OPEN ACCESS
Water Economics and Policy, Vol. 8, No. 4 (2022) 2240012 (27 pages)

DOI: 10.1142/S2382624X22400124

A Hydro-Economic Model to Calculate the Resource Costs of Agricultural Water Use and the Economic and Environmental Impacts of Their Recovery

Francesco Sapino[*,§], C. Dionisio Pérez-Blanco[*,†] and Pablo Saiz-Santiago[‡]

[*]*Department of Economics and Economic History and*
IME Multidisciplinary Business Institute, Universidad de Salamanca
C/Francisco Tomás y Valiente s/n, 37007 Salamanca, Spain

[†]*Euro-Mediterranean Center on Climate Change*
RAAS Division, Via della Libertà, 12, 30121 Venezia, Italy

[‡]*Duero River Basin Authority, C/Muro, 5, 47004 Valladolid, Spain*
[§]*fsapino@usal.es*

Received 19 July 2022
Revised 4 November 2022
Accepted 18 November 2022
Published 21 February 2023

In this paper, we estimate the resource costs of agricultural water use and simulate the environmental and economic impacts of their recovery. To this end, we develop a socio-hydrology-inspired, dynamic, protocol-based modular approach that interconnects economic and hydrologic modeling via two-way feedback protocols. The hydrologic module is populated with the AQUATOOL model, the Decision Support System used in Spanish river basins; while the economic module is populated with an ensemble of four Mathematical Programming Models (MPMs) that capture human agency and responses. This allows us to sample uncertainty and provide a range for resource costs estimates and the environmental and economic impacts of their recovery, rather than a point estimate. Methods are illustrated with an application to the Órbigo Catchment, a sub-basin of the Douro River Basin in Spain. Our results suggest significant resource costs (a 34–62% increase in existing charges, depending on the model) with non-trivial impacts on income (2–27% reduction) and the environment (water savings range between 6% and 69%), while the impact on tax revenue is ambiguous yet potentially significant (between −2.3 million EUR/year and 5 million EUR/year).

Keywords: Resource cost; multi-model ensemble; mathematical programming; water policy; WFD.

1. Introduction

Water is essential for life, food production, and a key input for virtually all economic activities (Unesco 2021). However, water demand is growing at a pace that cannot be met by the increasingly volatile and overall diminishing supply (IPCC 2021). Climate change, population growth, and changing distributions of wealth are expected to intensify water scarcity and droughts in already water-stressed areas, putting water users under further pressure. This holds particularly true for irrigation, which represents 70% of global freshwater withdrawals (FAO 2021), concentrates the least valuable uses of the resource (less than 7% of the world's Gross Domestic Product) (World Bank 2020), and is accordingly targeted as the main source of much-needed water savings (OECD 2015; UNDRR 2021). A key policy to achieve such savings, as substantiated in the fourth Dublin Principle, the first Rio Principle, and recent flagship reports, is water charging — also referred to as water pricing[1] (UN 1992a,b; UNESCO 2021).

Water pricing is widely regarded and used as an instrument for cost recovery; but it is also a behavioral incentive that can align individual decisions with key societal objectives such as environmental sustainability and economic efficiency (Delacámara *et al.* 2014; Dinar *et al.* 2015), as well as building environmental and economic resilience through an appropriate linkage between current uses and future water availability — where misleading price tags can lead to overdraft, increased exposure to extreme events, and non-trivial disinvestments into the future, e.g. through the loss of natural capital or perennial crops (Loch *et al.* 2020a). To this end, it is critical that the price conveys information on the full cost of water use, including any externalities and opportunity costs that may emerge now or into the future (Adamson and Loch 2021; Dinar and Subramanian 1997; Tsur and Dinar 1997). While it has been often assumed that mature water economies with full-fledged water markets will deliver price tags that fully internalize all costs (Randall 1981), reality has proved otherwise (Loch *et al.* 2020b), making government intervention necessary to address negative externalities, improve market outcomes, and ensure that the opportunity costs of water use (including resource costs) are well understood. In this context, several governments worldwide have integrated water pricing instruments into their legal *acquis* to recover costs and reallocate water towards other productive uses and the environment (Bogardi *et al.* 2021). This is the case of EU and its Member States through the Water Framework

[1] In the policy arena, the term 'charge' (a levies introduced administratively) and the term "price" (the exchange value of a good) or "pricing", as is the case in the WFD (European Commission 2000) are commonly used interchangeably; this is also common in the scientific literature (see, for example, Dinar *et al.* 2015; Olmstead and Stavins 2007).

Directive (WFD) (European Commission 2000), which aims to put the "right price tag" on water through the full recovery of water use costs, including not only the financial costs of the resource typically charged to users (O&M and investment payback), but also the environmental (costs of associated negative environmental effects) and resource costs (forgone opportunities of alternative water uses) of water. However, whereas O&M and investment costs of water are measured and (at least partially) recovered across several countries, environmental and resource costs are rarely measured by water institutions. Moreover, the relatively low-cost recovery levels achieved in agriculture (often through flat rates that fail to reduce demand) make it unlikely that any relevant share of environmental or resource costs is charged to users (Bhaduri *et al.* 2021; EEA 2020).

Two key **barriers** explain the limited institutional measurement and recovery of environmental and resource costs. *First*, agricultural economics research and modeling have long argued that bridging the gap between observed water prices and the full cost of water use could substantially reduce agricultural income without significantly reducing water use due to the inelastic water demand of farmers (Berbel and Expósito 2020; Cornish and Perry 2003; Molden *et al.* 2010). In this context, partial cost recovery plays the role of an implicit subsidy to irrigators to prevent income losses and distributive imbalances (Rey *et al.* 2019). On the other hand, some have questioned whether this equity target could be alternatively addressed through a more environmentally sustainable instrument such as a decoupled subsidy, possibly funded via higher cost recovery (Young 2014). Moreover, the inelastic response of irrigators to prices predicted in traditional economic models has been challenged by recent research that finds a more elastic demand curve for irrigation water once intensive margin adaptation (deficit and supplementary irrigation) is considered (Graveline and Mérel 2014; Sapino *et al.* 2022), albeit inelastic responses can be still observed for perennials, particularly until minimum water requirements to ensure survival are met (Loch *et al.* 2020a). *Second*, and critical for this research, the lack of standardized accounting and monetization frameworks for environmental and resource costs remains a major barrier that hinders their recovery (Barraqué 2020; EEA 2013; UNESCO 2021). While several studies have provided definitions and conceptual frameworks for the assessment of environmental and resource costs (Berbel and Expósito 2020; EEA 2013; WATECO 2003), the number of studies that attempt to measure them and assess their impact empirically is significantly more limited — and appears biased towards the assessment of environmental costs (see e.g. Chaikaew *et al.* 2017; García de Jalón *et al.* 2017; Pérez-Blanco *et al.* 2021).

Applied studies on resource costs in the agricultural sector are limited. Pulido-Velazquez *et al.* (2013) and Pulido-Velázquez *et al.* (2006) estimate the resource cost of water use for all economic sectors in the Andra and Tous systems (Spain) by coupling a Decision Support System (DSS) model with a linear mathematical programming model (MPM) that simulates the optimal land allocation that maximizes users' net profit. By comparing the observed and the optimal allocation, the authors calculate the marginal opportunity cost of water — which ranges between 0.02 and 0.3 EUR/m^3 in the Andra system (Pulido-Velázquez *et al.* 2006) and between 0.02 and 0.75 EUR/m^3 in the Tous system depending on the type of water use. Alamanos *et al.* (2020) estimate the resource cost of water in Lake Karla (Greece) using two alternative methods: (i) as the expected foregone income using an agronomic model that proportionally allocates the water deficit among existing crops in the area; and akin to Pulido-Velazquez *et al.* (2013) and Pulido-Velázquez *et al.* (2006), (ii) as the "benefit differential between the existing water use and the optimum water use" using a linear MPM (Alamanos *et al.* 2020). The authors find resource cost estimates in the range of 10.5–22.4 million EUR/year. Looking at the demand side, Martin-Ortega *et al.* (2011) estimate the environmental and resource *benefits* of water in the Guadalquivir River Basin (Spain) under scarcity conditions through a choice experiment that yields a Willingness To Pay (WTP) ranging between 7 and 10 EUR/household, suggesting that water services hold a high value for society and pointing towards the need for water reallocations from agricultural to urban and environmental uses.

Four major **research gaps** emerge from the literature above. *First*, there is a weak integration between human and water systems in models. Most resource cost estimates rely on full-fledged economic *or* hydrological models, which ignore the feedbacks between human and water systems that are necessary to understand and interpret the human-modified water cycle (Sivapalan *et al.* 2014). Those few studies that couple human and water systems into hydroeconomic models rely on linear representations of the human system, which are subsequently integrated into the architecture of the hydrological model through piecewise equations. This approach "oversimplifies" human agency by failing to capture the relevant non-linearities that characterize individuals' adaptive behavior (Di Baldassarre *et al.* 2017), which can hinder our ability to predict and understand the evolving trajectories of coupled human and natural systems (UNDRR 2019). Two examples of nonlinearities in human agency can be found in Olmstead and Stavins (2007), which find growing price elasticities in water demand; and in Adamson *et al.* (2017), who find that shifts in water availability can induce non-trivial and sometimes abrupt changes in output decisions or water trading.

Second, and closely connected to the first point, coupled human-water systems are characterized by complex non-mechanistic dynamics and cascading uncertainties, which makes it challenging to accurately value and monetize the impact that a specific water use may have on others. However, and despite this limitation, resource cost studies typically offer single point predictions, which may lead to "unreliable" and unprecise estimates (Puy *et al.* 2022).

Third, while all studies produce resource cost estimates, no study offers an analysis of the economic and environmental impacts of *implementing* resource costs recovery.

Finally, the models developed by researchers to assess resource costs remain concealed in the academic arena and are generally not adopted by decision-makers to inform policy design (Berbel and Expósito 2020). This disconnection between scientific advances and actual decision-making is often attributed to the non-trivial time, monetary and knowledge barriers faced by decision-makers, which constrain the breadth and scope of their policy assessments (Driscoll *et al.* 2011; Nkiaka and Lovett 2019). Rather than adopting brand-new methods and models, which would demand a significant amount of limited resources, decision-makers typically follow a pragmatic approach in which they try to complete as many tasks and achieve as many objectives as possible with the resources and expertise available. This has hampered the measurement and recovery of resource costs, which require economic modeling and expertise that is typically not available in river basin authorities dominated by engineering technicians (Di Baldassarre *et al.* 2019; Sivapalan and Blöschl 2015). In this context, it is critical to design *actionable* science that allows for more effective integration of state-of-the-art economics research into the day-to-day operations of river basin authorities. One way to achieve this is by developing modular hierarchies in which the DSS already used by decision-makers are complemented by additional modules that incorporate new functions (e.g., an economic module to measure resource costs), to "respond progressively to the scale of the analysis, budgets, capacity, and timeframes of the river basin authority or competent body" (Acreman and Ferguson 2010).

From these research gaps emerges **the research question this paper is set to address**, namely: can we design actionable human-water system models that quantify the uncertainty involved in the environmental and economic assessment of resource costs, to inform robust decision-making?

To address this question, this paper develops a socio-hydrology-inspired, dynamic, protocol-based modular approach that interconnects economic and hydrologic modeling via two-way feedback protocols. The hydrologic module is populated with the AQUATOOL model, the DSS used in Spanish river basin authorities, which makes the proposed framework *actionable* and facilitates its uptake by decision-makers and other relevant stakeholders (Andreu *et al.* 1991); while the economic module is populated with an ensemble of four MPMs that captures human agency and responses (Sapino *et al.* 2020). The coupling between the human water system builds on recent work by Pérez-Blanco *et al.* (2021), which is expanded to couple multiple MPMs, rather than just one, to the hydrological module. This allows us to sample uncertainty and provide a range rather than a point prediction. In a first simulation, the proposed modeling framework is used to estimate the resource costs of agricultural water use; which are subsequently used as inputs for a second simulation that assesses the economic and environmental (through water-saving estimates) impacts of implementing resource costs recovery. Methods are illustrated with an application to the Órbigo Catchment, a sub-basin of the Douro River Basin in Spain.

2. Background to the Case Study

2.1. *Resource costs and water charges in the EU context*

In its Article 9, the WFD states that "water pricing policies provide adequate incentives for users to use water resources efficiently", while asking the Member States to "take account of the principle of recovery of the costs of water services, including environmental and resource costs" (European Commission 2000). The WFD identifies three key cost categories that should be "fully" recovered through pricing (European Commission 2000): financial, environmental, and resource costs. Resource costs were originally defined by the WATECO (*WAT*er and *ECO*nomics) Working Group of the WFD Common Implementation Strategy as the "cost of forgone opportunities that other users suffer due to the depletion of the resource beyond its natural rate of recharge or recovery" (European Commission 2003). Later definitions substantiate the notion of resource costs as an opportunity cost for alternative uses, including a more recent definition by the Working Group ECO2, in which resource costs are defined as "the opportunity cost or forgone benefits in the best alternative use" (Heinz *et al.* 2007).

Full-cost recovery of water use, including resource costs, is mandatory in the EU — albeit the WFD also states that "disproportionate" costs on users caused by the implementation of cost recovery should be avoided (European Commission 2000). However, resource (and environmental) costs are typically not included in water prices (OECD 2017). This implicit subsidy, coupled with other explicit subsidies to agriculture (e.g., for the modernization of irrigation systems), is often cited in the literature as a key factor explaining irrigation expansion and growing water demand and consumption across the EU despite diminishing supply and growing scarcity (Pérez-Blanco *et al.* 2020; Rey *et al.* 2019).

2.2. *The Órbigo Catchment in Spain*

Methods are illustrated with an application to the Órbigo Catchment in NW Spain, a historically water-abundant catchment within the larger Douro River Basin.

Annual water supply in the Órbigo has shown a consistent decrease over the past 40 years and now totals 1,436.5 million m^3 (average over the 1980–2018 period), or nearly three times as much as the average withdrawals of 484.9 million m^3 (DRBA 2016). This worsening supply-demand imbalance is reflected in the WEI+ (the ratio of freshwater use to total renewable water resources), which has grown steadily over the period and is now estimated at 33.8%, above the water scarcity threshold of 20% and rapidly approaching the absolute water scarcity threshold of 40% (DRBA 2021; EEA 2016). The frequency and intensity of drought events have significantly increased as well over the period (DRBA 2017; MAGRAMA 2017).

In the Órbigo Catchment irrigation represent 96.7% of the total water demand (DRBA 2021), and our case study comprises 12 Agricultural Water Demand Units (AWDUs), the basic irrigation unit in Spain (Figure 1). AWDUs are "local irrigation communities with common hydrological (e.g., water source), spatial (i.e., territory), and administrative characteristics" (DRBA 2016). The irrigated area object of this study comprises 41,000 ha, which are dominated by maize (73%), wheat (8%), sugar beet (7%), sunflower (4%), alfalfa (4%), hop plant (1%), barley (1%), and other crops (2%). The average Gross Value Added (GVA[2]) in the Órbigo Catchment irrigated land is 771 EUR/ha, 93% of which comes from profit (716 EUR/ha, significantly lower than in other irrigated areas in Spain but still well above the average profit of 390 EUR/ha for rainfed agriculture) and the remaining 7% from labor income (55 EUR/ha). The 12 AWDUs are supplied by a large reservoir located in the headwaters of the catchment, the Barrios de Luna Reservoir. Barrios de Luna has a capacity of 308 million m^3 and supplies approximately 272 million m^3/year for irrigation, plus 80 million m^3/year for higher priority uses including urban, industrial, and environmental (to sustain environmental flows in the Órbigo and Luna rivers) (DRBA 2017). The imbalance between water supply and demand (annual demand exceeds water stock by 44 million m^3) becomes apparent during the increasingly recurrent and intense drought periods when low-priority agricultural uses can experience water restrictions. Attempts to enlarge and increase the reliability of water supply through additional water infrastructures have been unsuccessful thus far: the Omaña Reservoir project (200 million m^3)

[2]GVA is obtained as profit plus labor income. Profit (in EUR/ha) is obtained as price (in EUR/kg) times yield (in kg/ha), plus coupled subsidies, and minus the variable costs (in EUR/ha). Labor income (in EUR/ha) is obtained as hired labor (in numbers of working days/ha) times daily wage (in EUR/working day). Both labor and profit are attributes $\boldsymbol{Z}(\boldsymbol{X})$ whose relevance is explored by the ensemble of models, and their mathematical formulation is available in Annex I. The GVA is initially obtained per hectare and per crop, and then combined with the crop portfolio $\boldsymbol{X}$ to obtain the GVA at an AWDU level.

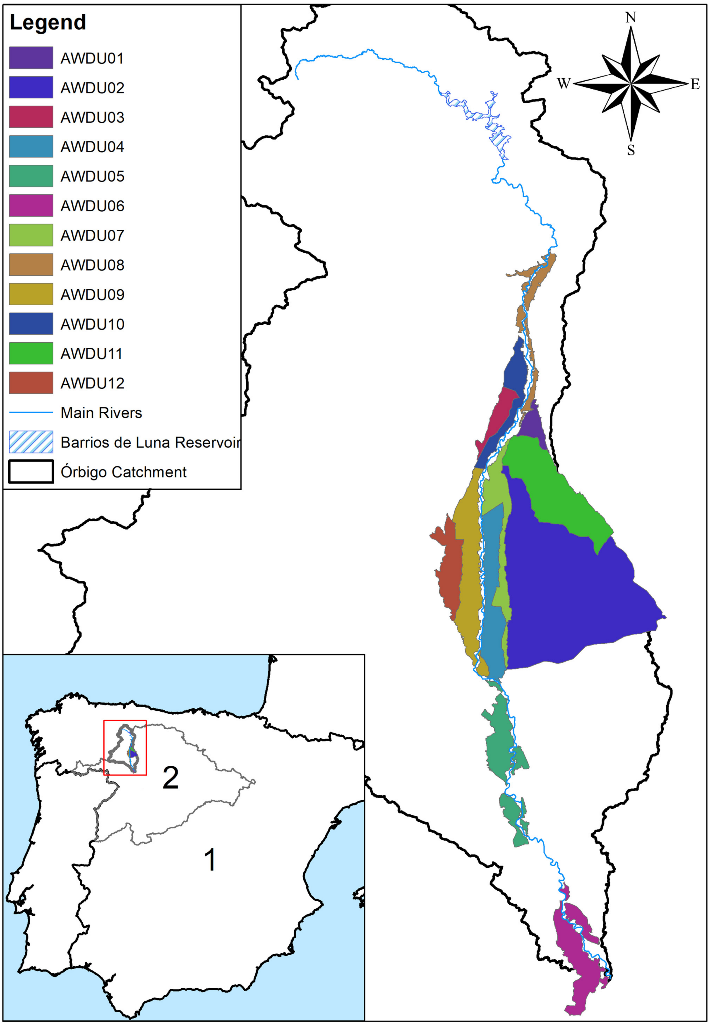

Figure 1. Study Area: The Órbigo Catchment

Source: Own elaboration.

was discarded in 1993 due to environmental concerns, while the smaller La Rial and Los Morales reservoirs project (23 and 11 million m^3, respectively) is on standby and both reservoirs were excluded from the latest river basin management plan for the 2021–2027 period due to concerns over their economic and financial sustainability (DRBA 2021)

As in other basins across Spain and the EU, agricultural water prices in the Órbigo Catchment are an instrument for financial cost recovery rather than for modulating the use of increasingly limited water resources. The average water charge in the Órbigo Catchment is estimated at 0.047 EUR/m^3, which allows a recovery of about 70% of the total financial costs (DRBA 2021). This cost recovery figure drops to 45% if recent estimates of environmental costs are included (Pérez-Blanco *et al.* 2021). No estimates of resource costs are available for the basin.

3. Materials and Methods

This paper develops a modular hierarchy of human and water systems to estimate the resource cost and then assess the effect of the full recovery of this cost in the agricultural sector. The water system is populated with the DSS used by Spanish river basin authorities, AQUATOOL; while the human system is populated with an ensemble of four conventional MPMs that reproduce the behavior of irrigators. The human-water model was developed under the auspices and with the collaboration of the Douro River Basin Authority in the context of one European and two national research projects in which the Universidad de Salamanca and the Douro River Basin Authority collaborate (see Acknowledgements Section for details on the projects). Model co-development efforts between scientists and stakeholders helped us to deliver an actionable modeling framework that can be readily used by stakeholders to inform actual decision-making. The following sections present each system/module and the models that populate them, the coupling protocol developed to integrate human and water systems, and the simulation setup adopted for the resource costs application.

3.1. *Hydrologic module*

AQUATOOL is the DSS model used by Spanish river basin authorities to advise decision-making at a basin level (Andreu *et al.* 1991). AQUATOOL is a complex interface including several modules, each addressing key aspects of the hydrological system: SIMGES (simulates watershed management), GESCAL (simulates water quality at basin scale), OPTIGES (optimizes watershed management), SIMRISK (risk assessment and management), EGRAF (shows the graphical results of the previous modules), and EXTOPO (exports spatial data to vector format) (PUV 2020). In this paper, the AQUATOOL and SIMGES modules were used for setup and simulation, respectively. These two modules import and manage information on many aspects of the water system, namely flows and stocks in surface and “groundwater bodies, discharge under natural conditions, river-aquifer interaction, infrastructures (reservoirs, canals, irrigation systems), water demand units (including AWDUs), conveyance, distribution, and application inefficiencies (and related return flows and non-beneficial consumption), evaporation from reservoirs, environmental flows, water rights, and water operation rules" (Pérez-Blanco *et al.* 2021).

A key input for AQUATOOL is the discharge series under natural conditions, which are produced by treating daily precipitation data from 1950 to 2015 with the EVALHID tool, which integrates several rainfall-runoff models (Lerma *et al.* 2017). This 1950–2015 series is then integrated with data from the SIMPA (*Sistema Integrado para la Modelación del proceso Precipitación Aportación*) model to obtain the final 1940–2018 series (CEDEX 2020). In case of discrepant values, we correct the series with more specific data records from reservoirs and monitoring stations. Following the guidance provided by MAGRAMA (2017) and observed by the Douro River Basin Authority, discharge series are subsequently adjusted to account for climate change impacts under a RCP4.5 scenario (an 11% reduction of the discharge in the whole basin).

Using discharge data inputs, AQUATOOL assesses the impacts on water availability in water bodies "through a longitudinal simulation that offers spatial information on surface and subsurface water flows on a monthly basis" (Pérez-Blanco *et al.* 2021). Then, a network optimization algorithm simulates the water allocations for each use following the management rules of the river basin authority. This algorithm follows a multi-objective optimization that includes (i) satisfying the environmental flows targets, (ii) minimizing water deficits among users, (iii) the maintenance of the minimum water stock in reservoirs, and (iv) achieving hydropower generation targets.

In this paper, we used the latest version of AQUATOOL which was set up and calibrated by the Douro River Basin Authority to produce the 2021 Douro River Basin Management Plan (DRBA 2020).

3.2. *Economic module*

Positive MPMs are widely used calibrated microeconomic models that represent agents' behavior and their responses to key *stimuli*. Agents can be defined at different scales, from farmers to entire regions, and in this case are the AWDUs — which are the agricultural water demand unit adopted in AQUATOOL. Agents in MPMs decide on the crop portfolio, timing, water application, and key investments aiming to maximize an objective or utility function subject to a series of constraints (e.g., land availability, water caps). This complex decision is usually reduced to a decision on the crop portfolio, where each alternative portfolio yields a unique combination of crops, timing, investments, and water application (Graveline 2016). The general formulation of the problem is as follows:

$$\text{Max}\, U(\boldsymbol{X}) = f(z_1(\boldsymbol{X}), \ldots, z_m(\boldsymbol{X})) \tag{1}$$

Subject to:

$$x_i \geq 0, \tag{2}$$

$$\sum_{i=1}^{n} x_i = 1 \tag{3}$$

$$\boldsymbol{X} \in F \tag{4}$$

$$\boldsymbol{X} \in \mathbb{R}^n, \tag{5}$$

$$z_1(\boldsymbol{X}), \ldots, z_m(\boldsymbol{X}) = \boldsymbol{Z}(\boldsymbol{X}) \in \mathbb{R}^m. \tag{6}$$

Agents in the model decide on the *crop portfolio* $\boldsymbol{X} \in \mathbb{R}^n$, a vector that represents the share of land allotted to each of the n individual crops available $x_i (i = 1, \ldots, n)$, to maximize their utility within the domain F. Utility $U(\boldsymbol{X})$ is a function of m attributes $z_1(\boldsymbol{X}), \ldots, z_m(\boldsymbol{X})$, including e.g., profit, risk avoidance, and management complexity avoidance. The attributes are defined so that "more-is-better", i.e., if the provision of one attribute increases and the provision of the remaining attributes remains constant, then total utility increases. Accordingly, "less-is-better" attributes (e.g., risk) are transformed into "more-is-better" attributes (e.g., risk avoidance). Annex I presents the mathematical formulation of the attributes employed in this ensemble, the data inputs, and a mathematical formulation of the constraints that conform the domain.

We assess uncertainty in human behavior and responses using a multi-model ensemble of MPMs consisting of two Positive Mathematical Programming (PMP) models (Howitt 1995; Júdez *et al.* 2002), one Weighted Goals Programming (WGP) model (Sumpsi *et al.* 1997) and a Positive Multi-Attribute Utility Programming (PMAUP) model (Gutiérrez-Martín and Gómez 2011). Differences across the MPMs considered in the ensemble stem from the *form* and *calibration* of the utility function (Sapino *et al.* 2020). Regarding the *form*, the utility functions used by MPMs can adopt a non-linear Cobb-Douglas (PMAUP), non-linear quadratic (PMP), and linear additive (WGP) form. The utility functions can also be single- (in the case of PMP) or multi-attribute (in the case of WGP and PMAUP) (Graveline 2016; Pérez-Blanco and Sapino 2022). Single-attribute utility functions consider only expected profit as the relevant attribute; whereas the multi-attribute utility functions also include risk avoidance and management complexity aversion. Another relevant difference between each MPMs regards the *calibration* method, which is discussed in Annex II. The calculation of the calibration residuals is presented in Annex III. Finally, the calibration results for the agents in the Órbigo Catchment using the four models above are presented in Annex IV.

3.3. *Coupling*

The coupling between the hydrologic and economic modules (Figure 2) adopts a time-variant and sequential fashion in six steps and using two protocols, as follows:

In Step 1, a discharge shock that accounts for future climate change impacts forces AQUATOOL.

In Step 2, AQUATOOL assesses water availability and runs a simulation to determine water allocation using its network optimization algorithm.

In Step 3, the *first protocol* is activated, and information on the water allocation for every agent/AWDU simulated in AQUATOOL is conveyed to the economic module.

In Step 4, the MPMs within the economic module simulate the crop portfolio responses of agents/AWDUs to the water allotments determined in AQUATOOL. MPMs produce key information on hired labor, profit, and effective water use per AWDU, from which we can estimate income changes through the GVA.

In Step 5, the *second protocol* is activated, and information on effective water use by AWDUs is conveyed to the hydrologic module. This second protocol is relevant because effective water use may be lower than water allocation to AWDUs, particularly during relatively water-abundant years, and this has implications for water flows and stocks in the water system. Note that while AQUATOOL operates at a monthly timescale, the MPMs in the economic module operate

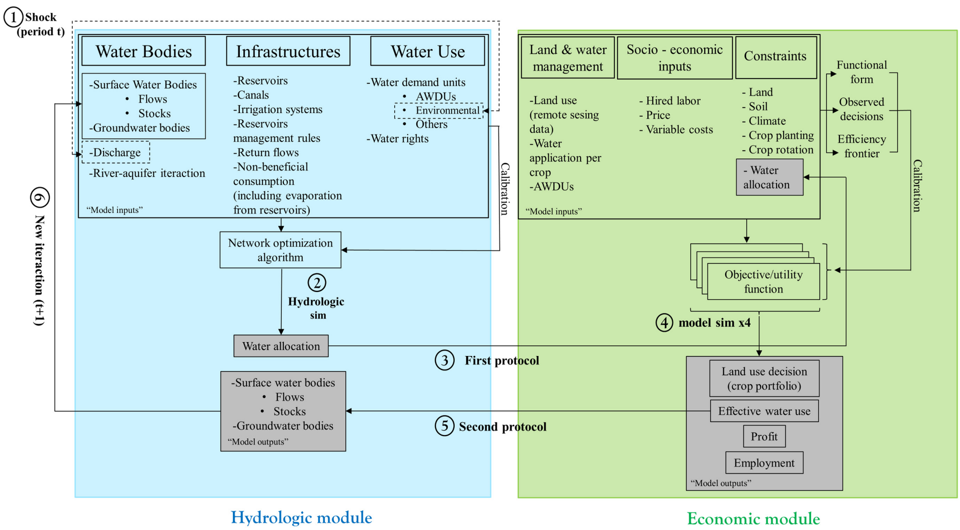

Figure 2. Flowchart of the Coupled Human and Water Systems

Source: Adapted from Pérez-Blanco *et al.* (2021).

on a yearly basis. Accordingly, the information on water use produced by the economic module is distributed over the months of the irrigation season and then imported to AQUATOOL.

In Step 6, AQUATOOL combines the information on effective water use by AWDUs and discharge data over the following months to reassess the status of the water system and estimate water availability and allocation in $t + 1$.

This process is repeated in sequence for a predefined time, which follow best practices in Spanish basins is set to 38 hydrological years[3] (1980/1981 to 2017/2018) (MAGRAMA 2017).

3.4. *Setup of the resource costs simulation*

Resource costs are obtained by comparing the results obtained in two alternative sets of simulations using the coupled hydroeconomic model presented above:

In the **first simulation (SIM00)**, we use the original model setup presented in the previous sub-section to estimate the total GVA per AWDU;

in the **second set of simulations (SIMr, where $\boldsymbol{r = 01, 02, \ldots, 12}$)** we run 12 independent simulations identical to that in 1), but in each of them we exclude one of the AWDUs (in SIM01 we exclude AWDU01, in SIM02 we exclude AWDU02, and so on). Again, we calculate the total GVA per AWDU; finally, the resource cost in EUR/m^3 for a given AWDU r is obtained as the difference between the total GVA in SIM r and the total GVA in SIM00 excluding AWDU r, divided by the total water allotted to AWDU r.

Note that resource costs can be obtained on a yearly basis (where resource costs will be higher during droughts, and low or even zero during water-abundant years) or as an annuity over the entire simulation period (arithmetic mean). In compliance with existing water charging mechanisms for the recovery of financial costs in Spanish river basins, the Douro River Basin Authority declared a preference for the latter mechanism (annuity), which we used to calculate resource costs in the Órbigo Catchment.

On the other hand, the annuity can be recovered through a user-specific charge or by applying the same annuity across all users. In compliance with existing water charging mechanisms, to prevent regressive impacts (see next section), and due to the technical challenge of measuring and monitoring resource costs for each user, the Douro River Basin Authority declared a preference for the latter mechanism.

[3]Discharge data were available for the entire time-series 1940–2018 (78 hydrological years), but in Spain hydrological studies normally adopt shorter discharge series starting from 1980. The short series should represent more accurately the current water regime, significantly modified by human activities (DRBA 2020; MAGRAMA 2017).

The Douro River Basin authority also manifested a preference for volumetric prices. Accordingly, we simulated the economic and environmental impacts of implementing resource costs recovery through a volumetric and homogeneous annuity payment across irrigators.

4. Results and Discussion

Figure 3 displays the resource costs per AWDU for the four MPMs explored in the economic ensemble. Average resource costs range between 0.016 EUR/m^3 and 0.029 EUR/m^3 (i.e., between 34% and 62% of the current water charge in the Órbigo Catchment of 0.047 EUR/m^3). The range of resource costs is significantly widened by the WGP model, which consistently estimates higher values. This is because the WGP model linearizes the objective function for its calibration, and as a result is more likely to lead to corner solutions: the agent specializes in a crop and once a tipping point is reached, this crop is abruptly replaced by another one (typically with a lower return). These abrupt changes in the crop portfolio translate into more pronounced changes in water use, profit, labor, and GVA in the WGP, which explains the differences in resource costs estimates when compared to other MPMs with nonlinear utility functions.

The two AWDUs with the lowest surface and water allocation, AWDU03 and AWDU08, are those showing the highest resource costs. This may come as a surprise if we consider that these two AWDUs are those showing the higher

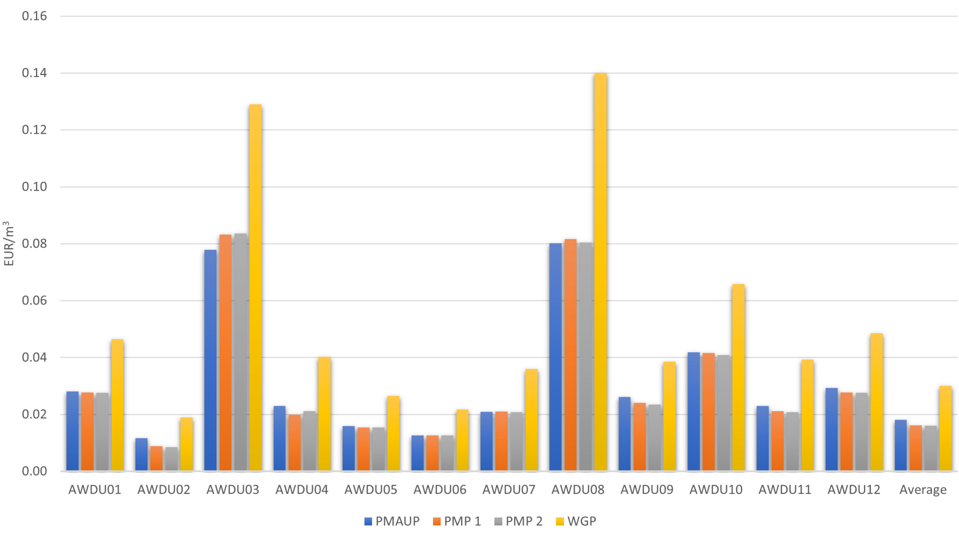

Figure 3. Resource Cost for AWDUs

Source: Own elaboration.

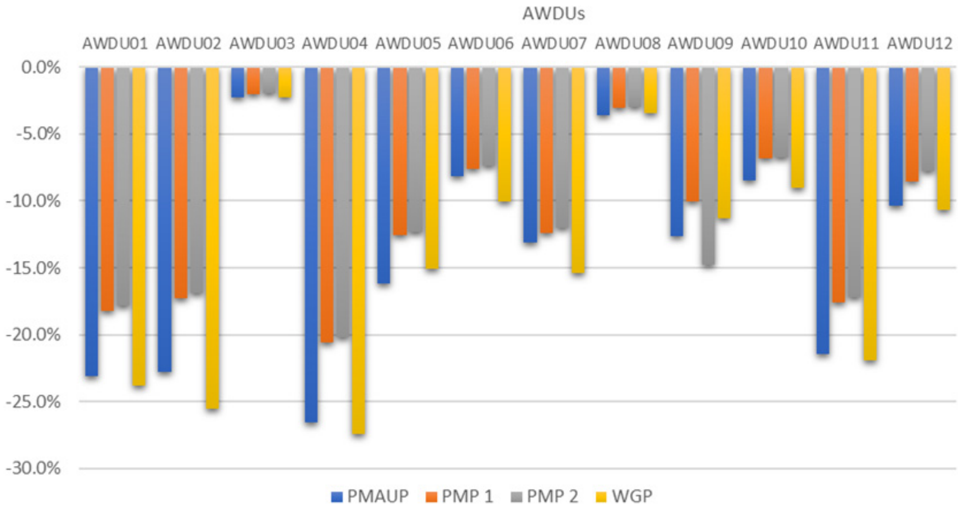

Figure 4. GVA Reduction Per AWDU Following Resource Costs Recovery

Source: Own elaboration.

GVA/m^3 of water use. This apparently paradoxical result is explained due to the limited water use of these AWDUs: when farmers elsewhere receive water resources from these two AWDUs, they initially use these resources to irrigate those crops with a higher return and then move progressively to crops with a lower return. Thus, the resource cost per unit of water used (EUR/m^3) for a given AWDU is likely to be higher the lower its water use, and will decrease as water use increases, creating a regressive effect that can penalize less intensive water users. Preventing this undesirable distributive impact calls for a volumetric and *homogeneous* (in EUR/m^3) charge across all AWDUs — the type of charge adopted here.

Figure 4 shows the GVA reduction following the recovery of resource costs through a homogeneous and volumetric annuity payment across agricultural water users. Unsurprisingly, the WGP model that estimates (and charges) a higher resource cost also yields the larger GVA losses in more than half of the AWDUs, albeit in this case differences are less marked than the results in Figure 3. PMP models predict lower GVA losses for most of the AWDUs as compared to the two multi-attribute models (PMAUP and WGP). The difference in the impact between PMP 1 and PMP 2 is explained by the different calibration mechanisms and the dual value used by the two models (see Annex II): PMP 1 dual value is associated with the land constraint for every crop, while PMP 2 dual value uses the average value of land rent price (typically lower). The GVA impacts predicted by the PMAUP model typically range between those predicted by the PMP models and the WGP. Among AWDUs, AWDU03 and AWDU08 show the lowest GVA losses (below 5%). Recall that these are the AWDUs with the most profitable crops and the highest ability to pay for water. In our simulations for AWDU03 and AWDU08, the resource costs estimated (all MPMs) are not high enough to induce significant changes in the crop portfolio, and only marginally erode profit and GVA. Importantly, in all the remaining AWDUs GVA losses are at least 2× larger than in AWDU03 and AWDU08 and can exceed 25% of baseline GVA. This nonlinear outcome is in line with the findings of Adamson *et al.* (2017) and Olmstead and Stavins (2007). Importantly, when a water charge has a "disproportionate" cost, for example through a significant reduction in profit and GVA that exacerbates inequality or triggers farm exit, the EU WFD allows an exception to the principle of full cost recovery (European Commission 2000). Alternatively, complementary compensation mechanisms such as decoupled subsidies may be adopted to prevent inequitable redistributions of income/farm exit while ensuring more efficient water allocations.

Figure 5 shows water savings (in %) following the recovery of resource costs through a homogeneous and volumetric annuity payment. The PMAUP and WGP models predict the highest water savings for most of the AWDUs, which are largely achieved through the substitution of irrigated maize with rainfed cereals. Since the dual values used by PMP models in their calibration tend to penalize changes in the crop portfolio, PMP models report lower water savings as compared to the PMAUP and WGP models. The WGP model predicts zero water savings in

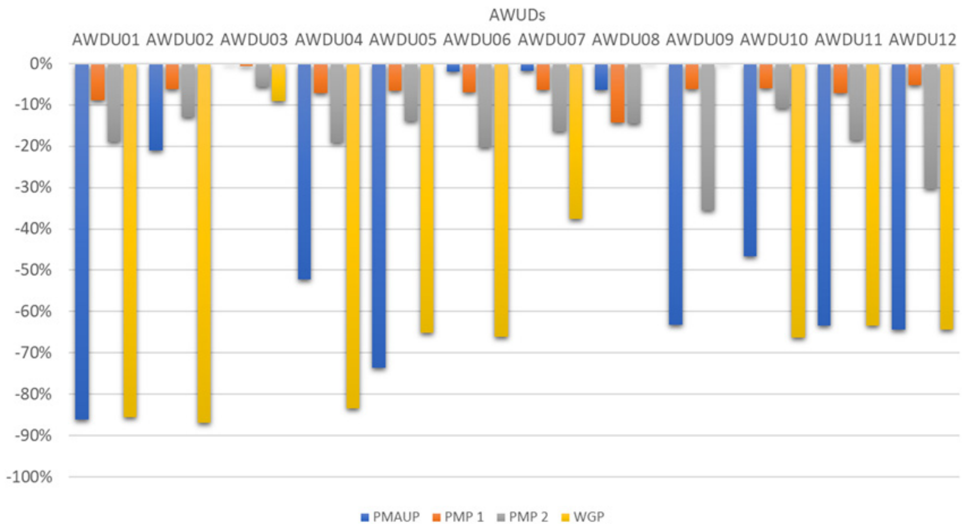

Figure 5. Water Savings Per AWDU Following Resource Costs Recovery
Source: Own elaboration.

AWDU08 and AWDU09, while PMAUP predicts a very small saving in AWDU03 (only 2.3 m^3/ha or -0.04%). Zero water savings indicates a corner solution, a characteristic behavior of linear models: the crop portfolio remains unchanged until a tipping point is reached and all the area assigned to a specific crop is suddenly replaced. In these two AWDUs, the tipping point is not reached for any specific crop, which results in zero water savings. Unsurprisingly, the AWDUs with the highest expected income per hectare (AWDU03 and AWDU08) show the lowest water savings due to their relatively more inelastic demand. Overall, the recovery of resource costs yields significant water savings over the entire Órbigo Catchment at a relatively lower (yet non-trivial) economic cost: while GVA over the entire Órbigo Catchment is reduced by 17.56% following the recovery of resource costs, water use is reduced by 29.45% (ensemble average).

Finally, Figure 6 shows the net revenue raised through resource cost recovery. Tax revenues are predicted to increase in all AWDUs for the PMP models following full resource costs recovery, while the PMAUP and the WGP predict both higher and lower tax revenues depending on the AWDU. Reductions in the tax revenue occur when the negative effect on tax revenue from reduced water use offsets the positive effect from higher prices. Over the entire Órbigo Catchment, implementing full cost recovery is expected to change the tax revenue between -2.3 million EUR/year (-30%) and 5 million EUR/year ($+45.6\%$), depending on the model, with the two multi-attribute models (PMAUP and WGP) predicting a reduction in tax revenue and the two PMP models predicting an increase.

The results above substantiate the role of pricing as a behavioral incentive with the potential to align individual decisions with key societal objectives such as

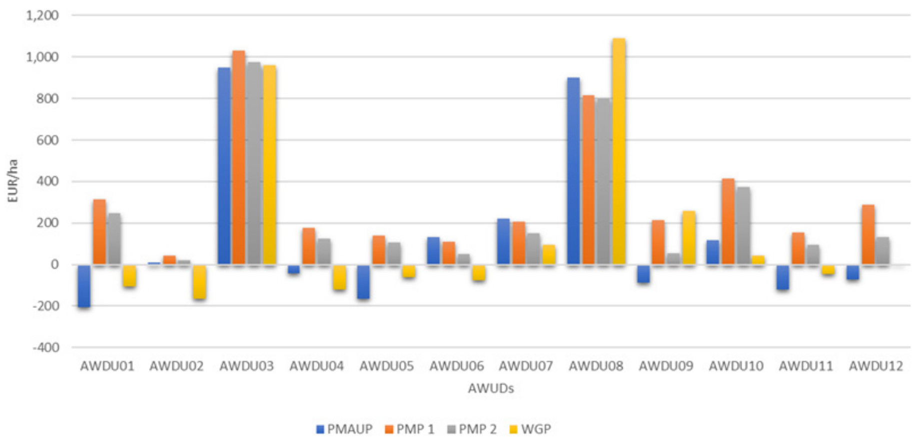

Figure 6. Changes in Tax Revenue Per AWDU Following Resource Costs Recovery
Source: Own elaboration.

environmental sustainability. WATECO defines resource costs as the "cost of forgone opportunities that other users suffer due to the depletion of the resource beyond its natural rate of recharge or recovery" (European Commission 2003). Giving back these opportunities to users requires, in the first place, preventing the depletion of the resource, which in the case of the overallocated Órbigo Catchment involves a reduction in water use that restores the balance between demand and an increasingly volatile and overall decreasing supply. This will enhance environmental performance (higher savings) and reduce economic performance (through GVA losses), with tax revenue potentially experiencing non-trivial changes as well (between −30% and 45.6%).

Noteworthy, and as noted above, "disproportionate" costs may trigger exceptions of the cost recovery principle (European Commission 2000). In addition, recent rulings from the European Court of Justice suggest that the precise instrument to achieve the good ecological status of water bodies is at the discretion of Member States (Jääskinen 2014), who may find more adequate alternatives to manage demand than charges, such as buyback, caps, or others (Rey *et al.* 2019). At the very least, if resource costs are to be recovered in the Órbigo Catchment, it appears sensible to accompany the implementation of higher charges with complementary measures (e.g., decoupled subsidies) that address the relevant and asymmetric income losses across AWDUs. Critically, a decrease in tax revenue implies that returns from resource cost recovery cannot be used to compensate those users that suffer the most from additional charging (recall that GVA losses per AWDU range from 2% to 27%). Mitigating these imbalanced income losses would require additional resources that will result in further budgetary pressures.

5. Conclusions

This paper presents a methodology to estimate the resource costs of agricultural water use and assess the environmental and economic impacts of implementing their recovery. To this end, we develop a socio-hydrology-inspired, dynamic, protocol-based, modular hydroeconomic model that interconnects human and water systems through two-way feedback protocols. The hydrologic module is populated with the AQUATOOL model, the DSS used by Spanish river basin authorities (Andreu *et al.* 1991); while the economic module is populated with an ensemble of four MPMs that captures human agency and responses (Sapino *et al.* 2020). Results for the Órbigo Catchment in NW Spain show significant resource costs (between 0.016 EUR/m^3 and 0.029 EUR/m^3 depending on the model, i.e., a 34–62% increase in existing charges of 0.047 EUR/m^3) that have a significant impact on the GVA (2–27% reduction) and the environment (water savings range between 6% and 69%), while tax revenue experiences only marginal changes (between −2.3 million EUR/year and 5 million EUR/year).

There are several ways in which the proposed modeling framework and research could be improved. *First*, additional ecological (e.g., an agronomic module) and human systems (e.g., a macroeconomic module) can be incorporated into the modeling framework to account for relevant feedbacks and cascading effects (e.g., changes in commodity and input prices in a macroeconomic context, which would in turn condition irrigators' decisions) (Parrado *et al.* 2020). The new modules could be populated with several models each, leading to a grand ensemble (i.e., an ensemble of ensembles) that more thoroughly samples risk and uncertainty and better informs robust decision-making. Including macroeconomic aspects into our model would allow us to reveal backward and forward linkages across sectors and help us track the roots of environmental stressors and their economic drivers (i.e., those using the production obtained through agricultural water use, such as tourism or food industries, but also society benefiting from enhanced food security or the repopulation and conservation of rural landscapes), as well as those benefiting from the charge, to better understand costs and benefits. It may also be that the benefits from enhanced water availability largely exceed the costs experienced by farmers facing cost recovery, or that collecting from those who benefit has an impact on the overall surplus well below than charging users (e.g., because charges are distributed across a much larger group and marginal utility losses will be lower). In this context, there exists "a possible system of compensations and collections such that everyone would be better off than before" (Hotelling 1938). But without information on who benefits and who pays, such adjustments would not in fact be made. The debate between social and private beneficiaries of water use (either direct or indirect) is not trivial in this context and is a solid justification as to why resource and other water use costs are not fully recovered. Tracking and identifying the drivers of water use can support the design of a more comprehensive policy mix that includes, for instance, cross-subsidization mechanisms to distribute the economic repercussions of cost recovery more efficiently and equally.

Second, the MPMs used could be improved to explore the relevance of additional attributes in multi-attribute models, and to allow for alternative adaptation strategies, notably adaptation at the intensive margin through deficit irrigation; albeit a key constraint to this is data availability and the limited number of models tackling this aspect — which would reduce the number of ensemble elements (Koundouri 2004; Loch *et al.* 2020a; Sapino *et al.* 2022). Additionally, attributes could be revisited to include new variables such as fixed costs, to assess farm exit scenarios in the longer run.

Third, the ensemble of MPMs can be expanded by considering additional models, and an ensemble of hydrologic models could be incorporated. The latter was explored alongside stakeholders, and finally discarded for this exercise due to two key factors: (i) an *ad-hoc* network optimization algorithm that accurately represents water reallocations by decision-makers should be programmed in the new models, which is resource and time consuming; and more importantly, (ii) the actionable nature of our modeling framework resides, to a large extent, on the use of AQUATOOL, which is the DSS used by decision-makers in Spanish river basins. The use of additional hydrologic models would demand more time and resources that are presently not available in the river basin authority.

Fourth, the number of scenarios considered, particularly climate change scenarios, could be expanded to further sample uncertainty and underpin robust decision-making. This was challenging to communicate to stakeholders, which are presently conducting river basin planning considering one climate change scenario (RCP4.5) following guidance from the relevant Spanish ministry (MAGRAMA 2017).

Fifth, while our paper estimates resource costs and assesses the economic and environmental impacts of their recovery assuming all else is kept equal, complementary policies such as decoupled subsidies may be necessary to address some of the (sometimes disproportionate) negative impacts of water charges. In line with the thinking of Tinbergen (1952), to address several policy objectives an equal number of policy instruments are necessary — one per objective. Moreover, in line with Mundell (1962) Assignment principle, each instrument should be used to target the objective to which it is best suited. Following the Tinbergen and Assignment principles, if charges target cost recovery, they may underperform in terms of water savings, which may require an additional instrument to ensure sustainable use, such as caps or buyback.

Sixth, the AQUATOOL model could be downscaled from a hydrologic (it now works at sub-catchment level) and water user perspective (e.g., using irrigation communities instead of AWDUs, which would provide higher granularity) (Fiseha *et al.* 2014).

Finally, aside from the modeling aspects that this paper targets, it is of critical importance to explore how the engagement of stakeholders can be further strengthened beyond the model co-development efforts conducted in this research (see Sec. 3). Indeed, co-development is part of a more comprehensive co-creation process that involves several science-policy interactions through (i) co-design of climate and socioeconomic scenarios (closely connected to the fourth point above) and strategies (see fifth point above), (ii) co-development of the model(s) to be used, (iii) co-evaluation of adaptation outcomes to identify strengths and vulnerabilities of alternative strategies, (iv) co-identification of the robust strategy to achieve the selected goal(s), and (v) co-implementation of the policy, including monitoring and adoption of corrective actions where needed (Prahalad and Ramaswamy 2000 2004). Co-creation process is intricate and typically requires the development of cohesive and lasting knowledge networks (Eikebrokk *et al.* 2021), as well as engagement mechanisms such as serious gaming (Solinska-Nowak *et al.* 2018). While the development of knowledge networks or serious gaming is out of the scope of our research, future co-creation processes can greatly benefit from the use of actionable models that are trusted by the key stakeholders involved in decision-making — an input that this paper delivers.

Acknowledgments

The research leading to these results has been developed with the support of the PRIMA Foundation's TALANOA-WATER Project (Talanoa Water Dialogue for Transformational Adaptation to Water Scarcity under Climate Change), of the Spanish Ministry for the Ecological Transition and Demographic Challenge through Fundación Biodiversidad (ATACC Project - Adaptación Transformativa al Cambio Climático en el Regadío), and of the Ministry of Science and Innovation's IRENE Project (Integrated socioeconomic and environmental modelling using remote sensing data for the management of unauthorized water abstractions).

Credit Author Statement

Francesco Sapino: Conceptualization, Formal analysis, Methodology, Data curation, Investigation, Software, Validation, Visualization, Writing — original draft; **C. Dionisio Pérez-Blanco**: Conceptualization, Formal analysis, Funding acquisition, Investigation, Resources, Project administration, Supervision, Writing — review & editing; **Pablo Saiz-Santiago**: Methodology, Software, Validation, Visualization.

Supplementary Material

The Supplemental Materials are available at: https://www.worldscientific.com/doi/suppl/10.1142/S2382624X22400124.

References

Acreman, MC and AJD Ferguson (2010). Environmental flows and the European Water Framework Directive. *Freshwater Biology*, 55, 32–48. https://doi.org/10.1111/j.1365-2427.2009.02181.x.

Adamson, D and A Loch (2021). Incorporating uncertainty in the economic evaluation of capital investments for water use efficiency improvements. *Land Economics*, 100119. https://doi.org/10.3368/wple.97.3.100119-0143R.

Adamson, D, A Loch and K Schwabe (2017). Adaptation responses to increasing drought frequency. *Australian Journal of Agricultural and Resource Economics*, 61, 385–403. https://doi.org/10.1111/1467-8489.12214.

Alamanos, A, D Latinopoulos and N Mylopoulos (2020). A methodological framework for an easy and reliable estimation of the full cost of irrigation water. *Water and Environment Journal*, 34, 529–539. https://doi.org/10.1111/wej.12556.

Andreu, J, J Capilla and E Sanchis (1991). AQUATOOL: A computer-assisted support system for water resources research management including conjunctive use, In *Decision Support Systems*, DP Loucks and JR da Costa (eds.), NATO ASI Series, pp. 333–355. Springer, Berlin, Heidelberg.

Barraqué, B (2020). Full cost recovery of water services and the 3T's of OECD. *Utilities Policy*, 62, 100981. https://doi.org/10.1016/j.jup.2019.100981.

Berbel, J and A Expósito (2020). The theory and practice of water pricing and cost recovery in the water framework directive. *Water Alternatives*, 13(3), 659–673.

Bhaduri, A, CD Pérez-Blanco, D Rey, MS Iftekhar, A Kaushik, A Escriva-Bou, J Calatrava, D Adamson, S Palomo-Hierro, KW Jones, H Asbjornsen, MA Altamirano, E Lopez-Gunn, M Polyakov, M Motlagh and M Bekchanov (2021). Economics of water security, In *Handbook of Water Resources Management: Discourses, Concepts and Examples, Handbook of Water Resouces*. Berlin (Germany): Springer International Publishing. https://doi.org/10.1007/978-3-030-60147-8_10.

Bogardi, JJ, T Tingsanchali, KDW Nandalal, J Gupta, L Salamé, RRP Nooijen, AG van Kolechkina, N Kumar and A Bhaduri (eds.) (2021). *Handbook of Water Resources Management: Discourses, Concepts and Examples.* Springer International Publishing, Berlin (Germany). https://doi.org/10.1007/978-3-030-60147-8.

CEDEX (2020). SIMPA. Sistema Integrado para la Modelación del proceso Precipitación Aportación [WWW Document]. URL http://ceh-flumen64.cedex.es/Hidrologia/pub/proyectos/simpa.htm [19 January 2021].

Chaikaew, P, AW Hodges and S Grunwald (2017). Estimating the value of ecosystem services in a mixed-use watershed: A choice experiment approach. *Ecosystem Services*, 23, 228–237. https://doi.org/10.1016/j.ecoser.2016.12.015.

Cornish, G and C Perry (2003). Water charging in irrigated agriculture; lessons from the field (Report No. OD150). HR Wallingford Ltd., Wallingford (UK).

Delacamará, G, T Dworak, CM Gomeź, M Lago, A Maziotis, J Rouillard and P Strosser (2014). Design and development of Economic Policy Instruments in European water policy (Report No. D5.2). 7th Framework Contract Project EPI-Water Project (GA 265213).

Di Baldassarre, G, F Martinez, Z Kalantari and A Viglione (2017). Drought and flood in the Anthropocene: Feedback mechanisms in reservoir operation. *Earth System Dynamics*, 8, 225–233. https://doi.org/10.5194/esd-8-225-2017.

Di Baldassarre, G, M Sivapalan, M Rusca, C Cudennec, M Garcia, H Kreibich, M Konar, E Mondino, J Mård, S Pande, MR Sanderson, F Tian, A Viglione, J Wei, Y Wei, DJ Yu, V Srinivasan and G Blöschl (2019). Sociohydrology: Scientific Challenges in

Addressing the Sustainable Development Goals. *Water Resources Research*, 55, 6327–6355. https://doi.org/10.1029/2018WR023901.

Dinar, A, V Pochat and J Albiac (eds.) (2015). *Water Pricing Experiences and Innovations, Global Issues in Water Policy*. Springer International Publishing. https://doi.org/10.1007/978-3-319-16465-6.

Dinar, A and A Subramanian (1997). *Water Pricing Experiences: An International Perspective* (Report No. WTP386). The World Bank.

DRBA (2016). Plan Hidrológico de la Cuenca del Duero 2015-2021 (River Basin Management Plan). Duero River Basin Authority, Valladolid (Spain).

DRBA (2017). Plan Especial de Sequía (Report). Douro River Basin Authority.

DRBA (2020). Esquema de temas importantes en materia de gestión de las aguas del Plan Hidrológico 2022-2027 (River Basin Management Plan). Duero River Basin Authority, Valladolid (Spain).

DRBA (2021). Plan hidrológico 2022–2027 (Propuesta) — Confederación Hidrográfica del Duero [WWW Document]. URL https://www.chduero.es/web/guest/plan-hidrologico-2022 [10 May 2021].

Driscoll, CT, KF Lambert and KC Weathers (2011). Integrating Science and Policy: A Case Study of the Hubbard Brook Research Foundation Science Links Program. *BioScience*, 61, 791–801. https://doi.org/10.1525/bio.2011.61.10.9.

EEA (2013). Assessment of cost recovery through water pricing (Report No. 16/2013). European Environment Agency, Copenhagen (Denmark).

EEA (2016). Water exploitation index [WWW Document]. Water exploitation index. URL http://www.eea.europa.eu/data-and-maps/indicators/water-exploitation-index [30 March 2016].

EEA (2020). European environment — state and outlook (Outlook No. 2020). European Environment Agency, Copenhagen (Denmark).

Eikebrokk, TR, NF Garmann-Johnsen and DH Olsen (2021). Co-creation in networks of SMEs: A conceptual model of the co-creation process. Vol. 181, pp. 360–366. https://doi.org/10.1016/j.procs.2021.01.179.

European Commission (2000). Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. (OJ L 327, 22.12.2000, p. 1).

European Commission (2003). Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Economics and the environment: the implementation challenge of the water framework directive. OPOCE, Luxembourg.

FAO (2021). FaoStat [WWW Document]. Food and Agriculture Organization of the United Nations. URL http://faostat.fao.org/[10 February 2021].

Fiseha, BM, SG Setegn, AM Melesse, E Volpi and A Fiori (2014). Impact of climate change on the hydrology of upper tiber river basin using bias corrected regional climate model. *Water Resources Management*, 28, 1327–1343. https://doi.org/10.1007/s11269-014-0546-x.

García de Jalón, S, M González del Tánago, C Alonso and D García de Jalón (2017). The environmental costs of water flow regulation: An innovative approach based on the ‘Polluter Pays’ Principle. *Water Resources Management*, 31, 2809–2822. https://doi.org/10.1007/s11269-017-1663-0.

Graveline, N (2016). Economic calibrated models for water allocation in agricultural production: A review. *Environmental Modelling & Software*, 81, 12–25. https://doi.org/10.1016/j.envsoft.2016.03.004.

Graveline, N and P Mérel (2014). Intensive and extensive margin adjustments to water scarcity in France's Cereal Belt. *European Review of Agricultural Economics*, 41, 707–743. https://doi.org/10.1093/erae/jbt039.

Gutiérrez-Martín, C and CM Gómez (2011). Assessing irrigation efficiency improvements by using a preference revelation model. *Spanish Journal of Agricultural Research*, 9, 1009–1020. https://doi.org/10.5424/sjar/20110904-514-10.

Heinz, I, M Pulido-Velazquez, JR Lund and J Andreu (2007). Hydro-economic modeling in river basin management: Implications and applications for the European Water Framework Directive. *Water Resources Management*, 21, 1103–1125. https://doi.org/10.1007/s11269-006-9101-8.

Hotelling, H (1938). The general welfare in relation to problems of taxation and of railway and utility rates. *Econometrica*, 6, 242–269. https://doi.org/10.2307/1907054.

Howitt, RE (1995). A calibration method for agricultural economic production models. *Journal of Agricultural Economics*, 46, 147–159. https://doi.org/10.1111/j.1477-9552.1995.tb00762.x.

IPCC (2021). IPCC Sixth Assessment Report (AR6) (No. WGII). Intergovernmental Panel on Climate Change, Geneva (Switzerland).

Jääskinen, N (2014). Case C-525/12 European Commission v Federal Republic of Germany.

Júdez, L, JM de Miguel, J Mas and R Bru (2002). Modeling crop regional production using positive mathematical programming. *Mathematical and Computer Modelling*, 35, 77–86. https://doi.org/10.1016/S0895-7177(01)00150-9.

Koundouri, P (2004). Current issues in the economics of groundwater resource management. *Journal of Economic Surveys*, 18, 703–740. https://doi.org/10.1111/j.1467-6419.2004.00234.x.

Lerma, N, J Paredes, A Solera and J Andreu (2017). Herramienta EvalHid para la evaluación de recursos hídricos (Manual No. 1.1). Polytechnic University of Valencia, Valencia, Spain.

Loch, A, D Adamson and C Auricht (2020a). (g)etting to the point: The problem with water risk and uncertainty. *Water Resources and Economics*, 32, 100154. https://doi.org/10.1016/j.wre.2019.100154.

Loch, A, D Adamson and NP Dumbrell (2020b). The Fifth Stage in Water Management: Policy Lessons for Water Governance. *Water Resources Research*, 56, e2019WR026714. https://doi.org/10.1029/2019WR026714.

MAGRAMA (2017). Evaluación del impacto del cambio climático en los recursos hídricos y sequías en España (Report). Ministerio de Agricultura y Pesca, Alimentación y Medio Ambiente, Madrid (Spain) [29 April 2019].

Martin-Ortega, J, G Giannoccaro and J Berbel (2011). Environmental and Resource Costs Under Water Scarcity Conditions: An estimation in the context of the European Water Framework Directive. *Water Resources Management*, 25, 1615–1633. https://doi.org/10.1007/s11269-010-9764-z.

Molden, D, T Oweis, P Steduto, P Bindraban, MA Hanjra and J Kijne (2010). Improving agricultural water productivity: Between optimism and caution. *Agricultural Water Management*, 97, 528–535. https://doi.org/10.1016/j.agwat.2009.03.023.

Mundell, R (1962). The appropriate use of monetary and fiscal policy for internal and external stability. *IMF Staff Papers*, 9, 70–79.

Nkiaka, E and JC Lovett (2019). Strengthening the science-policy interface for climate adaptation: Stakeholder perceptions in Cameroon. *Regional Environmental Change*, 19, 1047–1057. https://doi.org/10.1007/s10113-018-1441-4.

OECD (2015). Water Resources Allocation: Sharing risks and opportunities, In *OECD Studies on Water*. Paris: OECD Publishing. https://doi.org/10.1787/9789264229631-en.

OECD (2017). *Water Charges in Brazil: The Ways Forward*, *OECD Studies on Water*. OECD Publishing, Paris. https://dx.doi.org/10.1787/9789264285712-en.

Olmstead, S and R Stavins (2007). *Managing Water Demand – Price vs. Non-Price Conservation Programs (Report)*. Boston, MA: Pioneer Institute for Public Policy Research.

Parrado, R, CD Pérez-Blanco, C Gutiérrez-Martín and L Gil-García (2020). To charge or to cap in agricultural water management. Insights from modular iterative modeling for the assessment of bilateral micro-macro-economic feedback links. *Science of the Total Environment*, 742, 140526. https://doi.org/10.1016/j.scitotenv.2020.140526.

Pérez-Blanco, CD, A Hrast-Essenfelder and C Perry (2020). Irrigation technology and water conservation: A review of the theory and evidence. *Review of Environmental Economics and Policy*, 14, 216–239. https://doi.org/10.1093/reep/reaa004.

Pérez-Blanco, CD, L Gil-García and P Saiz-Santiago (2021). An actionable hydro-economic decision support system for the assessment of water reallocations in irrigated agriculture. A study of minimum environmental flows in the Douro River Basin, Spain. *Journal of Environmental Management*, 298, 113432. https://doi.org/10.1016/j.jenvman.2021.113432.

Pérez-Blanco, CD and F Sapino (2022). Economic sustainability of irrigation-dependent ecosystem services under growing water scarcity. Insights from the reno river in Italy. *Water Resources Research*, 58, e2021WR030478. https://doi.org/10.1029/2021WR030478.

Prahalad, CK and V Ramaswamy (2000). Co-opting customer competence. *Harvard Business Review*, 78, 79–90.

Prahalad, CK and V Ramaswamy (2004). Co-creation experiences: The next practice in value creation. *Journal of Interactive Marketing*, 18, 5–14. https://doi.org/10.1002/dir.20015.

Pulido-Velazquez, M, E Alvarez-Mendiola and J Andreu (2013). Design of efficient water pricing policies integrating basinwide resource opportunity costs. *Journal of Water Resources Planning and Management*, 139, 583–592. https://doi.org/10.1061/(ASCE)WR.1943-5452.0000262.

Pulido-Velázquez, M, J Andreu and A Sahuquillo (2006). Economic optimization of conjunctive use of surface water and groundwater at the basin scale. *Journal of Water Resources Planning and Management*, 132, 454–467. https://doi.org/10.1061/(ASCE)0733-9496(2006)132:6(454).

PUV (2020). Manuals — AquaTool [WWW Document]. Manuals — AquaTool. URL https://aquatool.webs.upv.es/aqt/en/manuals/ [19 January 2021].

Puy, A, R Sheikholeslami, HV Gupta, JW Hall, B Lankford, S Lo Piano, J Meier, F Pappenberger, A Porporato, G Vico and A Saltelli (2022). The delusive accuracy of global irrigation water withdrawal estimates. *Nature Communications*, 13, 3183. https://doi.org/10.1038/s41467-022-30731-8.

Randall, A (1981). Property entitlements and pricing policies for a maturing water economy. *Australian Journal of Agricultural and Resource Economics*, 25, 195–220. https://doi.org/10.1111/j.1467-8489.1981.tb00398.x.

Rey, D, CD Pérez-Blanco, A Escriva-Bou, C Girard and TLE Veldkamp (2019). Role of economic instruments in water allocation reform: Lessons from Europe. *International Journal of Water Resources Development*, 35, 206–239. https://doi.org/10.1080/07900627.2017.1422702.

Sapino, F, CD Pérez-Blanco, C Gutiérrez-Martín and V Frontuto (2020). An ensemble experiment of mathematical programming models to assess socio-economic effects of agricultural water pricing reform in the Piedmont Region, Italy. *Journal of Environmental Management*, 267, 110645. https://doi.org/10.1016/j.jenvman.2020.110645.

Sapino, F, CD Pérez-Blanco, C Gutiérrez-Martín, A García-Prats and M Pulido-Velazquez (2022). Influence of crop-water production functions on the expected performance of water pricing policies in irrigated agriculture. *Agricultural Water Management*, 259, 107248. https://doi.org/10.1016/j.agwat.2021.107248.

Sivapalan, M and G Blöschl (2015). Time scale interactions and the coevolution of humans and water. *Water Resources Research*, 51, 6988–7022. https://doi.org/10.1002/2015WR017896.

Sivapalan, M, M Konar, V Srinivasan, A Chhatre, A Wutich, CA Scott and JL Wescoat (2014). Socio-hydrology: Use-inspired water sustainability science for the Anthropocene. *Earth's Future*, 2, 225–230. https://doi.org/10.1002/2013EF000164.

Solinska-Nowak, A, P Magnuszewski, M Curl, A French, A Keating, J Mochizuki, W Liu, R Mechler, M Kulakowska and L Jarzabek (2018). An overview of serious games for disaster risk management — Prospects and limitations for informing actions to arrest increasing risk. *International Journal of Disaster Risk Reduction*, 31, 1013–1029. https://doi.org/10.1016/j.ijdrr.2018.09.001.

Sumpsi, J, F Amador and C Romero (1997). On farmers' objectives: A multi-criteria approach. *European Journal of Operational Research*, 96, 64–71. https://doi.org/10.1016/0377-2217(95)00338-X.

Tinbergen, J (1952). *On the Theory of Economic Policy*. North-Holland Pub. Co., Amsterdam (the Netherlands).

Tsur, Y and A Dinar (1997). The relative efficiency and implementation costs of alternative methods for pricing irrigation water. *World Bank Economic Review*, 11, 243–262.

UN (1992a). The Dublin statement on water and sustainable development [WWW Document]. International Conference on Water and the Environment, United Nations. URL http://www.inpim.org/files/ Documents/DublinStatmt.pdf [30 September 2013].

UN (1992b). Rio Declaration on Environment and Development [WWW Document]. United Nations. URL http://www.un.org/documents/ga/conf151/aconf15126-1annex1.htm [30 September 2013].

UNDRR (2019). Global Assessment Report on Disaster Risk Reduction (GAR). United Nations Office for Disaster Risk Reduction, Geneva (Switzerland).

UNDRR (2021). GAR Special Report on Drought 2021 — World (Special Report on Drought). United Nations Office for Disaster Risk Reduction, Geneva (Switzerland).

UNESCO (2021). *The United Nations World Water Development Report 2021: Valuing Water*. Paris: UNESCO.

WATECO (2003). Economics and the environment. The implementation challenge of the Water Framework Directive (Guidance document No. 1), Common Implementation Strategy for the Water Framework Directive (2000/60/EC). European Commission, Brussels (Belgium).

World Bank (2020). World Bank DataBank [WWW Document]. DataBank. URL https://databank.worldbank.org/databases/exchange-rates [16 January 2020].

Young, MD (2014). Designing water abstraction regimes for an ever-changing and ever-varying future. *Agricultural Water Management*, 145, 32–38. https://doi.org/10.1016/j.agwat.2013.12.002.