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Assessing farmers' adaptation responses to water conservation policies through modular recursive hydro-micro-macro-economic modeling

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ABSTRACT

Farmers' adaptation responses to water conservation policies involve a complex decision-making process that depends on a range of criteria, including water availability, profits, and risks, which are in turn dependent on (and might have consequences at) broader scale processes including water systems and the macroeconomy. The non-consideration of the complex interactions between and within natural and human systems often leads to unforeseen consequences and sub-optimal water policy design. There exists a fundamental need to improve our understanding of complex human-water (e.g. hydro-economic) and human-human (e.g. micro-macroeconomic) systems' interactions so to better inform policy-makers. This paper develops an innovative modeling framework for capturing the richness of interactions in complex human-water systems by: i) considering the rationale behind farmers' behavior and responses through microeconomic models; ii) assessing the complex interactions among economic sectors and regions within an economy through macroeconomic models; iii) simulating responses on water cycle dynamics within a river basin by means of hydrologic modeling; and iv) representing the interconnected dynamics and two-way feedback responses between human-water and human-human (through micro-macro-economic) systems. The proposed modeling framework operates through a recursive modular approach built from independent modules which are, in turn, connected through a set of protocols that control the exchange of information. Methods are illustrated considering an incremental agricultural water charging policy in the Spanish part of the Douro River Basin (DRB). Results show that local land use reallocations have an impact on the supply of irrigated (rainfed) agricultural commodities at the macroeconomic level, which further leads to higher (or lower) commodity prices that partially offset changes in crop profitability due to changing water charges and readjustments in the crop portfolio. These, in turn, result in non-linear responses in land and water use with non-trivial impacts on water system's dynamics, where evapotranspiration, surface runoff, and groundwater evapotranspiration are the main hydrologic components affected. We conclude that the integration of hydro-micro-macro-economic modules through a set of protocols can provide crucial information for promoting the efficient design of agricultural water policies.

1. Introduction

Water demand is expected to outstrip supply by 40 percent at the global level by 2030, causing GDP to decline by as much as 6 percent in water scarce areas (2030 Water Resources Group, 2019). Avoiding this "misery in slow motion" will demand transformational changes in water governance (Damania et al., 2017), particularly in agriculture, the largest consumptive and least productive sector using water resources

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(FAO, 2021). Yet, the effects of policy- and climate-induced human responses on the land surface and water systems dynamics and their feedbacks responses to both the economic and natural systems remain poorly understood (Pande and Sivapalan, 2017). Indeed, water is at the core of the most difficult sustainability challenges facing humans in the modern era, involving feedbacks across multiple scales, sectors, and agents (Sivapalan et al., 2014). To better understand hydrologic-related events in the Anthropocene (Crutzen, 2002), research efforts should focus on understanding the co-evolutionary dynamics of human societies and hydrologic systems (Baldassarre et al., 2017). In this context, there exists a fundamental need to improve our current understanding of complex human-water (e.g. hydro-economic) and human-human (e.g. micro-macroeconomic) systems' interactions so to better inform policy-makers (Sivapalan et al., 2012).

Traditional hydrology perceives impacts from socioeconomic systems either as external forcings to the water system or as boundary conditions (Blair and Buytaert, 2016). Hybrid hydroeconomic models typically represent the behavior of water users through piecewise exogenous benefit functions that relate water use to profit, which misses the often complex, non-linear responses of economic agents (Harou et al., 2009). Where full-fledged economic and hydrologic models interact, this typically happens at a microeconomic scale, which offers the necessary level of detail (e.g. land and water use) for a coupling with hydrologic models (Essenfelder et al., 2018; Esteve et al., 2015). This approach usually works in a one-way sequential fashion, with the outputs of one model being used as an input for the other model, thereby missing the multi-directional feedbacks between human and water systems (Di Baldassarre et al., 2013) and within human systems (i.e. micro-macroeconomic) (Pérez-Blanco and Standardi, 2019). For example, a policy that changes irrigators' water and/or land allocation responses will affect water systems' dynamics, which may in turn strengthen or loosen the water availability constraint to users (Essenfelder et al., 2018); moreover, on-farm adaptation to changing climate and water availability conditions will affect agricultural output, changing the demand of inputs and supply of agricultural commodities in the market, and causing effects on commodity prices that affect the decisions from irrigators and other economic agents both at the micro- and macro-economic scale (Parrado et al., 2020).

In this context, river basins are complex systems that contain many interacting components, among which nature and society are two examples. Both realms can be identified in terms of the individual modules they are composed of and defined by the relations between them (many of which can arguably be considered as complex systems themselves, such as agricultural and hydrological systems) (Ratter, 2012). Csete and Doyle (2002) define modules as components, parts, or sub-systems of a larger system that contain some or all of the following features: i) identifiable interfaces providing connection to other modules; ii) can be modified and evolved somewhat independently; iii) facilitate simplified or abstract modeling; iv) maintain some identity when isolated or rearranged, and; v) derive additional identity from the rest of the system. Of particular interest to the definition of modules in complex systems is the notion of connecting interfaces, which allow for the exchange of information between modules; indeed, connecting interfaces are a fundamental feature of complex systems which permits systems' functions that could not be achieved by isolated modules (Turnbull et al., 2018). Connections between modules can be managed through protocols, i.e. rules designed to manage relationships and processes between modules (Csete and Doyle, 2002).

In this paper, we argue that by making use of relatively simple yet detailed hydrologic and economic models currently available in the literature, it is possible to set up an internally consistent framework to represent the interconnected dynamics and feedback responses of human-water (i.e. hydrologic and economic modules) and human-human (i.e. micro- and macroeconomic modules) systems based upon notions of modularity and protocols. The main goal of consistently studying the interconnected dynamics of human-water and human-human systems is to better support experts' discussions and inform policy-makers on achieving sustainable development and on avoiding maladaptation. To this end, this paper develops an innovative modeling framework that: i) captures the rationale behind irrigators' behavior and responses through microeconomic models; ii) assesses the complex interactions among economic sectors and regions within the economy through macroeconomic models; iii) simulates the water cycle dynamics within a river basin by means of hydrologic modeling; and iv) enables the simulation of the interconnected dynamics and two-way feedback responses between human-water and human-human (i.e., micro-macro-economic) systems. The proposed modeling framework operates through a recursive modular approach that allows running each module independently but closely connected through a set of protocols. This enables the simulation of interconnected dynamics and feedback responses between human-water and human-human (i.e., micro-macro-economic) systems. The resulting framework is designed to be flexible and to allow each module to be populated with alternative models. The models used in this paper are: i) the Soil and Water Assessment Tool (SWAT), as the hydrological module (Arnold et al., 1998); ii) A multi-factor, non-linear Positive Multi-Attribute Utility Programming (PMAUP) model, as the microeconomic module (Essenfelder et al., 2018; Gutiérrez-Martín and Gómez, 2011); and iii) a Computable General Equilibrium (CGE) model calibrated at a regional level, as the macroeconomic module (Bosello and Standardi, 2015). The protocols' connection between the different modules is done through the land-use component of each module, while information exchanged among the models includes commodity prices, water availability and water allocation changes, among others. Methods are illustrated with a hypothetical, yet realistic, agricultural water charging policy in the Spanish part of the Douro River Basin (DRB).

2. Methods

For the purpose of illustrating the potential of the proposed hydro-micro-macro-economic recursive modular framework, a time-invariant setting that assesses changes in the equilibrium of the hydro-micro-macro-economic system through comparative statics was adopted. The static hydro-micro-macro-economic recursive modular framework (depicted in Fig. 1) works as follows:

- In step 1, a policy shock (in our case, water charging) is applied to the microeconomic model, whose solution provides changes in the reallocation of land among crops based on the preferences of economic agents.
- In step 2, land use changes as simulated by the microeconomic model in step 1 are aggregated and fed into the agricultural sector of the case study area's corresponding region in the macroeconomic model; a macroeconomic simulation is then performed with the new input information to find a new economic equilibrium and to provide a set of production quantities and commodity prices that are consistent with economy-wide effects.
- In step 3, changes in crop commodity prices in the relevant region are fed back into the microeconomic model, and economic agents in the case study area reassess their initial crop portfolio decision. Steps 1 to 3 occur iteratively until convergence is reached (Hasegawa et al., 2016; Ronneberger et al., 2009). Note that the coupling protocol between the micro- and macro-economic modules through land use and crop price changes yields a stable system, as shown in Parrado et al. (2019).
- In step 4, resulting land use choices and water use by economic agents are fed into the corresponding consolidated land areas of the hydrologic model, which simulates the effects of the socioeconomic system on the water dynamics of the river basin.
- In step 5, relevant effects on the water system (i.e. water availability for irrigation) are passed as spatially-distributed information to the corresponding microeconomic agent. If the water availability

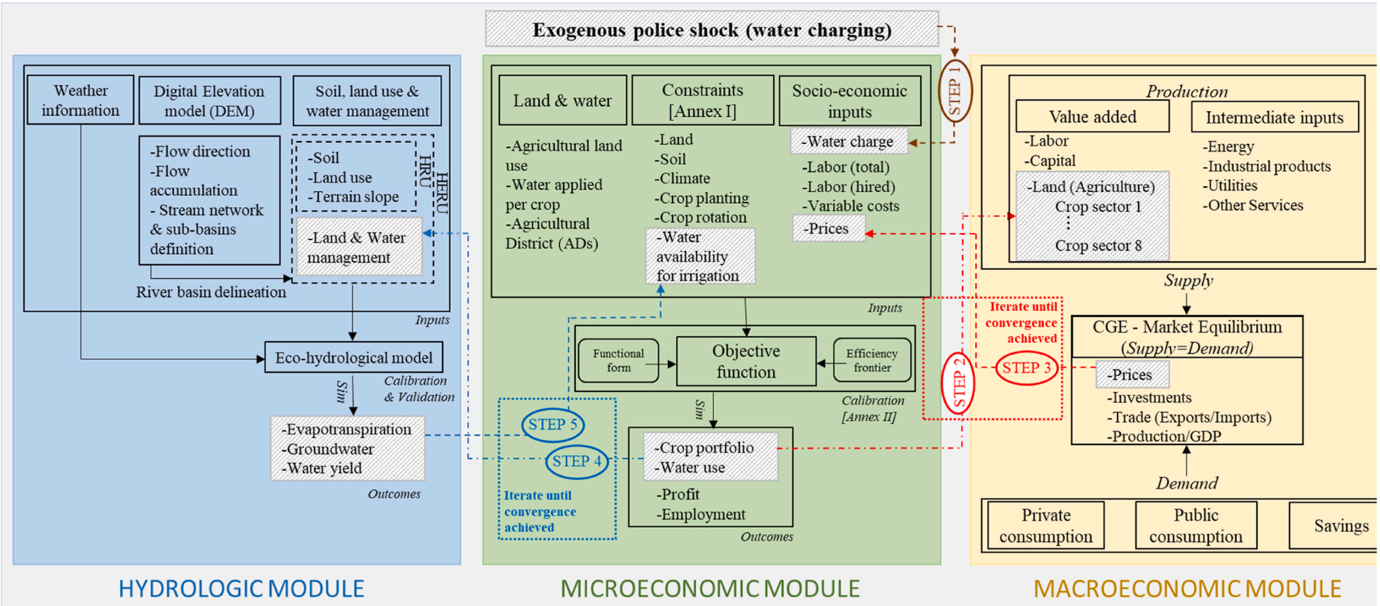


Fig. 1. Conceptual representation of the coupled hydro-micro-macro-economic modeling framework.

constraint strengthens following the hydrologic simulation, forcing economic agents to adapt, the iterative process in step 1 to 3 is repeated until convergence is reached (i.e. models predictions are stable and consistent).

The five main methodological steps are built-upon the protocol connections between the modules, which are conceptually represented in Fig. 1 and discussed in detail in Sections 2.1 and 2.2. Note that for climate-induced responses (e.g. climate change scenario analysis), simulations would run from the hydrologic (external climate forcing) to the microeconomic to the macroeconomic model (i.e. step 4 and 5 as first steps, followed by 1–3). The same rationale is valid for macro-economic induced responses, such as macroeconomic shocks due to COVID19, for instance. The overall simulation from steps 1 to 5 follows the rules established in the protocols' connection and is repeated in successive iterations until convergence of results provided by both water and human systems is reached.

Convergence between systems is assessed through a convergence test[1] (see Section 3) (Hasegawa et al., 2016; Ronneberger et al., 2009). Convergence refers to a situation when information exchanged between models does not result in meaningful changes between two succesive simulation iterations (Hasegawa et al., 2016; Ronneberger et al., 2009).

To check if the framework is in equilibrium (i.e., there is convergence), one should empirically test results by running at least two complete iterations (steps 1 to 5) and assessing the degree of change in the value of predetermined variables. The framework depicted in Fig. 1 has two convergence variables (one for the human-human/micro-macro economic coupling and another for the human-water coupling, which in our case are land allocation and crop prices as detailed in Section 2.2), which are subject to the convergence test. The framework is in equilibrium only when convergence is simultaneously achieved in the human-human/micro-macroeconomic and in the human-water coupling. In our application, the framework is assumed to be in equilibrium when the values of the convergence variables for the last two successive iterations experience a change below a predetermined threshold set at 0.00001% (Parrado et al., 2020). If convergence is not achieved, the recursive modular framework continues, and additional convergence tests are conducted after each complete iteration (steps 1 to 5) until the framework is in equilibrium. The efficiency of convergence tests is limited by the amount of information exchanged between models, which conditions the number of convergence variables to be considered. Accordingly, the more variables are included in the information exchange between models, the more computationally expensive and time consuming will become the convergence process. Thus, it is necessary to carefully assess how many variables are coupled between modules to keep tractability of the convergence process and the whole modelling framework, while accurately representing the relevant socio-ecological system.

2.1. *Modularity*

2.1.1. *The microeconomic module*

In microeconomic agricultural water management models, the agent (i.e. a farmer or a representative group of farmers) decides on the crop portfolio and timing, water withdrawals and capital investment, so to maximize its utility in accordance to one (single-attribute) or multiple (multi-attribute) utility-relevant attributes and a number of constraints defining a domain. Literature typically simplifies this decision-making process by representing each possible combination of crops, timing, water application and capital as a separate crop with unique characteristics, so that the utility maximization problem is reduced to a decision on the crop portfolio x within a domain $F(\mathbf{x})$ (Graveline, 2016):

[1] The static approach assumes economic agents know their best management alternatives by having *a priori* access to reliable and accurate information on future prices and hydro-meteorological variables. The assumption of *a priori* access to reliable and accurate information implies that even if agents' expectations are wrong, they are on average correct; in other words, "agents' expectations are not systematically biased and collectively use all relevant information in forming expectations of economic variables" (Muth, 1961). Without the assumption of *a priori* access to reliable and accurate information, the integration would be dynamic in time: one year run of each model following steps 1 → 2→4 → 5→3 without convergence tests, carrying the information forward in time. Note that the dynamic setting does not ensure convergence and therefore precludes a comparative statics exercise. Time-variant settings have received attention in the literature to illustrate complex adaptation pathways involving multiple policies and policy levers to support dynamically robust decision making (Haasnoot et al., 2013; Kwakkel et al., 2015). On the other hand, the time-invariant static approach is typically used to assess changes in the equilibria of systems following a shock through comparative statics (Bosello et al., 2012; Dixon et al., 2012; Taheripour et al., 2016).

$$\underset{x}{\text{Max}} \ \text{U}(\mathbf{x}) = \text{U}(z_1(\mathbf{x}); \ z_2(\mathbf{x}); z_3(\mathbf{x}) \ldots \ z_m(\mathbf{x})) \quad [1]$$

$$\text{s.t.:} \quad 0 \le x_i \le 1 \quad [2]$$

$$\sum_{i=1}^{n} x_i = 1 \quad [3]$$

$$x \in F(\mathbf{x}) \quad [4]$$

$$\mathbf{z} = \mathbf{z}(\mathbf{x}) \in R^m \quad [5]$$

where $\mathbf{x} \in R^n$ is the crop portfolio or decision variable, which is represented by a vector containing the land share devoted to each individual crop x_i $(i = 1, \ldots, n)$. Note that each crop *i* has a unique combination of utility-relevant attributes $\mathbf{z}(\mathbf{x})$ attached (notably profit, but also risk or management complexity aversion). Attributes are quantities of dimension one, obtained dividing their observed values by the maximum value they can possibly attain in the model (accounting for the domain). Increasing the provision of any given attribute improves agent's utility, provided the provision of the remaining attributes is kept constant ("more is better"). Convexity holds, i.e. increasing the provision of a utility-relevant attribute will reduce the provision of another utility-relevant attribute; otherwise there is no tradeoff and the choice between the two attributes becomes irrelevant, meaning one of them is not utility-relevant and can be discarded. The domain $F(\mathbf{x})$ is defined by a set of quantifiable constraints, including agronomic (e.g. crop rotation), policy (e.g. Common Agricultural Policy rules), information (e.g. know how), land (i.e. agricultural and irrigable area) and water availability constraints. The latter can be represented as:

$$\sum_{i=1}^{n} w_i \ \ x_i \le W_g \quad [6]$$

where water availability per hectare is denoted by W_g. w_i is the water required by crop x_i, per hectare (a constant, i.e. adaptation at the intensive margin/deficit irrigation is not considered). A detailed description of the model constraints is available in Annex I in the online supplementary material.

For the calibration of the microeconomic module, this paper relies on a Positive Multi-Attribute Utility Programming (PMAUP). PMAUP models have been used extensively to calibrate agents' objective functions and assess their responses to climatic and policy shocks such as droughts (Pérez-Blanco et al., 2017), irrigation restrictions (Essenfelder et al., 2018), crop price volatility (Gutiérrez-Martín et al., 2014), insurance policies (Pérez-Blanco et al., 2016) or water charging (Parrado et al., 2019). The calibration procedure is described in detail in Gómez-Limón et al. (2016) or Gutiérrez-Martín and Gómez (2011), and is also available in Annex II in the online supplementary material. Data inputs and calibration results are available in Annex III.

2.1.2. The macroeconomic module

To account for the economy-wide feedbacks at the macroeconomic scale, we use a regionalized CGE model for the 17 NUTS2[2] regions of Spain (Bosello and Standardi, 2015; Parrado et al., 2019). The theoretical structure is based on the Global Trade Analysis Project (GTAP) model (Hertel, 1997). The neoclassical structure implies that in each region investments are saving-driven, factors of production are fully employed and perfect competion holds in the markets. The behavior of the representative agents (household, government, firms and factors) is driven by the changes in the relative prices which clear the markets, meaning that for each commodity the supply is equal to its demand, thus creating the new equilibrium in the economic system. The CGE aggregated database includes 15 economic sectors and the 17 NUTS2 Spanish regions, the rest of EU28 and the rest of the world as shown in Table 1. There are 8 aggregate crop sectors which are mapped considering data from the PMAUP model. Data inputs for the macroeconomic module are described in Annex III.

2.1.2.1. Supply side. A representative firm in each sector minimizes output costs (y) under a Leontief technology between value added (va) and intermediate inputs (in):

$$\underset{va_{j,s},\, in_{j,s}}{\text{Min}} \left(\ pva_{j,s} va_{j,s} + pin_{j,s} in_{j,s} \right) \quad [7]$$

$$\text{s.t.:} \quad y_{j,s} = \min\left\{ va_{j,s}, in_{j,s} \right\} \quad [8]$$

where $pva_{j,s}$ and $pin_{j,s}$ are respectively the price of the value added composite (calculated as the weighted average of the prices of each value-added component: labor, capital and land) and the price of intermediate inputs in sector j of region s.

Value added is modelled though a Constant Elasticity of Substitution (CES) function which allows for substitution between primary factors (Labour, capital, land and natural resources). Labor and capital are used by all sectors, while land is specific to the agricultural sectors (sectors 1–9) and natural resources to the extractive sector (sector 10). The CES function depends on v_f primary factors, with sector-specific elasticity of substitution σ_j. Input augmenting or biased technical change is represented with the parameter $\eta_{f,j,s}$ for each primary factor f in sector j and region s.

$$va_{j,s} = F\left(\eta_{f,j,s}, v_{f,j,s}, \sigma_j\right) \quad ; \ \sigma_j > 0 \quad [9]$$

Primary factors are used domestically since they are not internationally tradable. Labor and capital are perfectly mobile across sectors within a region. In the standard version of the model land supply at the sectoral level is modelled through a Constant Elasticity of Transformation (CET) which allocates the (exogenous) overall regional land to agricultural sectors according to sector-specific land rents (Bosello and Standardi, 2015; Parrado et al., 2019). However, for this study we modify the land allocation among crops in order to be set exogenously

Table 1
Regional and sectoral aggregation of the regionalized CGE model.

Regions		Sectors	
Spain (disaggregation in NUTS 2)	1) Galicia 2) Asturias 3) Cantabria 4) Pais Vasco 5) Navarra 6) La Rioja 7) Aragon 8) Madrid	Crops	1) rice 2) wheat 3) other cereals 4) vegetables and fruits 5) oil seeds 6) sugar cane and beet 7) plant-based fibers 8) crops not elsewhere classified
	9) Castilla y Leon 19) Castilla y Mancha		
	11) Extremadura 12) Cataluña	Industry	9) livestock 10) extraction, fishing and forestry
	13) Valencia 14) Balears 15) Andalucía		11) food industry 12) rest of industry
	16) Murcia 17) Canarias 18) Rest of EU-28 19) Rest of the world	Services	13) utilities 14) construction 15) services

[2] The *Nomenclature des Unités Territoriales Statistiques* (NUTS), or nomenclature of territorial units for statistics, is "a hierarchical system for dividing up the economic territory of the EU" (Eurostat, 2020). In Spain, NUTS 1 refers to macro-regions; NUTS 2 to regions; and NUTS 3 to provinces.

according to changes in land use dictated by the PMAUP model following Ronneberger et al. (2009).

2.1.2.2. Demand side. Income from primary factors accrues to each regional representative household which disposes it following a Cobb-Douglas per capita utility function (Hertel, 1997).

$$U_s = Cons_s^{\omega_{Cons}} Gov_s^{\omega_{Gov}} Sav_s^{\omega_{Sav}} \quad [10]$$

where U_s represents the utility of the representative household determining the demand side of the CGE model, obtained as the aggregate of private consumption ($Cons_s$); government consumption (Gov_s) and savings (Sav_s) in region s; and the parameters ω are the associated budget shares. Private and government consumption represent the aggregate demand for the commodities produced in the different sectors of the economy, where commodities can be produced either domestically or imported from other regions. Savings represent the resources available for investment needs.

A global bank collects regional savings and then allocates these resources as investments among regions. Investments are mobile at the international level and the difference between regional savings and investments determines the trade balance.

Trade is a central aspect in the CGE model. Commodities can be exchanged in the domestic, intra-national and inter-national markets. To model trade at these three levels, we consider an upper bundle between domestic and imported goods and a lower bundle to source imports from all sources. We keep the GTAP formulation assuming imperfect substitution (Armington, 1969) between the domestic demand ($dd_{j,s}$) and the aggregate demand for imported products ($dm_{j,s}$) in region s and sector j via a CES function. In each economic sector, the representative household, government or firm minimize the total expenditure under the CES constraint on domestic and imported goods.

$$\underset{dd_{j,s},\, dm_{j,s}}{\mathrm{Min}} \left(pdd_{j,s} dd_{j,s} + pdm_{j,s} dm_{j,s} \right) \quad [11]$$

$$\text{s.t.:} \;\; dtot_{j,s} = G_1\left(dd_{j,s}, dm_{j,s}, \sigma_j^{Up}\right); \sigma_j^{Up} > 0 \quad [12]$$

where $dtot_{j,s}$ is the total demand and $pdd_{j,s}$ and $pdm_{j,s}$ are the prices associated with domestic and aggregate demand for imported goods, respectively.

Given the importance of the intra-national trade in this experiment and differently form GTAP which has no sub-country detail, in the lower level the aggregate amount of imports ($dm_{j,s}$) is sourced from the country or the sub-country region of origin through a Constant Ratio of Elasticities of Substitution and Homothetic (CRESH) constraint (Cai and Arora, 2015; Hanoch, 1971; Pant, 2007) which allows for more flexibility in the choice of product substitutability for each couple of spatial units.

$$\underset{imp_{j,s',s}}{Min} \sum_{s'} pimp_{j,s',s} imp_{j,s',s} \quad [13]$$

$$\text{s.t.:} \;\; dm_{j,s} = G_2\left(imp_{j,s},\, \sigma_{j,s}^{Lo}\right) \quad ; imp_{j,s} \in R^S \,,\; \sigma_{j,s}^{Lo} \in R^S \,,\; \sigma_{j,s',s}^{Lo} > 0 \quad [14]$$

where $imp_{j,s',s}$ is the bi-lateral trade flow from region/country s' to region/country s in sector j and $pimp_{j,s',s}$ is the associated price; $imp_{j,s}$ and $\sigma_{j,s}^{Lo}$ are two S-dimensional vectors (S being the number of country/regions in the CGE) representing respectively all the bi-lateral imports and elasticities of substitution of region/country s in sector j.

2.1.3. The hydrologic module

Hydrologic modeling is useful for supporting water management by providing quantified information regarding the water dynamics and water-related processes in a river basin (Brutsaert, 2013). Commonly, hydrologic modeling uses mathematical constructs to simulate the water cycle of hydrologic-defined units. When applied at a river basin scale, these models are referred to as regional hydrologic models. Mathematical models of river basin hydrology can be employed to address a wide range of environmental and water resources issues, such as surface runoff and soil erosion modeling, reservoir management, and groundwater recharge dynamics. Depending on the complexity, level of detail, and application, models may be classified as eco-hydrologic models. Eco-hydrologic models are characterized by the simulation of not only the hydrologic cycle, but also hydrologic-related processes, such as vegetation and crop dynamics, the computation of the nutrients' cycling throughout a river basin, and the provision of ecosystem services. A well-documented and extensively used eco-hydrologic model is the SWAT model (Arnold et al., 1998).

SWAT is a conceptual, semi-distributed eco-hydrologic model operating at a river basin scale. The model offers the capability of assessing different river basin-related management processes and operations, such as agriculture-related practices and irrigation methods, while also providing tools for the assessment of their impacts on a sub-basin unit scale (Arnold et al., 1998; Neitsch et al., 2011). SWAT is a versatile model, being applied under an extensive range of studies, including the evaluation of land-use and climate change scenarios, alternative best management practices, and the simulation of sediments, nutrients and pesticides transport throughout a river basin, both for academic purposes or governmental/engineering applications (Abbaspour et al., 2007; Bressiani et al., 2015; Ullrich and Volk, 2009; Zhang et al., 2007).

Being a semi-distributed hydrologic model, SWAT assumes a river basin to be composed of a mosaic of smaller spatially defined units, known as sub-basins. In turn, a sub-basin is further subdivided into smaller units known as hydrological response units (HRUs) (Neitsch et al., 2011; Winchell et al., 2007). An HRU can be understood as the lumped land areas within a sub-basin that is comprised of unique land cover, soil, slope and management combination, which, together, comprise the main inputs to the SWAT model (see Annex III for a detailed description of the data inputs). The subdivision into sub-basins and HRUs enables the SWAT model to not only reflect differences in the hydrologic cycle for various crops and soils, both temporally and spatially, but also influences coming from the implementation of specific land management practices and water-related policies (Krysanova and Arnold, 2008).

Of particular interest to the present research, SWAT allows the definition of a variety of land management practices that may take place in each HRU in order to translate the different properties and actions taken for the management of different land-use types (Neitsch et al., 2011). When considered under a perspective of socio-hydrology, however, the model lacks a socio-economic component capable of accounting for both the self-organization of people in the landscape (Sivapalan et al., 2012) and the feedbacks between human and water systems (Sivapalan et al., 2014). As a consequence, human actions are generally assumed to be an exogenous forcing to the natural system. Recognizing the fact that socio-economic agents may not only affect natural processes but also adapt to changing environmental conditions, some eco-hydrologic processes as simulated by the SWAT model are of particular interest when studying the dynamics of human-water systems in a river basin, the first and most obvious being a river basin's water balance, computed by SWAT as follows:

$$SW_t = SW_0 + \sum_i^t \left(R_{day,i} + Q_{irr,i} - Q_{surf,i} - E_{a,i} - \omega_{seep,i} - Q_{gw,i}\right) \quad [15]$$

Where i is the index for the simulation step [day]; t is the final simulation step [day]; SW_t is the soil water content [mmH2O]; SW_0 is the initial soil water content [mmH2O]; $R_{day,i}$ is the amount of precipitation that reaches the soil surface [mmH2O]; $Q_{irr,i}$ is the amount of water added to the soil profile by irrigation [mmH2O]; $Q_{surf,i}$ is the amount of surface runoff [mmH2O]; $E_{a,i}$ is the amount of actual evapotranspiration [mmH2O]; $\omega_{seep,i}$ is the amount of water entering the vadose zone from the bottom of soil profile [mmH2O], and; $Q_{gw,i}$ is the amount of return flow[mmH2O]. Groundwater dynamics in SWAT is represented by a lumped module for each individual sub-basin that accounts for two separate groundwater storages, named shallow and a deep aquifers. Both the shallow and the deep aquifers may be used as sources for irrigation and may eventually contribute to streamflow (e.g. as baseflow or as irrigation abstractions). In the considered case study area, however, irrigation water comes mostly from surface water. All these eco-hydrologic processes belong to the land phase of the hydrologic cycle, and can be influenced by variations in land-use and management by economic agents. The resulting information is then spatially connected in a watershed during the routing phase of the SWAT model. The combination of both phases spatially characterizes eco-hydrological processes that might act as a constraint to the microeconomic module, such as water availability for irrigation. The processes that control the flow of information between modules are governed by a set of rules known as protocols.

The SWAT model is calibrated and validated for the Douro River Basin in the context of the project AGUAMOD (AGUAMOD, 2021). A total of 22 stream gauge stations have been used to calibrate the SWAT model, with data covering the period from 1994 to 2013. The period from 1994 to 2010 is used for calibration of the model, while validation is performed using data from 2011 to 2013. Calibration and validation is done through the use of the SWAT-CUP software and the Sequential Uncertainty Fitting procedure (SUFI2) (Abbaspour, 2012). Results indicate a Nash-Sutcliffe (NSE) ranging from 0.55 to 0.94 and Percent Bias (PBIAS) ranging from −5.70 to 14.10 for calibration, and NSE ranging from 0.51 to 0.92 and PBIAS ranging from −12.40 to 12.40 for validation, all under a monthly time-scale. Detailed information regarding the data source and the calibration of the SWAT model can be found in the Supplementary Material (Tables A.III.3 and A.III.4).

2.2. Protocols

Protocols define the rules for the exchange of information on the coupling variables between the different modules. This allows for translating and mapping information between modules, so that relevant inputs can be read by the destination module. The protocols developed here are shown schematically as the methodological steps in Fig. 1. The main spatial element connecting all the modules is land use, while protocols ensure that information is translated into a readable format from a source module to the destination module. Protocols are activated whenever a new scenario simulation is initiated and until convergence is achieved between the incumbent set of modules (micro-macro-economic system in steps 1–3 and human-water system in steps 1–5). Convergence is a major challenge in the development of the set of rules or *protocols* connecting and allowing information exchange between modules. The literature typically assesses convergence empirically, adopting a trial-and-error approach where protocols are fine-tuned and conflicting module components/agents that prevent convergence are substituted/discarded (Hasegawa et al., 2016; Ronneberger et al., 2009). Available research on convergence in multi-model and multi-system applications suggests two steps to achieve convergence. In the first step, "data values translated between the models must be verified" (Hasegawa et al., 2016). Typically, fluctuations will be observed in the first iterations due to the differences in the initial assumptions between the models, followed by relative stabilization if convergence is observed. Convergence criteria in our recursive modular framework are explained in the following two sections. After the second step, convergence can be explored through a convergence test. Convergence tests "assess the response of the coupled system to a series of shocks to, then, audit its convergence behavior" in succeeding iterations (Ronneberger et al., 2009). In our research, this is done in Section 3, where a policy shock is applied to the recursive, modular hydro-micro-macro-economic model. Aside from conventional empirical convergence tests, Parrado et al. (2019) use a a simplified dynamic system model to show that application of the coupling protocol through land use and crop price changes in the micro-macro-economic system yields a stable equilibrium.

2.2.1. Coupling human-human (micro-macro-economic) systems

In step 1 of the recursive modular framework, a new water policy shock g (in our case, charging) impacts microeconomic agents. In each policy shock scenario, agents reassess their choices so to maximize utility within the corresponding domain The resulting utility-maximizing crop portfolio is defined as $\mathbf{x}^*_{g,p_0}$, where p_0 represents the set of crop prices observed in the baseline. Since in the microeconomic model crop prices are exogenous, we do not observe any change in this variable for now.

Optimal crop portfolios before ($\mathbf{x}^*_{0,p_0}$) and after the policy shock ($\mathbf{x}^*_{g,p_0}$) are used to calculate land use changes for every crop in the microeconomic model. Individual crops in the microeconomic model are then aggregated into crop sectors j following the aggregation in the CGE model (see Table 1 for a list of the CGE crop sectors) to obtain the land use per crop sector $x^*_{g,p_0,j}$ and the percentage land use changes per crop sector, or $\gamma_{g,p_0,j}$:

$$\gamma_{g,p_0,j} = \gamma_j * \left(\frac{x^*_{g,p_0,j}}{x^*_{0,p_0,j}} - 1\right) *100 \tag{16}$$

where γ_j is a fixed coefficient that represents the fraction of the total land use of crop sector j in the case study's corresponding region (NUTS2) that belongs to the agents considered in the microeconomic simulation. γ_j is calculated as the ratio of total land use by the crop sector j in the area belonging to the microeconomic agents to the total land use by the crop sector j in the corresponding region for the base year 2015 (observed values). The γ_j ratio is used to circumvent the spatial divergence between the regionalized (NUTS2) macroeconomic model and the microeconomic model, which is calibrated for representative agricultural water users at a sub-regional level.

In step 2, the percentage land use changes per crop sector j, or $\gamma_{g,p_0,j}$, are applied to the agricultural land use allocation xm in the corresponding region s in the CGE model, so to replicate microeconomic agents' choices in a macroeconomic context.

$$xm_{g,p_0,j,s} = F\left(\gamma_{g,p_0,j}, xm_{0,p_0,j,s}\right) \tag{17}$$

A macroeconomic simulation is then performed with the new input information to find a new economic equilibrium and to provide a set of production quantities and prices that are consistent with economy-wide effects.

In step 3, changes in crop prices in the relevant region s are fed back into the microeconomic model, and agents in the case study area reassess their initial crop portfolio decision. This will result in a new optimal solution to the problem in Eqs. (1)–(5), namely the utility-maximizing crop portfolio $\mathbf{x}^*_{\mathbf{g},\mathbf{p}_1}$. Next, we repeat the process above to obtain the updated percentage land use changes per crop sector j, or $\gamma_{g,p_1,j}$, that will feed the CGE in the next iteration.

$$\gamma_{g,p_1,j} = \gamma_j * \left(\frac{x^*_{g,p_1,j}}{x^*_{0,p_0,j}} - 1\right) *100 \tag{18}$$

where p_1 represents the updated crop prices provided by the CGE model in iteration 1. Again, the percentage land use changes per crop sector $\gamma_{g,p_0,j}$ are applied to the agricultural land use allocation xm in the corresponding region s in the CGE model:

$$xm_{g,p_1,j,s} = F\left(\gamma_{g,p_1,j}, xm_{0,p_0,j,s}\right) \tag{19}$$

Steps 1 to 3 occur iteratively until the sum over all crop sectors of the absolute value of the differences between the current and the previous iterations for the changes in crops' prices in the CGE and land use changes in the PMAUP model is below 0.00001%, where convergence between the two models is assumed.

2.2.2. Coupling human-water systems

The protocol connection between the eco-hydrologic and microeconomics modules is done in a modular and sequential fashion, through the land-use component of each module and using a common spatial element, hereinafter referred to as 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 hydrologic units and socio-economic agents. As a result, each 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. Crop choices, land use management, water withdrawals, and water availability are examples of information that are exchanged between the two modules. By combining physical and economic spatial information, HERUs enable not only the identification of a common spatial unit among human and natural systems, but also provides the means for the exchange of information between them. The models used to illustrate the coupling protocol of human-water systems are the microeconomic PMAUP model (Gómez-Limón et al., 2016; Gutiérrez-Martín and Gómez, 2011) and the eco-hydrologic SWAT model (Arnold et al., 1998).

As HERUs are independent entities endowed with the capacity for decision-making, preferences and choices identified and simulated by the PMAUP component are passed to the SWAT model as land and water management actions, which in this study correspond to the crop portfolio choices defined in the PMAUP model (x) and related water use ($\sum_{i=1}^{n} w_i \ x_i$, where w_i are the water needs per crop i, which are constant—see Eq. (6)). This is described in step 4 in Fig. 1. Similarly, eco-hydrologic information that might constrain the decision-making of economic agents (i.e., water availability for irrigation practices, denoted by the constraint W_g in Eq. (6)) are transferred from the SWAT model to the PMAUP model, where W_g is determined by i) the irrigation water source of each single farmer and; ii) the amount of water available in that source and location as simulated by SWAT. This is described in step 5 in Fig. 1). In case eco-hydrologic information transferred from the SWAT model further constrains the decision-making of the microeconomic agents in the PMAUP model (i.e., W_g is strengthened), a new round of feedback responses between the PMAUP and SWAT models is performed until convergence is achieved (recall that in our application we adopt a time-invariant approach where temporal dynamics are not accounted for[3]). Note that due to the policy explored in our application (water charging), the feedback loop from the eco-hydrologic to the economic module achieves convergence after one iteration. This is because higher water charges will loosen rather than strengthen water availability W_g, and agents will use less water than allotted (i.e., the binding constraint is now the budget—due to the growing cost of water—instead of the water allocation). The eco-hydrologic-economic feedback loop would need more iterations to achieve convergence under alternative policies such as irrigation modernization subsidies, where the increased consumed fraction of water allocations can reduce return flows and water availability for downstream users (Pérez-Blanco et al., 2020).

2.3. Case study area and policy scenarios

Methods are illustrated with a hypothetical, yet realistic, agricultural water charging policy in the Spanish part of the DRB. The DRB is the largest river basin in the Iberian Peninsula, spanning an area of 98103 km^2 between Spain (80.4% of the territory) and Portugal (19.6%). Our assessment of water charging impacts in the DRB considers ten scenarios in which water charges are increased from 0 to 0.1 EUR/m^3 at intervals of 1 Eurocent (i.e. 10 scenarios/simulations).

2.3.1. Case study area

The Spanish part of the DRB spreads over 78888.5 km^2 and eight regions (NUTS2), among which the most relevant one is Castile and León, which accounts for 98.25% of the DRB's territory. Castile and León is the largest region in Spain, representing 18.6% of the Spanish territory, and one of the least populated, with 2418694 inhabitants (5.1% of the Spanish total). The GDP per capita is EUR 22649, slightly below the national average of EUR 24100. The Castile and León Region and by extension the DRB is highly dependent on agriculture, which represents 4.6% of the regional GDP, 6.7% of the regional employment and 15% of the total Spanish agricultural production. Significant economic impacts from irrigation rationing policies can be anticipated from this economic structure.

The DRB has a continental Mediterranean climate (Csb/Cfb Köppen climate classification), with the characteristic long, cold winters and short, hot summers of continental climate and the 3–4 months of summer aridity characteristic of Mediterranean climate. Most of the DRB is occupied by a large plateau (*Meseta Central*), surrounded by mountains. Average rainfall ranges between 450 and 500 mm/year, which is considerably lower across the central plateau where most agricultural activity is located, and higher in the mountains. The climate and the geography of the DRB have favored the development of an extensive rainfed agriculture dominated by cereal production. Rainfed crops represent 90% of the 5.7 million ha of agricultural land in the DRB and include cereals such as wheat, barley, rye and oats; legumes such as carob and chickpeas; sunflower; and vineyard. Most relevant irrigated crops include sugar beet, cereals such as maize and barley, sunflower, potatoes, alfalfa and vegetables. The main source of irrigation waters are superficial sources (DRBA, 2020). The relevant territorial unit for administrative purposes in the agricultural sector of the DRB and elsewhere in Spain are the Agricultural Districts (ADs, in Spanish: *comarcas*), which are also the agents in the microeconomic module/PMAUP model. ADs are intermediate administrative levels between the province and the municipality level that have a "high degree of homogeneity from an agricultural perspective" and are used as the relevant planning unit in agricultural and water policy in Spain (MAPAMA, 2019). Fig. 2 presents the case study area and its most relevant features.

The significant income gap between irrigated and non-irrigated crops, accentuated by the introduction of technical innovations throughout the process of agricultural production, and the relative abundance of inexpensive water sources in the DRB have favored the expansion of irrigated land around the Douro, Esla, Órbigo, Pisuerga and Tormes rivers during the last three decades, and irrigation now represents 3432.6 million m^3 out of the 3862 million m^3 of water annually withdrawn by consumptive uses in the DRB (88.9%). On the other hand, climate change is diminishing water availability, which has fallen from an average of 14231.4 million m^3/year for the 1940/41–2005/06 period to 12777.3 million m^3/year for the 1980/81–2005/06 period (−10.2%); this trend will be aggravated in the future, with reductions in water availability up to −25% in 2040–2070 and −50% in 2070–2100

[3] If the integrated framework adopts an alternative dynamic setting, some additional considerations should be made to circumvent this temporal divergence: i) the microeconomics model simulates the preferences of farmers (i.e. crop portfolio) at the beginning of the crop season based on available information; ii) decisions made by farmers at the beginning of a crop season are invariant throughout that year (i.e. one year run of each model); iii) the decisions taken by farmers are passed to the hydrologic model at the beginning of the year, and parameters are updated accordingly (e.g. switch to a new crop type, or reduction in the amount of water withdrawn for irrigation, etc.); and iv) the feedbacks from the hydrologic to the microeconomic model are passed as an average yearly value at the end of that particular year.

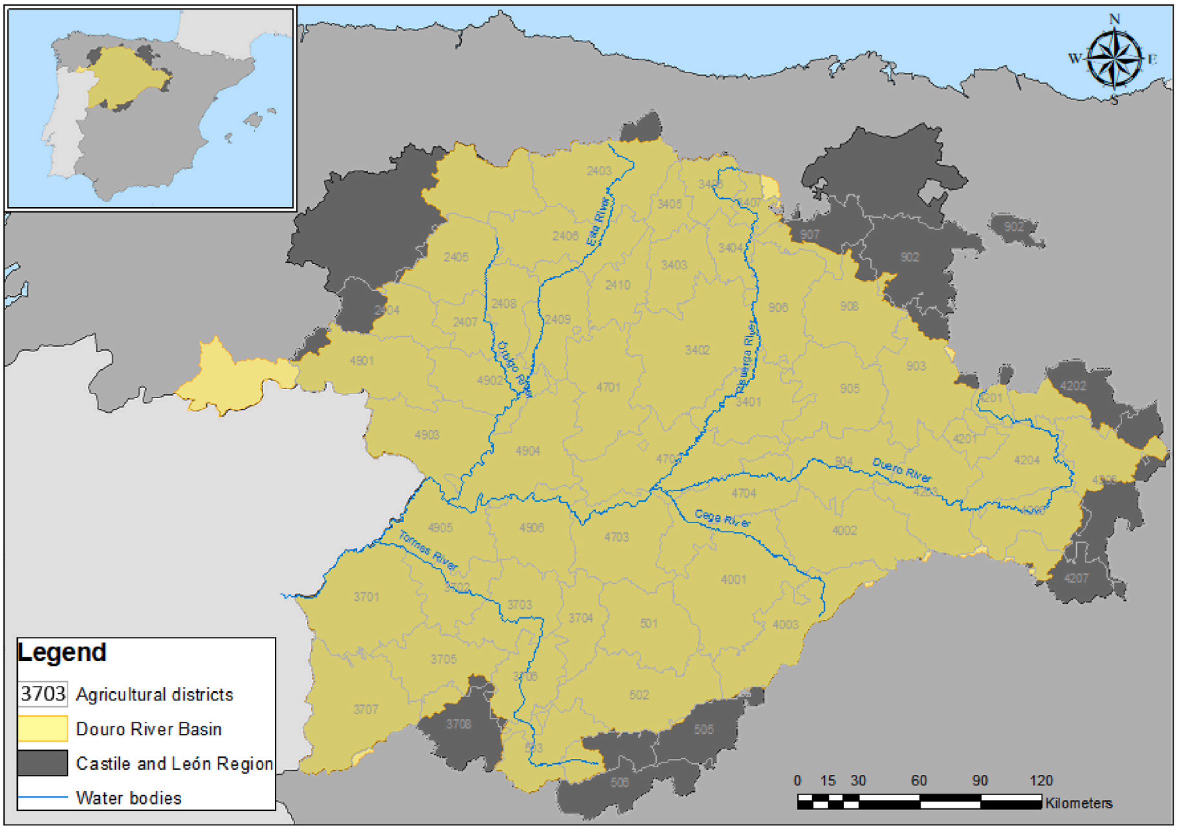


Fig. 2. Location of the DRB and the Castile and León Region and detail of the ADs. *ADs have a numeric label that corresponds to the coding they are assigned in river basin planning documentation (see e.g., DRBA, 2020).*

according to predictions from the Spanish Ministry of Environment (MAGRAMA, 2017). Although the DRB has historically invested heavily in reservoirs to enhance water availability and security, to the point of becoming one of the basins in Spain with the highest storage capacity, water works have failed to meaningfully expand the supply base (Parrado et al., 2020). The compounded effect of growing demand and diminishing supply has increased the frequency of drought events: after only 25 years with recorded droughts in the period 1940–2000 (one drought every 2.4 years), droughts have hit the basin in 15 years since the turn of the century (one drought every 1.2 years). Structural scarcity is also a growing concern: the average Water Exploitation Index (WEI) of the DRB, measured as the ratio of water depletion (i.e. withdrawals minus return flows) to renewable resources, is estimated at 18.7%, and is progressively getting closer to the 20% warning threshold that distinguishes a non-stressed from a water scarce basin (EEA, 2016). Actually, during the year 2013 the WEI was on average 24%, and in the worst month hit 44% (DRBA, 2016).

2.3.2. Policy scenarios

Following EU guidelines (EC, 2015a, 2015b, 2009; OJ, 2000), the DRB is now in the process of designing water reallocation policies that ensure the good ecological status of its water bodies, in which the management of irrigation water through charging is expected to play a central role. Accordingly, our application considers a series of scenarios where water charges are increased from 0.0 to 0.1 EUR/m^3 at intervals of 0.01 EUR/m^3 (10 charging scenarios). This represents a charge increase of 50–500% (the average water charge in the DRB is 0.02 EUR/m^3), consistent with the scenarios considered by the Douro River Basin Authority in the third river basin planning cycles (DRBA, 2020).

3. Assessing the implications of agricultural water charging through modular recursive hydro-micro-macro-economic modeling

3.1. Implications on the human-human (micro-macro-economic) system

We run a series of simulations in which water charges are increased from 0 to 0.1 EUR/m^3 at intervals of 1 Eurocent (i.e. 10 simulations). Agents in the microeconomic module respond to higher water charges revising their crop portfolio decisions, which modifies the supply of agricultural commodities and inter-regional trade flows in the macroeconomic module, thus affecting crop prices. In turn, crop price feedbacks from the macroeconomic module constrain agents to revise again their crop portfolio decision. This iterative feedback loop is repeated until convergence is achieved (a convergence test of the model is available in Annex IV). Fig. 3 reports changes in the equilibrium crop portfolio choices and crop prices per crop category, for different water charging values. The situation in the baseline scenario is available in Annex III in the online supplementary material, which reports the calibration results of each model (note that the baseline scenario obtained through the calibration of the models may differ from observed data, which are also available in Annex III, due to calibration errors).

Incremental water charges reduce the profitability (attribute z_1 in the microeconomic model) of irrigated crops, a key attribute driving utility and agent's decisions. As water charges increase, agents replace marginal irrigated crops by rainfed crops (Fig. 3a). This involves the substitution of irrigated cereals such as wheat, maize and barley (the latter two included in the category 'other cereals'), and sunflower ('oil seeds'), by rainfed cereals, notably barley, rye and oats. Accordingly, the surface of 'other cereals' consistently increases along higher water charges. Trends in the substitution of irrigated by rainfed crops are not uniform. In the range 0–0.05 EUR/m^3, irrigated crops at the margin can

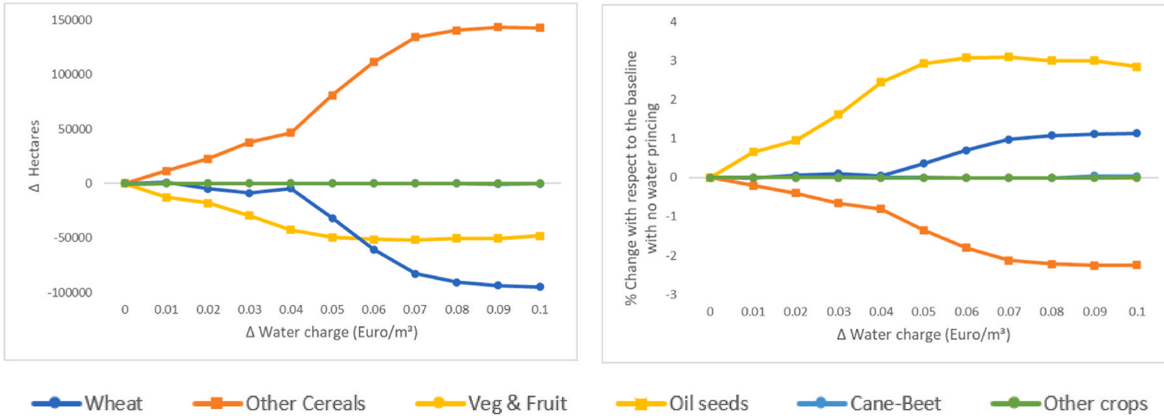


Fig. 3. Equilibrium crop portfolio choices (a) and crop prices (b), for selected water charging simulations. *The curves of Veg&Fruits, Cane-Beet and Other crops experience marginal surface and price changes and are all hidden behind the curve of Other crops.*

absorb the water charge increase to a large extent, and changes in the irrigated surface of sunflower, maize, barley and wheat are relatively small. However, a water charge increase of 0.05 EUR/m^3 or higher significantly undermines the competitiveness of these irrigated crops relative to rainfed crops, and the irrigated surface experiences a more pronounced reduction. The irrigated area of vegetables and fruits, sugar beet (main crop in the 'cane-beet' category) and alfalfa (main crop in the 'other crops' category) remains essentially constant for all water charging simulations except for the last one (0.1 EUR/m^3), where there is a slight reduction. These relatively water-intensive crop categories display a considerably higher profitability than existing alternatives in the baseline without water charging, and remain competitive after absorbing the water charge shock in the simulations considered.

Importantly, land use reallocations among crops induce scarcity or excess supply of agricultural commodities, which causes crop prices and profitability to increase (water-intensive crops whose surface is reduced) or decrease (rainfed crops whose surface increases) (Fig. 3b). The impact of this trend is particularly relevant in the case of sunflower, whose surface initially decreases along incremental water charging in the range 0–0.05 EUR/m^3, then stabilizes in the range 0.05–0.07 EUR/m^3 as oil seed prices grow, and finally increases in the range 0.07–0.1 EUR/m^3. Critically, these results suggest the importance of considering macro-economic price feedbacks and also that the "small open economy" approach (Schöb, 1998) that assumes perfectly elastic demand, often used in microeconomic agricultural water allocation models (Graveline, 2016), is unfit for basin-wide applications, as shown here for the case of the DRB.

Finally, Fig. 4 provides a spatially disaggregated account of profit and water use changes under incremental water charges in the DRB's ADs. Average profit and water use decrease consistently along incremental water charges, although the former at a higher rate than the latter: the average cost of water conservation in terms of foregone profit is estimated at 0.008 EUR/m^3 in the 0.01 EUR/m^3 water charging simulation, rising up to .04 EUR/m^3 in the 0.1 EUR/m^3 water charging simulation. This is consistent with simulated crop portfolio responses, which predict a substitution of increasingly valuable irrigated crops (from irrigated cereals and sunflower to sugar beet and vegetables and fruits) by rainfed crops as water charges grow.

3.2. Implications on the human-water system

As water charges increase and ADs opt-out to reduce the amount of

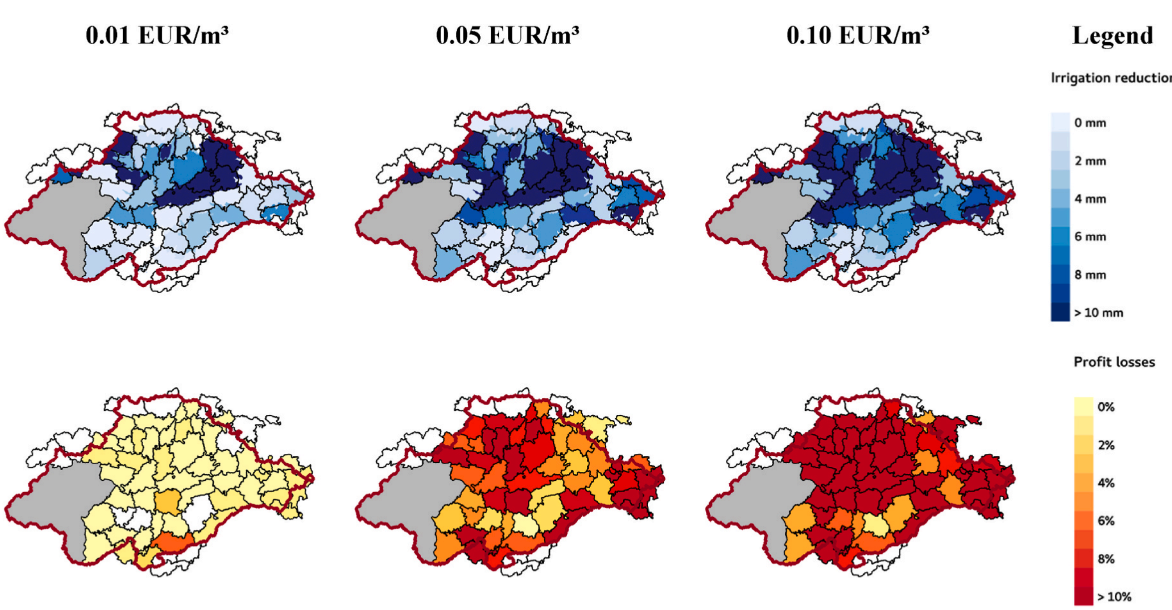
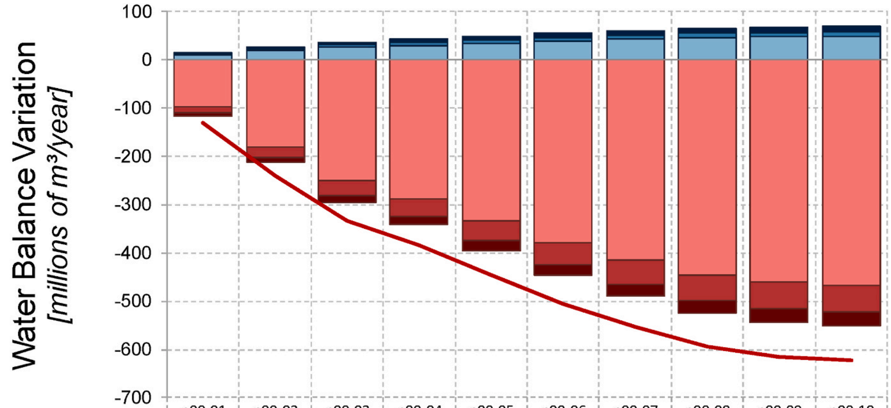


Fig. 4. Heterogeneity across space in water conservation and profit losses. Irrigation reduction is expressed in mm of water as averaged over the respective ADs. Profit losses are expressed in relative values with respect to the baseline scenario. The grey-shaded area indicates the portion of the DRB that is located in Portugal, and thus not considered in this study.



	p00.01	p00.02	p00.03	p00.04	p00.05	p00.06	p00.07	p00.08	p00.09	p00.10
Lateral Flow	2.5	4.6	6.4	7.3	8.5	9.6	10.5	11.3	11.7	11.9
Baseflow	1.8	3.3	4.6	5.3	6.1	6.9	7.6	8.2	8.5	8.6
Groundwater Evap.	10.2	18.8	26.2	30.2	35.0	39.7	43.5	46.7	48.4	49.1
Percolation	-6.0	-11.0	-15.3	-17.6	-20.4	-23.1	-25.3	-27.2	-28.1	-28.5
Surface Runoff	-11.7	-21.5	-29.9	-34.5	-40.0	-45.2	-49.5	-53.1	-55.0	-55.8
Evapotranspiration	-97.7	-180.3	-250.1	-288.5	-334.4	-378.3	-414.5	-444.8	-460.7	-467.4
Irrigation Applications	-130.2	-240.5	-333.5	-384.7	-446.0	-504.5	-552.7	-593.2	-614.3	-623.3

Fig. 5. Land phase water balance variation with respect to the baseline scenario (i.e. no water charging policy) of the main hydrologic components with respect to incremental water charging policy in the DRB.

water withdrawals and switch to less water demanding crops in order to mitigate profit losses, land management changes affect the circulation of water resources basin-wide (Fig. 5).

The reduction of the amount of irrigation water withdrawals predicted in the human-human (micro-macro-economic) system follows a logarithmic functional form, varying from 130.2 million m^3 for a water charge of 0.01 EUR/m^3 to 623.3 million m^3 for a 0.10 EUR/m^3 water charge. Those reductions directly impact the water balance of the DRB by altering several hydrologic processes, the main being evapotranspiration and surface runoff. As irrigation systems are often designed to convert irrigated water to evapotranspiration with minimal losses, it is not a surprise that evapotranspiration is the main hydrologic cycle's component affected by the implementation of a water charging policy. In fact, the reduction in evapotranspiration is estimated to vary between 97.7 million m^3 for a water charge of 0.01 EUR/m^3 to 467.4 million m^3 for a 0.10 EUR/m^3 water charge. Moreover, as less water is withdrawn from surface water sources and applied over land areas, processes such as surface runoff and percolation are expected to be reduced, the later potentially leading to reductions in the recharge rate of aquifer systems. In contrast, natural processes may offset the reduction in irrigation water applications, in particular the groundwater evapotranspiration process. As irrigation applications are reduced due to the incremental water charging policy, soil moisture may be gradually reduced, leading to a negative pressure in the soil profile. As a consequence, water may naturally move upward from the underlying soil layers and aquifer systems. The increased groundwater evapotranspiration and the reduced percolation rates can potentially lead to unforeseen pressures on groundwater sources (Condon et al., 2020); this however, is out of the scope of the current study and would require a spatially-explicit groundwater flow processes model to fully capture the dynamics

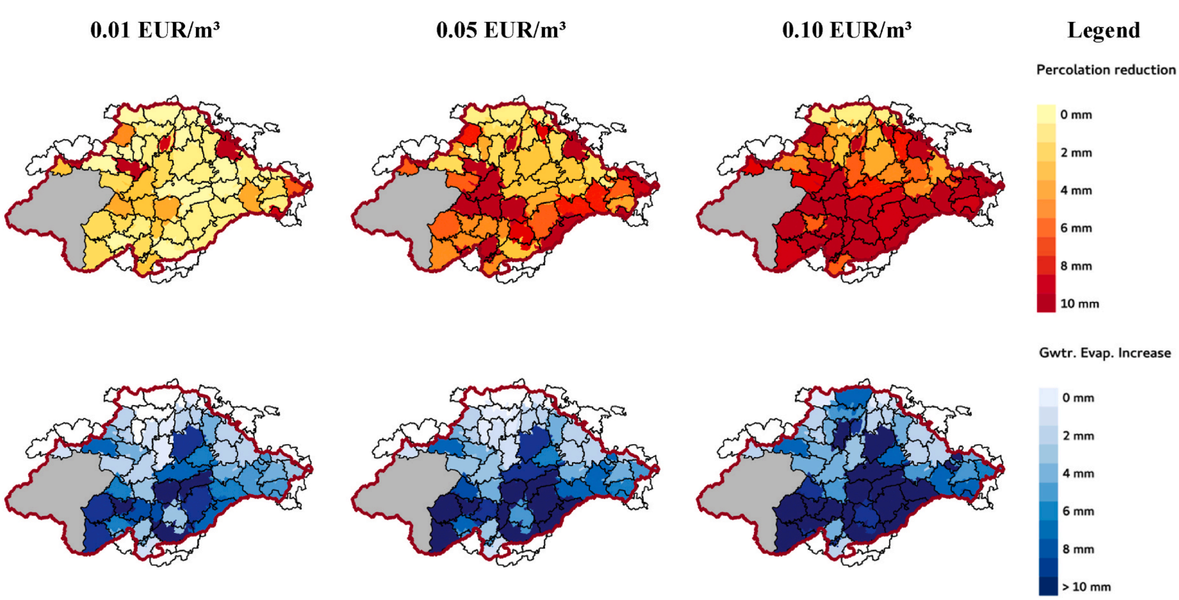


Fig. 6. Spatial incremental pressures on groundwater sources due to the reduced percolation and increased groundwater evapotranspiration dynamics. Values are expressed in mm of water as averaged over the respective subbasins. The grey-shaded area indicates the portion of the DRB that is located in Portugal, and thus not considered in this study.

between land surface and groundwater dynamics. The spatial increment of pressures on groundwater sources due to the reduced percolation and increased groundwater evapotranspiration dynamics is spatially disaggregated by AD in Fig. 6.

In summary, irrigators' responses through land (and related water) reallocations can significantly affect hydrologic processes, such as the alteration in the potential to which rainfall is converted to runoff. For instance, soil water content variations due to changing irrigation patterns can directly affect runoff generation; a similar thinking is valid for land cover and land use changes due to the adoption of different crop portfolios. Ultimately, the amount of water that can be reduced from irrigation withdrawals following the implementation of the assessed policy depends on the preferences of the ADs, which in turn are subject to a set of criteria and constrains (see Section 2). Indeed, as shown in Fig. 6, decisions taken by ADs are not spatially homogeneous, and localized decisions can affect hydrologic process occurring in their own properties as well as in their neighbors' lands, particularly in downstream areas. In this sense, a holistic, multi-factor perspective, capable of accounting not only for eco-hydrologic consequences but also implications in the micro-and-macro-economy, is required to evaluate the impacts of water policy interventions in complex human-water systems.

4. Conclusions

This paper develops a protocol-based recursive modular framework for capturing the richness of interactions in complex human-water systems by assessing the two-way feedbacks between hydrologic, microeconomic and macro-economic systems. Methods are illustrated with an application to water charges in irrigated agriculture in the Spanish part of the DRB. We conclude that the integration of hydrologic, micro- and macro-economic modules through a set of protocols can provide additional information about the systemic complexity of human-water systems and potentially improve the design of agricultural water policies, notably by accounting for both the price feedbacks between micro- and macro-economic systems and the non-linear responses of economic agents in water systems.

The coupling framework developed in this paper is designed to be replicable and flexible, capable of including alternative micro-, macroeconomic and hydrologic models that are better suited to represent water demand and/or supply challenges elsewhere. Standard agricultural microeconomic models that can be incorporated to the coupling framework include Expected Utility (von Neumann and Morgenstern, 1953), Linear Programming (Paris, 2015), Positive Mathematical Programming (Howitt, 1995), Multi-criteria Decision Models (Pereira et al., 2003; Sumpsi et al., 1997) and Positive Multi-Attribute Utility Programming models (Essenfelder et al., 2018; Gutiérrez-Martín and Gómez, 2011). Macroeconomic standard models that can be incorporated to the coupling framework include Computable General Equilibrium (CGE) (Hertel and Liu, 2016) and Input Output (IO) models (Oosterhaven and Bouwmeester, 2016). Ideally, macroeconomic models should be regionally-calibrated (NUTS 2 scale or similar) to increase the spatial disaggregation of the shares of value added, and accordingly of labor, capital, natural resources and land, and the accuracy of the coupling with microeconomic models (Carrera et al., 2015; Koks et al., 2015). It should be noted that CGE models are capable of representing land use and of simulating the price dynamics that allow for the recursive programming in step 3; while alternative IO models offer a one-way sequential coupling using a proxy for land use change (e.g. changes in gross value added) (Pérez-Blanco et al., 2017). Finally, a large pool of hydrologic models is available in the literature (see e.g. Texas A&M University, 2018), but not all of them can be integrated in the coupling framework above. To be compatible with economic models in the modeling framework proposed in this paper, hydrologic models must meet the following criteria: i) be spatially distributed (i.e. fully or semi-distributed models, to be capable of spatially representing different microeconomic agents); and ii) have a land management module (i.e. be capable of translating the crop portfolio choices taken by the different microeconomic agents into hydrological responses). A non-exhaustive list of models satisfying these criteria includes: the Soil and Water Assessment Tool – SWAT (Arnold et al., 1998); Annualized Agricultural Non-Point Source Pollution Model – AnnAGNPS (Young et al., 1989); Areal Nonpoint Source Watershed Environment Response Simulation – ANSWERS 2000 (Bouraoui and Dillaha, 2000); Agricultural Policy/-Environmental eXtender Model – APEX (Gassman et al., 2009); US Army Corps of Engineers - Hydrologic Engineering Center - Hydrologic Modeling System – HEC-HMS (US Army Corps of Engineers, 2015); European Hydrologic System – MIKE-SHE (Refsgaard and Storm, 1995); and Soil and Water Integrated Model – SWIM (Krysanova et al., 2005).

We envision several ways in which the proposed framework and research could be improved. First, given that the flexible hydro-micro-macro-economic framework developed in this paper can be populated with alternative models at the level of each module (see above), this enables the development of multi-model ensembles at each system level. A natural follow-up to our research is therefore the development of multi-system and multi-model ensembles that thoroughly sample parameter and structural uncertainties in models, as well as cascading uncertainties across systems. Coupled with scenario discovery methods that sample scenario uncertainty (e.g., considering multiple water charging scenarios), the resultant multi-system ensemble can be used to build a large dataset of plausible futures; which can help detect vulnerabilities to proposed adaptation strategies and identify robust adaptation strategies that display a satisfactory performance under most plausible futures. Second, individual models within each module can be also improved by incorporating recent scientific developments within each discipline. For example, recent developments in microeconomic modeling dislodge land use choices from water use choices, meaning crop portfolio choices need not be linearly related to water application (Graveline and Mérel, 2014; Loch et al., 2019; Sapino et al., 2022). This makes possible the representation and assessment of adaptation responses at the intensive margin (deficit/supplementary irrigation), beyond the extensive (shift to less water intensive crops) and super-extensive (shift to rainfed crops) margin adaptations studied in conventional microeconomic models. Note that these new microeconomic models could be incorporated to our modular recursive framework with relatively minor changes in the set of protocols. Third, higher granularity can be achieved in human system models through a more detailed representation of agents in micro- (e.g. at Agricultural Water Demand Unit instead of AD level, which in the DRB would yield 10x more agents) and macro-economic (e.g. at NUTS3—province—instead of NUTS2 level—region, which in Spain would yield 50 instead of 17 agents) models, although this would involve also a more computationally-demanding process. Fourth, additional modules could be added to the modeling framework, e.g., through the coupling with climate and/or crop models to assess climate change impacts on water availability (climate and hydrological models) and on crop yields (climate and crop models). Finally, additional relevant protocols across systems could be added to enrich the framework and its interactions, e.g. to incorporate water use separated from land use decisions in the coupling between the micro-economic and the hydrologic model; or to incorporate macro-micro-economic linkages through price changes in agricultural inputs.

CRediT authorship contribution statement

C. Dionisio Pérez-Blanco: Conceptualization, Methodology, Validation, Resources, Writing – original draft, Writing – review & editing, Supervision, Project administration, Funding acquisition. **Ramiro Parrado:** Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Writing – original draft, Writing – review & editing, Supervision. **Arthur H. Essenfelder:** Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Writing – original draft, Writing – review & editing, Supervision. **José Bodoque:**

Methodology, Formal analysis, Investigation, Project administration, Funding acquisition. **Laura Gil-García:** Software, Investigation, Data curation. **Carlos Gutiérrez-Martín:** Methodology, Formal analysis, Investigation, Validation. **Julián Ladera:** Software, Investigation, Data curation. **Gabriele Standardi:** Methodology, Software, Investigation, Data curation, Writing – review & editing.

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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2022.132208.

References

Abbaspour, K.C., 2012. SWAT-CUP 2012: SWAT Calibration and Uncertainty Programs - A User Manual. Swiss Federal Institute of Aquatic Science and Technology, Dübendorf (Switzerland).

Abbaspour, K.C., Yang, J., Maximov, I., Siber, R., Bogner, K., Mieleitner, J., Zobrist, J., Srinivasan, R., 2007. Modelling hydrology and water quality in the pre-alpine/alpine Thur watershed using SWAT. J. Hydrol. 333, 413–430. https://doi.org/10.1016/j.jhydrol.2006.09.014.

AGUAMOD, 2021. AGUAMOD. website [WWW Document]. Proj. Website. URL. http://www.aguamod-sudoe.eu/es/.

Armington, P.S., 1969. A theory of demand for products distinguished by place of production. IMF Staff Pap. 16, 159–178.

Arnold, J.G., Srinivasan, R., Muttiah, R.S., Williams, J.R., 1998. Large area hydrologic modeling and assessment Part I: model development. J. Am. Water Resour. Assoc. 34, 73–89. https://doi.org/10.1111/j.1752-1688.1998.tb05962.x.

Baldassarre, G.D., Martinez, F., Kalantari, Z., Viglione, A., 2017. Drought and flood in the Anthropocene: feedback mechanisms in reservoir operation. Earth Syst. Dyn. 8, 225–233. https://doi.org/10.5194/esd-8-225-2017.

Blair, P., Buytaert, W., 2016. Socio-hydrological modelling: a review asking "why, what and how?". Hydrol. Earth Syst. Sci. 20, 443–478. https://doi.org/10.5194/hess-20-443-2016.

Bosello, F., Standardi, G., 2015. A Sub-national CGE Model for the European Mediterranean Countries (Working Paper No. RP0274), CMCC Research Papers. CMCC, Lecce (Italy).

Bosello, F., Nicholls, R.J., Richards, J., Roson, R., Tol, R.S.J., 2012. Economic impacts of climate change in Europe: sea-level rise. Clim. Change 112, 63–81. https://doi.org/10.1007/s10584-011-0340-1.

Bouraoui, F., Dillaha, T., 2000. ANSWERS-2000: non-point-source nutrient planning model. J. Environ. Eng. 126, 1045–1055. https://doi.org/10.1061/(ASCE)0733-9372(2000)126:11(1045).

Bressiani, D. de A., Gassman, P.W., Fernandes, J.G., Garbossa, L.H.P., Srinivasan, R., Bonumá, N.B., Mendiondo, E.M., 2015. A review of soil and water assessment tool (SWAT) applications in Brazil: challenges and prospects. Int. J. Agric. Biol. Eng. 8, 1–27. https://doi.org/10.3965/j.ijabe.20150803.1765.

Brutsaert, W., 2013. Hydrology: an Introduction, eighth ed. Cambridge University Press, Cambridge, United Kingdom.

Cai, Y., Arora, V., 2015. Disaggregating electricity generation technologies in CGE models: a revised technology bundle approach with an application to the U.S. Clean Power Plan. Appl. Energy 154, 543–555. https://doi.org/10.1016/j.apenergy.2015.05.041.

Carrera, L., Standardi, G., Bosello, F., Mysiak, J., 2015. Assessing direct and indirect economic impacts of a flood event through the integration of spatial and computable general equilibrium modelling. Environ. Model. Software 63, 109–122. https://doi.org/10.1016/j.envsoft.2014.09.016.

Condon, L.E., Atchley, A.L., Maxwell, R.M., 2020. Evapotranspiration depletes groundwater under warming over the contiguous United States. Nat. Commun. 11, 873. https://doi.org/10.1038/s41467-020-14688-0.

Crutzen, P.J., 2002. Geology of mankind. Nature 415, 23. https://doi.org/10.1038/415023a.

Csete, M.E., Doyle, J.C., 2002. Reverse engineering of biological complexity. Science 295, 1664–1669. https://doi.org/10.1126/science.1069981.

Damania, R., Desbureaux, S., Hyland, M., Islam, A., Moore, S., Rodella, A.-S., Russ, J., Zaveri, E., 2017. Uncharted Waters: the New Economics of Water Scarcity and Variability. World Bank Publications, S.l.

Di Baldassarre, G., Viglione, A., Carr, G., Kuil, L., Salinas, J.L., Blöschl, G., 2013. Socio-hydrology: conceptualising human-flood interactions. Hydrol. Earth Syst. Sci. 17, 3295–3303. https://doi.org/10.5194/hess-17-3295-2013.

Dixon, P., Rimmer, M.T., Wittwer, G., 2012. Buybacks to restore the Southern Murray-Darling basin. In: Wittwer, G. (Ed.), Economic Modeling of Water, Global Issues in Water Policy. Springer Netherlands, pp. 99–118. https://doi.org/10.1007/978-94-007-2876-9_6.

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

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).

EC, 2009. Guidance on Groundwater Status and Trend Assessment (Technical Report No. 2015– 026, No. 18), CIS Guidance Document. European Commission, Brussels (Belgium).

EC, 2015a. Technical Report on Groundwater Associated Aquatic Ecosystems (Technical Report No. 2015– 093). European Commission, Brussels (Belgium).

EC, 2015b. Ecological Flows in the Implementation of the Water Framework Directive (Technical Report No. 2015– 086, No. 31), CIS Guidance Document. European Commission, Brussels (Belgium).

EEA, 2016. Water exploitation index [WWW Document]. Water Exploit. Index. URL. http://www.eea.europa.eu/data-and-maps/indicators/water-exploitation-index, 3.30.16.

Essenfelder, A.H., Pérez-Blanco, C.D., Mayer, A.S., 2018. Rationalizing systems analysis for the evaluation of adaptation strategies in complex human-water systems. Earth's Future 6, 1181–1206. https://doi.org/10.1029/2018EF000826.

Esteve, P., Varela-Ortega, C., Blanco-Gutiérrez, I., Downing, T.E., 2015. A hydro-economic model for the assessment of climate change impacts and adaptation in irrigated agriculture. Ecol. Econ. 120, 49–58. https://doi.org/10.1016/j.ecolecon.2015.09.017.

Eurostat, 2020. Eurostat database [WWW Document]. Eurostat Database. URL. http://epp.eurostat.ec.europa.eu/portal/page/portal/statistics/themes, 10.5.13.

FAO, 2021. FaoStat [WWW Document]. Food Agric. Organ. U. N. URL. http://faostat.fao.org/, 2.10.21.

Gassman, P., Williams, J., Wang, X., 2009. The Agricultural Policy Environmental Extender (APEX) Model: an Emerging Tool for Landscape and Watershed Environmental Analyses (Report No. 41), CARD Technical Reports. CARD.

Gómez-Limón, J.A., Gutiérrez-Martín, C., Riesgo, L., 2016. Modeling at farm level: positive multi-attribute utility programming. Omega. https://doi.org/10.1016/j.omega.2015.12.004.

Graveline, N., 2016. Economic calibrated models for water allocation in agricultural production: a review. Environ. Model. Software 81, 12–25. https://doi.org/10.1016/j.envsoft.2016.03.004.

Graveline, N., Mérel, P., 2014. Intensive and extensive margin adjustments to water scarcity in France's Cereal Belt. Eur. Rev. Agric. Econ. 41, 707–743. https://doi.org/10.1093/erae/jbt039.

Gutiérrez-Martín, C., Gómez, C.M., 2011. Assessing irrigation efficiency improvements by using a preference revelation model. Spanish J. Agric. Res. 9, 1009–1020. https://doi.org/10.5424/sjar/20110904-514-10.

Gutiérrez-Martín, C., Pérez-Blanco, C.D., Gómez, C.M., Berbel, J., 2014. Price volatility and water demand in agriculture. A case study of the Guadalquivir river basin (Spain). In: Economics of Water Management in Agriculture, pp. 319–348. Bournaris, T., J. Berbel, B. Manos, D. Viaggi, Boca Raton (US).

Haasnoot, M., Kwakkel, J.H., Walker, W.E., ter Maat, J., 2013. Dynamic adaptive policy pathways: a method for crafting robust decisions for a deeply uncertain world. Global Environ. Change 23, 485–498. https://doi.org/10.1016/j.gloenvcha.2012.12.006.

Hanoch, G., 1971. CRESH production functions. Econometrica 39, 695–712. https://doi.org/10.2307/1909573.

Harou, J.J., Pulido-Velazquez, M., Rosenberg, D.E., Medellín-Azuara, J., Lund, J.R., Howitt, R.E., 2009. Hydro-economic models: concepts, design, applications, and future prospects. J. Hydrol. 375, 627–643. https://doi.org/10.1016/j.jhydrol.2009.06.037.

Hasegawa, T., Fujimori, S., Masui, T., Matsuoka, Y., 2016. Introducing detailed land-based mitigation measures into a computable general equilibrium model. J. Clean. Prod. 114, 233–242. https://doi.org/10.1016/j.jclepro.2015.03.093. Towards Post Fossil Carbon Societies: Regenerative and Preventative Eco-Industrial Development.

Hertel, T.W. (Ed.), 1997. Global Trade Analysis: Modeling and Applications. Cambridge University Press, Cambridge; New York.

Hertel, T.W., Liu, J., 2016. Implications of Water Scarcity for Economic Growth (OECD Environment Working Papers). Organisation for Economic Co-operation and Development, Paris.

Howitt, R.E., 1995. Positive mathematical programming. Am. J. Agric. Econ. 77, 329–342. https://doi.org/10.2307/1243543.

Koks, E.E., Carrera, L., Jonkeren, O., Aerts, J.C.J.H., Husby, T.G., Thissen, M., Standardi, G., Mysiak, J., 2015. Regional disaster impact analysis: comparing Input-Output and Computable General Equilibrium models. Nat. Hazards Earth Syst. Sci. 3, 7053–7088. https://doi.org/10.5194/nhessd-3-7053-2015.

Krysanova, V., Arnold, J.G., 2008. Advances in ecohydrological modelling with SWAT—a review. Hydrol. Sci. J. 53, 939–947. https://doi.org/10.1623/hysj.53.5.939.

Krysanova, V., Hattermann, F., Wechsung, F., 2005. Development of the ecohydrological model SWIM for regional impact studies and vulnerability assessment. Hydrol. Process. 19, 763–783. https://doi.org/10.1002/hyp.5619.

Kwakkel, J.H., Haasnoot, M., Walker, W.E., 2015. Developing dynamic adaptive policy pathways: a computer-assisted approach for developing adaptive strategies for a deeply uncertain world. Clim. Change 132, 373–386. https://doi.org/10.1007/s10584-014-1210-4.

Loch, A., Adamson, D., Auricht, C., 2019. (g)etting to the point: the problem with water risk and uncertainty. Water Resour. Econ., 100154 https://doi.org/10.1016/j.wre.2019.100154.

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).

MAPAMA, 2019. Infraestructura de Datos Espaciales (IDE) [WWW Document]. Minist. Agric. Pesca Aliment. URL. https://www.mapa.gob.es/es/cartografia-y-sig/ide/descargas/agricultura/default.aspx.

Muth, J.F., 1961. Rational expectations and the theory of price movements. Econometrica 29, 315–335. https://doi.org/10.2307/1909635.

Neitsch, S.L., Arnold, J.G., Kiniry, J.R., Williams, J.R., 2011. Soil & Water Assessment Tool Theoretical Documentation. Version 2009 (No. TR-406). Texas A&M AgriLife, USDA Agricultural Rsearch Service, College Station, TX, USA.

OJ, 2000. Water Framework Directive 2000/60/EC. Council Directive.

Oosterhaven, J., Bouwmeester, M.C., 2016. A new approach to modeling the impact of disruptive events. J. Reg. Sci. 56, 583–595. https://doi.org/10.1111/jors.12262.

Pande, S., Sivapalan, M., 2017. Progress in socio-hydrology: a meta-analysis of challenges and opportunities. WIREs Water 4, 1–18. https://doi.org/10.1002/wat2.1193.

Pant, H., 2007. GTEM: Global Trade and Environment Model (ABARE Technical Report). Australian Bureau of Agricultural and Resource Economics and Sciences, Canberra (Australia).

Paris, Q., 2015. An Economic Interpretation of Linear Programming, first ed. 2015 edition. Palgrave Macmillan, Houndmills, Basingstoke, Hampshire ; New York City, NY.

Parrado, R., Pérez-Blanco, C.D., Gutiérrez-Martín, C., Standardi, G., 2019. Micro-macro feedback links of agricultural water management: insights from a coupled iterative positive Multi-Attribute Utility Programming and Computable General Equilibrium model in a Mediterranean basin. J. Hydrol. 569, 291–309. https://doi.org/10.1016/j.jhydrol.2018.12.009.

Parrado, R., Pérez-Blanco, C.D., Gutiérrez-Martín, C., Gil-García, L., 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. Sci. Total Environ. 742, 140526 https://doi.org/10.1016/j.scitotenv.2020.140526.

Pereira, L.S., Gonçalves, J.M., Campos, A., Fabiao, M., 2003. Demand and delivery simulation and multi-criteria analysis for water saving. In: Water Savings in the Yellow River Basin. Issues and Decision Support Tools in Irrigation. China Agriculture Press, Beijing, China, pp. 247–274.

Pérez-Blanco, C.D., Standardi, G., 2019. Farm waters run deep: a coupled positive multi-attribute utility programming and computable general equilibrium model to assess the economy-wide impacts of water buyback. Agric. Water Manag. 213, 336–351. https://doi.org/10.1016/j.agwat.2018.10.039.

Pérez-Blanco, C.D., Delacámara, G., Gómez, C.M., 2016. Revealing the willingness to pay for income insurance in agriculture. J. Risk Res. 19, 873–893. https://doi.org/10.1080/13669877.2015.1042505.

Pérez-Blanco, C.D., Koks, E.E., Calliari, E., Mysiak, J., 2017. Economic impacts of irrigation-constrained agriculture in the lower Po basin. Water Econ. Policy, 1750003. https://doi.org/10.1142/S2382624X17500035, 04.

Pérez-Blanco, C.D., Hrast-Essenfelder, A., Perry, C., 2020. Irrigation technology and water conservation: a review of the theory and Evidence. Rev. Environ. Econ. Pol. 14, 216–239. https://doi.org/10.1093/reep/reaa004.

Ratter, B.M.W., 2012. Complexity and Emergence – key concepts in non-linear dynamic systems. In: Glaser, M., Krause, G., Ratter, B., Welp, M. (Eds.), Human-Nature Interactions in the Anthropocene: Potentials of Social-Ecological Systems Analysis. Routledge, New York, NY, USA.

Ronneberger, K., Berrittella, M., Bosello, F., Tol, R.S.J., 2009. KLUM@GTAP: introducing biophysical aspects of land-use decisions into a computable general equilibrium model a coupling experiment. Environ. Model. Assess. 14, 149–168. https://doi.org/10.1007/s10666-008-9177-z.

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

Schöb, R., 1998. Ecological tax reforms and the environment: a note. Bull. Econ. Res. 50, 83–89. https://doi.org/10.1111/1467-8586.00053.

Sivapalan, M., Savenije, H.H.G., Blöschl, G., 2012. Socio-hydrology: a new science of people and water. Hydrol. Process. 26, 1270–1276. https://doi.org/10.1002/hyp.8426.

Sivapalan, M., Konar, M., Srinivasan, V., Chhatre, A., Wutich, A., Scott, C.A., Wescoat, J. L., 2014. Socio-hydrology: use-inspired water sustainability science for the Anthropocene. Earth's Future 2, 225–230. https://doi.org/10.1002/2013EF000164.

Sumpsi, J., Amador, F., Romero, C., 1997. On farmers' objectives: a multi-criteria approach. Eur. J. Oper. Res. 96, 64–71. https://doi.org/10.1016/0377-2217(95)00338-X.

Taheripour, F., Hertel, T.W., Narayanan, B., Sahin, S., Markandya, A., Mitra, B.K., 2016. Economic and land use impacts of improving water use efficiency in irrigation in South Asia. J. Environ. Protect. 1571. https://doi.org/10.4236/jep.2016.711130, 07.

Texas A&M University, 2018. Hydrologic models [WWW Document]. Hydrol. Models. URL. http://hydrologicmodels.tamu.edu/models.htm, 3.2.18.

Turnbull, L., Hütt, M.-T., Ioannides, A.A., Kininmonth, S., Poeppl, R., Tockner, K., Bracken, L.J., Keesstra, S., Liu, L., Masselink, R., Parsons, A.J., 2018. Connectivity and complex systems: learning from a multi-disciplinary perspective. Appl. Netw. Sci. 3, 1–49. https://doi.org/10.1007/s41109-018-0067-2.

Ullrich, A., Volk, M., 2009. Application of the Soil and Water Assessment Tool (SWAT) to predict the impact of alternative management practices on water quality and quantity. Agric. Water Manag. 96, 1207–1217. https://doi.org/10.1016/j.agwat.2009.03.010.

US Army Corps of Engineers, 2015. HEC-HMS Hydrologic Modeling System. User's Manual - Version 4.1 (User's Manual No. CPD-74A V. 4.1). US Army Corps of Engineers, Davis, California (US).

von Neumann, J., Morgenstern, O., 1953. Theory of Games and Economic Behavior. Princeton University Press, Princeton (US).

2030 Water Resources Group, 2019. The 2030 Water Resources Group Annual Report (Report No. 2019). World Bank, Washington D.C. (US).

Winchell, M., Srinivasan, R., Di Luzio, M., Arnold, J., 2007. ArcSWAT Interface for SWAT2005 User's Guide. Texas A&M AgriLife, USDA Agricultural Rsearch Service, Temple, TX, USA.

Young, R.A., Onstad, C.A., Bosch, D.D., Anderson, W.P., 1989. AGNPS: a nonpoint-source pollution model for evaluating agricultural watersheds. J. Soil Water Conserv. 44, 168–173.

Zhang, X., Srinivasan, R., Hao, F., 2007. Predicting hydrologic response to climate change in the Luohe river Basin using the SWAT model. Am. Soc. Agric. Biol. Eng. 500 https://doi.org/10.13031/2013.23154, 901–310.