models and data that can significantly increase the time and budget needed for the study. To circumvent these limitations, benefit transfer methods that rely on existing models to develop metamodels are often used to estimate the economic benefits of wetlands. For instance, researchers might use a benefit transfer approach by applying valuation estimates from one well-studied wetland area to another with similar characteristics. A notable example is the use of valuation studies from the Everglades in Florida to estimate the economic benefits of wetland restoration projects in the Mississippi Delta (Johnston and Rosenberger, 2010). Another example involves transferring ecosystem service values derived from wetlands in the Chesapeake Bay to similar wetlands in the Great Lakes region, allowing policymakers to estimate benefits without conducting extensive new surveys (Plummer, 2009). These examples demonstrate how benefit transfer methods can provide valuable, cost-effective insights into the economic contributions of wetlands, aiding in conservation and policy-making efforts.

The transfer value method, also known as benefit transfer method, leverages existing data from similar assets (in our case, wetlands) where the value of relevant services has already been determined through an ad-hoc model, and transfers it to a target asset located elsewhere (Westra and Boutwell, 2013). In order to determine the target asset's value, the information from the original valuation study is transferred and adjusted for variations in size, ecosystem services, and socioeconomic considerations relevant to the target asset (Westra and Boutwell, 2013). This can be done using two different approaches: unit value transfer and function transfer (Van Zanten et al., 2023). Unit value transfer adjusts mean or median values (sometimes complemented with intervals) from the original study to the target asset, while function transfer uses an estimated valuation function tailored to the target asset's specific context (Ready and Navrud, 2006). Due to the limited requirements in terms of time, budget, or data which often constrain the ability to conduct original valuation studies, the benefit transfer method is one of the most widely adopted methods to value the services provided by environmental assets (Westra and Boutwell, 2013). The benefit transfer method can also quantify uncertainty ranges by using multiple valuation studies adopting alternative inputs and model parameterization and structures. On the other hand, the accuracy and reliability of this method hinges on the similarity between the study and policy sites, and it relies heavily on the availability and quality of existing valuation studies (Wilson and Hoehn, 2006). These limitations can be addressed using repositories that incorporate a wealth of detailed information from diverse studies, including on wetland creation and restoration studies, such as the Environmental Valuation Reference Inventory (EVRI, 2023). Additionally, wetlands' values are often context-specific, influenced by local ecological conditions, community dependence on wetland services, and regional policy frameworks, which transferred values may not fully capture (Westra and Boutwell, 2013). Issues related to aggregation and scaling can further distort the true economic value of wetlands in the valuation of the target asset (Jenkins et al., 2010). Despite these limitations, the transfer value method remains a vital component of the toolkit for wetland economic valuation, enabling more informed and effective environmental management and policy-making (Richardson et al., 2015). To enhance its robustness, efforts should be made to compile high-quality, contextually relevant valuation studies and refine transfer techniques to better account for site-specific differences (Richardson et al., 2015).

In recent years, numerous studies have estimated the economic values of the ecosystem services provided by wetlands, such as water purification, water availability, risk reduction, climate change mitigation, and recreation services, *inter alia* (Dubgaard, 2004). These studies include both primary studies using ad-hoc valuation methods (see e.g., Imdad et al., 2023; Pelletier et al., 2021) and benefit transfer studies. Our study conducts an original benefit transfer that includes those studies relevant to the context of the Flumen Watershed and then validates this information with the data provided in existing benefit transfer studies that are relevant to our case study.

3.3.1. Data

To conduct our original benefit transfer study, we generated a comprehensive database of primary studies leveraging EVRI, a searchable repository of empirical studies on the economic value of environmental assets. The first step involved a search on primary wetland valuation studies, which yielded 162 studies encompassing all types of wetlands and their ecosystem services. The second step involved filtering these cases to focus on those wetlands that provided essential ecosystem services to the case of the Flumen Watershed wetland restoration project based on the documentation from (CREAMAgua, 2011), namely: water quality improvement, water availability, biodiversity enhancement, and mitigation of extreme events. This step yielded 21 relevant studies. Noteworthy, these studies also included additional ecosystem services provided by wetlands, for a total of 22 ecosystem services categorized into cultural services, supporting services, provisioning services, and regulating services (Table 3).

For each of the 21 selected case studies, the *per annum* economic value of one hectare of wetland was obtained and subsequently updated to 2020 EUR using the Consumer Price Index and the exchange rate. After adjusting the economic value of wetlands to EUR per hectare for the year 2020, the economic value was calculated using unit value transfer method as the mean of the studies in the sample, considering all ecosystem services. Upper and lower thresholds were also obtained by ranking studies from higher to lower value and then obtaining the thresholds for the first (25 % of the studies with the lower valuation) and third quartile (25 % of the studies with the higher valuation).

Table 3
Group of ecosystem services included in the transfer value approach.

Category	Wetlands services
Cultural Services	Aesthetic value (source of inspiration for art and culture) Physical and mental well-being (walking, playing sports)Sense of belonging to the place (spiritual needs) Tourism
Supporting Services	Habitat support for animal and plant speciesMaintenance of genetic diversity (different animal and plant species)Resilience (e.g. future food availability) Preservation of unique habitatsPreservation of unique species
Provisioning Services	Availability of food (agricultural and wild crops) Availability of raw materialsAvailability of medicinal and pharmaceutical resources (e.g. medicinal plants) Availability of fresh water
Regulating Services	Mitigation of extreme events (natural disasters) Local climate and air qualityAbsorption of carbon dioxide (greenhouse gases) from the atmosphereWastewater treatment (reduction of pollution generated by human waste)Biological control (insect pests, pests) Water quality Erosion prevention and maintenance of soil fertilityPollination (essential for the development of seeds, plants, fruits) Conservation of resources (e.g. plants for pharmaceutical research, tourism)

3.3.2. Calibration

The benefit transfer method described above yielded considerable variation in the economic valuation of wetlands. The minimum economic value was observed in a study in Canada, where wetlands primarily providing recreational ecosystem services were valued at 208.38 EUR/ha/year; while the maximum value was observed in a New Zealand study, with wetlands valued at 19,686.17 EUR/ha/year (detailed results for each study are available in Annex III). This wide range is frequently observed in benefit transfer studies and is explained by factors including the diverse geographical regions of the original studies, prevailing climatic conditions, and the number of ecosystem services provided and/or valued. Notably, while the LIFECREAMAgua project identifies five critical ecosystem services in the Flumen Watershed (water quality improvement, water availability, biodiversity enhancement, and mitigation of extreme events), the 21 studies selected valued 22 different ecosystem services, which can amplify the gap between studies valuing a reduced number of ecosystem services (e.g., the study in Canada focused on recreational services) and more comprehensive studies (e.g., the study in New Zealand).

Using the unit value transfer approach, the average economic value of wetlands from the 21 studies in the sample is estimated at 4,748 EUR/ha/year. Upper (first quartile) and lower (third quartile) thresholds are estimated at 631.58 EUR/ha/year and 6,071,57 EUR/ha/year.

Next, we validate the values reported in our original benefit transfer study with the data provided in already existing benefit transfer studies that are relevant to our case study site. For the benefit transfer studies on wetland restoration/creation identified in the literature, we updated the reported values to 2020 EUR following the procedure described in Section 3.3.1. Table 4 synthesizes the values reported in previous benefit transfer studies on wetland creation and/or restoration available in the literature. The reported values fall within the range of values found for our ad hoc benefit transfer study, including the Brander et al. (2010) benefit transfer study conducted in Spain, and most reported values fall within the upper and lower thresholds defined for our study.

Table 4
Economic valuation of wetland restoration/creation: Results of previous benefit transfer studies.

Author and year	Area	Country	Intervention	Min value (EUR/ha)	Max value (EUR/ha)
Jenkins et al. (2010)	Mississippi Alluvial Valley	US	Wetland restoration	1,398.64	1,448.34
Wilson (2010)	British Columbia's Lower Mainland	Canada	Wetland conservation	2,876.48	4,293.59
Pattison et al. (2011)	Black River Sub-watershed	Canada	Wetland restoration & conservation	109.4 million (conservation)12.3 million (restoration) (Net Present Value over a 30-y period)	
Brander et al. (2010)	Spain	EU 27	Wetland conservation	235.79 (Finland) to 80,982 (Malta). Spain: 6,996.84 EUR	

3.4. Model coupling

The coupling of the three models is implemented via protocols. Protocols establish guidelines for exchanging information regarding the coupling variables among the different models presented in Sections 3.1–3. These guidelines facilitate the translation and mapping of information between models, ensuring that the destination model can interpret the necessary inputs by converting information into a format that the destination model can understand.

The primary element linking the models is land use by agriculture and wetlands. Protocols are triggered at the start of a new scenario simulation and operate dynamically throughout the entire climatic series (see Section 3.5). Two protocols are developed to couple the water (SWAT+) and human (PMP and benefit transfer) system models: one bidirectional protocol that interconnects the PMP and SWAT+ models (Section 3.4.1), and one sequential protocol that works from the benefit transfer model to the SWAT+ model (Section 3.4.2). A schematic representation of the protocols developed here is provided in Fig. 2.

3.4.1. PMP-SWAT+ protocol

The bidirectional protocol between the PMP and SWAT+ leverages the concept of Hydrologic-Economic Representative Units (HERUs) (Essenfelder et al., 2018). HERUs are defined as "the lowest level spatially-disaggregated entities endowed with decision-making capacity, resulting from the combination of [SWAT+] HRUs and [PMP] socioeconomic agents", where each resulting HERU is "a spatially-homogeneous hydrologic-economic entity comprising common behavioral preferences at an individual or at a group of individuals level and representing homogeneous land cover, land management, and soil characteristics for the hydrological-economic simulations" (Essenfelder et al., 2018). By merging physical and economic spatial data, HERUs not only identify a common spatial unit between human and water systems but also facilitate the exchange of information between them. Specifically, HERUs allow the exchange of information between PMP and SWAT+ on crop choices, land use management and water withdrawals (from PMP to SWAT+), and water availability (from SWAT+ to PMP).

The process of developing HERUs is explained in detail in Essenfelder et al. (2018), and briefly summarized in the following lines. In the SWAT+ model, HRUs are the most fundamental computational entities, defined by areas with uniform land use, management practices, topography, and soil characteristics (Neitsch et al., 2011). Similarly, in mathematical programming models such as PMP the fundamental computational entities are socioeconomic agents, either as individual farmers or representative groups of farmers (e.g., at a municipality level, as is the case in this study). Both HRUs and socioeconomic agents can be spatially identified; however, they typically represent different spatial units. Socio-economic agent boundaries are often determined by political or socio-economic factors, whereas HRUs boundaries are based on physical and land management features. By overlaying these spatial units, a new spatial element can be identified to capture both biophysical and socioeconomic processes. This is done by integrating the socioeconomic agents in PMP into SWAT+ using a raster overlay of socioeconomic agent maps and land use maps before the HRU analysis phase in SWAT+ (Winchell et al., 2007). This overlay creates unique combinations of land use and socioeconomic agent codes, which are then added to the SWAT project database as unique plant codes (e.g., CPNM in the "crops" table). Plant-specific parameters are imported from their respective land uses (e.g., maize, wheat) to generate these unique codes in the SWAT+ database.

The bidirectional protocol leverages HERUs to dynamically exchange information between the PMP and SWAT+ models on a yearly basis, as follows: water availability constraints are imported from SWAT+ into the PMP model, which in turn determines land (and water) use and management operations (i.e., crop portfolio) under the newly defined constraints; this crop portfolio data is next imported from the PMP into SWAT+, specifying crops to be planted in each HERU and the irrigation water applied. The trigger to these interactions can be physical (e.g., water scarcity) but also political (e.g., a pricing policy that constrains farmers to irrigate less).

3.4.2. Benefit transfer to SWAT+ protocol

The protocol to import the estimated benefits of wetlands obtained with the transfer value approach into SWAT+ *is one-way sequential. First, the economic value of wetlands is obtained in 2020 EUR on an annual and per hectare basis (EUR/ha/year), using the benefit transfer methods described in Section 3.3. Second, climatic and management scenarios (Section 3.5) are used to run simulations with* SWAT+ *to assess their impacts on water availability for wetlands (as well as other uses such as agriculture), and their surface, for every year in the climatic series. Finally, the surface area of wetlands (in hectares) for each year in the series is multiplied by the economic value obtained in the first step to calculate the annual economic benefit from wetlands.*

3.5. Scenarios

The hydroeconomic modeling framework simulates how climatic conditions and water management in the Flumen watershed influence water dynamics, including the development of wetlands (focusing on the area covered by wetlands and the volume of water stored) and how these conditions affect water availability for irrigation and economic return (including environmental benefits and costs to irrigation). Two types of scenarios are considered in our analysis: climatic and management scenarios. The climate change scenarios correspond to the Representative Concentration Pathways (RCP) adopted by the Intergovernmental Panel on Climate Change (IPCC) to project future greenhouse gas concentrations and are described in Section 3.5.1. Management scenarios include the wetland scenario, where wetlands are developed affecting water allocation among uses, and the counterfactual scenario where no wetlands are developed and the water allocation rules existing prior to the LIFE CREAMAgua project apply (Section 3.5.2).

3.5.1. Climate change scenarios

The study considers three climate change scenarios, namely the **RCP 2.6, 4.5, and 8.5**, generated using five alternative Coupled Model Intercomparison Project Phase 6 (CMIP6) global climatic models: ACCESS-ESM1-5, CNRM-CM6-1, GFDL-ESM4, HadGEM3-GC31-LL, and MPI-ESM1-2-LR. The outcomes from these models are downscaled using the Long Ashton Research Station Weather Generator (LARS-WG), a robust stochastic weather generator equipped to integrate data from the CMIP6 and provide projections based on various RCPs (Lotfi et al., 2022), which has been extensively utilized in climate impact studies due to its ability to replicate observed weather statistics, generate future climate scenarios, and simulate daily weather data (Semenov and Barrow, 2002). The current version of LARS-WG8 incorporates climate projections from the latest CMIP6 ensemble used in the IPCC 6th Assessment Report (Semenov, 2021). LARS-WG8 employs a semi-empirical approach, wherein statistical properties derived from historical weather data are used to generate synthetic weather data under both current and future climate conditions.

In this study, LARS-WG has been first calibrated using historical weather data of the Flumen watershed from 1951 until 2019, including daily maximum and minimum temperatures and precipitation. This process ensures that the generated data accurately reflects observed climatic conditions. LARS-WG8 has been next used to generate synthetic daily weather data for future periods (2021–2040, 2041–2060, 2061–2080, 2081–2100) based on the selected RCP scenarios 2.6, 4.5, and 8.5. This process involves running LARS-WG8 with the calibrated parameters and the integrated CMIP6 data. The generated data includes projections of daily maximum and minimum temperatures and precipitation, which are crucial for subsequent climate impact assessments. This synthetic data helps in understanding potential changes in climatic variables over the 21st century. Finally, the generated climate scenarios are integrated with the impact assessment model SWAT+ . This integration allows for the evaluation of climate change impacts on hydrological processes and water availability in the Flumen watershed. The combined use of LARS-WG8 and SWAT+ facilitates a comprehensive understanding of climate change effects at regional scales.

3.5.2. Management scenarios

Two management scenarios are considered, namely the *wetland scenario* where the sixteen wetlands introduced in the LIFE CREAMAgua project with a total area of 300 ha and 1 m average depth of water are integrated into SWAT+; and the *counterfactual scenario* where the wetlands introduced in the LIFE CREAMAgua project are removed from SWAT+ . By simulating the wetland and counterfactual scenario and comparing their environmental and socioeconomic performance (including costs and benefits), we can estimate the sustainability and net benefit/cost of wetland creation/restoration policies.

Wetland scenario. In the wetland scenario, the water allocation rule to users (economic, including agriculture; and environmental, including wetlands) gives priority to wetland water demand, and thus is conditional to the wetland operation rule. The *wetland operation rule* in SWAT+ builds on the standards defined in the LIFE CREAMAgua project and aims to capture the hydrodynamic behavior of the wetlands while maintaining their ecological functions (CREAMAgua, 2011). The operation rule is based on the volume of water stored in wetlands, precipitation, evapotranspiration and the average depth of water. Considering the average depth of water at the principal spillway and the emergency spillway are both set at 1 m, one hectare of wetland can store 10,000 cubic meters (m^3) of water. This calculation assumes that the depth of water is consistently maintained at least at 1 m, which is essential for preserving the ecological functions of the wetlands (Papa and Frappart, 2021). The wetland operation rule used to model the hydrodynamics of the wetlands is formulated (Eq. (7) to maintain the ecological status of wetlands by determining the amount of water that they release into the river by number of days (SWAT+, 2021).

$$R = \begin{cases} \text{below\_emer} & \text{if} \quad P_{vol} < V < E_{vol} \\ \text{above\_emer} & \text{if} \quad V > E_{vol} \\ 0 & \text{if} \quad V \le P_{vol} \end{cases} \tag{7}$$

where *below_emer* is the volume (m^3) of stored water in wetlands (less than the required volume); above_emer is the volume (m^3) of stored water in wetlands (more than the required volume); V is the volume of stored water in wetlands (stored on a HERU); P_{vol} is the precipitation volume; E_{vol} is the evapotranspiration volume; and R are the days of water release.

Note that the larger the number of release days (R), the larger the amount of water available for irrigation ($W_{irrigation}$), which affects the water allocation rule below.

The water allocation rule under the wetland scenario considers the availability of water relative to water demand for both irrigation and wetland maintenance, as follows (CREAMAgua, 2011):

- Sufficient water for both irrigation and wetlands. If the water available for irrigation ($W_{irrigation}$) exceeds the irrigation water demand ($D_{irrigation}$, i.e., the amount of water that is demanded in a normal hydrological year without droughts) and the combined water stored by wetlands and available for irrigation ($W_{wetlands}+W_{irrigation}$) also exceeds the irrigation water demand, there is no reduction in water usage for agriculture.

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$$\text{If } W_{irrigation} > D_{irrigation} \text{ and } W_{wetlands} + W_{irrigation} > D_{irrigation} \tag{8}$$

$$\Delta W_{agriculture} = 0$$

- Wetland conservation under water scarcity. If the water available for irrigation ($W_{irrigation}$) is less than the irrigation water demand ($D_{irrigation}$), but the combined water stored by wetlands and available for irrigation ($W_{wetlands}+W_{irrigation}$) exceeds the irrigation water demand, the reduction in agricultural water usage is caused by the need to protect wetlands.

$$\text{If } W_{irrigation} < D_{irrigation} \text{ and } W_{wetlands} + W_{irrigation} > D_{irrigation} \tag{9}$$

$$\Delta W_{agriculture} = D_{irrigation} - W_{irrigation}$$

- Severe Water Scarcity. If the water available for irrigation ($W_{irrigation}$) is less than the irrigation water demand ($D_{irrigation}$), and the combined water stored by wetlands and available for irrigation ($W_{wetlands}+W_{irrigation}$) still does not meet the irrigation water demand, the reduction in agricultural water usage is caused by both water scarcity and the need to maintain wetlands.

$$\text{If } W_{irrigation} < D_{irrigation} \text{ and } W_{wetlands} + W_{irrigation} < D_{irrigation} \tag{10}$$

$$\Delta W_{agriculture} = D_{irrigation} - (W_{wetlands} + W_{irrigation})$$

Counterfactual scenario. In the counterfactual scenario, the *wetland operation rule* defaults to 0, while the water allocation rule is the same as in the wetland scenario, only in this case $W_{wetlands}$=0.

4. Results

A series of hydroeconomic simulations is run in which the socioeconomic and environmental performance of the wetland and counterfactual scenarios are estimated for each climate change scenario. In a first step SWAT+ simulates, for each year *t* in the synthetic weather data series and each of the management scenarios in Section 3.5.2 (wetland v. counterfactual), the water allocation for irrigation and wetlands and the surface and water volume of the latter (as well as other relevant water dynamics features). Leveraging the benefit transfer to SWAT+ protocol, the wetland area surface obtained in SWAT+ is combined with the economic value of wetlands in EUR/ha reported in Section 3.3.2 to calculate the benefits provided by wetlands in year *t* (this value totals 0 in the counterfactual scenario without wetlands). Leveraging the PMP-SWAT+ protocol, the water allocation for irrigation obtained in SWAT+ is used to update the water availability constraint of irrigators (Eq. (6), who reassess their crop portfolio choices (Eq. 4–5) that are conveyed to SWAT+ HERUs. Crop portfolio choices are used to calculate actual water use and profit in year *t*. In year *t* + *1*, the process recommences, and SWAT+ simulates again water availability for irrigation and wetlands and wetlands area surface, starting from the agricultural and wetland land uses simulated at the end of year *t*.

We run the simulations above over the entire synthetic weather series to dynamically calculate the environmental and socioeconomic performance (including costs for irrigators and wetland benefits) under each climate change scenario for both the wetland and counterfactual scenario, including the wetland surface and volume stored in wetlands (wetland scenario simulations only), water allocation for irrigation (with different results for both the wetland and counterfactual scenario), wetland benefits (wetland scenario simulations only, obtained as the product of the wetland surface times wetland benefits obtained in Section 3.3), and irrigation profits (with different results for both the wetland and counterfactual scenario). The costs of wetlands are subsequently obtained as the difference between the foregone profit due to water allocation restrictions in the wetland scenario v. the foregone profit due to water allocation restrictions in the counterfactual scenario, for each climate change scenario. Comparing costs and benefits of wetlands we can assess their economic impact and feasibility. We report these results in detail in the figures below.

Figs. 3 and 4 show the biophysical responses of the Flumen Watershed to climate change (RCPs 2.6, 4.5 and 8.5) by assessing the surface area of wetlands (wetland scenario only) and the water allocation for irrigation (as a percentage of the water allocated in a non-drought year where there is sufficient water to address agricultural and wetlands demands), respectively. Significant variations in the wetland surface area are observed, fluctuating in response to changes in precipitation and evapotranspiration patterns predicted by the climatic models. A declining trend over the wetland surface series is observed across all climate change scenarios, which is mimicked by the wetland water volume series that are driven by wetland surface. The declining trend in wetland surface and storage is particularly pronounced under the higher emission scenarios, RCP4.5 and RCP8.5. In contrast, RCP2.6 exhibited a more stable, though still declining, pattern in water surface and storage volumes, suggesting a lower but still significant impact on wetland hydrology.

Significant variations in agricultural water use were also projected for both the wetland and counterfactual scenario, driven by the availability of water resources as influenced by climatic changes and wetland demand. Supply-demand imbalances in agriculture are significant under the more severe climate scenarios RCP4.5 and RCP8.5 where water resources become more constrained; and are aggravated by the conservation/restoration of wetlands under the wetland scenario (i.e., higher restrictions are observed in the wetland scenario).

Fig. 5 reports the benefits of wetland creation/restoration policies obtained in the wetland scenario, which are calculated by multiplying the wetland area surface under climate change obtained in SWAT+ (Fig. 3) by the economic value of wetlands in EUR/ha reported in Section 3.3.2, for each possible combination of climate change scenarios (RCP 2.6, 4.5 and 8.5), climate change models (ACCESS-ESM1-5, CNRM-CM6-1, GFDL-ESM4, HadGEM3-GC31-LL, and MPI-ESM1-2-LR), and economic value in EUR/ha/year (first quartile: 631.58, third quartile: 6,071.57 and the average: 4,748). Since the economic benefits of wetlands reported in Fig. 5 is the result of multiplying the area surface

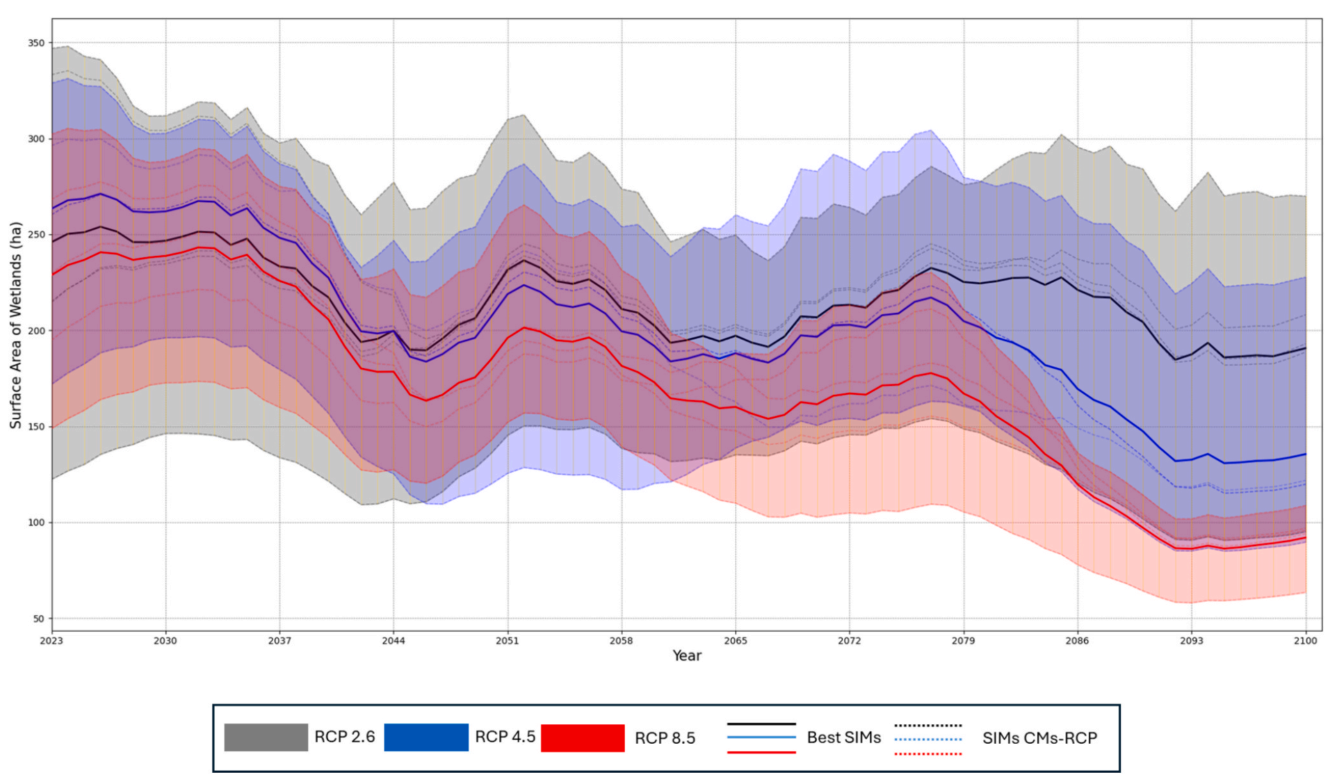

Fig. 3. Surface area of wetlands within Flumen watershed under climate change, wetlands scenario (in the counterfactual scenario the area of wetlands is 0). Each line represents the wetland area under one climate change model (ACCESS-ESM1-5, CNRM-CM6-1, GFDL-ESM4, HadGEM3-GC31-LL, and MPI-ESM1-2-LR), while each color corresponds to a climate change scenario (RCP 2.6, 4.5 and 8.5). Surface is measured in total hectares.

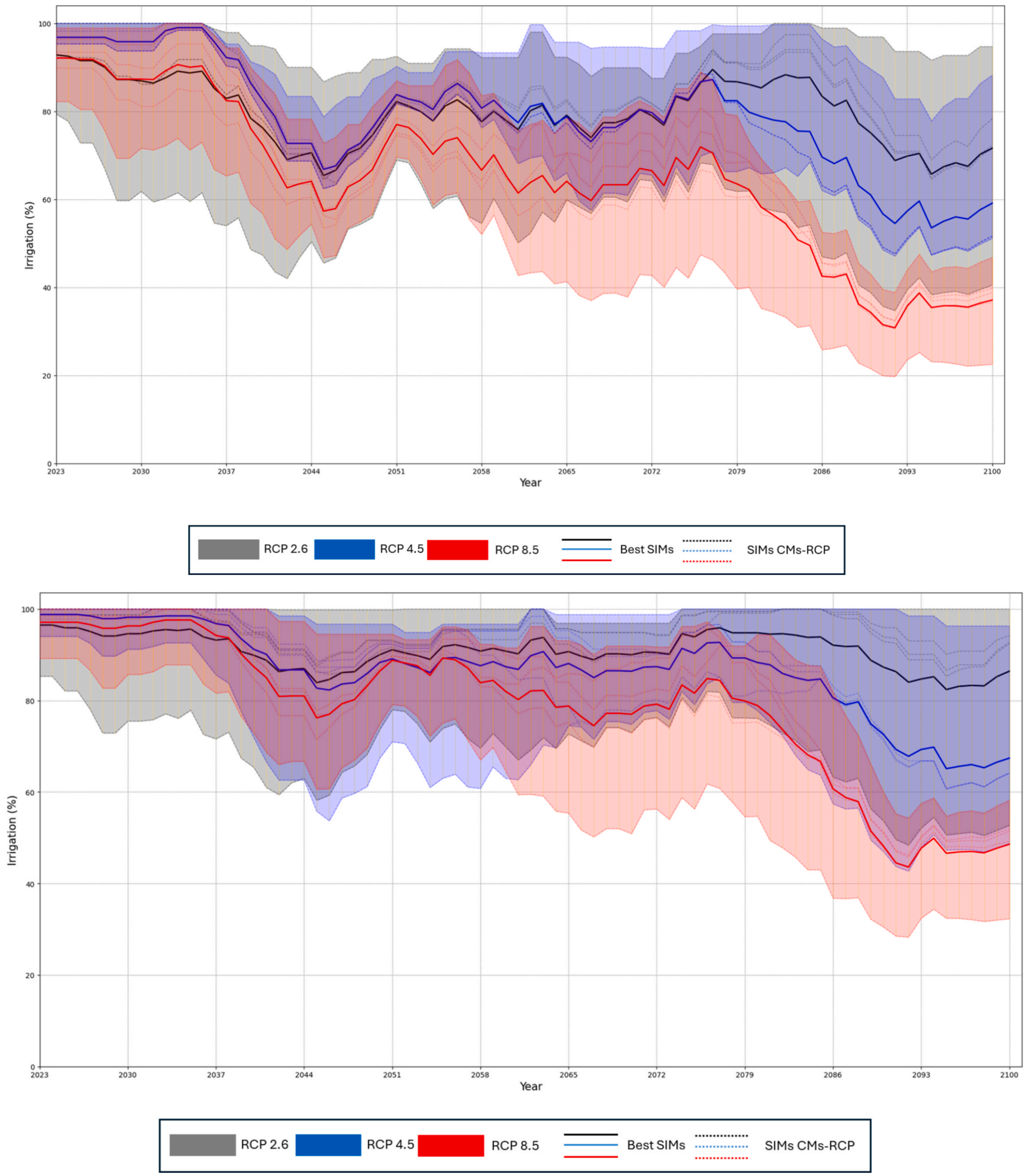

Fig. 4. Water allocation for irrigation within Flumen watershed under climate change, wetland (a) and counterfactual (b) scenario. Each line represents the water allocation to irrigation under one climate change model (ACCESS-ESM1-5, CNRM-CM6-1, GFDL-ESM4, HadGEM3-GC31-LL, and MPI-ESM1-2-LR), while each color corresponds to a climate change scenario (RCP 2.6, 4.5 and 8.5). Water allocation is defined as a percentage of the water allocated in a non-drought year where there is sufficient water to address agricultural and wetlands demands.

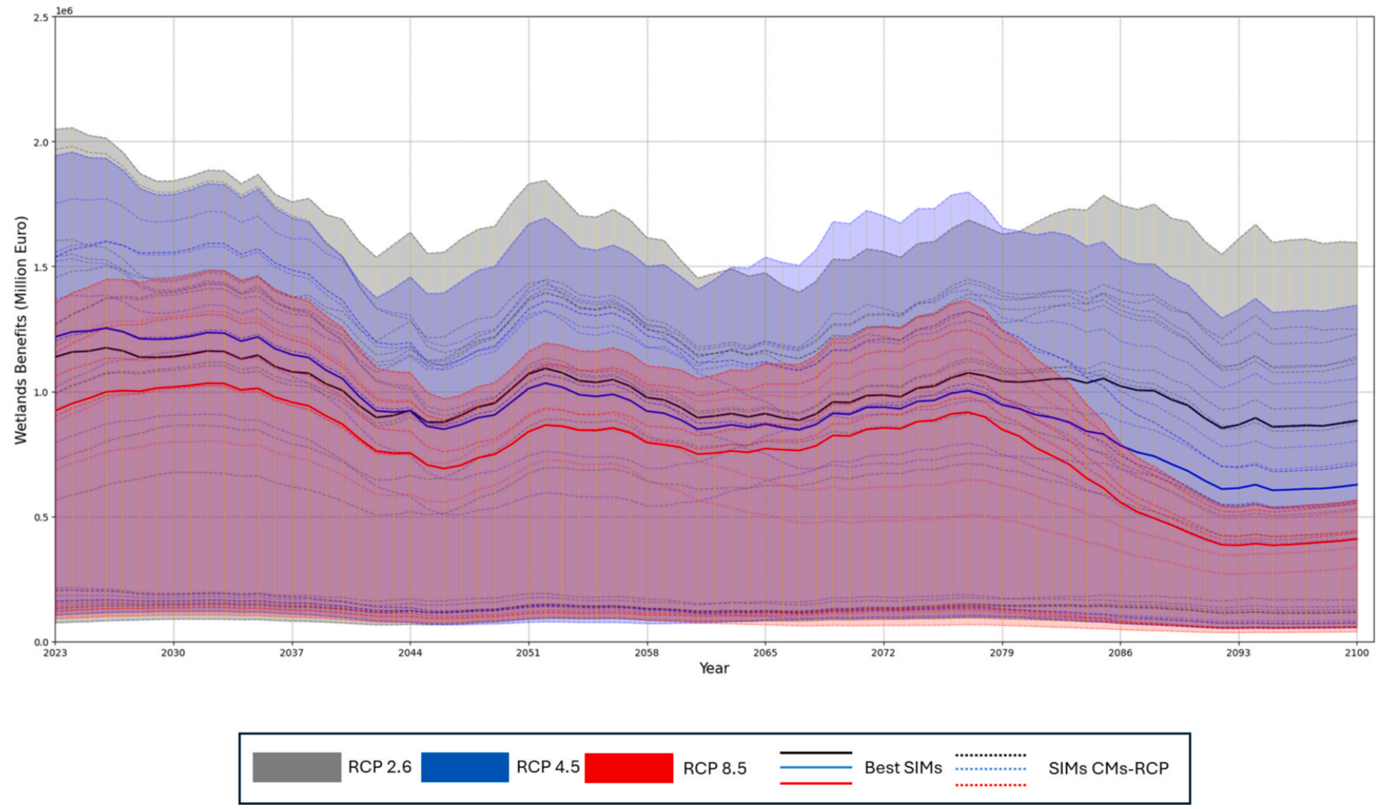

Fig. 5. Wetland benefits within Flumen watershed under climate change, wetland scenario. Each line represents the wetland benefits under each possible combination of climate change models (ACCESS-ESM1-5, CNRM-CM6-1, GFDL-ESM4, HadGEM3-GC31-LL, and MPI-ESM1-2-LR) and wetland economic value in EUR/ha/year (first quartile: 631.58, third quartile: 6,071.57 and the average: 4,748), while each color corresponds to a climate change scenario (RCP 2.6, 4.5 and 8.5). Wetland benefits are measured in EUR/year.

estimated in SWAT+ on a yearly basis (Fig. 3) by a constant value in EUR/ha, changes in the benefits of wetlands are driven by the evolution of their surface area. Accordingly, we observe a declining trend in the economic benefits of wetlands over the series that matches that of their area surface and is particularly pronounced in RCP4.5 and RCP8.5.

Fig. 6 shows the evolution of irrigators' profit (in EUR/ha) under climate change, for both the wetland and counterfactual scenarios. Irrigation profits are a function of water availability for irrigation (Fig. 4), which is recurrently reduced due to water shortages and constrains (Eq. (6) irrigators to revise their crop portfolio (Eq. 4–5); and the adaptive responses of socioeconomic agents in the PMP, which are often nonlinear and abrupt. For example, marginally decreasing water availability from 100 % to 70 % of the initial water allocation in municipalities Monflorite-Lascasas and Poleñino leads to a reduction in the surface of low value-added crops, which slightly reduces the gross variable margin by 15.07 %; while marginally decreasing water availability from 70 % to 40 % of the initial water allocation in the same municipality causes an abrupt response of irrigators, who start reducing the surface of higher value-added crops and cause a reduction in the gross variable margin of 17.75 %.

Due to the development of wetlands, the reductions in water availability and irrigation profit under climate change represented in Fig. 4 are more severe in the wetland scenario than in the counterfactual scenario (see Fig. 6). Fig. 7 builds on the results from Fig. 6 above to report the costs of wetland creation/restoration policies, which are obtained as the excess foregone profit for irrigators in the wetland scenario as compared to the (lower) foregone profit for irrigators in the counterfactual scenario. Costs show a growing trend over time, driven by the higher frequency and intensity of water allocation constraints that is particularly pronounced in RCP4.5 and RCP8.5.

Finally, Fig. 8 compares the wetland benefits (Fig. 5) and wetland costs (Fig. 7) to produce a dynamic cost-benefit analysis that illustrates the tradeoffs involved in wetland creation/restoration policies for the Flumen Watershed under each climate change scenario. Fig. 8 shows that wetland benefits consistently exceed wetland costs over the three climate change scenarios for most combinations of climate change models and wetland economic value (see Section 3.3.2), albeit the gap between benefits and costs progressively closes as climate change reduces water inputs to the basin. Climate change causes both a reduction in the surface area of wetlands and related wetland benefits, and a reduction in water availability for irrigation that depresses irrigation profit and increases wetland costs. Accordingly, over the last years of the series, wetland costs start exceeding wetland benefits – albeit only for the most pessimistic combinations of climate change models and economic values of wetlands. Overall, for any discount rate of 0 % or higher, the present discounted value of wetland restoration/conservation projects is significantly above zero for most combinations of climate change models and economic values of wetlands—the only exception being the most pessimistic combinations of climate change models, values of wetlands and discount rate (i.e., first quartile wetland value, RCP8.5, <1% discount rate), where the value is negative but close to 0. This suggests a positive and robust performance of the wetland creation/restoration project in the Flumen Watershed. Noteworthy, beyond climatic inputs our estimates can also be sensitive to changes in other environmental or socioeconomic variables over time, such as higher prices due to the reduced supply of agricultural commodities and wetland ecosystem services, and this can affect our results through impacts on economic costs (higher foregone agricultural profit) and benefits (higher wetland value) of wetland creation/restoration. We address this aspect in the following section.

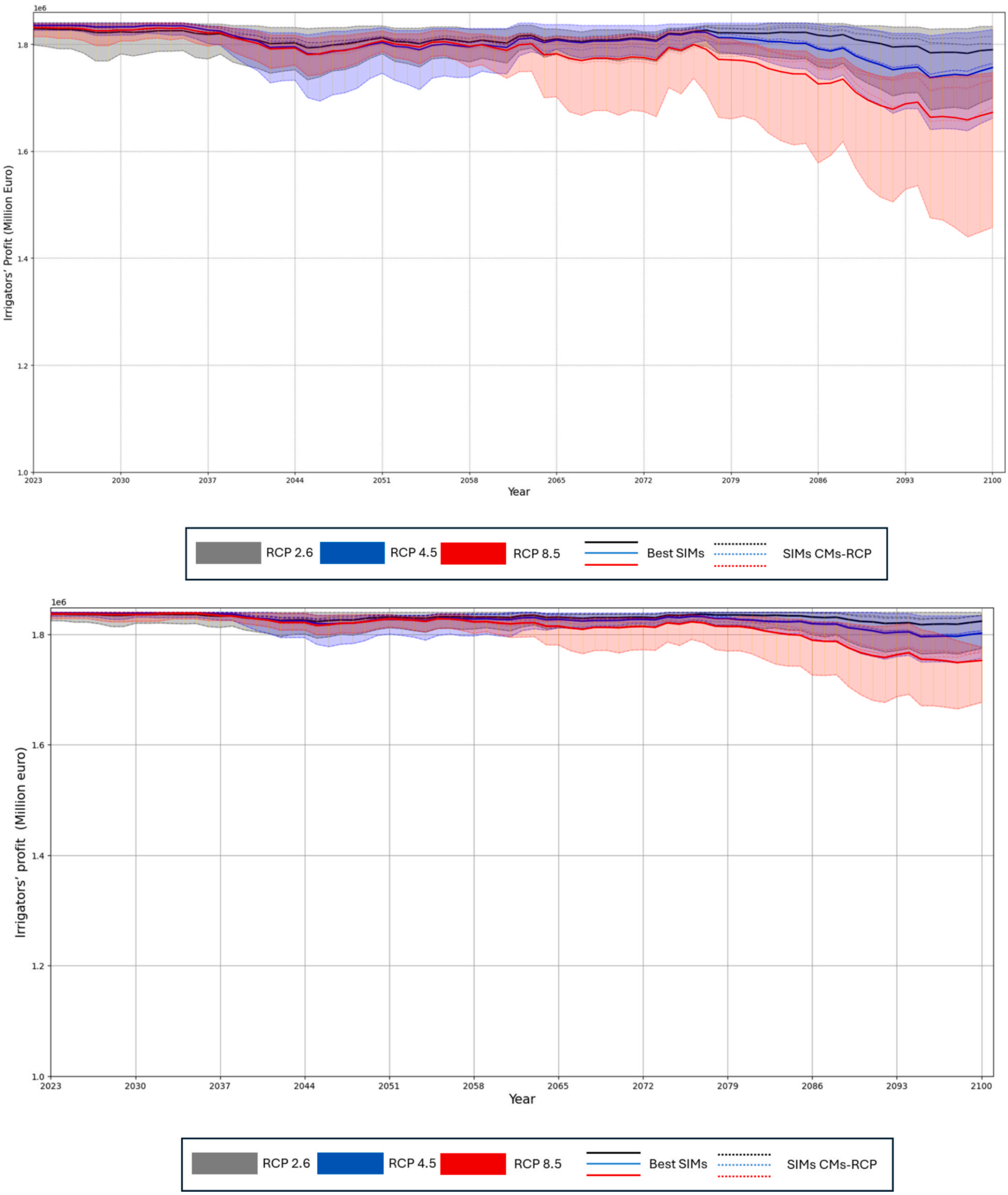

Fig. 6. Irrigators' profit within Flumen watershed under climate change, wetland (a) and counterfactual (b) scenario. Each line represents irrigation profit under one climate change model (ACCESS-ESM1-5, CNRM-CM6-1, GFDL-ESM4, HadGEM3-GC31-LL, and MPI-ESM1-2-LR), while each color corresponds to a climate change scenario (RCP 2.6, 4.5 and 8.5). Irrigation profits are measured in EUR/year.

5. Discussion

We envision several avenues for future scientific research to further develop and expand the proposed hydroeconomic modeling framework. *First*, improvements to individual models within the hydroeconomic modeling framework can be introduced leveraging recent scientific advancements in the relevant field. For example, new developments in microeconomic modeling separate land use decisions from water use decisions, introducing two decision variables (land and water use and management) instead of just one (land use and management, or crop

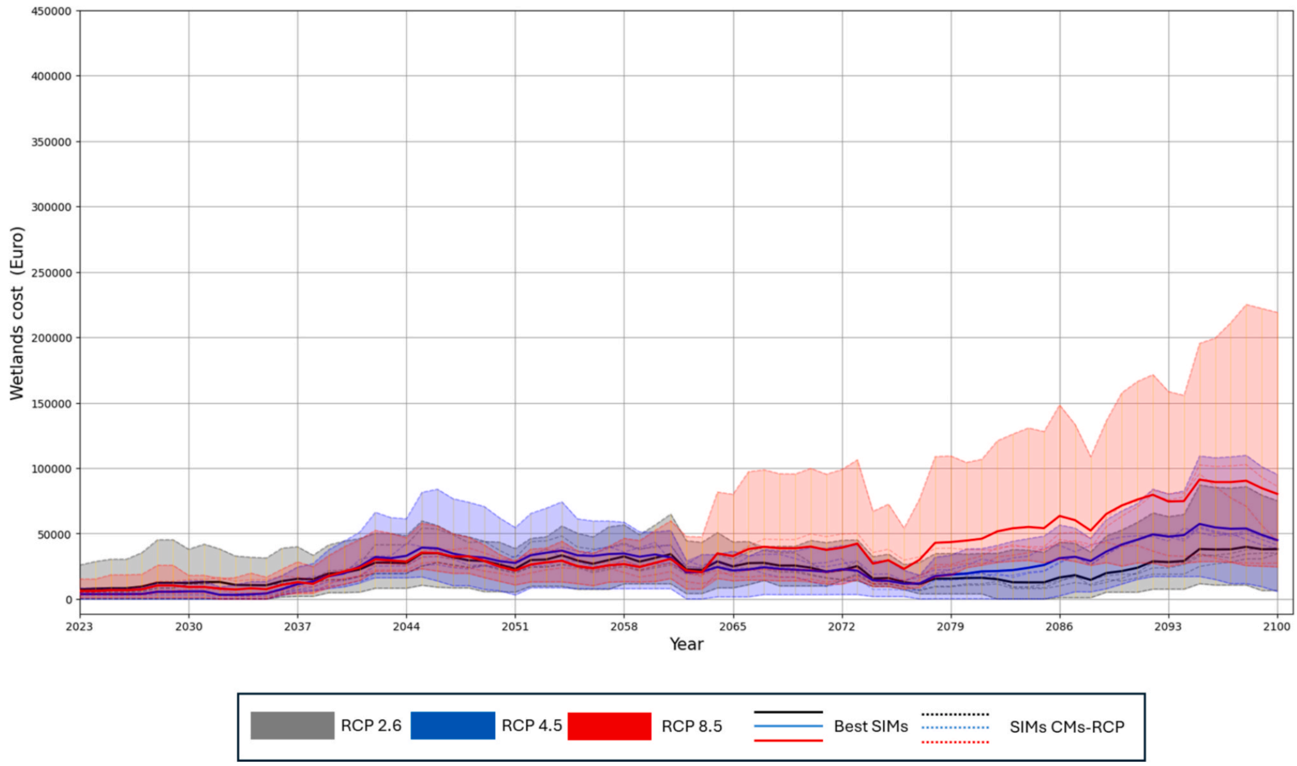

Fig. 7. Wetland costs within Flumen watershed under climate change, wetland scenario. Each line represents wetland costs under one climate change model (ACCESS-ESM1-5, CNRM-CM6-1, GFDL-ESM4, HadGEM3-GC31-LL, and MPI-ESM1-2-LR), while each color corresponds to a climate change scenario (RCP 2.6, 4.5 and 8.5). Wetland costs are measured in EUR/year.

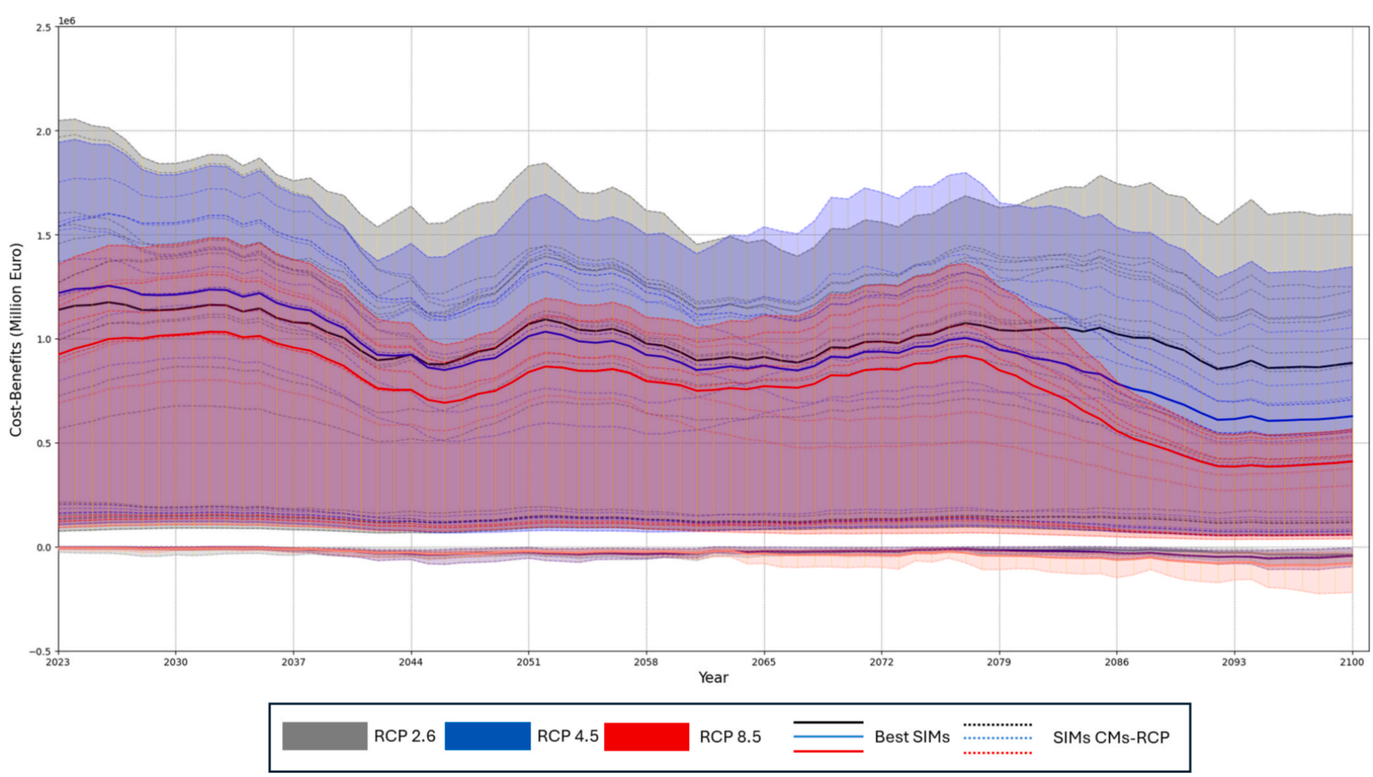

Fig. 8. Cost-benefit analysis of wetlands within Flumen watershed under climate change, wetland scenario. Each line represents wetland costs (under each climate change model) and benefits (under each possible combination of climate change models and wetland economic value), while each color corresponds to a climate change scenario (RCP 2.6, 4.5 and 8.5). Wetland costs and benefits are measured in EUR/year. The benefits are depicted in the positive range of the graph, whereas the costs are represented in the negative range.

portfolio). This is a departure from traditional PMP and other mathematical programming models, where agents only decide on land use, and allows for the evaluation and representation of adaptation strategies at the intensive margin such as deficit or supplementary irrigation, extending beyond the extensive (switching to less water-intensive crops) and super-extensive (shifting to rainfed crops) adaptations typically considered in conventional PMP and other mathematical programming models (Graveline and Mérel, 2014; Sapino et al., 2022). Additionally, the development and integration of microeconomic models that incorporate behavioral economics elements (e.g., availability bias, loss aversion, regret) could further refine the understanding of farmer decision-making processes under economic and climatic stressors (Koundouri et al., 2023; Wuepper et al., 2023). These and other innovations could also enhance the detail of the integrated hydroeconomic modeling framework, for example by adding additional protocols that introduce further detail in the model (e.g., dividing the land use-based protocol in Section 3.5.1 into two different protocols conveying information on water use and land use).

Second, uncertainty could be more thoroughly quantified, albeit a carefully designed strategy is necessary to circumvent the curse of dimensionality where the computational cost of uncertainty quantification exponentially increases with the number of variables to study. Each type of uncertainty (input uncertainty arising from the variability and potential inaccuracies in the data fed into the models; parameters uncertainties within models emerging from inaccuracies in the estimation of coefficients based on historical data or expert judgment; and structural uncertainties emerging from the choice of model structure, such as the selection of equations and relationships that describe hydrological and economic processes) presents unique challenges and requires specific strategies to mitigate its impact on model outcomes (Reichert and Mieleitner, 2009). Strategies such as ensemble modeling (running multiple models or multiple runs of the same model with different initial conditions and parameters to generate a range of possible outcomes), scenario analysis (creating and evaluating multiple plausible future scenarios to understand the potential impacts of different socio-economic, climatic, and policy changes), and sensitivity analysis (systematically varying model parameters to identify which parameters have the most significant impact on model outputs) are often used to quantify the impact of uncertainties on model predictions and to provide more reliable guidance for water resource management (Athey et al., 2019), and have been employed in this paper (ensemble forecasting for benefit transfer model, sensitivity analysis for the SWAT+ model, and scenario analysis through climatic and management scenarios). Uncertainty quantification efforts in our modeling framework could be further underpinned for example by comparing the results of the SWAT+ with other models like MODFLOW or HEC-HMS, which could offer insights into the relative strengths and weaknesses of each model, enhancing the robustness of hydroeconomic analyses (González-López et al., 2023a,b). Alternative socioeconomic models and model setups could be also adopted to assess changes and uncertainties in key variables such as prices. Notably, as wetland area and agricultural production decrease over time, their relative scarcity will increase their prices, affecting the valuation of the economic costs (higher foregone agricultural profit) and benefits (higher wetland value) of wetland creation/restoration. However, a comprehensive quantification of the three sources of uncertainty (input, parameter, structural) across each element of the hydroeconomic model, particularly where full-fledged socioeconomic and hydrologic models are integrated, is often unfeasible due to high computational costs. This explains why no hydroeconomic model in the literature has conducted a global sensitivity analysis (González-López et al., 2023a,b). Further research in this direction is necessary to ensure models can inform robust NBS and other policies, for example by applying machine learning techniques to develop emulators or metamodels that can thoroughly quantify uncertainty at a significantly lower computational cost, and thus uncover patterns and interactions that are not apparent through traditional modeling approaches (Saltelli et al., 2020). This could lead to a more comprehensive quantification of uncertainty that better informs the design of strategies towards adapting to climate change impacts on water resources – albeit at the expense of reducing our ability to understand simulation results.

Third, improvements in the accuracy of data inputs could also enhance our predictions. Notably, exploring the integration of remote sensing technologies could provide a more dynamic and precise input for water use assessments in agricultural areas. Moreover, by combining these data with climate change projections, it would be possible to more accurately forecast future irrigation demands and evaluate water resource management strategies under varying climatic conditions.

Finally, additional relevant system models could be integrated into the modeling framework, for example crop models that calculate the impact of climate change on crop yields over time. This information could be obtained by downscaling yield data from Global Gridded Crop Models available in CMIP6 or other model intercomparison projects such as the Agricultural Model Intercomparison Project (AgMIP, 2023). Incorporating new models will reveal new plausible futures and can potentially identify nonlinear changes with significant impact on the socioeconomic and environmental performance of policies; but will also further exacerbate the curse of dimensionality and the computational cost of uncertainty quantification.

6. Conclusions

This study presents a dynamic hydroeconomic modeling framework that integrates one hydrological (SWAT+) and two socioeconomic models (PMP and benefit transfer) to evaluate the socioeconomic (including costs and benefits) and environmental performance of wetland creation/restoration projects over time. The potential of the proposed modeling framework is illustrated with an application to the Flumen Watershed in Spain, where we assess the environmental and economic sustainability of the LIFE CREAMAgua wetland creation and restoration project.

Results illustrate significant tradeoffs between wetlands and irrigation in wetland creation/restoration, with wetland benefits offsetting the costs created for agriculture over the time series under most setups considered—albeit the difference is reduced over time as the effects of climate change intensify and further constrain water availability for environmental and economic uses. These results provide robust evidence for the economic and environmental sustainability of wetland conservation and restoration programs in the Flumen Watershed.

The study highlights the relevance of considering both costs and benefits, in addition to environmental indicators, in NBS performance assessments, as well as of the importance of conducting dynamic evaluations that allow for the study of climatic changes and their impacts on NBS performance, including under uncertainty. When the benefits of wetland restoration/conservation exceed costs over time and under most plausible futures, evidence can be deemed robust enough to justify investments in wetland restoration/conservation. Note that robustness necessitates a comprehensive uncertainty quantification that accounts for multiple plausible futures rather than point predictions that oversimplify the complexity of decision making in complex human-water systems under uncertainty. Moreover, in some instances evidence will not be overwhelmingly in favor of or against the investment (i.e., benefits may exceed costs in some plausible futures, while the opposite may occur in some others). In these cases, further analysis will be required to underpin decision making, notably through heuristic methods that complement mechanistic approaches like the one proposed in this paper with expert judgement that rely on the expertise of decision makers and key stakeholders.

CRediT authorship contribution statement

Noureddine Bouzidi: Writing – original draft, Visualization, Validation, Software, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **C. Dionisio Pérez-Blanco:** Writing – review & editing, Visualization, Validation, Supervision, Resources, Methodology, Investigation, Formal analysis, Conceptualization.

Declaration of competing interest

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

Annex l. : Dataset for the hydroeconomic model and the benefits transfer model.

Input	Source
Weather information	Global Data | SWAT | Soil & Water Assessment Tool (tamu.edu)
Digital Elevation model (DEM)	EarthExplorer (usgs.gov)
Soil map	Global Data | SWAT | Soil & Water Assessment Tool (tamu.edu)
Land use	Superficie Declarada PAC. Gobierno de Aragón (aragon.es)
	DESCARGAS ICEARAGON
	GeoPortal Sitebro (chebro.es)
	Centro de Descargas del CNIG (IGN)
Water management	eportal.mapa.gob.es/websiar/Inicio.aspx
	Inicio - Portal CHEbro
Socio-economic inputs	ECREA: Estudios de costes y rentas de las explotaciones agrarias (mapa.gob.es)
	Encuesta sobre Superficies y Rendimientos Cultivos (ESYRCE) (mapa.gob.es)
Case studies of wetlands benefits	EVRI

Annex ll: PMP model: domain, calibration procedure, data, and calibration results

The set of constraints that conform the domain F used in the calibration and simulation of the model is described in the following paragraphs.

Land availability. Available agricultural land is assumed constant and equals the summation of observed agricultural land uses (3).

$$\sum_{i=1}^{n} x_i \leq 1 \tag{11}$$

Water availability. It is assumed that water abstraction licenses remain constant before and after every simulation run, i.e.:

$$\sum_{i=1}^{n} w_i x_i \leq W \tag{12}$$

where w_i is crop i's specific water requirements and W is the total water allotment in the study area.

Climate and soil. Since each agricultural area/climatic region has its soil and climatic characteristics, agents in the model can only grow those crops that are observable in the database (Essenfelder et al., 2018).

$$\sum_{i=1}^{n} y_i x_i = 0 \mid y_i \in \{0, 1\} \tag{13}$$

where $y_i = 0$ means the crop is observable and $y_i = 1$ means the crop is not observable in the area.

Crop-specific constraints. Some crops in the portfolio have an upper and/or lower area bound because of specific policy restrictions.

$$\varphi_i x_i \leq (1 + b_i) x_i^0 \mid \varphi_i \in \{0, 1\};\ 0 \leq b_i \leq 1 \tag{14}$$

where φ_i is a binary vector that (de)activates the constraint, b_i is the upper bound set (in percentage) and x_i^0 is the observed share of land devoted to crop i. Equation (11) refers to the upper bound constraint; in the lower bound, the inequality would be the opposite and the right-hand side of the equation would be a subtraction. This restriction could be used to set a minimum/maximum threshold for ligneous trees of ± 5 %, to prevent large (dis)investments with potentially large impacts on e.g. carbon sequestration, whose economic value is not accounted for in the models, which focus on yearly market variables (notably profit) (Essenfelder et al., 2018).

Annex lll: Benefit transfer studies.

Authors	Year	Country	Euro/ha/year (2022)
Thompson et al.	2010	Canada	208.38
Troy et al.	2009	Canada	219.12
Danielson et al.	1996	United States	260.09
Wilson	2008	Canada	341.57
Wilson	2010	Canada	378.8
De Bruin et al.	2009	Netherlands	631.58
Beran	1995	United States	669.11
Voora et al.	2008	Canada	897.45
Camacho-Valdezbles et al.	2014	Mexico	1226.12
Jenkins et al.	2010	United States	1423.49
Anielski et al.	2005	Canada	1618.33

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(*continued*)

Authors	Year	Country	Euro/ha/year (2022)
Emerton	2003	Sri Lanka	2564.33
Campbell E.T.	2018	United States	2751.46
Gregg et al.	2018	Australia	4217.06
Turner	1991	United States	5511.46
Davis et al.	2010	United States	6071.57
Yang et al.	2009	Canada	6894.95
Moffette et al.	2015	Canada	9990.69
Morris et al.	2011	United Kingdom	14566.24
Randall et al.	1996	United States	19600.02
Ndebele et al.	2014	New Zealand	19686.17

Data availability

Data will be made available on request.

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