Sustainable water management in the agricultural sector under deep uncertainty

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Sustainable water management in the agricultural sector under deep uncertainty

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Contents

List of Abbreviations Used in the Thesis

Abbreviation	Definition
%	Percentage
ABM	Agent-Based Model
AdBPo	Autorità di Bacino distrettuale del fiume Po
AgMIP	Agricultural Model Intercomparison and Improvement Project
CAPRI	Common Agricultural Policy Regionalised Impact Modelling System
CES	Constant Elasticity of Substitution
CIMIP	Coupled Model Intercomparison Project
DMDU	Decision Making under Deep Uncertainty
DSS	Decision Support System
EU	European Union
FAO	Food and Agriculture Organization
GDP	Gross Domestic Product
HEPEX	Hydrologic Ensemble Prediction EXperiment
IPCC	Intergovernmental Panel on Climate Change
ISIMIP	Inter-Sectoral Impact Model Intercomparison Project
MPM	Mathematical Programming Model
OECD	Organisation for Economic Co-operation and Development
Ph.D.	Doctor of Philosophy
PM	Positive Model
PMAUP	Positive Multi-Attribute Utility Programming
PMP	Positive Mathematical Programming
PWS	Payment for Watershed Scheme
RDM	Robust Decision Making
TAMAL	Take severAl Models And combine them alL
TAP	Transformational Adaptation Policy
TAPAS	Take A Previous model and Add Something
UN	United Nations
UNDRR	United Nations Office for Disaster Risk Reduction
UNEP	United Nations Environmental Programme
UNESCO	United Nations Educational, Scientific and Cultural Organization
WCD	World Commission on Dams
WFD	Water Framework Directive
WPM	Water Programming Model
WRI	World Resources Institute

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Organización de la tesis doctoral - Organization of the doctoral thesis

De acuerdo con el RD 99/2011, de 28 de enero, por el que se regulan las enseñanzas oficiales de doctorado, la Comisión de Doctorado y Posgrado de la Universidad de Salamanca establece como posible formato de presentación de Tesis Doctoral, la modalidad de Tesis por Compendio de Artículos/Publicaciones, publicados o aceptados en revistas especializadas y de prestigio. Así, la presente Tesis Doctoral se presenta bajo esta modalidad, optando a la mención de Doctorado Internacional. Las publicaciones incluidas en este compendio son los siguientes:

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Francesco Sapino[1]; C. Dionisio Pérez Blanco[1]; Carlos Gutiérrez Martín[2]; Vito Frontuto[3]. *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, 01/08/2020.
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C. Dionisio Pérez Blanco[1]; **Francesco Sapino**[1], Pablo Sainz-Santiago[5]. *First-degree price discrimination water bank to reduce reacquisition costs and enhance economic efficiency in agricultural water buyback.* Ecological Economics. 205, 01/03/2023.
DOI: https://doi.org/10.1016/j.ecolecon.2022.107694
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Abstract

The world is experiencing a global water crisis, caused by the compounded effects of overexploitation, population growth, mismanagement, and climate change. The design of water-adaptation strategies to address this crisis has traditionally relied on consolidative models that offer decision-makers point predictions on economic impacts, water conservation, etc. However, in recent decades, nonlinearities (e.g., in climate change, adaptive behavior) have challenged the reliability of these models and overwhelmed existing policies, which have systemically failed to achieve their targets due to new correlations across complex and interconnected socioeconomic and ecological systems that were not previously anticipated. In this context, planning for the future is characterized by a high degree of uncertainty, or deep uncertainty. Under deep uncertainty, we cannot associate probabilities to outcomes (as with risk), and therefore we cannot individuate with confidence a single strategy that is expected to outperform the alternatives. Instead of looking for optimality, under deep uncertainty, decision-makers should prioritize robustness, i.e., the identification of the strategy(ies) that achieve the objective of sustainable and equitable economic growth in the most plausible futures. This calls for water reallocations from economic to environmental uses in order to guarantee the good ecological status of ecosystems, complemented by reallocations among economic uses to enhance both efficiency and equity. Most reallocation strategies will target the agricultural sector, which is the largest water user and concentrates the marginal (i.e., least valuable) use of the resource. This thesis presents a modeling framework to design and inform robust adaptation strategies in the agricultural sector, structured in 5 chapters. In the first chapter, we introduce the topic, state the objective, and present the structure of the thesis. The second chapter presents the socioeconomic methodology used to assess farmers' behavior (Mathematical Programming Models – MPMs) and introduces a new model that allows deficit irrigation as an adaptation strategy to water scarcity, usually not considered in conventional MPMs. In the third chapter, we introduce a novel multi-model ensemble of MPMs to sample uncertainty and thus inform robust decisions. In chapter four, we explicitly include the water systems coupling the multi-model ensemble of MPMs with a decision support system model used to manage water at basin level. Finally, chapter five includes the conclusions and recommendations of the author. The main purpose of our modeling framework is to deliver actionable science. Thus, it is designed to be modular, updatable, and ready to apply by policymakers. Thanks to the collaboration with Italian and Spanish regulatory authorities, we have applied our modeling framework to 5 policy cases: we tested the performance of two pricing policies, and a water

bank to buyback water for the environment, we assessed a pecuniary compensation scheme designed to sustain irrigation-dependent ecosystem services, and we calculated the resource cost of agricultural water. Our multi-model ensemble can inform the identification and adoption of robust strategies that contribute to the targets of equitable and sustainable economic and welfare growth. Moreover, the explicit inclusion of the water system allows for the consideration of the co-evolution of the water and human systems, in order to avoid unfavorable outcomes triggered by the possible two-way feedback between these systems.

Resumen

El mundo está experimentando una crisis mundial del agua, causada por los efectos combinados de la sobreexplotación, del crecimiento demográfico, de la mala gestión y del cambio climático. Tradicionalmente las estrategias de adaptación para hacer frente a esta crisis se han basado en modelos de consolidación que ofrecen a los responsables políticos predicciones puntuales sobre los impactos económicos, la conservación del agua, etc. Sin embargo, en las últimas décadas, las no linealidades (por ejemplo, en el cambio climático, el comportamiento adaptativo, etc.) han desafiado la fiabilidad de estos modelos y abrumado las políticas existentes, que sistemáticamente no han logrado sus objetivos debido a nuevas correlaciones entre complejos e interconectados sistemas socioeconómicos y ecológicos que no se habían previsto previamente. En este contexto, la planificación para el futuro se caracteriza por un alto grado de incertidumbre o incertidumbre profunda. Bajo incertidumbre profunda, no podemos asociar probabilidades a los resultados (como con el riesgo), y por lo tanto no podemos destacar una sola estrategia que se espera actúe mejor de las alternativas con confianza. En lugar de la estrategia que mejor funciona, la incertidumbre obliga a los responsables políticos a dar prioridad a la robustez, es decir, a través de la identificación de la(s) estrategia(s) que logre(n) el objetivo de un crecimiento económico sostenible y equitativo bajo los futuros más plausibles. Esto requiere reasignaciones de agua de usos económicos a usos ambientales, para garantizar el buen estado ecológico de los ecosistemas, complementado con reasignaciones entre usos económicos para mejorar la eficiencia y la equidad. La mayoría de las estrategias de reasignación se dirigirán al sector agrícola, que es el mayor usuario de agua y concentra el uso marginal (es decir, el menos valioso) del recurso. Esta tesis presenta un marco de modelos para diseñar e informar estrategias de adaptación robustas en el sector agrícola, estructurado en 5 capítulos. En el primer capítulo presentamos el tema, el objetivo y la estructura de esta tesis. El segundo capítulo presenta la metodología socioeconómica utilizada para evaluar el comportamiento de los agricultores (Modelos de Programación Matemática - MPMs) y luego introduce un nuevo modelo que permite el riego deficitario como una estrategia de adaptación a la escasez de agua, generalmente no considerada en los MPMs convencionales. En el siguiente capítulo, presentamos un novedoso conjunto multimodelo de MPMs para reducir la incertidumbre y así fundamentar decisiones sólidas. A continuación, en el capítulo cuatro, incluimos explícitamente el sistema hidrológico juntando el conjunto de MPMs con un modelo de gestión del agua utilizado a nivel de cuenca. Finalmente, el capítulo cinco incluye las conclusiones y recomendaciones del autor. El propósito principal de nuestro sistema de modelos es ofrecer ciencia aplicable por los responsables políticos, por lo tanto, el sistema es modular, actualizable y listo para ser utilizado. Gracias a la colaboración con algunas

autoridades reguladoras italianas y españolas, aplicamos nuestro sistema de modelos a 5 casos de estudio: evaluamos el rendimiento de dos políticas de precios y un banco de agua para comprar agua para el medio ambiente, Evaluamos un esquema de compensación pecuniaria diseñado para sostener los servicios ecosistémicos dependientes del riego, y calculamos el costo de los recursos de agua agrícola. Nuestro sistema de modelos puede ayudar en la identificación y adopción de estrategias sólidas que contribuyan a los objetivos de crecimiento económico y de bienestar equitativo y sostenible. Además, la inclusión explícita del sistema hidrológico permite considerar la co-evolución de los sistemas humanos y natural, para evitar resultados desfavorables desencadenados por la posible retroalimentación bidireccional entre estos sistemas.

Chapter 1. Introduction

"We think we'll always have enough, especially in the developed world where fresh water is only ever a tap turn away. We don't realize that water is a finite resource that the planet is in serious danger of running out of"
Richard Mills, Extreme Heatwave's Global Warming Catastrophe, 2022.

1.1. Context: water scarcity and deep uncertainty

Freshwater is the most essential resource for life and almost every economic activity. Due to its historic abundance in many populated areas of the world, humans have used it carelessly, both in quantitative and qualitative terms. Freshwater, however, is finite, vulnerable, and unevenly distributed across time (dry and wet seasons) and space (dry and wet regions). More recently, population growth and economic development have generated an unprecedented increase in water demand. Over the last 100 years, freshwater use has grown by 600% and it is still growing today at a 1% annual rate. On the other hand, water availability in water-scarce regions is decreasing due to climate change[1] (UNESCO, 2020). As a result, freshwater scarcity and water stress, i.e., when the demand overcomes the renewable water supply, are growing almost everywhere in the world and are affecting human activities and the environment.
Conventional engineering policies that aim to expand the supply base are a key pillar upon which the greatest human civilizations have been built, allowing the necessary water and food supply to develop, e.g., the first large cities. The key target of engineering policies is to build and expand storage, transportation, and delivery infrastructures in order to adapt the water supply to the growing demand (e.g., the number of large dams has globally exploded from 5,000 in 1950 to 45,000 in 2000) (WCD, 2000). Most recently, however, this approach has shown its limitations. Several water economies have reached a mature phase characterized by inelastic supplies with incremental costs, which cannot meet the expanding demand (Loch et al., 2020; WCD, 2000). This has led to an increase in the frequency and intensity of drought events: 10% of the global population already lives in areas with high or critical water stress (UNESCO, 2021), and the 2030 Water Resources Group (2020) predicts that under current water-use trends, the world will face a 40% water deficit in 2030. Furthermore, climate change endangers freshwater availability even more, and vulnerability is becoming evident also in some of the wetter regions of the world (Damania et al., 2017). Climate change is affecting and will affect freshwater resources in different ways: reducing the annual stream flow of rivers (Figure 1), increasing the periods of drought (Figure 2), and modifying trends and intensity of precipitations (Figure 3). Climate change, therefore, is exacerbating water volatility and scarcity, which will be further aggravated by other compounded effects, such as growing evapotranspiration demand from crops, a pressure that already stressed basins (Figure 4) cannot sustain (Caretta et al., 2022). Despite this dire

[1] Note that in this thesis “climate change” means human induced climate change or global warming, i.e., the effect of the rising concentration of greenhouse gases in the atmosphere caused by human activities on earth.

situation, water continues to be treated as a plentiful resource in many water-stressed parts of the world.

This “perfect storm” of stable or decreasing supply, increasing demand, and ineffective policies is at the origin of an unprecedented global water crisis. This crisis could cost some regions “up to 6% of their gross domestic product while spurring migration and sparking conflict” (UNESCO, 2020), a figure that exceeds the losses during the first year of the COVID-19 pandemic (–3.4% globally) (UN, 2022). The effects of the water crisis are particularly concerning for the food production system since the irrigation sector, globally, is responsible for 72% of freshwater use and is highly dependent on this resource to feed billions of people worldwide (FAO and UN Water, 2021). Irrigated agriculture, while covering only 20% of total cropland (see Figure 5) (FAO, 2021), is responsible for 40% of the total agricultural production (Cherlet et al., 2018). Moreover, irrigated land is still expanding, with a 117% increase from 1961 to 2009, compared to an almost stable rain-fed land over the same period (Dinar and Schwabe, 2015). This trend is expected to continue as an adaptation strategy to climate change (IPCC, 2021). Despite being such an important sector for human wellbeing and the largest consumer of freshwater by far, agriculture most often produces the least valuable use of this resource in terms of GDP – Gross Domestic Product (less than 7% globally), and is the least regulated and poorly managed user (World Bank, 2022). Accordingly, agriculture is the sector targeted by the majority of water policies, as the main source of much-needed water savings (OECD, 2015; UNDRR, 2021).

Addressing the current water crisis calls for urgent and radical actions from the water management side through Transformational Adaptation Policies (TAPs), i.e., systemic (applied at a much larger scale) and/or paradigm shifts (truly new or never applied before in a certain location) in water resources management that integrate the adaptation of ecosystems and economic sectors to climate change. TAPs recognize the interconnection and cascading effect between different systems and coordinate the development and management of water, land, and related resources, leveraging both water-allocation systems and economic instruments (OECD, 2021; UNEP, 2021; UNESCO, 2020).

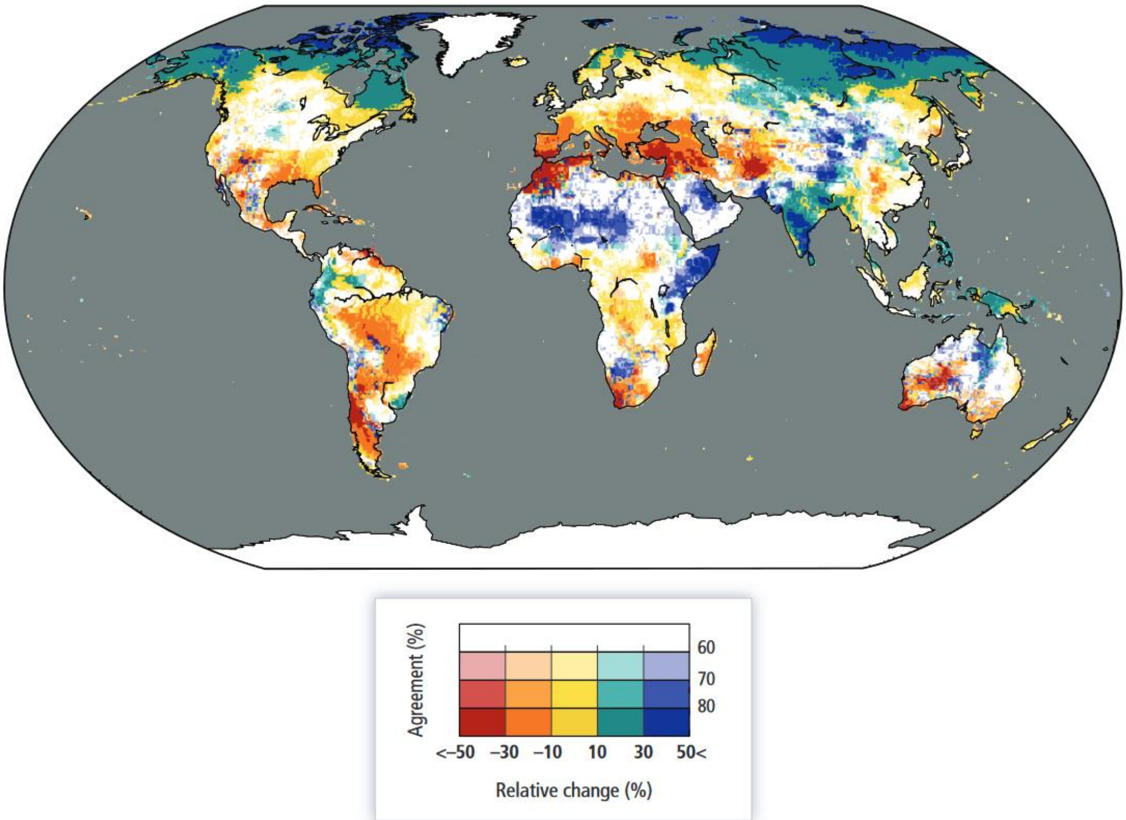


Figure 1: Percent change of mean annual river streamflow for a global mean temperature rise of 2 °C (1980 - 2010). Source: Jiménez Cisneros et al., (2014).

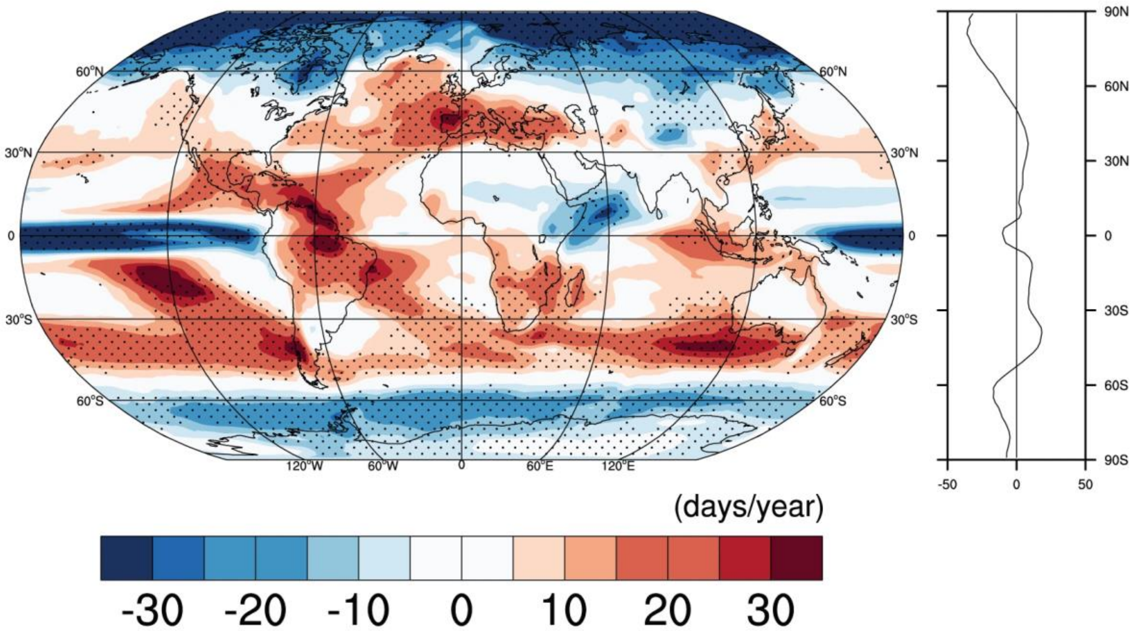


Figure 2: Average mean changes in the frequency of dry days (days/year) by 2060–2089, relative to the historical period 1960–1989, using the RCP8.5 forcing scenario. Source: Polade et al., (2014)

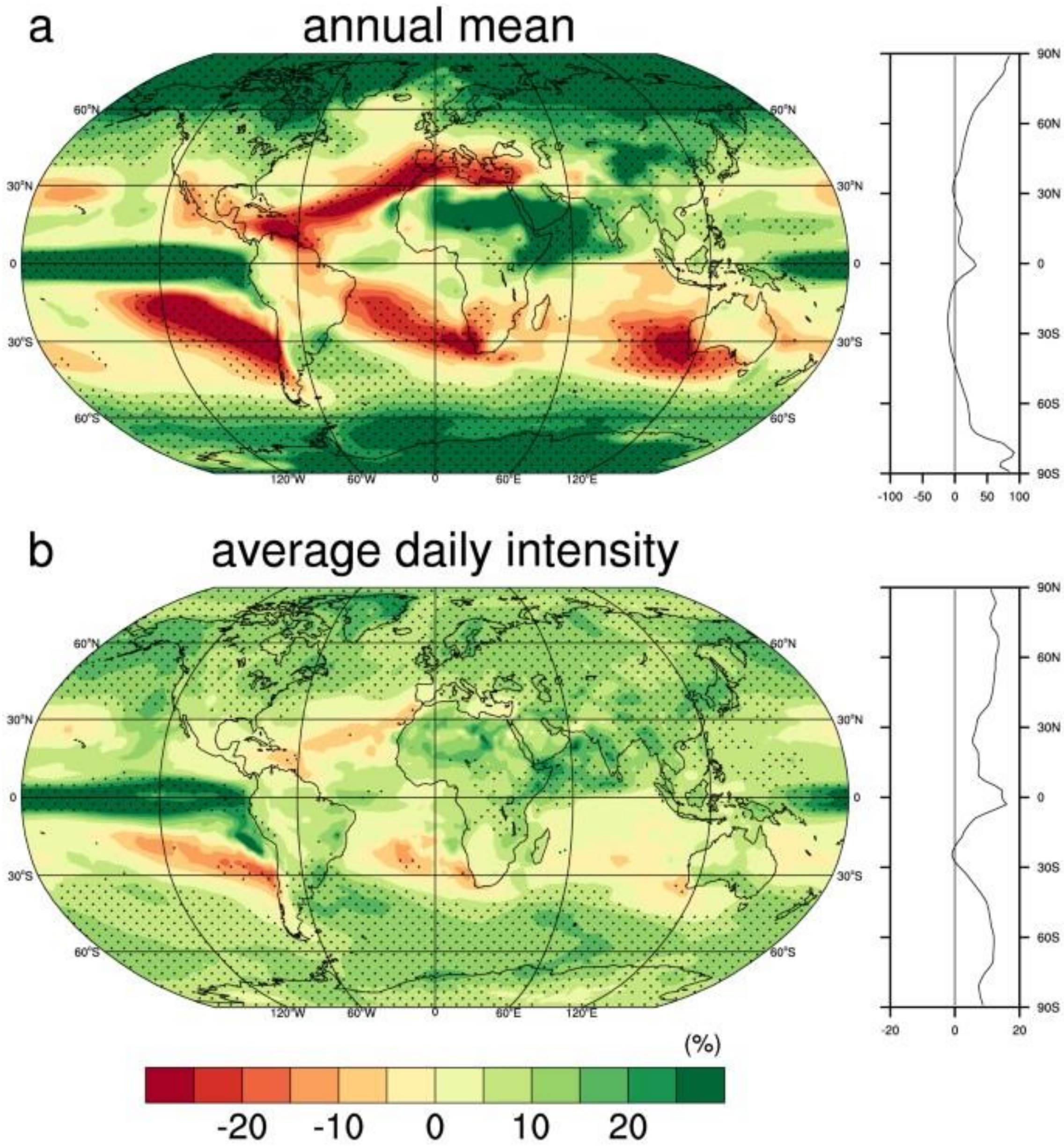


Figure 3: Average percent change in (a) annual mean precipitation; (b) precipitation intensity during precipitating days. Values are computed over the period 2060–2089, relative to the historical period 1960–1989, using the RCP8.5 forcing scenarios. Source: Polade et al., (2014).

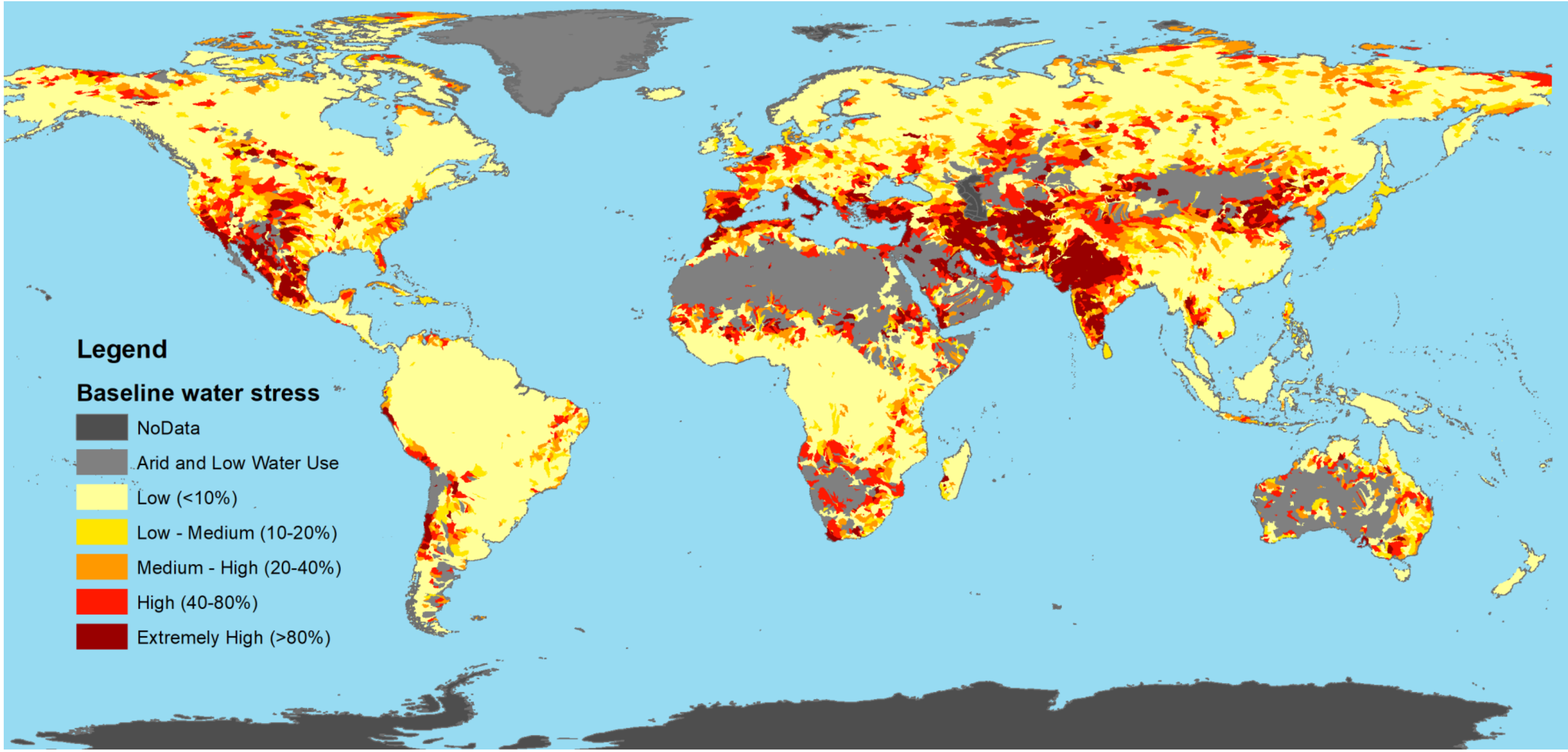


Figure 4: Water stress per catchment. Source: Aqueduct Global Maps 3.0 Data (WRI, 2019).

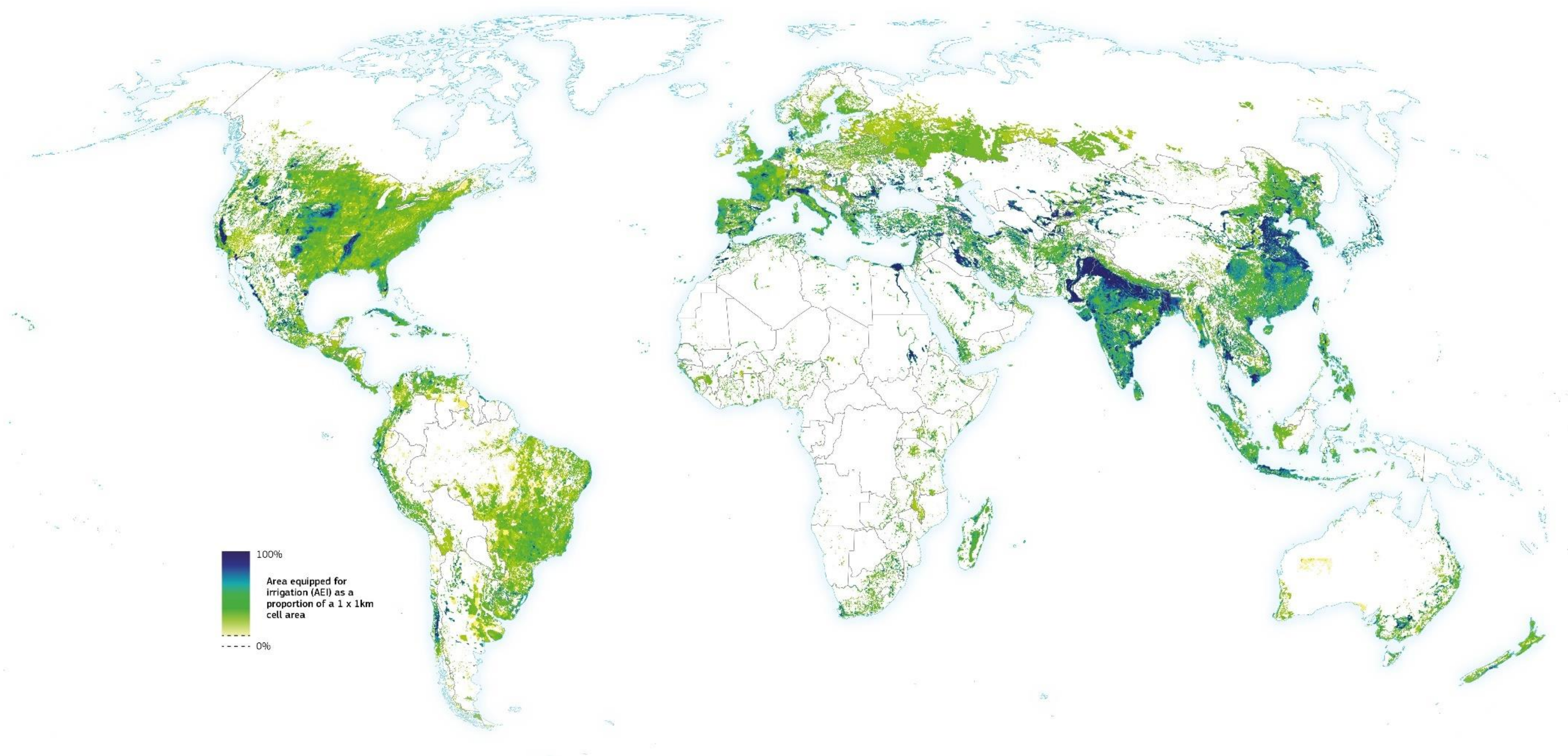


Figure 5: Global irrigated and rainfed cropland area. Source: Cherlet et al., (2018, p. 56).

However, TAPs have proven to be ineffectual. New threats and correlations across complex and interconnected socioeconomic and ecological systems are emerging in ways that had not been anticipated or more quickly than previously thought possible (UNESCO, 2020). Consistently, decision-makers are “surprised and overwhelmed” by the systemic nature of water scarcity and climate change, which can trigger and aggravate other ecological (e.g., pests) and socioeconomic threats (e.g., commodity price fluctuations) via feedback loops and cascading impacts across systems (UNDRR, 2021). In this context, planning for the future is characterized by a high degree of uncertainty, or **deep uncertainty**, a situation where experts and policy makers do not know or cannot agree on: (i) the external context of the system and its evolution; (ii) how the systems work and interrelate; and/or (iii) how to evaluate the desirability of alternative and potentially conflicting outcomes (Lempert et al., 2006).

While standard decision-making accounts for probabilistic risk, it typically does not account for uncertainty. Risk is the calculable and controllable part of what we do not know, while uncertainty is incalculable and uncontrollable (Knight, 1921). Risk is manageable because we can associate probabilities with it, while uncertainty is a broader concept with which we cannot associate reliable probabilities (Marchau et al., 2019). We can think of risk as a lower form of uncertainty. Uncertainty arises from the impossibility of identifying (all) the external changes that are relevant to describing the development of a system and assessing its responses to these external changes (Marchau et al., 2019). Instead of foreseeing the best prediction, under deep uncertainty, decision-makers should prioritize robustness, i.e., strategies that perform better under many plausible futures (see box 1 for more details on uncertainties and robust decision-making techniques). In the case of agricultural water management, uncertainty arises in both human and natural systems because of behavioral variability (human behavior, non-rationality, cognitive dissonance, non-standard behavior (Walker et al., 2003)); societal variability (macroeconomic behavior, markets, and societal processes); epistemic uncertainty (incomplete scientific or technical knowledge); and future conditions (related to the application of a model developed in the actual conditions that are associated with the unknown conditions of the future).

Box 1: Deep Uncertainty and Decision Making under Deep Uncertainty (DMDU)

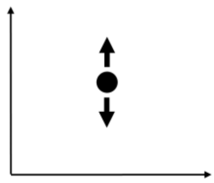
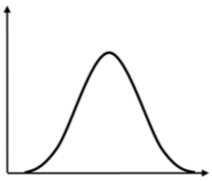
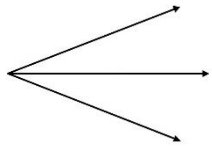
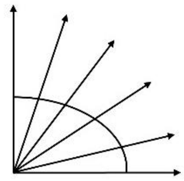
A situation is deeply uncertain if it is not clear or there is no agreement on: “(1) the appropriate models to describe the interactions among a system’s variables, (2) the probability distributions to represent uncertainty about key variables and parameters in the models, and/or (3) how to value the desirability of alternative outcomes” (Lempert et al., 2003). When a decision must be made under deeply uncertain conditions, it is not possible to infer the probability of success of the selected strategy, although it is possible to identify different levels of uncertainty (Figure 6) and accordingly select the better strategy to deal with it.

Figure 6: different levels of uncertainty from knowledge to no-knowledge. Source: adapted from Marchau et al., (2019); and Walker et al., (2003)

In the first level of uncertainty (deterministic uncertainty), the future is clear enough to be correctly modelized and it is possible to obtain a certain estimation of the outcome. In this case, the decision-makers carry out the better strategy to reach a clear objective. This is the case of some natural phenomena, e.g., in a reasonably stable system, it is possible to define the renewable quantity of water that is possible to extract from a water body. Statistical uncertainty is a situation in which statistics describes adequately the situation: it is possible to calculate precisely the uncertainty in every aspect of the problem, from the model to the outcome and a probability value can be assigned (Walker et al., 2003). This is the uncertainty of natural sciences: the measurement uncertainty associated with all data (sampling error, inaccuracy, or imprecision) is a clear example of this level of uncertainty. This kind of uncertainty is usually reduced with the advancement of knowledge and research. In the following level of uncertainty, scenario uncertainty, there is a range of possible outcomes, but it is not known, or it is not clear, how the outcomes were reached and, evidently, it is not possible to assign any probability. What is known, though, is how the system can react to some possible mechanism, but how, when, and the intensity in which they will occur remains unknown. Decisions are made considering alternative scenarios (the possible outcomes and how they are reached) instead of likelihood (Walker et al., 2003). Finally, there is a last level of uncertainty, before ignorance, that can be addressed with a series of techniques. The systemic uncertainty is the one in which the possible outcomes and how the systems interact are a wide range of possibilities, at which it is not possible to assign probabilities. Most of all, it is not possible to establish clear relations between the interactions within the system and with other systems. Finally in the two last levels of ignorance, recognized and total, the former refers to a kind of ignorance that can be resolved eventually, e.g., with further research, while the latter refers to an irreducible ignorance, of which "neither research nor development can provide sufficient knowledge about the essential relationships" (Walker et al., 2003).

In the following table (Table 1), are summarized the different levels of uncertainty. In case of high degrees of uncertainty, scenario, and systemic uncertainty according to Table 1, several techniques were developed in recent decades to allow decision-makers to individuate the path(s) that will produce the best performance, i.e., that allow achieving the objectives even if the future is deeply uncertain. The DMDU is the recent field of literature that is developing to deal with these issues. DMDU techniques are necessary and valid only when it is not possible to consider the past "a reasonably reliable predictor of the future", the basic assumption of classical statistics and probability (Marchau et al., 2019). However, if it is not the case, i.e., if the past can be a reliable predictor, statistics and probability techniques should be preferred. DMDU techniques include Robust Decision Making (RDM), Dynamic Adaptive Planning, Dynamic Adaptive Policy Pathways, Decision Theory, and Engineering Options Analysis (Marchau et al., 2019). The paradigm of DMUD is "monitor-and-adapt", this is a systematic approach that deals with uncertainty being always prepared to change the strategy when an unpredictable event changes the simulated path chosen to reach the desired outcome.

Table 1: Source: adapted from Marchau et al., (2019, p. 9)

	Deterministic uncertainty	***Statistical uncertainty***	***Scenario uncertainty***	***Systemic Uncertainty***
Context	Clear future 	Alternative futures (with probabilities) 	A few plausible futures 	Many plausible futures 
System model	A single deterministic system model	A single stochastic system model	A few alternative system models	Many alternative system models
System outcomes	A point estimation for outcome	A confidence interval for outcome	A limited range of outcomes	A wide range of outcomes

Under deep uncertainty, it is not foreseen the best predictor, but the one that yields the better decisions in this condition; this set of concepts, processes, and tools is known as RDM (Lempert, 2019). RDM combines different techniques and performs recursive stress tests over many possible future paths; this allows the decision-makers to identify relevant scenarios and adaptive strategies. This combination of models, scenarios, and recursive runs is also known as "exploratory modeling", it is a useful tool under deep uncertainty as it maps "a wide range of assumptions onto their consequences without privileging one set of assumptions over another" (Lempert, 2019). Exploratory models are extremely valuable if missing data, uncertainty, and competing objectives do not allow to validate a single model. Rather than using models to predict, RDM uses them to obtain a large dataset of plausible scenarios (the results of every run) and evaluates them with visualization and statistical methods to identify possible tipping points or problems that do not allow to reach the policy goals. Running models many times and in different conditions is not a sophisticated or accurate solution *per se*, but its results can be surprisingly useful if the appropriate scenarios and models are implemented, i.e., the ones that give insights to address the problems object of the study (Lempert, 2019). RDM is a suitable technique to advise policymakers under deep uncertainty, despite the high costs in terms of time, computational power, and complexity.

The uncertainty of climate change forced geophysical scientists (physicists, climatologists, hydrologists, etc.) to use model combinations through ensembles because classic models, even if globally consistent in average terms, "disagree on the magnitude, and in many cases even the sign, of change at a regional scale" (Schewe et al., 2014). The high degree of uncertainty led these researchers to rely on a modeling framework that consists of ensembles considering different disciplines, sectors, and scales, e.g., CIMIP (Coupled Model Intercomparison Project), (2022), ISIMIP (Inter-Sectoral Impact Model Intercomparison Project), (2022), AgMIP (Agricultural Model Intercomparison and Improvement Project), (2022), and HEPEX (Hydrologic Ensemble Prediction EXperiment), (2022). While geophysical researchers have embraced this approach, agricultural economics still relies on single models to produce point predictions that are vulnerable to uncertainty and may lack robustness.

This calls for an urgent change of direction in (agricultural) water management that explicitly quantifies uncertainty across coupled human-water systems to inform the identification and adoption of robust TAPs.

1.2. Thesis objective

The main objective of this work is to develop and test a multi-model and multi-system modeling framework that can inform the identification and adoption of robust TAPs that contributes to the Integrated Water Resources Management targets of equitable and sustainable economic and welfare growth. The thesis is policy-oriented and aims to deliver actionable science through the development of modeling frameworks that policymakers can readily apply to address water scarcity and test robust TAPs in the agricultural sector, e.g., building upon the models already used by decision-makers in river basins. This overarching objective builds upon three specific activities:

I. First, we analyzed the available methodologies used in agricultural economics to address water issues in the agricultural sector. Through this analysis, we identified and addressed a methodological gap of a specific MPM, therefore **expanding the literature on socioeconomic modeling** with a new model to assess farmers' behavior. This innovation, from a strict modeling perspective, is not a real new model; rather, it relies on the TAPAS (Take A Previous model and Add Something) approach (Frenken, 2006), thus improving an existing model.

II. Second, this work introduces an innovative **ensemble of socioeconomic models** to address deep uncertainty and advise robust decisions. Inspired by Frenken, (2006), the TAPAS approach is stressed to address uncertainty by introducing the TAMAL (Take severAl Models And combine them alL) approach. The objective of TAPAS is to innovate by improving an existing and well-established methodology, while the TAMAL approach combines different models to create an artificial world where it is possible to consider multiple plausible futures, to address possible unpredicted outcomes. The TAMAL approach is based on RDM (see box 1) and assesses robust decisions in the socioeconomic modeling.

III. Finally, following sociohydrology and hydroeconomic literature (Harou et al., 2009; Heinz et al., 2007; Sivapalan et al., 2014), we integrate the ensemble of MPMs into a **multi-systems modeling framework** that considers the co-evolution of the human-water systems and their bidirectional feedback. To this end, first, we developed a modular framework to integrate hydrologic and economic systems coupling a hydrologic model and the ensemble of socio-economic models. This framework is therefore fitted to sample modeling uncertainty and foresee robustness while simultaneously considering the bidirectional feedback between human and water systems. The coupling between the different systems is always modular, i.e., every model remains independent, to allow more specific rules “that help represent some degree of realistic agent […] behavior” (Erfani et al., 2014) but are capable of exchanging information with the other models.

1.3. Thesis Outline

This thesis is organized into five chapters, and every chapter has an introduction that states the objective of the successive pages and explains why it is relevant to the main objective of this study. Chapter 2 argues the rationale for using MPMs and briefly reviews the literature on these models, successively expanded in the publications included in this thesis. This chapter included the first publication of this thesis, where the first activity is completed. Chapter 3 includes two publications in which a multi-model ensemble of MPMs was built and used to advise robust decisions. Two different policies were tested: a pricing policy to assess a reform of the water licenses in the Piedmont region, Italy (3.2) and the introduction of a PWS (Payments for Watershed Services) scheme in the Reno River land reclamation and irrigation board, Emilia-Romagna, Italy (3.3). In chapter 4 the complexity of the system is increased by adding a hydrologic layer to the ensemble of MPMs. Chapter 4 includes two publications in which the modular framework that integrates human and water systems is presented: the multi-model ensemble of MPMs is coupled with the hydrologic DSS (Decision Support System) model AQUATOOL. In section 4.2, the modeling framework is used to calculate the resource cost of agricultural water to accurately apply the EU WFD (Water Framework Directive), while the second application (4.3) proposes an innovative water bank system used to maintain the minimum environmental flows, (partially) without increasing the public budget.

Finally, the last chapter depicts the conclusions of this work, evaluates how the main goal was achieved, and includes the recommendations of the author about future research that should be addressed to improve the actual modeling framework.

Chapter 2. Behavioral socioeconomic modeling: the adaptation strategies of farmers

"All models are wrong, but some models are useful"
George Box, 1979.

2.1. Introduction

The previous chapter pointed out the importance of changing the classic approach of agricultural economics modeling to address uncertainty challenges through ensembles of socioeconomic models. To do so, this chapter analyzes the different modeling approaches used in the agricultural sector to advise decision-makers and then presents an innovative methodology. Because this thesis focuses on water management, the methodology analyzed and proposed always refers to the case of water as the relevant input to be managed, but the same methodology remains valid to analyze adaptive responses to every other production input and/or political constraint (e.g., fertilizer use, labor).

During the 20th century, the classic economic approach to evaluate agricultural water policies was the development of models based on production economics to evaluate the allocation and demand for input and how it would change. Several empirical models were developed: econometrics models, field experiments, hedonic pricing, contingent valuation, and MPMs (possibly the most common). MPMs are optimization models, that, according to classic economics, are appropriate to describe the economic problem of making the best use of finite resources (Mills, 1984). These models aim to reproduce the behavior of economic agents and their adaptation pathways after a change/shock (e.g., the reduction of an input, the water in the case of this study). The objective is to maximize a function that represents humans' will in a simplified mathematical way.

These models reduce the complexity of humans' choices to a simple decision on the allocation of land to the different crops, according to the mathematical function that describes their behavior. Every choice in the crop portfolio represents, therefore, a unique combination of input applied in fixed proportions per unit of land (Arata et al., 2017; Gómez-Limón et al., 2016; Graveline et al., 2014). This simplification allows us to simulate adaptive responses to shocks (e.g., a drought) or policies (e.g., caps or pricing) through extensive margin (i.e., land reallocations toward less water-intensive crops) and superextensive margin adjustments (i.e., land reallocations from irrigated to rainfed agriculture), i.e., water is used in fixed proportion (constant water requirement per crop). However, some more complex models also allow intensive margin adjustments (i.e., deficit irrigation), a relevant management option in water-stressed areas (Koundouri, 2004). In this case, the agents will choose how to allocate land and water to the different crops, and a water-production function is necessary for each crop.

MPMs are widely used in agroeconomics, and different modeling techniques can be implemented depending on the data availability and the case study. Thus, it is useful to classify them into macro categories to opportunely identify, if possible, the best approach according to the issue that should be assessed. According to their mathematical specification, MPMs can use a linear or non-linear objective function. The former approach assumes a linear substitution between inputs, which can cause certain "rigidity" in responses ("over-specialized responses" and "jumpy behavior"), while the latter introduces non-linearities that usually represent smoother and more realistic changes (Graveline, 2016). Despite these issues, linear models are widely used, especially when there is a large amount of available data, and it is possible to characterize farmers' behavior at farm level (Bartolini et al., 2007; Graveline et al., 2012). At a higher scale (e.g., regional), non-linear models should be preferable, but still, it is not a binding condition (Buysse et al., 2007; Graveline, 2016).

The objective function (independently from its (non-)linear mathematical specification) is commonly characterized only by profit, but some authors have developed widener and more elaborated multi-attribute objective functions that also consider attributes other than profit (e.g., risk aversion) (Gómez-Limón et al., 2016; Rausser and Yassour, 1981). Multi-attribute objective functions are inspired by Theory of Planned Behavior of Ajzen, (1991), according to which humans' behavior is extremely complex and driven by "multiple attributes of objects (including but not limited to profit) and farmers' beliefs regarding these attributes" (Pérez-Blanco et al., 2017). Both single- and multi-attribute objective functions are equally valid and there is no agreement in the literature to establish which represents farmers better (Buysse et al., 2007; Graveline, 2016). Hence, single-attribute and multi-attribute objective functions are also used to classify MPMs. Usually, multi-attribute models need more data, and their availability is a typical constraint to their implementation (Pérez-Blanco et al., 2017).

Finally, MPMs can be classified as normative or positive models (Graveline, 2016). **Normative models** are MPMs that are not calibrated, i.e., they follow a deductive approach that assumes an *a priori* objective function that represents the agent's behavior. For example, if the modeler assumes that the outcome foreseen by the economic agent is to maximize their profit (a common practice in the economic field), the agent will choose the combination of inputs that yields the highest profit, e.g., in a simple case in which agents are irrigators that can cultivate only two crops and do not have any crop-specific constraints (policy caps, water restrictions, etc.), they will grow only the crop that has the higher expected profit. This means that normative models do not guarantee the reproduction of the observed reality if the modeler does not have perfect knowledge of every farmer (almost impossible in real-life modeling). **Positive models** (PMs) have a slightly different approach: they assume that the observed situation is the optimal one and that the model should be "adapted" to represent it in the best possible way. This hypothesis lays its roots in the classic economic theory according to which economic agents will always foresee their utility, i.e., if a model cannot reproduce the observed reality, it is because of a lack of knowledge. PMs rely on this assumption and perform a calibration procedure to elicit the parameters of the objective function that better reproduces the observed input allocation, i.e., the observed crops' distribution. The closer the results of the simulated model are to the observed situation, the better the calibration. The difference between the calibration and the observed situation is the calibration error.

Some models have a perfect calibration per se, i.e., the calibration procedure mathematically elicits the parameters that result in a null error, while others follow other rationales and can have a positive calibration error. Arguably, models that perfectly calibrate are intrinsically better, but the rationale beyond the calibration procedures that set the error to zero is still an open debate (Gutiérrez-Martín and Gómez, 2011; Heckelei et al., 2012; Mérel et al., 2011). Furthermore, Cloke et al., (2013) argue that calibration errors between different families of models (i.e., between models with different calibration procedures) are not comparable because they are independent, so it is not possible to use the calibration error as a metric to find the best model. Moreover, to the best of our knowledge, in the literature, there are no ex-post evaluations of MPMs' performance, so it is not possible to rank these models according to their precision.

Due to the impossibility of identifying an intrinsically better model, in this thesis, three different families of positive MPMs were used: linear positive modeling, Positive Mathematical Programming (PMP), and Positive Multi-Attribute Utility Programming (PMAUP). These models use different objective functions (single- and multi-attribute), specifications (linear and non-linear), and different calibration procedures that are discussed in 2.2 and 3.2. The models selected to populate the ensemble are among the most common MPMs, and they were the best option with the available data of the different case studies.

After the calibration, these models are ready to simulate the effects/outcome of a policy or water scarcity in the agricultural sector. Classic MPMs usually rely on extensive and super-extensive margin adjustment, so agents in the model can only allocate land to the different crops to maximize their objective function. These two adaptation strategies were criticized for their unrealistic predictions due to the exclusion of the well-established strategy of reducing water applications at a sub-optimal level when the resource is constrained (Koundouri, 2004). In the literature on linear and PMP models, researchers addressed the problem including a (continuous or piecewise) water-production function or coupling economic and agronomic models to account for the intensive margin adjustment (see the literature review in 2.2). In the case of PMAUP models, however, no example of this integration was found in the literature. The first publication included in this thesis (2.2) fills this methodological issue. In this publication, intensive adjustment adaptation strategy is allowed through the integration of a continuous crop-water production function into the objective function, and the performance of this new model is compared with a classic PMAUP model. The methodology is illustrated with a policy application in an irrigation area of the

Jucar River basin in Spain. The results of this study reveal that, when the available data allow the calibration of a crop-water production function, this should be preferred to the classic approach that could under- or over-estimate the elasticity of the water demand. Unfortunately, this approach is extremely data-intensive because it requires site-specific data of the water production function for every crop, which are usually not available. In almost all the study cases of this work, these data were only partially available (e.g., only for a few crops) or not available. With the increase in data availability and agronomics models, MPMs that use water-production functions should be preferred to conventional ones.

2.2. Influence of crop-water production functions on the expected performance of water pricing policies in irrigated agriculture

2.2.1. Resumen

Los Modelos de Programación del Agua (en inglés WPM – Water Programming Models) de la economía agraria han evidenciado que los regantes en áreas con escasez de agua tienen una respuesta bastante inelástica a los precios del agua, por lo tanto, las políticas de precio del agua no parecen ser eficientes para su ahorro. En este estudio se plantea la hipótesis de que las previsiones de ahorro de agua de las políticas de precio estén significativamente subestimadas por cuestiones estructurales de los modelos de simulación, es decir debido a la exclusión del riego deficitario entre las posibles decisiones de los agentes en los WPM convencionales. Para probar nuestra hipótesis, desarrollamos un modelo que integra una función de producción continua de cultivos que depende del agua aplicada en un WPM positivo de múltiples atributos. Esto nos permite evaluar las respuestas de los agentes a la variación de precio del agua a través también del riego deficitario. El modelo se ilustra con una aplicación a la zona de regadío de El Salobral-Los Llanos en España. Los resultados muestran que la incorporación del riego deficitario como opción de adaptación hace que la curva de demanda de agua sea significativamente más elástica en comparación con un modelo alternativo donde se excluye el riego deficitario entre las posibles respuestas de los agentes. Se concluye finalmente que ignorar el riego deficitario puede llevar a una subestimación significativa de la rentabilidad de los precios del agua para el ahorro de agua.

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Influence of crop-water production functions on the expected performance of water pricing policies in irrigated agriculture

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ABSTRACT

Agricultural economics Water Programming Models (WPM) has found that irrigators in water scarce areas have a rather inelastic response to water prices, making water pricing cost-ineffective towards water saving. We hypothesize that the predicted water saving performance of pricing is significantly underestimated by issues of model structure, due to the exclusion of deficit irrigation from the set of decision variables available to agents in conventional WPM. To test our hypothesis, we develop a model that integrates a continuous crop-water production function into a positive multi-attribute WPM, which allows us to assess agents' adaptive responses to pricing through deficit irrigation. The model is illustrated with an application to the El Salobral-Los Llanos irrigated area in Spain. Our results show that incorporating deficit irrigation as an adaptation option makes the water demand curve significantly more elastic as compared to an alternative model setting where deficit irrigation is precluded. We conclude that ignoring deficit irrigation can lead to a significant underestimation of the cost-effectiveness of water pricing towards water saving.

1. Introduction

1.1. Rationale

Reconciling growing freshwater demand with finite supply is one of the great policy challenges of our time (WEF, 2020c). Given that agriculture represents 70% of global water withdrawals, which contribute to 6.4% of the world's Gross Domestic Product (FAO, 2021a; World Bank, 2020d), governments are increasingly constrained to adopt agricultural water saving policies to reallocate irrigation water towards higher value-added economic uses, households and the environment. One such policy are water charges, often referred to as pricing, which are defined as an administrative levy imposed on irrigators to recover the costs of water use.[1] Theoretical and conceptual research has long argued that putting the "right price tag" on water can efficiently reallocate irrigation water towards other uses (Dinar and Subramanian, 1997; Tsur and Dinar, 1997). Echoing these results, several governments worldwide have integrated innovative water pricing instruments into their legal bodies to save water (Dinar et al., 2015a). For example, Article 9 of the EU Water Framework Directive states: "[…] water pricing policies provide adequate incentives for users to use water resources efficiently, and thereby contribute to the environmental objectives of this directive" (OJ, 2000).

However, the claim that agricultural water pricing can save water for other uses, including the environment, is not substantiated by empirical evidence. Virtually no water scarce area uses pricing as a water conservation/reallocation tool (the objective being mostly financial, through—partial—cost recovery of capital investments), which means

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[1] From an economic standpoint, charges are levies introduced administratively, while prices refer to the exchange value of any good arising from an interaction between supply and demand in a market environment. In the policy arena, though, many discussions on introducing 'charging' mechanisms for natural resources use the term "price" or "pricing", as is the case in the Water Framework Directive (OJ, 2000). This use of the term "prices" as a synonym of "charges" is also common in the scientific literature (see e.g., Dinar et al., 2015a; Olmstead and Stavins, 2007). For the sake of simplicity, in this paper we use both terms interchangeably throughout.

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Table 1
Type of crop water production functions used in WPM for the representation of farmers' behavior.

Authors	Economic calibration	Crop water production function	Development of the crop water production function	Application
Adamson et al. (2007)	No calibration	Piecewise	Expected yield penalties function of salinity	Basin-wide water reallocation to maximize agricultural income under alternative states of nature
Connor et al. (2009)	non-linear WPM	Continuous	Quadratic yield function of applied water and rain, calibrated with observed yield values.	Capital losses (death of permanent crops) under climate change
Connor et al. (2012)	PMP	Piecewise	Quadratic yield function of water and salinity, PMP calibration	Costs of salinity and climate change in the agricultural sector
Cortignani and Severini (2009)	PMP	Piecewise	Expected yield calculated with FAO's CropWat	Optimal water reallocation through agricultural profit maximization under changing water availability scenarios
Finger and Schmid (2008)	No calibration	Continuous	Yield function of water and nitrogen estimated with robust regression	Costs of climate change and related water availability uncertainty
Frisvold and Konyar (2012)	USARM	Continuous	Nested CES production function, PMP calibration	States-wide adaptation to large reduction in water supplies.
García-Vila and Fereres (2012)	non-linear WPM	Continuous	4 crops yield response to water application	Profit maximization by farmers under climate change
Graveline et al. (2012)	LP	Piecewise	Different expected yield values	Changes in utility through incremental/ decremental provision of profit and risk aversion
Graveline and Mérel (2014)	PMP	Continuous	Yield obtained as function of water with parameter estimated with non-linear least squares method, calibrated with agronomic function	Homogeneous reduction in water availability to all crops.
Howitt et al. (2009)	SWAP	Continuous	CES production function, PMP calibration	Optimal water reallocation through agricultural profit maximization
Kampas et al. (2012)	No calibration	Continuous	Yield function, fixed at optimal water application	Changes in agriculture water demand through water pricing
Loch et al. (2020b)	No calibration	Piecewise	In the different state of nature the yield is different	Basin-wide water reallocation to maximize agricultural income under alternative states of nature
Medellín-Azuara et al. (2010)	PMP	Continuous	Yield CES function with scaling parameter for different scenarios	Farmers response to external shocks or new policy
Medellín-Azuara et al. (2012)	PMP	Continuous	Nested CES function with scaling parameter for different scenarios	Farmers and regional responses to external shocks or new policy
Ortega Álvarez et al. (2004)	No calibration	Piecewise	Yield production function based on FAO's methodology	Basin-wide water reallocation to maximize agricultural income
Peña-Haro et al. (2010)	No calibration	Continuous	Quadratic crop production function, obtained with inputs from GEPIC model	Profit maximization by farmers subject to max. nitrate concentration
Peña-Haro et al. (2014)	No calibration	Continuous	Quadratic crop production function depends on water and nitrogen; depends on GEPIC model agronomic simulation	Profit maximization by farmers subject to max. nitrate concentration
Reca et al. (2001)	No calibration	Piecewise	Yield production function based on FAO's methodology	Basin-wide water reallocation to maximize agricultural income

Note: PMP: Positive Mathematical Programming; SWAP: Statewide Agricultural Production Model; LP: Linear Programming; USARM: U.S. Agricultural Resource Model; CES: Constant Elasticity of Substitution.
Source: Own elaboration with inputs from the papers listed in the first column of the table.

that field data on the water saving performance of pricing is limited and inconclusive (Dinar et al., 2015a; Rey et al., 2018). Moreover, applied research through agricultural economics Water Programming Models (WPM), which we define here as "a system of equations including an objective function and a set of constraints including resource constraints as a minimum" that is used to represent the behavior of individual economic agents such as irrigators (Graveline, 2016), has found that water pricing can only achieve relevant savings in water scarce areas at disproportionate costs due to the inelastic response of irrigators to higher prices (including in the initial stretches of the demand function) (see e.g. Berbel et al., 2007; Cornish et al., 2004; Dinar and Subramanian, 1997; Molden et al., 2010; Montilla-López et al., 2017; Pérez-Blanco et al., 2015; Steenbergen et al., 2007). This has led some to conclude that water pricing is cost-ineffective and even "irrelevant" as a water saving instrument (Berbel et al., 2007).

Our research hypothesis is that the predicted water saving performance of agricultural water pricing policies is significantly underestimated by issues of model structure, and specifically by the exclusion of deficit irrigation from the set of decision variables available to agents (irrigators) in conventional WPM applied to pricing.

In real life, irrigators decide on the crop portfolio, timing, investments (e.g. irrigation system) and water application, so to maximize their expected utility derived from one or a set of utility-relevant attributes (e.g. profit) provisioned through the production of agricultural goods, subject to a series of policy and resource constraints (i.e. feasible region). Conventional WPM reduce this complex choice to a decision on the crop portfolio, where each feasible choice represents a unique combination of crop, timing, investments, and water application. Note that this approach does not include an explicit crop-water production function; instead, water input is applied in fixed proportions to land (Arata et al., 2017; Gómez-Limón et al., 2016; Graveline et al., 2014). While this simplification allows for simulating adaptive responses to water conservation policies at the extensive margin (land reallocations towards less water intensive crops) and super-extensive margin (land reallocations from irrigated to rainfed agriculture), it does not allow for simulating intensive margin adjustments through deficit irrigation, a relevant management option in water stressed areas (Koundouri, 2004). Without water stress, it is reasonable to assume that economic agents will always apply water to the point where the marginal utility equates the marginal cost of water and thus maximize utility (i.e., no intensive margin adjustment will ever take place, only super-extensive and extensive margin adjustments); however, under scarcity conditions where water is a binding constraint, intensive margin adjustments are likely to be observed as agents now aim to determine the point at which each additional unit of water is providing the maximum attainable utility from its application, which needs not match

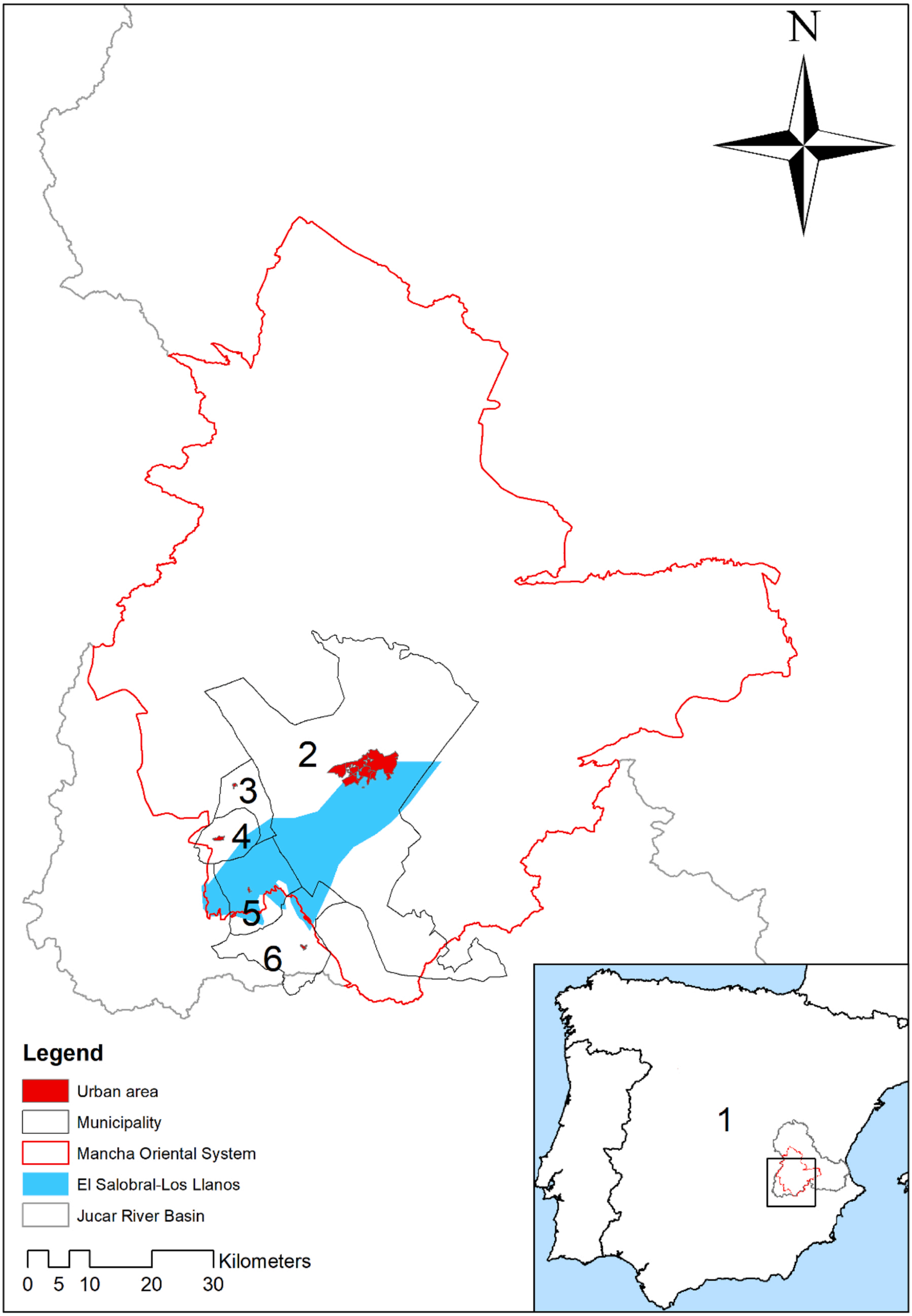

Fig. 1. Case study area. Legend: 1 Spain, 2 Albacete, 3 La Herrera, 4 Balazote, 5 Pozuelo, 6 Peña de San Pedro. Source: own elaboration.

the point at which marginal utility and costs are equated (which may be unfeasible under the new constraint). Accordingly, ignoring intensive margin adjustments can lead to biased estimates of human responses; biased estimates of the water saving potential and the economic impacts of pricing; and misleading policy recommendations (Frisvold and Konyar, 2012). This calls for the integration of crop-water production functions that reflect the biological processes occurring in the agricultural system into economic models assessing the impacts of pricing.

1.2. Literature review of agricultural economics WPM that integrate crop-water prediction functions

An expanding literature explores the integration of crop-water production functions in WPM to allow for intensive margin adjustments. Table 1 presents the WPM that incorporate a crop-water production function, classified in accordance with i) the calibration of the WPM, ii) the type of crop-water production function (functional form and development), and iii) the relevant policy issues addressed by these models.

Among *non-calibrated WPM*, Reca et al. (2001) develop an expected utility model composed of three sub-models: the first optimizes the water application for every crop through a continuous crop-specific water production function inspired in FAO's CROPWAT (FAO, 1992); the second approximates the crop water production function "to a discrete function considering a series of interval of irrigation depths" (Reca et al., 2001), and maximizes profit over the irrigated area subject to existing constraints (i.e. optimizing a linear objective function defined as crop-specific prices times yield minus per ha costs, multiplied by the crop's area); while the third optimizes water allocation over the entire water system taking into account water availability and maximum and minimum flows. A similar approach is followed by the MOPECO model (Ortega Álvarez et al., 2004), which integrates FAO's production function into an economic optimization model to determine the seasonal net irrigation depths, simulate water distribution and maximize the total profit at farm level. Finger and Schmid (2008) use regression methods to estimate the parameters of a continuous crop water production function for Swiss corn and wheat under different water availability scenarios. Crop production functions are subsequently integrated into the objective function of a linear economic model that maximizes profit, and the optimization problem is solved under alternative climate and water availability scenarios. Authors use this information to estimate the certainty equivalent for each climate and water availability scenario considered and reveal the minimum payoff agents would be willing to make to avoid climate change and related water availability uncertainty. Kampas et al. (2012) build a quadratic and continuous crop production function where yield depends on water and nitrates application. The crop water production function is calibrated for key crops in a case study area in Thessaly (Greece), namely corn and cotton; while water is used in fixed proportions to land for the remaining crops. This results in a model that combines non-linear (corn and cotton) with linear optimization (other crops). García-Vila and Fereres (2012) develop a non-linear programming model that uses FAO's AquaCrop model to simulate yield responses to water application for the four most relevant crops in Santaella, Spain (namely cotton, corn, potato and sunflower), and optimizes farmer profit under alternative price and climate change scenarios. Peña-Haro et al., (2014, 2010) develop an integrated economic-agronomic model featuring a non-linear quadratic production function where yield is a function of water and nitrogen application, which is integrated in a normative economic model that maximizes irrigators profit subject to groundwater quality constraints (maximum nitrate concentration). Loch et al. (2020b) integrate a piecewise crop-water production function into a WPM that aims to optimally reallocate water so to maximize a linear function of profit at a catchment level, under alternative states of nature that represent hypothetical water availability conditions. For permanent crops, the model divides the crop water production function into two components: the minimum amount of input necessary to guarantee crop survival and 'productive watering' that returns effective crop yield.

Among *calibrated WPM*, Graveline et al. (2012) integrate a piecewise crop water production function into a Linear Programming (LP) model to simulate stepwise changes in utility through the incremental/decremental provision of the two utility-relevant attributes considered, namely profit and risk aversion. Connor et al. (2009) develop a model for profit maximization with a quadratic crop water production function that distinguishes between annual and perennial crops, so to account for future penalization if the trees die or are damaged. The yield function is calibrated with observed values of water application and yield in the Murray-Darling Basin in Australia. Cortignani and Severini (2009) integrate a piecewise crop water production function into the objective function of a Positive Mathematical Programming (PMP) model; yield responses to water application are obtained from FAO's CropWat. Connor et al. (2012) introduce a piecewise quadratic crop water production function in a PMP model to evaluate the impact of salinity and climate change in the agricultural sector. The model is calibrated following the method proposed by Röhm and Dabbert (2003), which follows the classical PMP approach of adding quadratic components in the cost function from dual values' constraints while allowing for higher elasticity of substitution between groups of similar crops. The objective function distinguishes between annual and perennial crops to account for a future penalization in case of not reaching the minimum water supply that guarantees perennial crop survival. Frisvold and Konyar (2012) and Medellín-Azuara et al., (2012, 2010) integrate a continuous CES production function that assess crop yield responses to different inputs, including water, in a classical PMP and in the USARM[2] models, so to account for the elasticity of substitution between inputs. Frisvold and Konyar (2012) nest the crop production function in two steps: in the first they include land and water, and in the second chemicals, fertilizers, labor, capital and energy input. Medellín-Azuara et al. (2010) consider five inputs (land, applied water, supplies, a water capital bundle, and a composite input called effective water) and calibrate the parameters of a CES objective function for land, supplies, and effective water. Medellín-Azuara et al. (2012), add a new step to the calibration procedure of the CES production function above to consider the "substitution relationship between water and capital irrigation investment". In the SWAP model, Howitt et al. (2009) use a PMP model that adopts a CES objective function to maximize profit along 4 inputs (land, water, labor and supplies). The model has a multistage calibration process to specify the CES and exponential cost function parameters. Finally, Graveline and Mérel (2014) build on previous works by Mérel et al., (2014, 2011) to shift the non-linear components of the PMP objective function away from the cost function and into the production function. While conventional PMP models add a quadratic component to the cost function, so to introduce a non-linear component in the objective function that bounds the solution of the utility maximizing problem to observed decisions, Graveline and Mérel (2014) calibrate non-linear CES crop-water production functions to explicitly specify the "elasticities of substitution between land and water and calibrate them to replicate a set of exogenous agronomic crop yield responses to water application". Following this approach, the non-linearity in the objective function now comes from "decreasing return to scale at the crop level, rather than increasing marginal cost" (Graveline and Mérel, 2014). Applying this approach authors identify the shadow value of water, while the shadow value of land is set exogenously from the observed agricultural land value.

Three key commonalities emerge from our literature review of agricultural economics WPM that integrate a crop-water production function. The first commonality is that there are no applications of

[2] A PMP based model developed to simulate market and policy shocks in the US agricultural sector; it considers effects in land reallocation, water use, yield and production, labor and net farm income.

integrated WPM and crop-water production functions that assess the impacts of water pricing. Most applications research the impacts of water availability constraints and optimal basin-wade reallocations.

The second commonality is that, although most applications of agricultural economics WPM that integrate crop-water production functions rely on calibrated models (Graveline, 2016), non-calibrated models account for almost half of the papers in our review on integrated WPM and crop-water production functions.

The third commonality is that all the crop-water production functions in our literature review on agricultural economics WPM that integrate crop-water production functions study the relationship between crop yield and *water applied*, instead of the (more stable) relationship between crop yield and crop evapotranspiration that is typically reported in agronomic models (Steduto et al., 2007). This is because while irrigators can control water application, they cannot control crop evapotranspiration (which is a function of water applied and technology, but also of variables out of control of the irrigator such as wind or solar radiation). Thus, agricultural economics models that explore intensive margin adjustments use water applied as the argument of the objective function (together with land allocation) (Graveline, 2016).

Crop-water production functions relating water applied to yield are site specific and depend on several local factors (soil type, topography, irrigation method, farm management practices, precipitation regime, percentage of crop water requirements satisfied by rainfall). Fereres and Soriano (2007) and Trout and DeJonge (2017) show how small amounts of applied water increase yield linearly until a threshold is reached, from which the relationship becomes curvilinear because part of the water applied does not contribute to crop evapotranspiration due to increased deep percolation, runoff or evaporation, and less effective use of precipitation that reduces the efficiency of water application. Thus, when studying the relationship between crop yield and water applied, the use of a nonlinear concave crop-water production function is more realistic. In our literature review, the relationship between crop yield and water applied is approximated using either piecewise functions obtained from process-based crop simulation models or continuous functions parameterized using statistical methods (typically quadratic). Independently of the form used (piecewise or continuous), all crop-water production functions in the review are deterministic. Use of deterministic crop-water production functions is instrumental to integrate agronomic modeling and data into the structure of WPM, where all variables in the objective function (including yields, but also e.g. revenues and costs) are defined as a deterministic function of the decision variables (crop portfolio, water application).

1.3. Contribution of this research

The contribution of this paper to the scientific literature is twofold. *First*, we integrate a continuous crop-water production function into a positive WPM with a multi-attribute utility function as objective of the optimization process – known as Positive Multi-Attribute Utility Programming (PMAUP) (Gómez-Limón et al., 2016; Gutiérrez-Martín and Gómez, 2011). To the best of our knowledge, this is the first time a crop-water production function is integrated into a multi-attribute WPM. Adding new modeling approaches to the literature on integrated WPM and crop-water production functions can improve our understanding of irrigators' adaptive responses; and is instrumental towards the development of ensemble experiments that sample parameter and structural uncertainties arising from model choice. Ensemble experiments can be used to compare simulation results of the proposed integrated model against those of other integrated models in Table 1, under alternative model settings (i.e., exploring alternative functional forms and parameterization of the crop-water production functions and utility functions) and scenarios. The result is a large database of simulations in which each simulation represents the performance under one plausible future. This information can be used to identify futures where proposed policies meet or miss their objectives, explore potential tipping points, and inform the development of robust policies that show a satisfactory performance under most conceivable futures (Saltelli and Funtowicz, 2014; Sapino et al., 2020).

Second, we use our newly developed model to assess the water saving and economic performance of water pricing considering all three possible adaptive responses: extensive, super-extensive and intensive margin adjustments. The net effect of considering the option to adapt through the intensive margin is revealed through a comparison with a classic PMAUP model where the continuous agronomic production function is substituted by point values that represent expected yield under irrigated and/or rainfed agriculture, thus allowing only for extensive and super-extensive margin adjustments. Methods are illustrated with an application to the agricultural area of El Salobral-Los Llanos domain (SLD) located on the overallocated Mancha Oriental System (MOS), a major aquifer within the Júcar River Basin (south-eastern Spain).

2. Background to the case study: El Salobral-Los Llanos domain in the mancha oriental aquifer (Spain)

2.1. Water use and pressures

The SLD comprises part or the totality of the municipalities of Albacete, Balazote, La Herrera, Peña de San Pedro and Pozuelo (Fig. 1). The SLD has an extension of 420 km^2, of which 337 km^2 are devoted to agriculture (80%) and 100 km^2 are irrigated. Most relevant irrigated crops in the area include wheat, barley, corn, onion, garlic and almond, which have an average water allotment available of 1500 m^3/ha. About 90% of water withdrawals in the SLD come from irrigated agriculture, most of which are met through groundwater extractions from the MOS. On top of agricultural water demand, water bodies within the SLD supply water to a population of circa 5000 inhabitants (about 10% of the total demand) (Peña-Haro et al., 2014).

The MOS in the Júcar River Basin is one of the largest groundwater bodies in Spain (7260 km^2), encompassing parts of the provinces (NUTS3[3]) of Albacete, Cuenca, and Valencia. In the last three decades, a significant increase in agricultural water demand and withdrawals has been observed in the MOS through the development of an intensive irrigated agriculture that represents a significant share of the employment and value added of the region. At present, over 80,000 ha of land equipped with modern technologies are irrigated in the MOS, mostly with groundwater. Because of irrigation expansion, the aquifer has been subject to an intensive groundwater overexploitation since the 1980 s, which has resulted in a continued reduction in the piezometric levels, especially in the southern area where the SLD is located. Stream–aquifer interaction with the Júcar River has been substantially affected as a result of aquifer overdraft: previously, the MOS discharged water into the Júcar River and enhanced its streamflow, while today these dynamics have been reversed and the Júcar River recharges the MOS (Sanz et al., 2011). Groundwater overdraft has led to a significant streamflow reduction in the Júcar River with non-trivial environmental consequences, such as the drying of a significant reach of the Júcar River in the summers of 1994 and 1995, which in turn has caused significant conflicts with downstream uses (Apperl et al., 2015). This situation is expected to be exacerbated by future climate and land use change scenarios (Pulido-Velazquez et al., 2015). Some measures have been recently proposed to reduce agricultural water use and restore the balance in the overexploited MOS Aquifer, notably water pricing (Peña-Haro et al., 2014).

[3] The Nomenclature of Territorial Units for Statistics (in French: Nomenclature des unités territoriales statistiques, or NUTS) is "a geocode standard for referencing the subdivisions of countries for statistical purposes" used by the EU (Eurostat, 2020a). NUTS3 is equivalent in Spain to provinces.

2.2. Water pricing

In Spain, irrigators relying on surface water bodies pay river basin authorities a Water Use Fee (in Spanish: *Tarifa de Utilización del Agua*) and a Regulation Fee (*Canon de Regulación*) designed to recover the investment and maintenance costs of conveyance and water storage infrastructure operated by the public administration (e.g. reservoirs, large canals, water transfers), for which cost recovery levels range from low to moderate (EEA, 2013). Water Use Associations (WUA) can also price water through an additional fee to recover the investment and maintenance costs of storage and distribution infrastructures operated by the WUA (e.g. canals within the WUA). Most irrigators across the MOS rely on groundwater bodies and do not use water storage and distribution infrastructure, and typically do not belong to any WUA, which makes them exempt from the payment of all the above-mentioned fees (Water Use, Regulation and WUA fees). On the other hand, the falling piezometric levels of aquifers are increasing the energy costs of groundwater pumping, which now are 0.1 EUR/m^3 in average (up to 0.2 EUR/m^3 for the deepest extractions) in the SLD and other irrigated areas in the MOS (ITAP, 2020; JCRMO, 2009; JRBA, 2016).

Aside from the fees to recover the investments in water storage and distribution infrastructures, and the variable costs of pumping groundwater and operating irrigation systems, no additional levies on agricultural water use exist in the irrigated areas of the MOS. This fails to comply with Article 9 of the EU Water Framework Directive, which calls for the implementation of pricing policies with a double role: cost recovery (financial instrument) and demand management (economic instrument to favor economic efficiency in water use). Those water prices should recover the "environmental and resource costs" of the resource on top of the financial costs from the construction and operation of irrigation system (OJ, 2000). Environmental costs are defined as the damage that water uses impose on ecosystems, which can be measured e.g., as the welfare loss experienced by those who enjoy those ecosystems[4]; while resource costs are defined as the opportunity cost (foregone economic benefits) of water allocation over space and time (Heinz et al., 2007; Pulido-Velazquez et al., 2008; WATECO, 2003). Both of these costs are present and significant in the overallocated and overexploited MOS, but are not recovered (Peña-Haro et al., 2014). As a result, the water charge applied is significantly lower than the theoretical water charge that would allow for full cost recovery, and insufficient to effectively curb down demand (Olmstead and Stavins, 2007; Pulido-Velazquez et al., 2013). In our application of the model to our case study area, we simulate a pricing instrument that applies incremental water charges to recover the resource and environmental costs of water use.

3. Methods and data

3.1. PMAUP model setting

This paper integrates, for the first time to the best of our knowledge, a continuous crop-water agronomic production function into a PMAUP model. PMAUP modeling builds on the Theory of Planned Behavior (Ajzen, 1991), which argues that agent's responses stem from a "summary of psychological evaluations based on farmers' beliefs on the goodness or badness of an object", which "can be associated to multiple attributes that are often conflicting" (e.g. expected profit v. risk) (Gómez-Limón et al., 2016). Accordingly, PMAUP models feature a characteristic multi-attribute utility function that typically includes measures of profit, risk and management complexity, albeit other attributes can be explored (Bartolini et al., 2007; Gómez-Limón et al., 2016; Pérez-Blanco and Standardi, 2019; Rausser and Yassour, 1981). Agents in our PMAUP model decide on the allocation of land and water (i.e. the decision variables) so to maximize utility through the provision of the above-mentioned utility-relevant attributes within a feasible region conformed by a series of constraints (e.g. water availability, land availability):

$$\underset{x,w}{\mathrm{Max}} U = U(\boldsymbol{Z}(\boldsymbol{X},\boldsymbol{W})) = \prod_{p=1}^{m} z_{\mathrm{p}}^{a_{\mathrm{p}}}(\boldsymbol{X},\boldsymbol{W}) \tag{1}$$

s.t.

$$\sum_{i=1}^{n} x_i = 1.0 \le x_i \le 1 \tag{2}$$

$$\sum_{i=1}^{n} w_i x_i \le \mathrm{WA} \tag{3}$$

$$\boldsymbol{X},\boldsymbol{W} \in F \in R^n \tag{4}$$

$$\boldsymbol{Z}(\boldsymbol{X},\boldsymbol{W}) \in R^{\mathrm{m}} \tag{5}$$

Where U is the utility or objective function, which in our case adopts a Cobb-Douglas specification, as it is common practice in the PMAUP literature[5] (see e.g. Sapino et al., 2020); **X** is the crop portfolio vector, which contains information on the fraction of land allocated to each crop i, x_i; **W** is the water application vector, which contains information on the water applied to each crop per hectare, w_i; WA represents the average water availability per hectare; $\boldsymbol{Z}(\boldsymbol{X}, \boldsymbol{W})$ is the vector of attributes, a function of the decision variables in vectors $\boldsymbol{X}$, $\boldsymbol{W}$, which contains information on the provision of each utility-relevant attribute z_p (all attributes are defined so that "more-is-better", i.e. all else equal increasing the provision of a given attribute yields a utility gain); m is the number of individual attributes z_p and parameters a_{p} considered; and F is the feasible region, which includes the following constraints:

- *Land availability* (see Eq. (2) above). Available agricultural land is assumed constant in all simulations considered.
- *Water availability* (see Eq. (3)). In all simulations, water application has a maximum bound to the observed water allotment per hectare in the case study area.
- *Climate, soil, know-how.* Due to the specific climatic and soil characteristics, and irrigator's know-how, the crop portfolio is restricted to those crops that are already present in the area and observable in the database (which are also the only crops for which historical data is available and ad-hoc continuous crop-water production function can be calibrated) (Essenfelder et al., 2018).

$$\sum_{i=1}^{n} y_i x_i = 0 \,\Big|\, y_i \in \left\{0, 1\right\} \tag{6}$$

where $y_{\mathrm{i}} = 0$ means the crop is observable and $y_{\mathrm{i}} = 1$ means the crop is not observable in the area.

[4] An alternative way of setting water charges is to define safe minimum standards for the quantitative status of water bodies, and then impose a set of prices sufficient to achieve these standards. While not Pareto-efficient, such approach can achieve safe minimum standards at a minimum cost for the economy (Baumol and Oates, 1971).

[5] Multiplicative functions such as the Cobb-Douglas are regarded as a superior alternative to additive forms in multi-attribute modeling (Sampson, 1999). Cobb-Douglas functions comply with the Inada (1963) conditions and guarantee the existence of a global optimum, provided the efficiency frontier is convex. Since attributes' parameters (a_{j}) are all lower than one, Cobb-Douglas functions are also consistent with the neoclassical postulate of decreasing marginal utility. Attributes' parameters can be also interpreted as a weight or indicator of the relative importance of each attribute in driving agents' behavior (for an early discussion on this see e.g. Bronfenbrenner, 1944; Brown, 1957).

- *Crop-specific constraints*. Some crops in the portfolio have an upper and/or lower area bound. In our application to the SLD, this restriction is used to set a minimum/maximum threshold for Almond trees of ± 5%. Although the pricing policy instrument is designed to work in the long run and it could result in more than ± 5% crop portfolio changes, this may lead to significant (dis)investments with impacts not accounted for in our models, which rely on yearly market variables (notably profit) (Essenfelder et al., 2018). For example, perennials add value outside the agricultural business itself through carbon sequestration and amenities (e.g., landscape value) that would be lost if perennials are substituted by annuals. On the other hand, perennials involve non-trivial investment costs that are not captured by yearly market variables such as profit and would not be recovered and remain as sunk costs if the perennial is replaced by an annual crop (Loch et al., 2020a) .[6] Accurately representing long run changes in the surface of permanent crops would demand the inclusion of other relevant variables (e.g., carbon prices, Payment for Ecosystem Services) and is beyond the scope of this paper. Alternatively, a minimum (maximum) bound for ligneous trees is common practice in the literature (e.g. Gutiérrez-Martín and Gómez, 2011; Parrado et al., 2019).

Note that the optimization problem above is resolved for a representative hectare, and all output variables are expressed in units per hectare.

3.2. PMAUP model calibration

In order to elicit the parameters of the utility function (α), we adapt the PMAUP calibration method originally developed by Gutiérrez-Martín and Gómez (2011) for the case of a single decision variable $\boldsymbol{X}$, to the case of a PMAUP with a crop-water production function and two decision variables $\boldsymbol{X}, \boldsymbol{W}$.

Following standard economic theory, the parameters of the objective function are elicited by means of equalizing the opportunity cost of trading one unit of attribute z_k off for one unit of attribute z_p, i.e. the slope of the efficient frontier or Marginal Rate of Transformation (MRT_{kp}), to the willingness to give up one unit of attribute z_k in exchange for a unit of attribute z_p, i.e. the slope of the indifference curve of the utility function or Marginal Rate of Substitution (MRS_{kp}). Repeating this process for every possible combination of two individual attributes z_k, z_p within the finite set of attributes in the vector $\boldsymbol{Z}$, yields a system of equations that, after being solved, provides the parameter values.

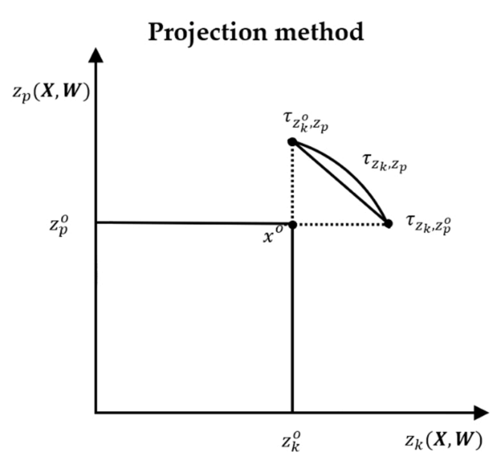

Fig. 2. Graphical representation of the approximation of the efficient frontier in the two-dimensional space pk using the projection method.
Source: own elaboration.

The MRS_{kp} is conditional on the specification used for the objective function. Under a Cobb-Douglas specification as the one used in this paper, the MRS_{kp} can be obtained as follows:

$$MRS_{kp} = -\frac{\partial U / \partial z_p}{\partial U / \partial z_k} = -\frac{\alpha_p}{\alpha_k}\frac{z_k}{z_p} \tag{7}$$

The main challenge in the calibration of PMAUP models concerns the elicitation of the efficient frontier to calculate the MRT_{kp}. The efficient frontier is defined as the maximum value of attribute z_p that can be achieved for a given value of attribute z_k given a series of restrictions, and vice versa. Marginal displacements along the efficient frontier give information on the opportunity cost between attributes, i.e. the cost of increasing the provision of attribute z_k in terms of attribute z_p, also known as the MRT_{kp}. Note that the term 'opportunity cost' implies convexity; if an efficient frontier is not found to be convex, there is no tradeoff between attributes and one of them should be removed from the attribute set. Since efficient frontiers "cannot be analytically defined using a closed function" (Gutiérrez-Martín and Gómez, 2011), numerical methods are typically used to estimate them. PMAUP literature reports several alternative methods to approximate the efficient frontier, among which the projection method developed by Gutiérrez-Martín and Gómez (2011) is the most commonly used (see e.g. Essenfelder et al., 2018; Gutiérrez-Martín et al., 2014; Parrado et al., 2019).

In fact, it is not necessary to know every point of the efficient frontier, but only those landing points between the efficient frontier and the utility function, where the MRT_{kp} will be equaled to the MRS_{kp}, as well as the slope of the efficient frontier (MRT_{kp}) in these points. The projection method starts by resolving two optimization problems in the two-dimensional space kp that calculate the maximum value that attribute z_k can achieve when z_p equals its observed value (i.e. z_p^o), and vice versa, for every $p \neq k$:

$$\underset{x,w}{Max}\; z_p(\boldsymbol{X},\boldsymbol{W}) \tag{8}$$

s.t.:

$$z_k(\boldsymbol{X},\boldsymbol{W}) = z_k^o(\boldsymbol{X},\boldsymbol{W})\ \forall k \neq p \tag{9}$$

And

$$\underset{x,w}{Max}\; z_k(\boldsymbol{X},\boldsymbol{W}) \tag{10}$$

s.t.:

$$z_p(\boldsymbol{X},\boldsymbol{W}) = z_p^o(\boldsymbol{X},\boldsymbol{W})\ \forall p \neq k \tag{11}$$

By solving the two optimization problems above we project the observed attribute values to the efficient frontier, thus obtaining two points (τ) within the efficient set: τ_{z_k, z_p^o} and $\tau_{z_k^o, z_p}$ (see Fig. 2).

These two points are subsequently connected through a hyperplane to approximate the MRT_{kp} through the slope β_{kp}^τ as follows:

$$MRT_{kp} = \beta_{kp}^\tau = \frac{z_p - z_p^o}{z_k - z_k^o} \tag{12}$$

Next the MRS_{kp} and the approximated MRT_{kp} are equalized for every combination of two attributes z_k and z_p to elicit the parameters of the objective function, as follows:

$$MRS_{kp} = -\frac{\partial U / \partial z_p}{\partial U / z_k} = -\frac{\alpha_p}{\alpha_k}\frac{z_k}{z_p} = \beta_{kp}^\tau = MRT_{kp}.\forall p \neq k \tag{13}$$

[6] Note that in Spanish Drought Management Plans, perennials are allotted a high priority: perennials are guaranteed the minimum amount of water needed for their survival (not the amount of water towards achieving a positive yield, though), and only after this amount of water is satisfied, annuals can receive any water.

$$\sum_{p=1}^{m} \alpha_p = 1 \tag{14}$$

The system of equations above is resolved for every possible attribute set (i.e. for alternative vectors $\boldsymbol{Z}$) considered, to elicit the corresponding objective function parameters. Next, for each set of attributes considered and related objective function parameters, we resolve the optimization problem in Eqs. (1)–(5) and obtain the simulated land (X^*) and water application (W^*) vectors, and the corresponding provision of attributes (z_p^*; p = 1,…,m). We next assess the performance of each set of attributes considered to represent observed behavior using three calibration residual metrics: i) e_x, the distance between the observed (x^o) and simulated land allocation (x^*), ii) e_w, the distance between the observed (w^o) and simulated water application (w^*), and iii) e_τ, the distance between the observed (z_p^o) and simulated attributes provision (z_p^*), which are then aggregated to calculate the average calibration residual e_m.

$$e_x = \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(\frac{x_i^o - x_i^*}{x_i^o}\right)^2} \tag{15}$$

$$e_w = \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(\frac{w_i^o - w_i^*}{w_i^o}\right)^2} \tag{16}$$

$$e_\tau = \sqrt{\frac{1}{m}\sum_{p=1}^{m}\left(\frac{z_p^o - z_p^*}{z_p^o}\right)^2} \tag{17}$$

$$e_m = \frac{\sqrt{e_x^2 + e_w^2 + e_\tau^2}}{3} \tag{18}$$

Of all the attribute sets explored and their related set of parameters and objective functions, the relevant attribute set and objective function that is used in the simulation exercises is the one that minimizes the average calibration residual e_m.

3.3. PMAUP model attributes

The attributes explored are selected based on a literature review on the Multi-Attribute Utility Theory (Bartolini et al., 2007; Gómez-Limón et al., 2016; Pérez-Blanco and Standardi, 2019; Rausser and Yassour, 1981), and include expected profit, risk avoidance and hired labor avoidance, a proxy of management complexity. Expected profit is the only attribute considered in single-attribute WPM, and critical towards explaining agents' choices. Risk avoidance reflects on the fact that agents are willing to sacrifice a fraction of expected profit so to reduce its variability. Finally, hired labor avoidance is a proxy of management complexity avoidance, which reflects on the fact that economic agents are willing to sacrifice a fraction of expected profit so to reduce the management complexity involved in their choices. The relevance of risk avoidance and management complexity avoidance attributes is visible in agents' choices, who rarely select a single profit maximizing crop, but rather a crop portfolio that balances the provision of utility-relevant attributes. All attributes are defined so that "more-is-better", i.e. all else equal increasing the provision of a given attribute yields a utility gain. Below we formally describe each of the attributes explored.

Expected profit (z_1) is obtained as the summation of the expected per hectare gross margin (π_i) of each crop i times the fraction of land allocated to that crop (x_i). The expected gross margin per hectare π_i is obtained as price (p_i, in EUR/kg) times yield (y_i, in kg/ha) plus coupled farm subsidies (s_i) minus variable costs (c_i, in EUR/ha). In the classical PMAUP model, p_i, y_i, s_i and c_i are obtained as the average values of longitudinal data on prices, yields, subsidies and variable costs, respectively. The innovation presented in this work is the integration of continuous crop-water production functions into the PMAUP model, transforming the expected profit attribute as follows:

$$z_1(\boldsymbol{X}, \boldsymbol{W}) = \sum_i x_i \pi_i(w_i) = \sum_i x_i(p_i y_i(w_i) + s_i - c_i(y_i(w_i))) \tag{19}$$

where $\boldsymbol{X}$ represents the *crop portfolio* vector, the first-choice variable in the model that allows for extensive and super-extensive margin adjustments; $\boldsymbol{W}$ is the *water application* vector, the second choice variable in the model that allows for intensive margin adjustments; and $y_i(w_i)$ and $c_i(y_i(w_i))$ are the crop-water production function and the variable costs function, respectively, which now are variable and depend on the decision of how much water to apply. Consistent with the majority of papers in our review, we approximate the crop-water production function through a quadratic function that adopts the following form:

$$y_i(w_i) = a_i w_i^2 + b_i w_i + d_i \tag{20}$$

Where $y_i(w_i)$ is the yield in kg per hectare, and a_i, b_i and d_i are the parameters of a quadratic function determining yield responses to alternative water application levels w_i (in m^3/ha), for a given crop i. If the crop can be cultivated under rainfed agriculture, d_i is positive and equals expected yield under rainfed agriculture; otherwise, it is zero or negative (Peña-Haro et al., 2014). For our case study in the SLD, a_i, b_i, and d_i were elicited using data from field experiments, combined with simulation outputs from a process-based agronomic model (Peña-Haro et al., 2014). The rationale for the use of simulations on top of field experiments in the calibration of the crop-water production function comes from the need to account for the complex impact and variability of different factors governing crop growth and yield other than water. If a researcher calibrates a production function using only a field experiment on water-yield relationship, she would be ignoring other factors that are responsible for the variation in crop yields from year to year (temporal variability) and across space (spatial variability). In fact the same plot, cultivated year after year in an identical way, without a priori limitations of any element (nutrients, water, other), has a temporal variability in yields due to climatic conditions, soil, etc. Similarly, there is also spatial variability across plots. To account for this variability, the literature on crop-water production functions complements field experiments with simulation models. By combining field experiments with process-based crop growth models, it is possible to capture the interacting effects of farmers' intra-seasonal irrigation decision-making, stochastic weather conditions, and physical and socio-economic water supply constraints on seasonal crop yield response to water. A recent review and application (through Aquacrop-OS) on how to simulate crop-water production functions using field experiments and process-based agronomic models is available in Foster and Brozović (2018). We adopt a similar approach in our model, where we use the process-based GIS-based Environmental Policy Integrated Climate (GEPIC) model (Liu et al., 2007) to account for temporal and spatial variability (GEPIC is the distributed GIS version of EPIC). The calibration results for the crop-water production functions of irrigated crops in the SLD are available in Section 4.1.1. Note that the production functions adopted in our paper are local and only have validity in the context of the area where they have been developed.

The variable costs function $c_i(y_i(w_i))$ adopts a linear form with respect of yield (in kg/ha), as follows:

$$c_i(y_i(w_i)) = e_i y_i(w_i) + f_i \tag{21}$$

Variable costs include plants and seeds, fertilizers, phytosanitary products, spare parts and repair services, subcontracting, hired labor and other supplies. Although national statistics only report two measures of variable costs per year and crop (under rainfed and irrigated agriculture), variable costs will typically be higher (lower) the higher (lower) the yield (e.g. more labor during harvest). We adjust the crop-water production function to account for this by making the variable costs of crop i a linear function of yield with a fixed (f_i, a parameter representing the minimum threshold for variable costs) and a variable

Table 2
PMAUP model data inputs.

Variable	Abbreviation used	Data provider	Ref. year	Granularity
Crop portfolio (% over total surface)	x_i		2015	Hectares per crop at municipality level (NUTS4)
Crop yield (kg/ha) and water applied (m^3/ha) (crop-water production function, annual crops)	y_i, w_i	Adapted from Fabeiro Cortés et al. (2003); ITAP (2005) and Peña-Haro et al. (2014)	2000 and 2009	Water basin level
Crop yield (kg/ha) and water applied (m^3/ha) (crop-water production function, permanent crops)	y_i, w_i	Adapted from Jiménez et al. (2004) and Peña-Haro et al. (2014)	2000 and 2009	Water basin level
Crop yield (kg/ha) (only for the variance and covariance matrix)	y_i	Adapted from MAGRAMA (2015)	2008–2015	Agricultural District (*Comarca*)
Prices (EUR/kg)	p_i	MAGRAMA (2015)	2008–2015	National (NUTS1)
Costs (EUR/ha) and subsidies (EUR/ha)	c_i, s_i	Adapted from MAPA (2019)	2008–2015	Agricultural District
Number of working days (days/ha)	N_i	Adapted from MAPA (2019)	2008–2015	Region (NUTS2)

Source: Own elaboration. MAGRAMA (2015)

(e_i) component, calculated as follows:

$$e_i = \left(c_{i,\ irrigated} - c_{i,\ rainfed}\right) / \left(\left(\max\left(y_i(w_i)\right)\right) = min\left(y_i(w_i)\right)\right) \tag{22}$$

$$f_i = c_{i,\ rainfed} \tag{23}$$

where $c_{i,\ irrigated}$ are the average values of longitudinal data on variable costs for crop i under irrigated agriculture, $c_{i,\ rainfed}$ are the average values of longitudinal data on variable costs for crop i under rainfed agriculture, $\max(y_i(w_i))$ is the maximum yield attainable in the crop-water production function for crop i and $\min(y_i(w_i))$ represents rainfed yield for crop i. If no rainfed alternative is available for crop i, $\min(y_i(w_i))$ and $c_{i,\ rainfed}$ equal 0.

Risk avoidance (z_2) is obtained as the profit variability (measured through the variance and covariance matrix) attached to the profit maximizing combination of land ($\overline{X}$) and water inputs ($\overline{W}$) minus the profit variability attached to the land (X) and water input (W) allocation chosen by the agent (recall attributes are defined so that "more-is-better") (Gómez-Limón et al., 2016):

$$z_2\ (\boldsymbol{X}, \boldsymbol{W}) = \overline{\boldsymbol{X}}^t VCV\left(\boldsymbol{\pi}\left(\overline{\boldsymbol{W}}\right)\right)\overline{\boldsymbol{X}} - \boldsymbol{X}^t V\ CV(\boldsymbol{\pi}(\boldsymbol{W}))\boldsymbol{X} \tag{24}$$

where $VCV(\boldsymbol{\pi}(\boldsymbol{W}))$ is the variance and covariance matrix of the gross variable margin, and π is a vector that contains the per hectare gross margin of each crop π_i. Note that information on yields under deficit irrigation for the crops in the SLD is available for a maximum of two years. While this makes possible to calibrate a crop-water production function $y_i(w_i)$ using a combination of field data and agronomic model simulations (Peña-Haro et al., 2014) (see Section 4.1), insufficient longitudinal data on crop yield for alternative water application levels precludes the calculation of a variance and covariance matrix that differentiates between crop and water application levels. Alternatively, we can obtain the variance and covariance matrix using observed longitudinal data on yield per crop available in official statistics, which is obtained as total irrigated (rainfed if observed) crop production at an agricultural district level divided by the surface of that irrigated (rainfed) crop in that agricultural district (i.e. without distinguishing water application levels). This means we have only one (two, if the series is available for both rainfed and irrigated technique) longitudinal series per crop, instead of one longitudinal series per crop and water application level. Thus, risk avoidance is assumed to be the same for different levels of water applied for the same crop. This is a limitation of the model that can only be addressed with additional longitudinal data that is currently unavailable.

Hired labor avoidance (z_3) is measured as the difference between the labor requirements of the profit maximizing combination of land ($\overline{X}$) and water inputs ($\overline{W}$) minus the labor requirements of the alternative/simulated land (X) and water input (W) combination (recall attributes are defined so that "more-is-better"):

$$z_3\ (\boldsymbol{X}, \boldsymbol{W}) = \sum_i \overline{x_i} N_i(y_i(\overline{w_i})) - \sum_i x_i N_i(y_i(w_i)) \tag{25}$$

where:

$$N_i(y_i(w_i)) = g_i y_i + h_i \tag{26}$$

$$g_i = \left(N_{i,\ irrigated} - N_{i,\ rainfed}\right) / \left(\left(\max\left(y_i(w_i)\right)\right) - min\left(y_i(w_i)\right)\right) \tag{27}$$

$$h_i = N_{i,\ rainfed} \tag{28}$$

where $N_{i,\ irrigated}$ are the average values of longitudinal data on labor requirements (number of days) per hectare for crop i under irrigated agriculture, $N_{i,\ rainfed}$ are the average values of longitudinal data on labor requirements per hectare for crop i under rainfed agriculture. If no rainfed alternative is available for crop i, $\min(y_i(w_i))$ and $N_{i,\ rainfed}$ equal 0.

3.4. Data

Table 2 summarizes data inputs for the PMAUP and crop-water production function model and related data providers.

4. Results

4.1. Calibration results

4.1.1. Calibration of the crop-water production functions

Crop-water production functions for annual crops in the SLD (namely, wheat, barley, corn, onion, garlic) were generated combining field data and agronomic simulations using the GEPIC model (Liu et al., 2007), building on previous work by Peña-Haro et al. (2014) in the SLD.

Table 3
Calibration results of the crop-water production functions for the main crops in the SLD.

	a ($kg.\frac{ha}{m^6}$)	b ($\frac{kg}{m^3}$)	d ($\frac{kg}{ha}$)	Correlation
Wheat	-0.00111	4.9830	1788.0	0.72
Barley	-0.00081	3.5000	1700.0	0.96
Corn	-0.00033	5.5398	-5399.2	0.88
Garlic	-0.00010	2.7000	3511.3	0.98
Onion	-0.00159	27.7090	-35848.7	0.87
Almond	-0.00004	0.4553	302.2	0.96

Source: Own elaboration.

Table 4
Calibration results and calibration residuals of the PMAUP model.

Attribute (z_p)	z_1	z_2	z_3	e_m
Parameter value (α_p)	0.915	0.079	0.006	1.42%

Source: Own elaboration.

GEPIC is a distributed version of the EPIC model (Williams et al., 1983) through a loose coupling between ArcGis and the EPIC model. In our application to the SLD, GEPIC was calibrated using the outcomes of field experiments that assessed the effect of water applied on the yield of annual crops. Field experiments were conducted in 2000 and 2009 growing seasons at the experimental station "Las Tiesas" in the SLD (Fabeiro Cortés et al., 2003; ITAP, 2005). Paired values of crop yield per level of applied water in the field experiments v. modelled yield were compared using regression analysis in order to calibrate the production functions. Crop responses to different water application values in the different type of soils and climatic areas were simulated in order to generate enough variability to fit the coefficients of the crop-water production functions. Note that since the production of corn and onion is considered unfeasible under rainfed agriculture in the SLD, the *d* parameter adopts a negative value for these crops.

Since there are no process-based agronomic models that simulate ligneous crops, crop-water production functions for ligneous crops in the SLD (almond) were directly calibrated from observed data using the results of deficit irrigation field experiments developed by the ITAP near the SLD area (Jiménez et al., 2004). Accordingly, crop-water production functions for almond trees do not account for the complex impact and variability of different factors governing crop growth and yield other than water. We consider nonetheless that having a production function with limitations is preferrable to ignoring a relevant crop in the case study area, and therefore have chosen to include this production function in the model.

Calibration results for the crop-water production functions in the SLD are reported in the Table 3. The table also reports the correlation between the values estimated with the agronomic model and the ones simulated with the quadratic function: the values range from 0 (worst performance of the quadratic function) to 1 (best performance) (Peña-Haro et al., 2010). The reader is referred to Peña-Haro et al. (2014) for a more detailed discussion of the results, methods and a discussion on the precision of the calibrated model.

4.1.2. PMUAP model calibration

PMAUP model calibration results and residuals for the SLD are shown in Table 4.

The columns α_1, α_2 and α_3 display the parameter values of the Cobb-Douglas utility function for the attributes profit (z_1), risk avoidance (z_2) and hired labor avoidance (z_3), while e_m is the average calibration residual. Calibration results show that the most relevant attribute driving agents' decisions is profit. Risk avoidance has also a relevant role in explaining the behavior of irrigators in the SLD. The attribute measuring management complexity avoidance (z_3) is marginally relevant. Caution must be exercised in interpreting the results. For example, it cannot be inferred that a high risk avoidance parameter will yield a low profit variability, since choices are ultimately constrained by the feasible region. Nonetheless, attribute parameters offer valuable insights on agent's preferences and can serve to project behavior, provided calibration errors are low. In the case of the SLD, metrics for performance evaluation are satisfactory, with a "very low" average calibration residual (Pérez-Blanco et al., 2015).

4.2. Simulation results

We simulate a progressive increase of water prices in the SLD from 0 to 1 EUR/m^3 at 0.01 EUR/m^3 intervals. For each simulation run, we obtain the crop portfolio and water application responses by irrigators and calculate the compensating variation (i.e. monetized foregone utility), foregone income and water saved. Simulations are run using the model setting described in Section 3, which integrates the crop-water production function of irrigated crops into the objective function and allows for adjustments at the intensive, extensive and super-extensive margin (W-PMAUP). The results thus obtained are subsequently compared with those from an alternative classic PMAUP model setting (C-PMAUP) where the continuous agronomic production functions are substituted by point values that represent average expected production under irrigated and/or rainfed agriculture for each crop (i.e., a maximum of two points per crop), thus allowing only for extensive and super-extensive margin adjustments (Pérez-Blanco and Gutiérrez-Martín, 2017). The difference between the modeling outcomes from these two model settings reveals the net effect of intensive margin adjustments on the expected economic and environmental performance of water pricing. Note that only the W-PMAUP model is calibrated: the C-PMAUP model uses the same parameters obtained for the W-PMAUP, with different crop-water production functions (point values representing crop yield under rainfed and irrigated agriculture, instead of continuous crop-water production function). If we calibrated both models separately, considering a continuous production function for the W-PMAUP and point values for the C-PMAUP, calibration results would differ both because of the alternative model settings *and because of the different data inputs*. In reality, though, intensive margin adjustments are a feasible option, which is indeed frequently adopted by farmers in the SLD and elsewhere, meaning that any model calibrated on a database that

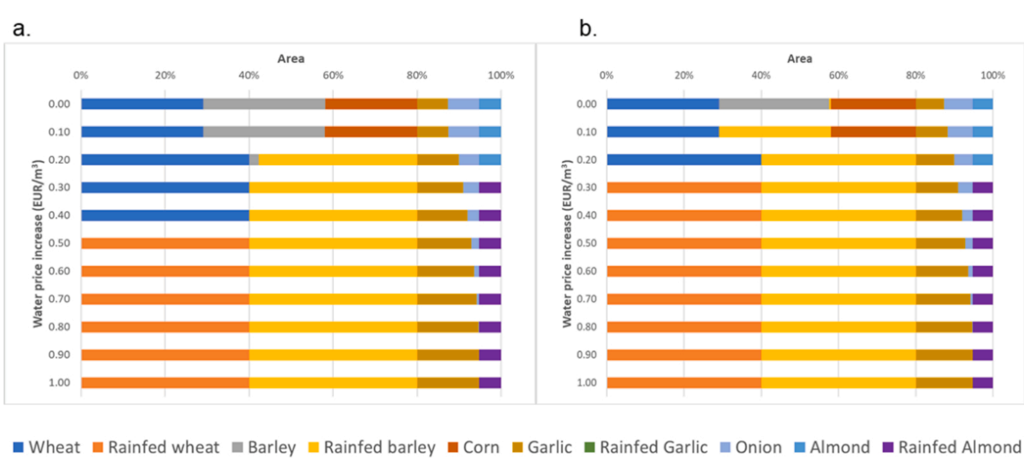

Fig. 3. Extensive and super-extensive margin adjustment (land allocation decisions) in W-PMAUP (a.) and in C-PMAUP model setting (b.) Source: own elaboration.

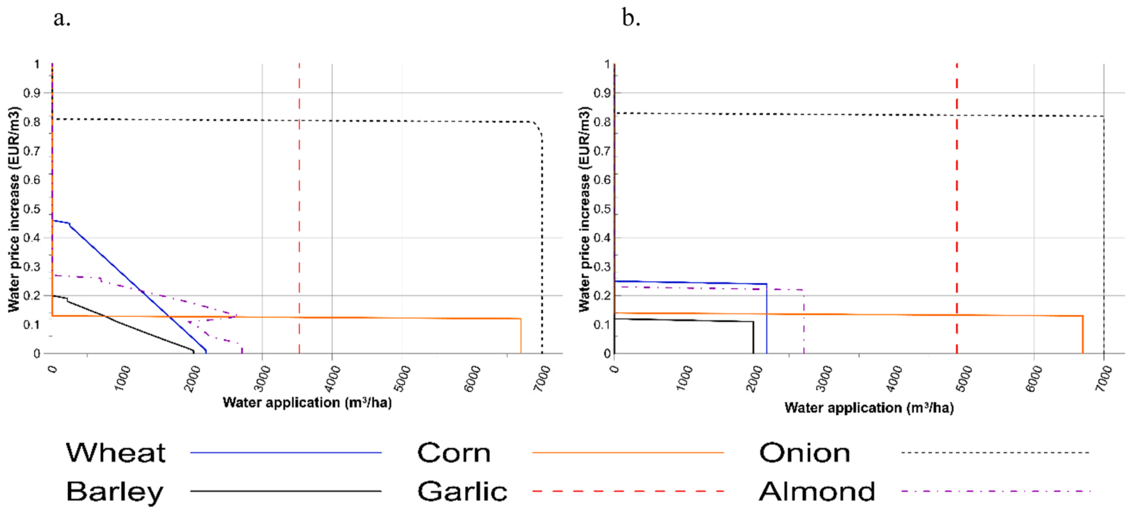

Fig. 4. Intensive margin adjustment (water application decisions) in W-PMAUP (a.) and C-PMAUP model setting (b.) Source: own elaboration.

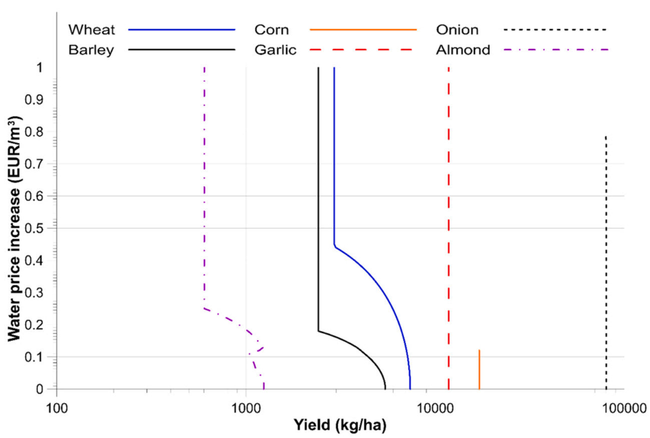

Fig. 5. Yield (kg/ha) per crop in the W-PMAUP model setting under alternative water prices (logarithmic scale).
Source: own elaboration.

excludes this option would incur in data measurement errors. Therefore, we use a single database (the one closer to observed irrigators' decisions, which includes continuous agronomic crop-water production functions) and, once the model is calibrated (W-PMAUP), we replace the continuous agronomic production functions by point values (C-PMAUP) to compare simulation outcomes with both settings, and thus reveal the net effect of intensive margin adjustments.

Fig. 3 and Fig. 4 represent agents' responses to incremental water pricing under the W-PMAUP and C-PMAUP model settings. In terms of land allocation to alternative crops (extensive and super-extensive margin adjustments), both model settings show similar responses in every simulation run (see Fig. 3); albeit dissimilarities arise regarding water application to crops (intensive water adjustment) (Fig. 4). Under the W-PMAUP model setting, agents can respond to higher prices by progressively decreasing the amount of water applied to irrigated crops (deficit irrigation). Deficit irrigation is observed in the W-PMAUP model for all crops with the exception of garlic, the most profitable crop in the case study area, which is fully irrigated in all simulations; and water intensive corn, for which irrigation abruptly stops in both models after a charge increase of 0.13 EUR/m^3. On the other hand, in the C-PMAUP model agents are constrained to apply water in fixed proportions to land, meaning that intensive margin adjustments are not possible and crops receive a constant amount of water inputs until abruptly interrupting irrigation and shifting to rainfed agriculture.

According to Graveline and Mérel (2014), there are three critical factors conditioning intensive margin adjustments: (i) water intensity (water-intensive crops are those that can contribute more significantly towards water saving); (ii) yield elasticity to water use (the higher yield elasticity to water use, the lower deficit irrigation is observed); and (iii) profitability (crops with higher profit will be less affected by deficit irrigation). It can be observed that these three factors explain water application responses to pricing in the W-PMAUP model setting. Garlic, the most profitable crop, can afford a price increase of up to 1 EUR/m^3 without applying deficit irrigation (i.e. a 600–1000% increase in the volumetric cost of water as compared to the observed groundwater pumping costs of 0.1–0.20 EUR/m^3); while in the case of onion, the second most profitable crop, water applied per hectare remains constant until a charge increase of 0.78 EUR/m^3 (a 490–780% water cost increase), at which point onion is replaced by other crops (with deficit irrigation briefly applied in the interim). Barley and wheat are the crops with the lowest yield elasticity to water use; accordingly, deficit irrigation is observed from the initial water price increases, until both crops eventually shift to rainfed agriculture, which happens at a water charge increase of 0.19 (barley) and 0.44 EUR/m^3 (wheat). Almond starts deficit irrigation in the initial simulation runs, with a slight rebound at a 0.12 EUR/m^3 price increase due to a substitution effect with corn. In the case of corn, the high yield elasticity to water application overtakes all the other effects, meaning that irrigated corn is abruptly substituted by less water-intensive crops without intermediary deficit irrigation at a charge increase of 0.12 EUR/m^3.

The possibility to adapt at the intensive margin in the W-PMAUP setting means crop yield per hectare is not constant anymore across the alternative price simulations, as happens in models where water input is applied in fixed proportions to land (C-PMAUP setting). Fig. 5 shows how different water application choices affect yield (in kg/ha) for each crop under the W-PMAUP model setting. For those crops that can be cultivated under rainfed agriculture (wheat, barley and almond), irrigation water is progressively diminished and yield reduces, until irrigation stops altogether and yield equates that under rainfed agriculture.

Fig. 6 shows the water demand curve representing the relationship between water prices and water application, for both the W-PMAUP and C-PMAUP model settings. Simulation results using the C-PMAUP model setting displays a "jumpy" behavior with (quasi-)inelastic responses in the initial and final stretches of the demand function. This outcome is consistent with those coming from the literature on agricultural water pricing under scarcity, where the model structures adopted also ignore deficit irrigation/intensive margin adjustments. On the other hand, the

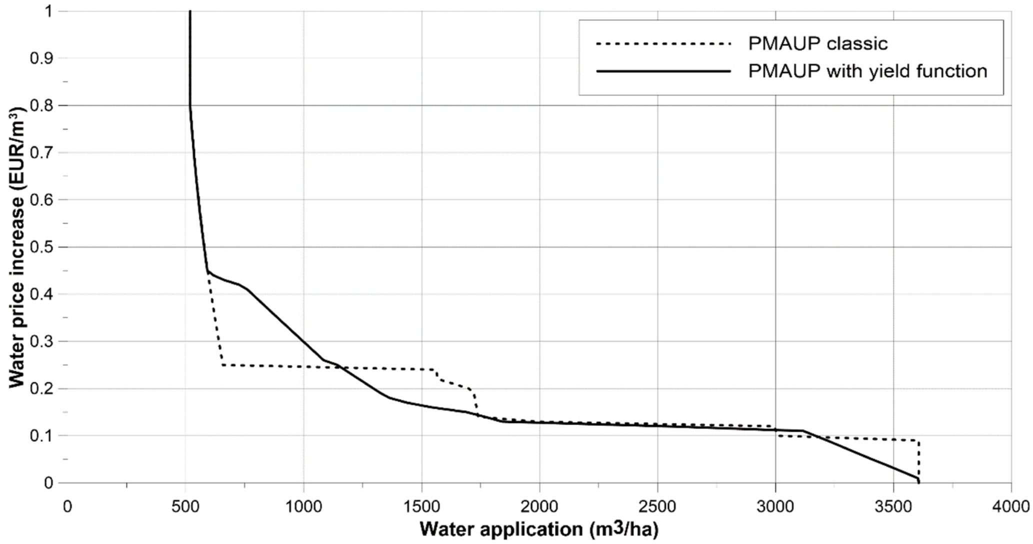

Fig. 6. Agricultural water demand curve.
Source: own elaboration.

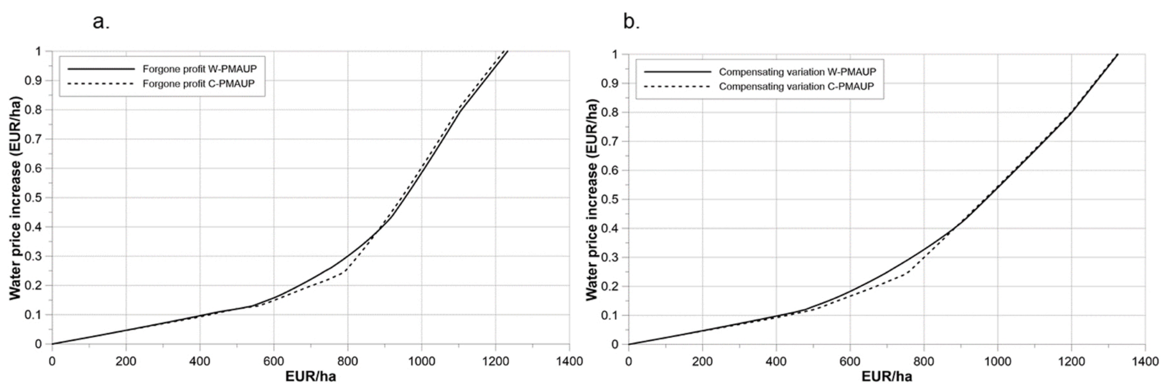

Fig. 7. Forgone profit (a.) and compensating variation (b.), water charges.
Source: own elaboration.

W-PMAUP model setting that allows for deficit irrigation displays a gradual reduction in water use along with higher prices in the initial and middle stretches of the demand function, and a quasi-inelastic response for a price increase of 0.43 EUR/m^3 or higher. This result suggests that having (or not) quasi-inelastic responses in the initial and middle stretches of the demand function is conditional on the model structure/settings choice. The upshot is that where intensive margin adaptation responses are possible in the model, water pricing is cost-effective towards water saving in the initial stretches of the demand function, which thus far have been often assumed to be inelastic. For example, a price increase of 0.09 EUR/m^3, which represents a non-trivial price increase of 45%– 90% in the SLD, is ineffective towards water saving in the C-PMAUP, but can save up to 390 m^3/ha (11% of the average water application of 3606 m^3/ha) in the W-PMAUP model setting. In addition, quasi-inelastic responses in the final stretches of the demand function, while appearing in both model settings, emerge at higher prices in the W-PMAUP (from 0.43 EUR/m^3) as compared to the C-PMAUP model setting (from 0.25 EUR/m^3).

Fig. 7 reports the economic impact of higher prices in terms of foregone profit (a.) and the monetized utility loss or compensating variation (b.), i.e. the amount of money the irrigator would need to achieve his initial utility following price increases. The compensating variation is consistently higher for the C-PMAUP model setting in all simulations, indicating a larger utility loss as compared to the W-PMAUP model setting, which is particularly visible in the charge increase interval between 0.1 EUR/m^3 and 0.4 EUR/m^3 (between 1% and 8% higher compensating variation in the C-PMAUP than in the W-PMAUP model setting). A similar outcome is observed for the foregone profit, which again is higher in the C-PMAUP model setting up to 0.4 EUR/m^3 increase, with a marked gap in the 0.1 EUR/m^3 - 0.4 EUR/m^3 interval (between 0.7% and 8.3% higher foregone income in the C-PMAUP than in the W-PMAUP model setting). The superior economic performance under the W-PMAUP setting is attributable to the increased number of adaptive responses available in the W-PMAUP as compared to the C-PMAUP and reveals the net effect of intensive margin adjustments on economic outputs.

5. Conclusion

This paper integrates a continuous crop-water production function into a PMAUP model to assess the influence of intensive margin adjustments on the expected water saving and economic performance of water pricing. The model is calibrated for an agricultural area in a water scarce agricultural area in southeastern Spain (El Salobral-Los Llanos in the Júcar River Basin), so to factor in climatic, soil and other local factors conditioning the crop yield-water input relationship. Results reveal non-trivial dissimilarities in the economic and water saving performance of the W-PMAUP (intensive, extensive and super-extensive margin adjustments) and C-PMAUP (only extensive and super-extensive margin adjustments) model settings. The quasi-inelastic responses in the initial and middle stretches of the C-PMAUP demand function, which are consistent with findings in the literature on water charges in water scarce areas, are transformed into elastic responses in the W-PMAUP, suggesting a more cost-effective contribution of water pricing towards water saving. Ignoring intensive margin adjustments also tends to overestimate the economic impact of water pricing, with a higher foregone profit and compensating variation obtained in the C-PMAUP model setting as compared to the W-PMAUP model setting in all simulation runs. The compensating variation (foregone profit) is up to 1%−8% (0.7%−8.3%) higher in the C-PMAUP than in the W-PMAUP model setting for the water price increase range 80%− 200%. Given that deficit irrigation is a commonly used adaptation response in the SLD, we argue that ignoring it may not accurately reflect the actual adaptation options available to irrigators, and recommend the use of WPM that integrate crop-water production functions.

The model proposed in this paper can be improved in several ways. Future efforts should aim at gathering longitudinal data on yields for alternative water application levels with high granularity through remote sensing, surveys, field experiments or other means, so to build increasingly accurate site-specific crop-water production functions and more detailed utility-relevant attributes (e.g. through the development of a variance and covariance matrix that differentiates between crop types and water application in the risk avoidance attribute). This will facilitate the application of the model and replication of the experiments elsewhere, a prerequisite to validate the preliminary findings obtained for our case study area in the SLD. Note that although calibration residuals are low in the case study area, performance may be less satisfactory elsewhere, which calls for exploring alternative/additional attributes (e.g. alternative definitions of the management complexity avoidance attribute) and more comprehensive and spatially detailed data (such as water use data) to better define the feasible region, e.g. leveraging on earth observation and in situ monitoring, digital data acquisition and management and predictive analytics. Furthermore, other input (e.g., fertilizer) could be introduced in the production function to consider other aspects of the complex mechanism that govern crop growth and yield, even though this goes beyond the object of this paper, which is to test the performance of a crop-water production function in saving water through pricing. Building multi-model ensemble experiments combining multiple WPM that allow for intensive margin adjustments can help us sample modeling uncertainty and establish confidence intervals for the environmental and economic performance of alternative water conservation policies (Sapino et al., 2020). Beyond multi-model ensembles, decision-making should be also informed by multiple scenarios that complement the pricing simulations in our model, which may in turn reveal far-reaching and under-recognized implications for our policy (Pannell, 2006). For example, our findings rely on the assumption that both water charges *and water allocations* are effectively enforced; however, where agents can resort to alternative sources of water (e.g., through illegal aquifer withdrawals), higher charges may lead to an increase in the use of alternative sources (which calls for complementary policies, such as use of remote sensing and penalties in the case of water theft) (Loch et al., 2020b). Coupling the proposed PMAUP model with a hydrologic model and a river basin management model is also necessary to assess the environmental impacts of water conservation across space, thus supporting the identification of potential distributive issues (e.g. upstream v. downstream water conservation). Finally, accurate estimates on the environmental and resource costs of water are needed to design realistic water pricing scenarios that substitute the hypothesized scenarios adopted in our simulations; albeit these costs are at present difficult to obtain given that there are "few standardized methods" to measure the economic value of water, and there are often "large differences between values obtained through different methods" (UN, 2021b).

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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References

Adamson, D., Mallawaarachchi, T., Quiggin, J., 2007. Water use and salinity in the Murray–Darling Basin: a state-contingent model*. Aust. J. Agric. Resour. Econ. 51, 263–281. https://doi.org/10.1111/j.1467-8489.2007.00380.x.

Ajzen, I., 1991. The theory of planned behavior. Organ. Behav. Hum. Decis. Process., Theor. Cogn. Self-Regul. 50, 179–211. https://doi.org/10.1016/0749-5978(91)90020-T.

Apperl, B., Pulido-Velazquez, M., Andreu, J., Karjalainen, T.P., 2015. Contribution of the multi-attribute value theory to conflict resolution in groundwater management – application to the Mancha Oriental groundwater system, Spain. Hydrol. Earth Syst. Sci. 19, 1325–1337. https://doi.org/10.5194/hess-19-1325-2015.

Arata, L., Donati, M., Sckokai, P., Arfini, F., 2017. Incorporating risk in a positive mathematical programming framework: a dual approach. Aust. J. Agric. Resour. Econ. 61, 265–284. https://doi.org/10.1111/1467-8489.12199.

Bartolini, F., Bazzani, G.M., Gallerani, V., Raggi, M., Viaggi, D., 2007. The impact of water and agriculture policy scenarios on irrigated farming systems in Italy: an analysis based on farm level multi-attribute linear programming models. Agric. Syst. 93, 90–114. https://doi.org/10.1016/j.agsy.2006.04.006.

Baumol, W.J., Oates, W.E., 1971. The use of standards and prices for protection of the environment. Swed. J. Econ. 73, 42–54. https://doi.org/10.2307/3439132.

Berbel, J., Calatrava, J., Garrido, A., 2007. Water pricing and irrigation: a review of the European experience, in: Molle, F., Berkoff, J. (Eds.), Irrigation Water Pricing: The Gap between Theory and Practice. CABI, Wallingford, pp. 295–327.

Bronfenbrenner, M., 1944. Production functions: Cobb-douglas, interfirm, intrafirm. Econometrica 12, 35–44. https://doi.org/10.2307/1905566.

Brown, E.H.P., 1957. The meaning of the fitted Cobb-Douglas function. Q. J. Econ. 71, 546–560. https://doi.org/10.2307/1885710.

Connor, J., Schwabe, K., King, D., Kaczan, D., Kirby, M., 2009. Impacts of climate change on lower Murray irrigation*. Aust. J. Agric. Resour. Econ. 53, 437–456. https://doi.org/10.1111/j.1467-8489.2009.00460.x.

Connor, J.D., Schwabe, K., King, D., Knapp, K., 2012. Irrigated agriculture and climate change: the influence of water supply variability and salinity on adaptation. Ecol. Econ. 77, 149–157. https://doi.org/10.1016/j.ecolecon.2012.02.021.

Cornish, G., Bosworth, B., Perry, C., Burke, J.J., Food and Agriculture Organization of the United Nations, 2004. Water charging in irrigated agriculture: an analysis of international experience (Report No. 28), FAO Water Reports. Food and Agriculture Organization of the United Nations, Rome (Italy).

Cortignani, R., Severini, S., 2009. Modeling farm-level adoption of deficit irrigation using Positive Mathematical Programming. Agric. Water Manag. 96, 1785–1791. https://doi.org/10.1016/j.agwat.2009.07.016.

Dinar, A., Pochat, V., Albiac, J. (Eds.), 2015a. 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., Subramanian, A., 1997. Water pricing experiences: an international perspective (Report No. WTP386). The World Bank.

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

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.

Eurostat, 2020a. Eurostat Database [WWW Document]. Eurostat Database. URL ⟨http://epp.eurostat.ec.europa.eu/portal/page/portal/statistics/themes⟩ (accessed 10.5.13).

Fabeiro Cortés, C., Martín de Santa Olalla, F., López Urrea, R., 2003. Production of garlic (Allium sativum L.) under controlled deficit irrigation in a semi-arid climate. Agric. Water Manag. 59, 155–167. https://doi.org/10.1016/S0378-3774(02)00125-7.
FAO, 2021a. FaoStat [WWW Document]. Food and Agriculture Organization of the United Nations. URL ⟨http://faostat.fao.org/⟩ (accessed 2.10.21).
FAO, 1992. CROPWAT, a computer program for irrigation planning and management by M. Smith. FAOIrrigation and Drainage Paper No. 26. Rome.
Fereres, E., Soriano, M.A., 2007. Deficit irrigation for reducing agricultural water use. J. Exp. Bot. 58, 147–159. https://doi.org/10.1093/jxb/erl165.
Finger, R., Schmid, S., 2008. Modeling agricultural production risk and the adaptation to climate change. Agric. Financ. Rev. 68, 25–41. https://doi.org/10.1108/00214660880001217.
Foster, T., Brozović, N., 2018. Simulating crop-water production functions using crop growth models to support water policy assessments. Ecol. Econ. 152, 9–21. https://doi.org/10.1016/j.ecolecon.2018.05.019.
Frisvold, G.B., Konyar, K., 2012. Less water: how will agriculture in Southern Mountain states adapt? Water Resour. Res. 48. https://doi.org/10.1029/2011WR011057.
García-Vila, M., Fereres, E., 2012. Combining the simulation crop model AquaCrop with an economic model for the optimization of irrigation management at farm level. Eur. J. Agron. 36, 21–31. https://doi.org/10.1016/j.eja.2011.08.003.
Gómez-Limón, J.A., Gutiérrez-Martín, C., Riesgo, L., 2016. Modeling at farm level: positive multi-attribute utility programming. Omega 65, 17–27. 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. Softw. 81, 12–25. https://doi.org/10.1016/j.envsoft.2016.03.004.
Graveline, N., Loubier, S., Gleyses, G., Rinaudo, J.-D., 2012. Impact of farming on water resources: assessing uncertainty with Monte Carlo simulations in a global change context. Agric. Syst. 108, 29–41. https://doi.org/10.1016/j.agsy.2012.01.002.
Graveline, N., Majone, B., Van Duinen, R., Ansink, E., 2014. Hydro-economic modeling of water scarcity under global change: an application to the Gállego river basin (Spain). Reg. Environ. Change 14, 119–132. https://doi.org/10.1007/s10113-013-0472-0.
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. Span. 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. Bournaris, T., J. Berbel, B. Manos, D. Viaggi, Boca Raton (US), pp. 319–348.
Heinz, I., Pulido-Velazquez, M., Lund, J.R., Andreu, J., 2007. Hydro-economic modeling in river basin management: implications and applications for the european water framework directive. Water Resour. Manag. 21, 1103–1125. https://doi.org/10.1007/s11269-006-9101-8.
Howitt, R.E., MacEwan, D., Medellín-Azuara, J., Lund, J.R., 2009. Economic Modeling of Agriculture and Water in California Using the Statewide Agricultural Production Model.
Inada, K.-I., 1963. On a two-sector model of economic growth: comments and generalization. Rev. Econ. Stud. 30, 119–127.
ITAP, I.T.A.P., 2020. Personal communication.
ITAP, I.T.A.P., 2005. S.A.
JCRMO, 2009. Aumento de los costes de las facturas eléctricas en los regadío de la Mancha Oriental. (Informe Tecnico). Junta Central de regantes de La Mancha Oriental.
Jiménez, M., Fuster, L., Ibáñez, R., 2004. ENSAYO DE EFICIENCIA DEL USO DEL AGUA EN EL ALMENDRO 19.
JRBA, J.R.B.A., 2016. Plan Hidrol. De. Cuenca 2015–2021.
Kampas, A., Petsakos, A., Rozakis, S., 2012. Price induced irrigation water saving: unraveling conflicts and synergies between European agricultural and water policies for a Greek Water District. Agric. Syst. 113, 28–38. https://doi.org/10.1016/j.agsy.2012.07.003.
Koundouri, P., 2004. Current issues in the economics of groundwater resource management. J. Econ. Surv. 18, 703–740. https://doi.org/10.1111/j.1467-6419.2004.00234.x.
Liu, J., Williams, J.R., Zehnder, A.J.B., Yang, H., 2007. GEPIC – modelling wheat yield and crop water productivity with high resolution on a global scale. Agric. Syst. 94, 478–493. https://doi.org/10.1016/j.agsy.2006.11.019.
Loch, A., Adamson, D., Auricht, C., 2020a. (g)etting to the point: the problem with water risk and uncertainty. Water Resour. Econ. 32, 100154 https://doi.org/10.1016/j.wre.2019.100154.
Loch, A., Pérez-Blanco, C.D., Carmody, E., Felbab-Brown, V., Adamson, D., Seidl, C., 2020b. Grand theft water and the calculus of compliance. Nat. Sustain. 3, 1012–1018. https://doi.org/10.1038/s41893-020-0589-3.
MAGRAMA, 2015. Hojas 1T de superficies municipales de cultivos (Database), Hojas 1T. Ministerio de Agricultura, Alimentación y Medio Ambiente, Madrid (Spain).
MAPA, 2019. ECREA: Estudios de costes y rentas de las explotaciones agrarias (Report). Ministerio de Agricultura, Pesca y Alimentación, Madrid (Spain).
Medellín-Azuara, J., Harou, J.J., Howitt, R.E., 2010. Estimating economic value of agricultural water under changing conditions and the effects of spatial aggregation. Sci. Total Environ., Spec. Sect.: Integr. Water Agric. Manag. Clim. Change 408, 5639–5648. https://doi.org/10.1016/j.scitotenv.2009.08.013.
Medellín-Azuara, J., Howitt, R.E., Harou, J.J., 2012. Predicting farmer responses to water pricing, rationing and subsidies assuming profit maximizing investment in irrigation technology. Agric. Water Manag., Irrig. Effic. Product.: Scales, Syst. Sci. 108, 73–82. https://doi.org/10.1016/j.agwat.2011.12.017.
Mérel, P., Simon, L.K., Yi, F., 2011. A fully calibrated generalized constant-elasticity-of-substitution programming model of agricultural supply. Am. J. Agric. Econ. 93, 936–948. https://doi.org/10.1093/ajae/aar029.
Mérel, P., Yi, F., Lee, J., Six, J., 2014. A regional bio-economic model of nitrogen use in cropping. Am. J. Agric. Econ. 96, 67–91. https://doi.org/10.1093/ajae/aat053.
Molden, D., Oweis, T., Steduto, P., Bindraban, P., Hanjra, M.A., Kijne, J., 2010. Improving agricultural water productivity: between optimism and caution. Agric. Water Manag. Compr. Assess. Water Manag. Agric. 97, 528–535. https://doi.org/10.1016/j.agwat.2009.03.023.
Montilla-López, N.M., Gutiérrez-Martín, C., Gomez-Limon, J., 2017. Impacto de la tarifación del agua de riego en el Bajo Guadalquivir. Itea. Inf. Técnica Económica Agrar. 113, 90–111. https://doi.org/10.12706/itea.2017.006.
OJ, 2000. Water Framework Directive 2000/60/EC, Council Directive.
Olmstead, S., Stavins, R., 2007. Managing Water Demand – Price vs. Non-Price Conservation Programs (Report). Pioneer Institute for Public Policy Research, Boston, MA.
Ortega Álvarez, J.F., de Juan Valero, J.A., Tarjuelo Martín-Benito, J.M., López Mata, E., 2004. MOPECO: an economic optimization model for irrigation water management. Irrig. Sci. 23, 61–75. https://doi.org/10.1007/s00271-004-0094-x.
Pannell, D.J., 2006. Flat earth economics: the far-reaching consequences of flat payoff functions in economic decision making. Appl. Econ. Perspect. Policy 28, 553–566. https://doi.org/10.1111/j.1467-9353.2006.00322.x.
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.
Peña-Haro, S., García-Prats, A., Pulido-Velazquez, M., 2014. Influence of soil and climate heterogeneity on the performance of economic instruments for reducing nitrate leaching from agriculture. Sci. Total Environ. 499, 510–519. https://doi.org/10.1016/j.scitotenv.2014.07.029.
Peña-Haro, S., Llopis-Albert, C., Pulido-Velazquez, M., Pulido-Velazquez, D., 2010. Fertilizer standards for controlling groundwater nitrate pollution from agriculture: El Salobral-Los Llanos case study, Spain. J. Hydrol. 392, 174–187. https://doi.org/10.1016/j.jhydrol.2010.08.006.
Pérez-Blanco, C.D., Delacámara, G., Gómez, C.M., 2015. Water charging and water saving in agriculture. Insights from a revealed preference model in a mediterranean basin. Environ. Model. Softw. 69, 90–100. https://doi.org/10.1016/j.envsoft.2015.03.006.
Pérez-Blanco, C.D., Gutiérrez-Martín, C., 2017. Buy me a river: use of multi-attribute non-linear utility functions to address overcompensation in agricultural water buyback. Agric. Water Manag. 190, 6–20. https://doi.org/10.1016/j.agwat.2017.05.006.
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.
Pulido-Velazquez, M., Alvarez-Mendiola, E., Andreu, J., 2013. Design of efficient water pricing policies integrating basinwide resource opportunity costs. J. Water Resour. Plan. Manag. 139, 583–592.
Pulido-Velazquez, M., Andreu, J., Sahuquillo, A., Pulido-Velazquez, D., 2008. Hydro-economic river basin modelling: The application of a holistic surface–groundwater model to assess opportunity costs of water use in Spain. Ecol. Econ., Spec. Sect.: Integr. Hydro-Econ. Model. Eff. Sustain. Water Manag. 66, 51–65. https://doi.org/10.1016/j.ecolecon.2007.12.016.
Pulido-Velazquez, M., Peña-Haro, S., García-Prats, A., Mocholi-Almudever, A.F., Henriquez-Dole, L., Macian-Sorribes, H., Lopez-Nicolas, A., 2015. Integrated assessment of the impact of climate and land use changes on groundwater quantity and quality in the Mancha Oriental system (Spain). Hydrol. Earth Syst. Sci. 19, 1677–1693. https://doi.org/10.5194/hess-19-1677-2015.
Rausser, G.C., Yassour, J., 1981. Multiattribute utility analysis: the case of filipino rice policy. Am. J. Agric. Econ. 63, 484–494. https://doi.org/10.2307/1240539.
Reca, J., Roldán, J., Alcaide, M., López, R., Camacho, E., 2001. Optimisation model for water allocation in deficit irrigation systems: I. Description of the model. Agric. Water Manag. 48, 103–116. https://doi.org/10.1016/S0378-3774(00)00126-8.
Rey, D., Pérez-Blanco, C.D., Escriva-Bou, A., Girard, C., Veldkamp, T.I.E., 2018. Role of economic instruments in water allocation reform: lessons from Europe. Int. J. Water Resour. Dev. 0, 1–34. https://doi.org/10.1080/07900627.2017.1422702.
Röhm, O., Dabbert, S., 2003. Integrating agri-environmental programs into regional production models: an extension of positive mathematical programming. Am. J. Agric. Econ. 85, 254–265. https://doi.org/10.1111/1467-8276.00117.
Saltelli, A., Funtowicz, S., 2014. When All Models Are Wrong. Issues in Science and Technology XXX.
Sampson, S.E., 1999. Axiomatic justification for a geometric quality aggregation function. Decis. Sci. 30, 415–440. https://doi.org/10.1111/j.1540-5915.1999.tb01616.x.
Sanz, D., Castaño, S., Cassiraga, E., Sahuquillo, A., Gómez-Alday, J.J., Peña, S., Calera, A., 2011. Modeling aquifer–river interactions under the influence of groundwater abstraction in the mancha oriental system (SE Spain). Hydrogeol. J. 19, 475–487. https://doi.org/10.1007/s10040-010-0694-x.
Sapino, F., Pérez-Blanco, C.D., Gutiérrez-Martín, C., Frontuto, V., 2020. An ensemble experiment of mathematical programming models to assess socio-economic effects of agricultural water pricing reform in the Piedmont Region, Italy. J. Environ. Manag. 267, 110645 https://doi.org/10.1016/j.jenvman.2020.110645.

Steduto, P., Hsiao, T.C., Fereres, E., 2007. On the conservative behavior of biomass water productivity. Irrig. Sci. 25, 189–207. https://doi.org/10.1007/s00271-007-0064-1.
Steenbergen, F., van, Cornish, G., Perry, C.J., 2007. Charging for Irrigation Services: Guidelines for Practitioners. IWA Publishing, London (UK).
Trout, T.J., DeJonge, K.C., 2017. Water productivity of maize in the US high plains. Irrig. Sci. 35, 251–266. https://doi.org/10.1007/s00271-017-0540-1.
Tsur, Y., Dinar, A., 1997. The relative efficiency and implementation costs of alternative methods for pricing irrigation water. World Bank Econ. Rev. 11, 243–262.
UN, 2021b. The United Nations World Water Development Report 2021 (Report). United Nations.
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).
WEF, 2020c. Global Risks 2020 (Report No. 15th Edition), Global Risks. World Economic Forum.
Williams, J.R., Renard, K.G., Dyke, P.T., 1983. EPIC: a new method for assessing erosion's effect on soil productivity. J. Soil Water Conserv. 38, 381–383.
World Bank, 2020d. World Bank DataBank [WWW Document]. DataBank. URL ⟨https://databank.worldbank.org/databases/exchange-rates⟩.

Chapter 3. Multi-model ensembles

"In a complex world, facing increasing environmental stresses and socioeconomic challenges, policy-makers have a tremendous need for reliable and robust decision-support tools to evaluate options and their inherent trade-offs."
Siwa Msangi, Applied Methods for Agriculture and Natural Resource Management, 2019

3.1. Introduction

In the previous chapter, we exposed that MPMs are the most suitable and used models to reproduce farmer choices and classified the principal families of these models in accordance with their objective function. However, conventional MPMs usually do not consider modeling and systemic uncertainty, and therefore, they can give misleading insights to policymakers. In this chapter, we argue the importance of considering modeling uncertainty when we use MPMs to advise decision-makers. To this end, we developed an innovative multi-model ensemble of MPMs capable of addressing robust decisions and we tested it in two study cases in Italy. The first publication (section 3.2) uses an ensemble of 5 MPMs to advise the water pricing reform designed by the regional authority of Piedmont, Italy. The second application (section 3.3) uses the same ensemble but in a slightly different policy approach: in this case, we assess the implementation of a PWS scheme to sustain irrigation-dependent ecosystem services. This second case was applied in the Reno River basin in the Emilia-Romagna Region, Italy. In this area, the irrigation is almost completely supplied by the Po River, which has shown an alarming increase in drought events in recent years, with 2022 as an emblematic example of the vulnerability of this supply (AdBPo, 2022). These applications are a valuable test of our ensemble approach to sampling modeling uncertainty through model spread, i.e., using the unweighted average of the results of each model. The unweighted average is considered the best estimation of an unknown variable if there is no evidence that one of the different models predicts better than the others. This approach allows us to advise robust policy through the application of the minimum regret principle and improves the robustness of the assessment compared to the classic approach of single-model point prediction.

3.2. An ensemble experiment of mathematical programming models to assess socio-economic effects of agricultural water pricing reform in the Piedmont Region, Italy.

3.2.1. Resumen

La Región del Piamonte, en el noroeste de Italia, ha puesto recientemente en marcha una ambiciosa y pionera reforma de los precios del agua destinada a integrar y aplicar eficazmente los principios de recuperación de costes, quien contamina paga y asequibilidad definidos por la Directiva Marco del Agua. Este artículo desarrolla un conjunto multimodelo que abarca 5 modelos de programación matemática (2 modelos de programación matemática positiva (PMP) , 2 modelos de programación de utilidad multiatributo positiva (PMAUP) y 1 modelo de programación lineal) que representan el comportamiento observado de los agentes socioeconómicos para: 1) simular los impactos de la reforma piamontesa de los precios del agua sobre el cambio del uso de la tierra, la reducción del consumo del agua y del beneficio de los agricultores, y la reducción de los ingresos de las tarifas del agua; 2) modelar la incertidumbre a través del conjunto; y 3) explorar posibles puntos críticos mediante el uso de técnicas de *scenario discovery*. Nuestra investigación sugiere que el reto clave para el complimiento de los objetivos reforma radica en la gestión de los cultivos de arroz, un cultivo extenso (17% de la superficie agrícola), exigente de agua y de relativamente bajo valor añadido que, sin embargo, ofrece servicios ecosistémicos significativos (p. ej., retención de agua) y de relevancia histórica y cultural para la región. El conjunto de modelos matemáticos sugiere que la agricultura arrocera disminuye rápidamente con un alza en el precio del agua entre 0.012 - 0.074 EUR/m^3 dependiendo del modelo. Antes de alcanzar este punto de inflexión, el precio del agua agrícola puede reducir las extracciones hasta un 1,7%-9,5%, mientras que reduce el beneficio entre 4,9% y 5,6% y logra un aumento de 57 a 65 veces en los ingresos por tarifas de agua.

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Research article

An ensemble experiment of mathematical programming models to assess socio-economic effects of agricultural water pricing reform in the Piedmont Region, Italy

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ABSTRACT

The Piedmont Region in NW Italy has recently deployed an ambitious and pioneering agricultural water pricing reform aimed at integrating and effectively enforcing EU's Water Framework Directive principles of cost recovery, polluter-pays and affordability. This paper develops a multi-model ensemble framework encompassing 5 mathematical programming models (2 Positive Mathematical Programming models, 2 Positive Multi-Attribute Utility Programming models and 1 Weighted Goal Programming model) that represent the observed behavior of socioeconomic agents to: 1) simulate the impacts of the Piedmontese water pricing reform on land use allocation and management, water conservation, profit and water tariff revenue; 2) sample modeling uncertainty through the ensemble spread; and 3) explore potential tipping points through use of scenario-discovery techniques. Our research suggests that the key challenge to the reform lies in the management of rice fields, an extensive (17% of the agricultural area), water-demanding and relatively low-added-value crop that nonetheless delivers significant ecosystem services (e.g. water retention) of historical and cultural relevance to the region. The ensemble experiment suggests that rice agriculture rapidly dwindles in the price range 0.012–0.074 EUR/m^3 depending on the model. Before reaching this tipping point, agricultural water pricing can reduce withdrawals up to 1.7%–9.5%, while reducing profit between 4.9% and 5.6% and achieving a 57- to 65-fold increase in water tariff revenue.

1. Introduction

Water scarcity and related crises are among the greatest global societal threats (WEF, 2019). In Europe, water scarcity is particularly felt in the closed or closing basins along the Mediterranean Basin, where inelastic water supply increasingly often falls short of commitments to fulfill growing demand. Restoring the balance in overallocated Mediterranean basins will necessitate demand-side policies that reallocate available resources from commercial uses to the environment while enabling economic growth and increasing social welfare. One such policy is pricing, the only demand-side instrument explicitly mentioned in the EU legal *acquis*. In its Article 9, the EU Water Framework Directive (WFD) states: "[…] water pricing policies provide adequate incentives for users to use water resources efficiently, and thereby contribute to the environmental objectives of this directive" (OJ, 2000). Despite this solid legislative basis, the implementation of pricing policies in the EU has been sluggish, also in the agricultural sector, the largest human water use (EEA, 2013). Agricultural water prices are often set independently of the volume used (e.g. on a per area basis) and present low cost-recovery ratios, which prevents incentive-pricing water conservation and reallocation to higher value uses. Although EU bodies have reacted to member states institutional paralysis with lawsuits, ruling from EU judiciary has been dichotomic (Jääskinen, 2014). As a result, 20 years after the adoption of the WFD, no member state in Southern Europe has *implemented* an agricultural water pricing reform that integrates the principles of cost recovery, polluter-pays and affordability as stated in the WFD (Rey et al., 2018).

The Piedmont Region in Northern Italy is set to change this trajectory. On July 24th, 2017, the Piedmont Region introduced two additional *ex-ante conditionalities* to access critical EU's Common

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Agricultural Policy (CAP) funding, namely, i) "harmonization of the methods for quantifying irrigation water withdrawals and effective collection, communication and management of this data", including the compulsory adoption of metering devices; and ii) "introduction of environmental and resource costs in the calculation of water prices", while "observing affordability principle" (Regione Piemonte, 2017). Users not observing the additional *ex-ante conditionalities* will not have access to critical CAP funding. It should be noted that in Italy, regional authorities issue, monitor and enforce water abstraction rights and set the corresponding prices; with national institutions and the relevant river basin authority (the Po River Basin Authority in the case of Piedmont) playing a secondary, advisory role (for a detailed description of the water allocation system in Italy and Piedmont the reader may refer to Santato et al., 2016). Under the current water abstraction regime, agricultural licenses are issued by the Piedmont Region for a maximum of 40 years, and prices are set on a per area basis (average charge: 1.22 EUR/ha) or based on the average flow rate capacity (0.56 EUR/l/s). Piedmont's agricultural water pricing reform is set to transition from the current pricing structure to a fully metered system (EUR/m^3). The new pricing structure, which is available in Regione Piemonte (2017), is based on a comprehensive methodology that first estimates the financial, environmental and resource costs of agricultural water use to then elicit the price increase that would enable a predefined cost recovery ratio; where the targeted cost recovery ratio is set based on a discretionary expert judgement that factors in affordability/disproportionate costs issues.

Recent institutional reports using this methodology foresee an average agricultural water price increase from a 0.00012 EUR/m^3 equivalent under the current pricing structure up to 0.013 EUR/m^3, which is expected to increase the contribution of agriculture to the region's water pricing revenues from less than 1% up to 32% (Frontuto et al., 2020). Noteworthy, the 0.013 EUR/m^3 price increase is set based on experts' opinion: beyond this point the impact of the pricing reform, albeit still moderate in terms of foregone income, is expected to have significant and potentially irreversible impacts on the structure of traditional irrigated agriculture (i.e. rice) and related (ecosystem) processes and services, which are regarded as *disproportionate costs* (Frontuto et al., 2020). Naturally this subjective pricing target needs to be further substantiated through a more profound assessment of irrigators' responses and their impact on economic and environmental (i.e. water conservation) performance, the tradeoffs observed between these two variables, and an analysis of disproportionate costs. To this end, Regione Piemonte and three academic institutions (Università di Torino in Italy, and Universidad de Salamanca and Universidad de Córdoba in Spain) have partnered to develop a comprehensive database and calibrate 5 mathematical programming models that represent the observed behavior of socioeconomic agents in an innovative multi-model ensemble experiment, in order to: 1) simulate the impacts of water pricing reform on land use allocation and management, water conservation, employment, profit and water tariff revenue; 2) sample uncertainty through the model spread (Cloke et al., 2013; IPCC, 2014); and 3) explore potential tipping points, with a focus on rice systems, through a scenario-discovery approach (Marchau et al., 2019). Unlike conventional consolidative modeling based on a single model and a complete probabilistic description of future scenarios, the ensemble experiment offers the advantage of providing policymakers with a more comprehensive overview of possible responses through stress test (alternative forcings/scenarios and models). Outputs from multi-model ensemble and scenario-discovery techniques can in turn be used to identify no-regret water pricing policies through robust decision making methods (Marchau et al., 2019).

The paper is structured as follows: Section 2 introduces the case study area, the Piedmont Region in the northwest of Italy; Section 3 presents the multi-model ensemble framework; Sections 4 and 5 present and discuss, respectively, the results achieved; and Section 6 concludes.

2. Case study area: The Piedmont Region in Italy

The Piedmont Region is located in the Northwest of Italy, has a population of 4,392,526 (2017) and spreads over 25,387 km^2 (Eurostat, 2017). The region is located within the Po River Basin District (PRBD), the largest (24% of Italian territory and 21% of its agricultural area) and most economically relevant Italian river basin (35% of Italian GDP and 30% of agricultural Gross Value Added (GVA)). The region comprises the upper stretches of the PRBD and 43% of its territory is classified as mountainous area (mostly the Alps), making the Piedmont Region a relatively water-abundant basin capable of supporting a water-intensive agriculture comprising 396,000 ha and largely based on annual crops such as rice, corn and cereal fodders (70% of Piedmont's agricultural area). Rice, which is supplied through the third largest artificial watercourse in Italy, the Canale Cavour, with a flow rate of 110 m^3/s and a length of 83 km, is the most iconic crop of the region. Piedmont's rice represents 52% of total production of Italy, which is in turn the largest rice producer in Europe (ISTAT, 2016), and is the largest water user in the region (nearly 31,500 m^3/ha on average) (Augusti et al., 2018), although its profit-to-water use ratio is relatively low compared to that of other crops in the region (1300 EUR/ha on average, as compared to e.g. 1200 EUR/ha profit and 3400 m^3/ha water use for corn) (INEA, 2018). Besides its market relevance, rice supplies relevant ecosystem services, most notably water retention services during the Po River's discharge peak in the spring, with subsequent water release throughout the summer season (about 150 m^3/s discharge in July), which is made available for other uses (Director of the Est Sesia Land Reclamation and Irrigation Board, 2019); but also historical and cultural services (rice production and the construction of related water draining and supply infrastructure in the Piedmont Region started in the mid-15th century) and aesthetic values, with rice fields defining the characteristic range of colors of the Piedmontese plains (blue in spring, green in summer, yellow in early autumn). Grassland is the most relevant crop in mountainous areas and represents 8% of Piedmont's agricultural area. Among permanent crops (12% of agricultural area) vineyard stands out, with the Piedmont Region representing 12% of Italian DOC[1] wine production (see Fig. 1).

Agriculture represents 3.5% of the Piedmontese GVA, above the national average of 2.8% (Banca D'Italia, 2018), and 75%+ of the regional water withdrawals, approximately 5000 million m^3/y (Regione Piemonte, 2018). Agricultural water use has significantly increased during the last 50 years due to wide-scale adoption of irrigated fodder and corn. This is coupled with a sustained reduction in average precipitation and in the number of rainfall days, and retreating Alps' glaciers (Regione Piemonte, 2018). As a result, the basin-wide water exploitation index (ratio of withdrawals to renewable resources) has increased from less than 20% (no water stress) to between 35% and 65% (severe water stress) over the last two decades (EEA, 2016). ARPA Piemonte (2018) estimates that half of Piedmont Region water bodies are already affected by water scarcity during the irrigation campaign and do not reach the good ecological status. Scarcity is amplified downstream in the PRBD, where more vulnerable regions dependent on the runoff generated within the Piedmont Region, such as the Emilia Romagna Region, are located.

3. Methods: the multi-model ensemble

Conventional consolidative modeling relies on a known set of possible states and related probabilities to identify a single optimal strategy through optimization in a single-model environment. Learning is acquired through observation, interpreting signals that constrain the set of possible states and updating probabilities according to Bayes' rule;

[1] *Denominazione di Origine Controllata* (controlled designation of origin): quality assurance for Italian wine.

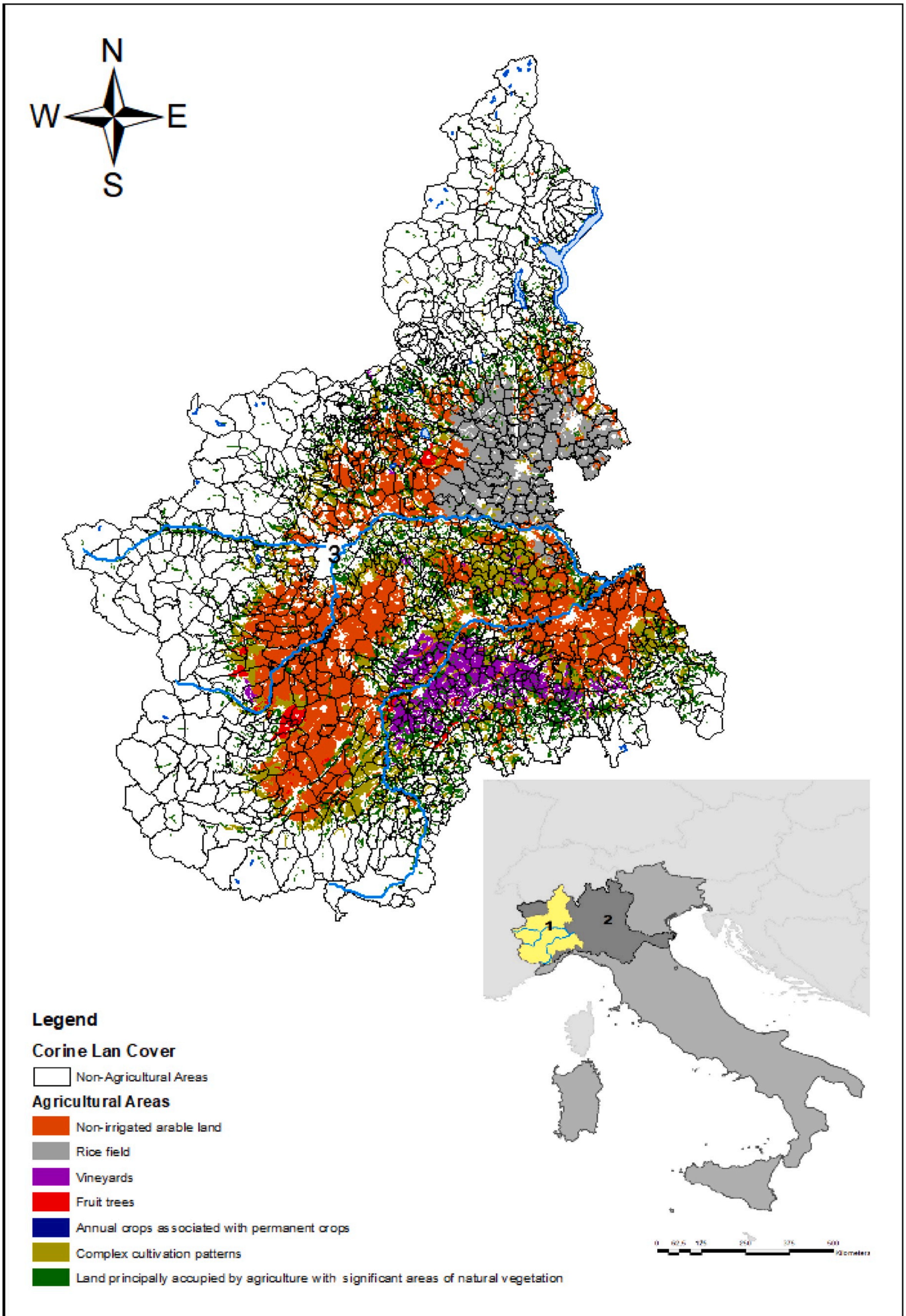

Fig. 1. Location of the Piedmont Region in Italy and detail of agricultural land use.
Legend: 1 Piedmont region, 2 Po River Basin District, 3 Po River.
Source: own elaboration from CORINE land cover (EEA, 2018).

and revising model design in order to better represent observed decisions. However, modeling errors arising from "parameter and structural uncertainties in the model design" (Tebaldi and Knutti, 2007) and the "impossible task" (Marchau et al., 2019) of accurately forecasting all possible future states in non-mechanistic complex socio-ecological systems imply that conventional consolidative modeling may not be capable of predicting contingencies arising from policy choices (Hino and Hall, 2017), including catastrophic and potentially irreversible outcomes (tipping points). In our research, modeling and scenario uncertainties are addressed through: i) a multi-model ensemble framework that samples modeling uncertainty through the model spread (this section); and ii) the use of scenario-discovery techniques to relate alternative simulation scenarios (water pricing scenarios in this case) to their implied consequences (see Section 4.1) (Marchau et al., 2019). The ensemble is populated with 5 positive economic calibrated models: 2 Positive Mathematical Programming (PMP) models, 2 Positive Multi-Attribute Utility Programming (PMAUP) models and 1 Weighted Goal Programming (WGP) model. The following sub-sections present the basics of the 3 modeling families and the 5 models considered. For a more detailed description of each ensemble component, the reader may refer to Howitt (1995) and Júdez et al. (2002) (PMP) (Gómez-Limón et al., 2016; Gutiérrez-Martín and Gómez, 2011); (PMAUP); and Sumpsi et al. (1997) (WGP).

3.1. Economic calibrated models: objective function and domain

Economic calibrated models for agricultural water management represent the pattern of yields, revenues and costs at different scales, from farm to agricultural district (Harou et al., 2009). In these models, agents (clusters in our application to the Piedmont Region, see Section 3.5) decide on crop mix and timing, investments and water application in an optimization framework that aims to maximize a single or multi-attribute objective function within a domain. This complex choice is usually "reduced to a decision on the crop portfolio", where each solution represents a "unique combination of crop, timing, investments and water application" (Pérez-Blanco et al., 2017). The general formulation of the utility maximization problem is as follows:

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

Subject to:

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

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

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

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

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

where $U(\boldsymbol{X})$ is the utility/objective function.

Agents in the model decide on the *crop portfolio* $\boldsymbol{X} \in \mathbb{R}^n$, a vector representing the fraction of land allotted to each one of the n individual crops available x_i $(i = 1, \ldots, n)$, so to maximize utility through the provision of utility-relevant attributes $z_1(\boldsymbol{X}), \ldots, z_m(\boldsymbol{X})$ (i.e. there are up to m relevant attributes), such as profit or avoided risk. Each attribute $z_1(\boldsymbol{X})$, …, $z_m(\boldsymbol{X}) \in \boldsymbol{Z}(\boldsymbol{X})$in the model is defined so that "more-is-better", i.e. increasing the provision of one attribute while keeping the provision of the remaining attributes constant increases utility. Accordingly, "less-is-better" attributes such as risk or management complexity are transformed into avoided risk/management complexity. Note that each crop portfolio $\boldsymbol{X}$ yields a *unique* provision of attributes $z_1(\boldsymbol{X})$, …, $z_m(\boldsymbol{X})$. Rational agents in the model will choose the crop portfolio that yields the provision of utility-relevant attributes that maximizes utility within the domain F.

The individual attributes that conform the attribute set $\boldsymbol{Z}(\boldsymbol{X})$ used in the calibration and simulation of the models are described in the following paragraphs. We explored the relevance of three attributes in the ensemble, namely: expected gross variable margin (z_1), risk avoidance (z_2) and total labor avoidance (z_3), a proxy for management complexity.

- Expected profit, measured as the expected gross variable margin (z_1). This is the only attribute considered in single-attribute mathematical programming models (PMP models in this ensemble). It is obtained as the summation of the expected per hectare gross margin of each crop π_i (obtained as price (in EUR/kg) times yield (in kg/ha) plus coupled subsidies minus the variable costs (in EUR/ha)) multiplied by that crop's land share (x_i):

$$z_1(\boldsymbol{X}) = \sum_i x_i \overline{\pi}_i \tag{7}$$

where $\overline{\pi}_i$ is the average gross margin for each crop i in the period 2008–2016, i.e. the summation of the observed gross margin of crop i for every year during the period 2008–2016, divided by the number of years with available data in the series. In the case of PMP models, an additional shadow cost is added to profit during calibration. Note that all variables used to calculate profit (prices, yield, subsidies, costs) are exogenous. In the case of prices, this implies that crops' demand is perfectly elastic. Such "small open economy assumption" (Schöb, 1998) is consistent with EU reports showing that "patterns of crop price variations are similar for all member states" (Kampas and Rozakis, 2017). Admittedly, regional differences in prices may arise, especially in face of asymmetric shocks such as the pricing policy discussed here. This could be modeled e.g. coupling the ensemble framework presented in this paper with a general equilibrium macroeconomic model (Parrado et al., 2019). The development of a multi-system ensemble goes beyond the scope of the present research; we nonetheless reflect on this in the conclusions, where we propose a multi-model and multi-system ensemble as a means to explicitly model crops' demand and prices endogenously, while accounting for modeling and scenario uncertainty.

- Risk avoidance (z_2), measured as the difference between the profit variability of the profit maximizing crop portfolio $\widehat{\boldsymbol{X}}$ and that of an alternative crop portfolio $\boldsymbol{X}$ (Bartolini et al., 2007):

$$z_2\ (\boldsymbol{X}) = \widehat{\boldsymbol{X}}^t VCV(\pi)\widehat{\boldsymbol{X}} - \boldsymbol{X}^t VCV(\pi)\boldsymbol{X} \tag{8}$$

where $VCV(\pi)$ is the variance and covariance matrix of profit in the time period for which data is available (2008–2016). The first term in the right-hand side of the equation, $\widehat{\boldsymbol{X}}^t VCV(\pi)\widehat{\boldsymbol{X}}$, yields the risk of the profit maximizing crop portfolio, while the second term, $\boldsymbol{X}^t VCV(\pi)\boldsymbol{X}$, yields the risk of the observed crop portfolio. Provided there is a tradeoff between risk and profit (the higher the profit, the higher the risk) (Gutiérrez-Martín and Gómez, 2011), risk avoidance ($z_2\ (\boldsymbol{X})$) will be positive.

- Total labor avoidance (z_3), a *proxy* for management complexity avoidance (Bartolini et al., 2007; Sumpsi et al., 1997) measured here as the difference between the total (family plus hired labor) expected (i.e. multi-annual average) labor requirements of the crop portfolio with the highest possible labor requirements within the domain, $\overline{\boldsymbol{X}}$, and those of an alternative crop portfolio $\boldsymbol{X}$.

$$z_3\ (\boldsymbol{X}) = \sum_i \overline{x_i} N_i - \sum_i x_i N_i \tag{9}$$

where N_i is the expected total labor requirements per hectare of crop i.

Note that in PMP models profit is the only utility-relevant attribute explored in the objective function, while the WGP and the two PMAUP models also explore the relevance of risk and management complexity aversion (multi-attribute). Accordingly, some of the constraints that conform the domain are not applicable/binding to all models, i.e. those referring to risk aversion and management complexity attributes do not apply in single-attribute PMP models.

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

- *Land availability.* Available agricultural land is assumed constant and equals the summation of observed agricultural land uses (see equations (2) and (3)).
- *Water availability.* It is assumed that water abstraction licenses remain constant before and after every simulation run, i.e.:

$$\sum_{i=1}^{n} w_i x_i \le w \tag{10}$$

where w_i is crop i's specific water requirements and w is the total water allotment in the Piedmont Region.

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

$$\sum_{i=1}^{n} y_i x_i = 0 \;\Big|\; y_i \in \{0, 1\} \tag{11}$$

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

- *Crop-specific constraints.* Some crops in the portfolio have an upper and/or lower area bound because of specific policy restrictions. In our application to the Piedmont Region, this restriction is used to set a minimum/maximum threshold for ligneous trees of ±5%. Admittedly, since the pricing policy instrument is designed to work in the long run, it could result in major crop portfolio changes involving permanent crops, which could eventually go beyond the 5% threshold. On the other hand, the reduction or expansion in the acreage of permanent crops beyond the 5% threshold would result in significant (dis)investments with potentially large impacts on e.g. carbon sequestration, whose economic value is not accounted for in the models, which focus on yearly market variables (notably profit) (Essenfelder et al., 2018). Accurately modeling agent's responses in terms of permanent crops necessitates the inclusion of other relevant variables, notably carbon prices and/or Payments for Ecosystem Services, which are at present being tested in the European context; yet, this is beyond the scope of this paper. Against this backdrop, setting a minimum/maximum threshold for ligneous trees is common practice in the literature (Gutiérrez-Martín and Gómez, 2011; Parrado et al., 2019).
- *Crop rotation.* In some cases, it is possible to observe that two or more crops rotate with each other. For example, if farmers in an area yearly rotate wheat with sunflower, aggregation over a sufficient number of farms (e.g. at a municipality level) typically results in a similar surface of wheat and sunflower (Gómez-Limón et al., 2016). Accordingly, the surface of wheat in the simulations cannot exceed the surface of sunflower, and vice versa. If the surface of sunflower (wheat) becomes binding and decreases below the surface of wheat (sunflower) (e.g. due to higher water prices), the surface of wheat (sunflower) must decrease to match that of sunflower.

3.2. Calibration

Economic calibrated models follow an inductive approach that aims to eliciting the parameters of an objective/utility function capable of reproducing observed agents' choices within a domain/set of constraints, in order to accurately predict future responses to policy shocks through simulation. Noteworthy, each modeling family considered explores one specific functional form for the objective function: additive (WGP), Cobb-Douglas (PMAUP) and quadratic (PMP).

The **WGP** approach used in our ensemble framework relies on the calibration method developed by Sumpsi et al. (1997) to elicit the parameters of a multi-attribute, additive objective function. Note that due to the definition of the attributes above, our application includes a non-linear component in the additive objective function through the risk attribute. WGP allows for both single- and multi-attribute specifications, which makes the approach consistent with the Theory of Planned Behavior (TPB) (Ajzen, 1991). The TPB argues that decision-making is driven by "the multiple attributes of objects (including but not limited to profit) and farmers' beliefs regarding these attributes" (Pérez-Blanco et al., 2017). TPB's theoretical construct is substantiated by a large body of empirical research on the relevance that attributes other than profit, such as risk aversion or management complexity aversion, have in explaining agent's behavior and choices (see e.g. Gómez-Limón et al. (2016)). On the other hand, use of an additive function may lead to over-specialized responses and even corner solutions: the agent sets the crop that delivers highest utility at the maximum level until a binding constraint prevents further specialization, which often results in a characteristic "jumpy behavior" (Graveline, 2016).

PMP is possibly the most popular economic calibrated model to assess the behavior of agricultural agents, and irrigators in particular (Graveline, 2016). PMP relies on non-linear objective functions to calibrate and accurately reproduce observed agent behavior. Through the use of non-linear functions, PMP avoids unrealistic outcomes such as corner solutions or abrupt discontinuities in agent's responses, yielding instead smooth calibration results (Howitt, 1995). Due to these obvious advantages, PMP has been consistently used to assess agricultural and water policies, including water pricing, in several regions worldwide (Graveline, 2016). PMP calibration uses "information contained in dual variables of calibration constraints, which bound the solution of the original linear programming problem to observed activity levels" to "specify a non-linear objective function such that observed activity levels are reproduced by the optimal solution of the new programming problem without bounds" (Heckelei and Britz, 2005). This is done in three steps: (i) an additional area constraint that bounds the model calibration results to observed choices is introduced in the domain and the dual values associated to the constraint for each crop obtained; (ii) these dual values are used to add a non-linear component to the utility function (typically a quadratic cost function, or shadow cost); and (iii) the utility non-linear function obtained in (ii) is maximized subject to a similar set of constraints to those considered in the original problem, which perfectly reproduces the observed agent's behavior (Henry de Frahan et al., 2007). The main critique to PMP modeling regards the challenge of providing an "economic or technological rationale for the non-linear terms in the objective function" (Heckelei et al., 2012). As a result, a modeler needs to resort to *ad-hoc* arguments to elucidate the outcomes of PMP models following a policy shock (Graveline, 2016). Moreover, while PMP has modeled risk aversion in a single-attribute environment through the use of mean-variance approach, its single-attribute approach struggles to explicitly measure and account for the utility-relevance of alternative attributes such as management complexity aversion. The ensemble framework in this paper relies on the classic calibration method (PMP_1) (Howitt, 1995) and a variation proposed by Júdez et al. (2002), that skips the first step using the average rent of land as dual value (PMP_2).

PMAUP models "build on the axioms of revealed preference to construct a multi-attribute objective function that is both consistent with an observed (and finite) set of choices and prices and suitable as a basis for empirical analysis" (Parrado et al., 2019). PMAUP replaces the dual variables that would traditionally be added to the objective function to make calibration possible in PMP with agent's preference parameters represented as shares of a non-linear (typically Cobb-Douglas) utility function, the arguments of which are competing attributes (e.g. profits v. avoided management complexity). PMAUP is a data and computationally intensive approach consistent with the TPB that has been used to empirically explore the relevance of attributes other than profit (Gómez-Limón et al., 2016; Gutiérrez-Martín and Gómez, 2011), particularly during the last decade, propelled by expanding frontiers in computational power and micro-data. Yet, since only observed behavior is used as an input and assumptions are limited (no engineering-based yield functions, no assumptions of fixed proportions, no limitation to profits as the sole relevant attribute of farmers), the calibration of PMAUP models is challenging where there is a large number of choice variables (several alternatives in the crop portfolio) and cross-sectional variation is low (time-series variation might be confounded with other trends), which may lead to some instability in the model calibration that is difficult to rationalize (e.g. abrupt changes in parameter values following the introduction of an additional attribute). The ensemble framework in this paper relies on two specific calibration methods: the projection method (Gutiérrez-Martín and Gómez, 2011) (PMAUP_1) and the iteration method (Gómez-Limón et al., 2016) (PMAUP_2).

3.3. Management of uncertainty and robust decision making

Apart from PMP, which is a special case where the estimated residual is adjusted to zero, all economic calibrated models considered in our ensemble yield calibration residuals, which can be used to assess the internal performance of each model (readers can refer to Annex I in the supplementary material for a complete description of the calibration residuals used in the ensemble models). This does not mean PMP models can perfectly forecast behavioral responses to policy shocks; there remain "significant" sources of uncertainty outside calibration residuals, including those models where residuals are adjusted to zero (Phillips et al., 2001). Note also that calibration residuals are not directly comparable between families of models, since modeling errors are independent (Cloke et al., 2013).

The difficulty in assigning a reasonable metric of uncertainty to each model is at the core of the use of multi-model ensemble frameworks. Admittedly, forecasts from different models do not have the same likelihood; however, since we do not know their probability and to the extent modeling errors are independent, we can explore uncertainty through the model spread (IPCC, 2014). It would be possible as well to apply Laplace "Principle of insufficient reason" to assume the ensemble behaves as a Bayesian System, and obtain a "best estimate" as the simple arithmetic mean of the forecasts from each model in every scenario considered. This approach may nonetheless assume more than is granted by available evidence (recall the likelihood of forecasts from different models is unknown) (Hino and Hall, 2017); to avoid maladaptation, this research adopts a robust decision making approach that minimizes regret.

Robust decision making is a method that uses results from several simulation runs (using alternative models and/or scenarios through scenario-discovery techniques) to connect policy makers with model(s) capable of exploring uncertainty, so to identify robust adaptation options as those that "perform well compared to alternatives" (Marchau et al., 2019) and "hedge against uncertainty" (Graveline, 2019). Robust decision making process typically follows an iterative process between researchers and stakeholders/policy makers in five steps (Marchau et al., 2019). *Step 1* involves the definition of the decision-making framework, which in our case involves the exploration of alternative agricultural water pricing strategies through simulation, leveraging on scenario-discovery and multi-model ensemble techniques. *Step 2* is the evaluation of the proposed pricing strategies (see Section 4.1). *Step 3* assesses vulnerabilities to pricing strategies, notably through the identification of potential tipping points (Section 4.2). *Step 4* assesses the tradeoffs between alternative strategies and involves the identification of robust policies that avoid tipping points and unfavorable surprises (decision-making process) (Section 4.2). The decision-making process can be implemented through the use of heuristics (expert judgement), through mechanistic constrained optimization algorithms, or through a combination of both (Marchau et al., 2019). If the 4 steps above do not yield a satisfactory outcome, an additional *Step 5* can be included to explore alternative adaptation strategies. This step was not necessary in our research, since the task commissioned from Regione Piemonte explicitly demanded an analysis of incremental volumetric water prices in the agricultural sector, and no alternative policy was considered.

3.4. Data

Data was collected for the period 2008–2016, with 2016 as the calibration year. The database includes 28 crops representing 95% of total irrigated surface in the region. All attributes are defined so that "more-is-better" and are quantities of dimension one (i.e. normalized) (Gómez-Limón et al., 2016). Table 1 summarizes data inputs and related data providers.

The only non-primary data source in the database are water withdrawals. The water abstraction license regime in Italy is "byzantine and substandard" (Santato et al., 2016): relevant data gaps on water withdrawals exist, and public statistics underestimate actual water use. According to Regione Piemonte (2018) estimates, total withdrawals from irrigation amount to 5000 million m^3/year; while ISTAT (2010) water use statistics (primary data source) set total withdrawals in the region at 0.7 billion m^3/year. To circumvent this data mismatch, we measured irrigation water withdrawals using the per crop irrigation water consumption estimates and irrigation efficiency data from Augusti et al. (2018), and obtained a figure of 4750 million m^3 annual withdrawals.

3.5. Economic agents and results aggregation

The decision variable (i.e. land use) is available at a municipality level. This means that we have 1204 potential economic agents for the models. During the robust decision-making process, policy makers and stakeholders argued in favor of aggregating these 1204 units into more tractable agents that could yield easy-to-understand results and better inform their decisions. In order to define tractable agents for the calibration and policy simulation, this paper follows the work by Gómez-Limón et al. (2012) and handles municipalities as local aggregation units that can be grouped into clusters. To this end, we first obtain

Table 1
Models' inputs and data providers.

ID	Data provider	Variable	Ref. year	Disaggregation
Agricultural land use	Sistemapiemonte (2018)	Crop portfolio	2016	Hectares per crop at a municipality level
Crop yields, prices and costs	INEA (2018)	Crop yields (kg/ha), prices (EUR/kg) and costs (EUR/kg)	2008–2016	Per crop and province (NUTS3)
Water withdrawals and consumption and irrigation technology	Adapted from Augusti et al. (2018)	Water withdrawals, water consumption (m3/ha) and irrigation technology (%)	2016	Per crop at regional level (water withdrawals and consumption); at regional level (irrigation efficiency)
Working days (labor)	INEA (2016)	Number of working days	2016	Per crop at a regional level (NUTS2)

Source: own elaboration.

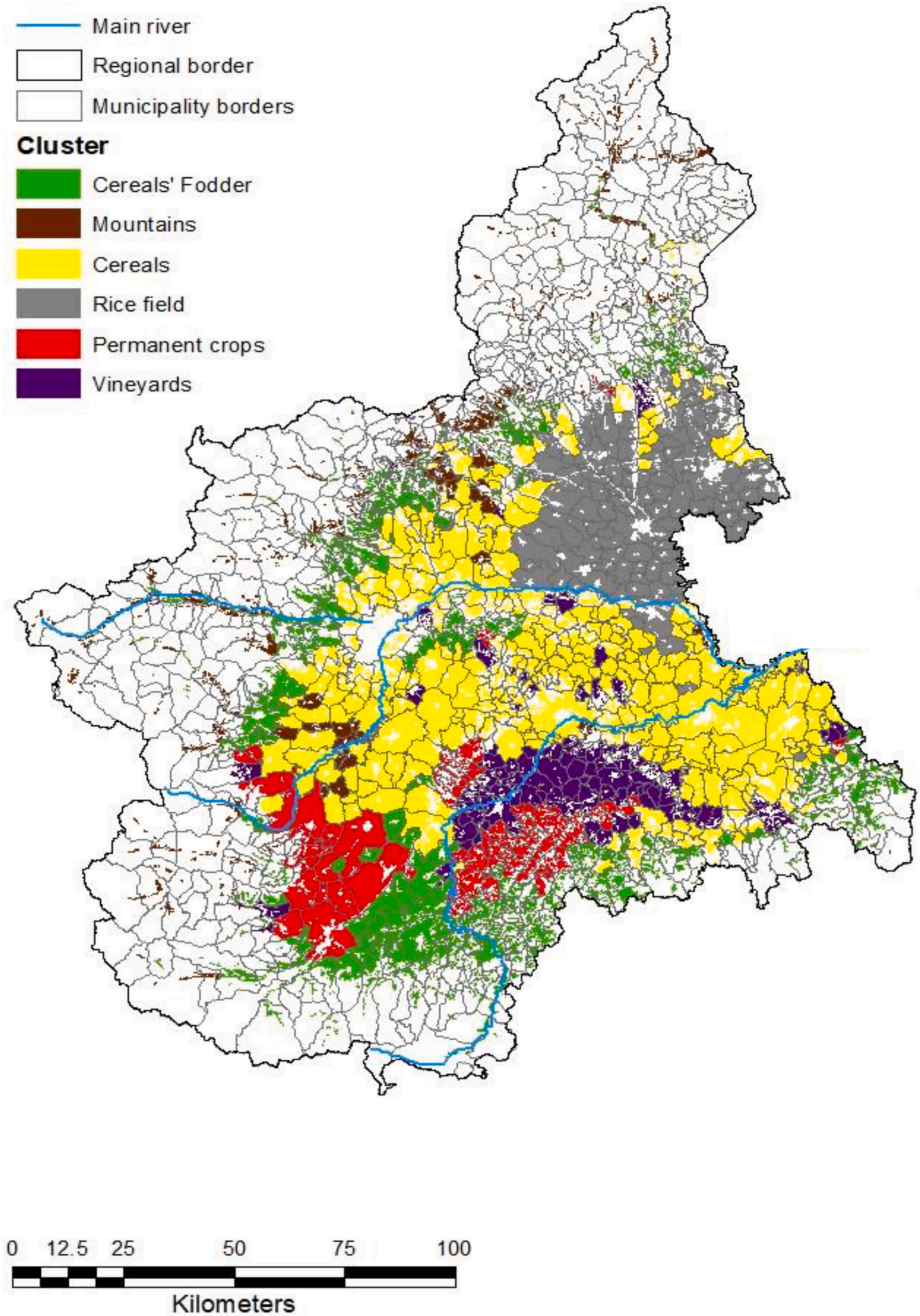

Fig. 2. Piedmont agricultural clusters.
Source: own elaboration.

information on the relevant data inputs described above for 1204 unique aggregation units/municipalities. We then employ a hierarchical aggregation procedure using the Euclidean distance measure and the Ward's agglomeration algorithm to maximize the internal homogeneity of clusters (Murtagh and Legendre, 2014). The results of the hierarchical clustering are usually presented in dendrograms from which, after visual inspection, the number of clusters is selected. The visual criterion can nonetheless be misleading and inefficient in identifying the optimal number of clusters, and this research adopts instead numerical criteria. Different indices have been proposed to find the optimal number of clusters. Following Charrad et al. (2014), we use a set of 30 indices instead of just one of them. The clustering procedure is performed using the NbClust package of the R software and is available in Annex II in the supplementary material, along with the list of the indices used. Note that the results of the different indices may not be univocal; when this happens, a simple majority rule is applied, which in our case led to 6 clusters as optimal grouping (Charrad et al., 2014) (see Fig. 2).

The resultant clusters are: C1 – Cereals and cereal fodders, which features cereals (29%), cereal fodders (60%) and permanent crops (6%); C2 – Mountains, largely devoted to the production of fodders (48%) and corn (42%); C3 – Cereals, including corn (29%), wheat (22%) and cereal fodders (28%); C4 – Rice (76% of land use in the cluster); C5 – Permanent crops, which encompasses vineyards (12%), other permanent crops (25%) and cereals (32%); C6 – Vineyards (48% of land use). These clusters are the agents of the mathematical programming models in the ensemble. Calibration results for these agents are presented in Annex III in the supplementary material.

Notably, simulation results using clusters as agents do not differ significantly from those obtained using individual municipalities as agents, which ensures consistence between the two aggregation levels while allowing for an easier-to-understand presentation of the results using a tractable number of agents (6 clusters v. 1204 municipalities).

Finally, results are aggregated at regional level as the weighted mean of the 6 clusters using clusters' land use shares as the weighting variable; i.e. attribute values for the Piedmont Region for every simulation run are obtained as the simulated attribute value for each cluster (per hectare), times the cluster's corresponding land use share. On the other hand, the crop portfolio at a regional level is obtained from the aggregation of the simulated crop portfolios for every cluster.

4. Results

4.1. Simulation

Once the five models are calibrated, they are used to run a number of simulations in which water prices are increased from 0 (baseline scenario) to 0.2 EUR/m^3 (i.e. 1666.7 times higher than the original price of 0.00012 EUR/m^3) at 0.002 EUR/m^3 intervals. Such pricing scenarios were co-developed with Regione Piemonte following a series of iterations (see Robust decision-making steps in Section 3.4). After every simulation run, agents in the model reassess their crop portfolio choices so to maximize their utility function within the domain. The result is a database representing the socio-economic effects of agricultural water pricing reform under multiple plausible futures, which is used to detect pricing policies that may potentially lead to contingencies/tipping points and underpin the implementation of a robust pricing policy.

Fig. 3 summarizes crop portfolio responses by ensemble component/model for relevant crops, namely fodders, corn, rice, wheat and grassland. The complete results including all crops are presented in Annex IV in the supplementary material. Fig. 3 also includes a "best estimate" obtained as the arithmetic mean of the forecasts from ensemble components in every scenario considered. It should be recalled that the "best estimate" is merely informative, since the likelihood of forecasts from different models is unknown, and the objective of this work is finding a robust pricing policy.

Overall, ensemble simulation results for the Piedmont Region show a

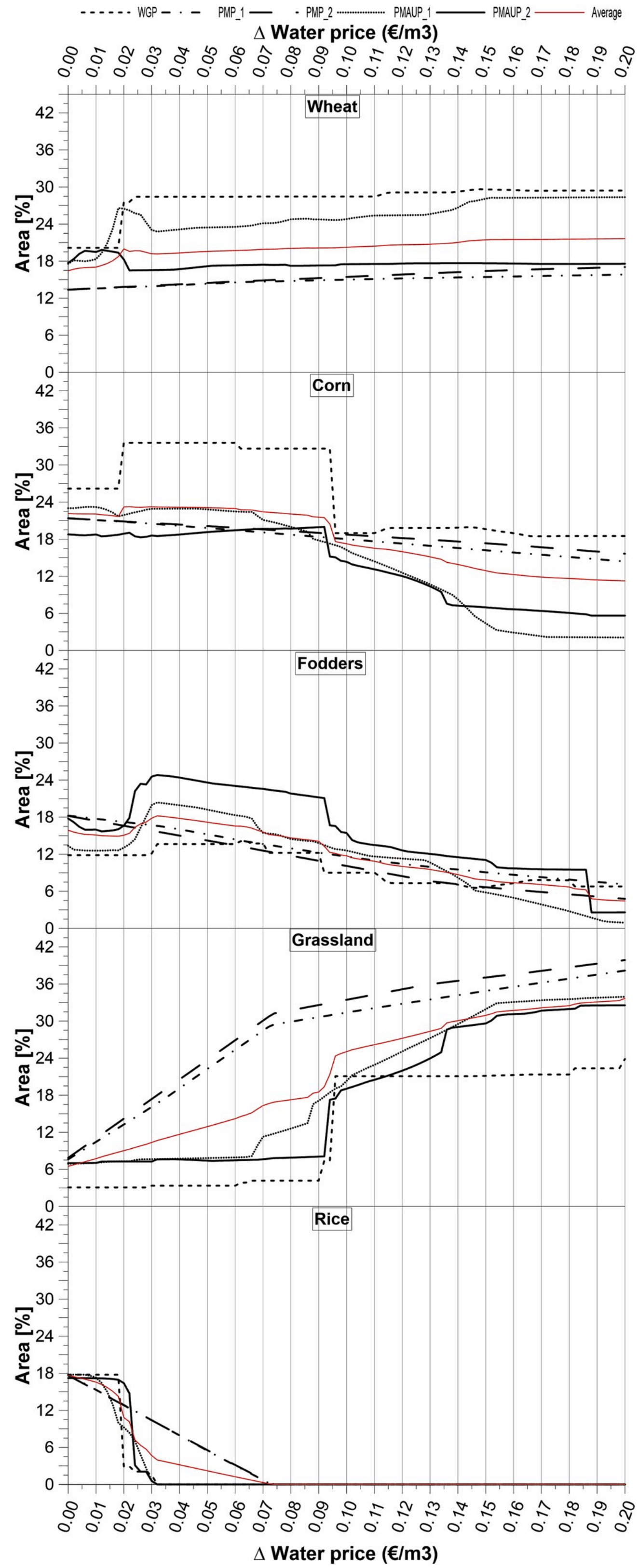

Fig. 3. Crop portfolio responses to incremental water prices for selected crops. Source: own elaboration.

trend towards the progressive substitution of water-intensive crops, such as corn and cereal fodders, by rainfed crops (wheat and grassland), although adaptation patterns to incremental prices may differ between models. For the case of cereal fodders, acreage reduction is constant and consistent across models. A similar trend is observed for corn, which nonetheless shows higher resilience to price increases and retains a relevant crop portfolio share throughout all simulations. Note that the land share of corn and cereal fodders (models WGP, PMAUP_1 and PMAUP_2) may experience some acreage expansion in the price range 0.012–0.032 EUR/m^3, as irrigators adapt to the new water price by substituting rice by less water intensive crops. Such land use changes are not observed in PMP models, where corn and cereal fodders experience a continued decrease and rice is substituted solely by grassland.

Rice responses to price shocks are heterogeneous and complex: while WGP and PMAUP models predict a significant reduction of rice beyond a price increase of 0.008 EUR/m^3 (PMAUP_1) and 0.018 EUR/m^3 (WGP and PMAUP_2), PMP models show a smooth reduction in acreage along price increases. In a series of interviews with representatives from the regional authority, river basin authority and Piedmontese land reclamation and irrigation boards, all stakeholders showed concern regarding this potential outcome, which had already been identified in a previous report as a critical barrier to the pricing reform (Frontuto et al., 2020) due to its potentially irreversible impact on the structure of traditional irrigated agriculture (i.e. rice) and related (ecosystem) processes and services of historical and cultural relevance to the region. Finally, like corn, cereal fodders and rice subside, grassland and wheat expand their acreage along with price increases.

Most water conservation is achieved at the 0.008–0.032 EUR/m^3 interval (0–0.074 EUR/m^3 for PMP models), when rice is replaced by less water-intensive corn/cereal fodders and rainfed crops in all the models, and water withdrawals fall from an average of 7000 m^3/ha to 1800 m^3/ha (see Fig. 4). Price increases below 0.012 EUR/m^3 slightly affect cereal fodders and yield modest water conservation figures. Price increases in the range of 0.034–0.19 EUR/m^3 for PMAUP and WGP models and 0.07–0.19 EUR/m^3 for PMP models lead to the gradual substitution of fodder and corn by rainfed crops, reducing water withdrawals from 1800 to 800 m^3/ha. Further price increases >0.19 EUR/m^3 meet an inelastic demand curve and are ineffective towards water conservation in the price range considered. This is explained because i) water withdrawals have already been removed from marginal lands and are now concentrated in highly productive areas capable of absorbing the price shock; and ii) agronomic restrictions, including crop rotations and planting constraints. Notably, although irrigators can reduce or expand permanent crops such as vineyard, we set a lower and upper bound of ±5% deviations from the original crop area. This is done to prevent significant capital (dis)investments, including the disruption in the provision of carbon sequestration services, which may conflict with other policies such as the Common Agricultural Policy (Essenfelder et al., 2018). Note that this constraint does not become binding until price increases beyond 0.19 EUR/m^3 due to the profitability of vineyards in the Piedmont Region.

Profit (see Fig. 5) falls consistently along price increases in both single-attribute PMP and multi-attribute PMAUP models, although the impact on PMP models is initially higher due to the presence of the quadratic cost function, which penalizes the shift towards less water intensive and/or rainfed crops that occupy a marginal area in the observed crop portfolio. WGP features a characteristic "jumpy" behavior where profit typically decreases but can also increase despite growing water prices.

The impacts of agricultural water pricing on employment (hired and family labor) is reported by the three multi-attribute ensemble components (PMAUP_1, PMAUP_2 and WGP). Initially employment increases along with prices, as rice is substituted by more labor-intensive corn. After a price increase of 0.096 EUR/m^3, when corn starts to decline consistently in all multi-attribute models, labor decreases as well (see Fig. 6). Note that information on employment (hired labor) is valuable to obtain information on GVA beyond profit/gross margin (the other component of GVA being labor income). For consistency among single- and multi-attribute models, this study reports information on profit and employment separately.

Water tariff revenue refers to the public revenue obtained directly from water pricing. Tariff revenue does not include other impacts on public revenue e.g. through a reduction in the income tax due to declining farmers' profits. Simulation results show that tariff revenue typically increases along with higher prices, although there are some significant exceptions where price increases trigger the substitution of water-intensive crops by less water intensive and rainfed crops (see Fig. 7). This is particularly visible for rice in the price range 0.012–0.074 EUR/m^3 for all models in the ensemble, and for corn in the price range 0.084–0.094 EUR/m^3 for multi-attribute PMAUP and WGP models. In

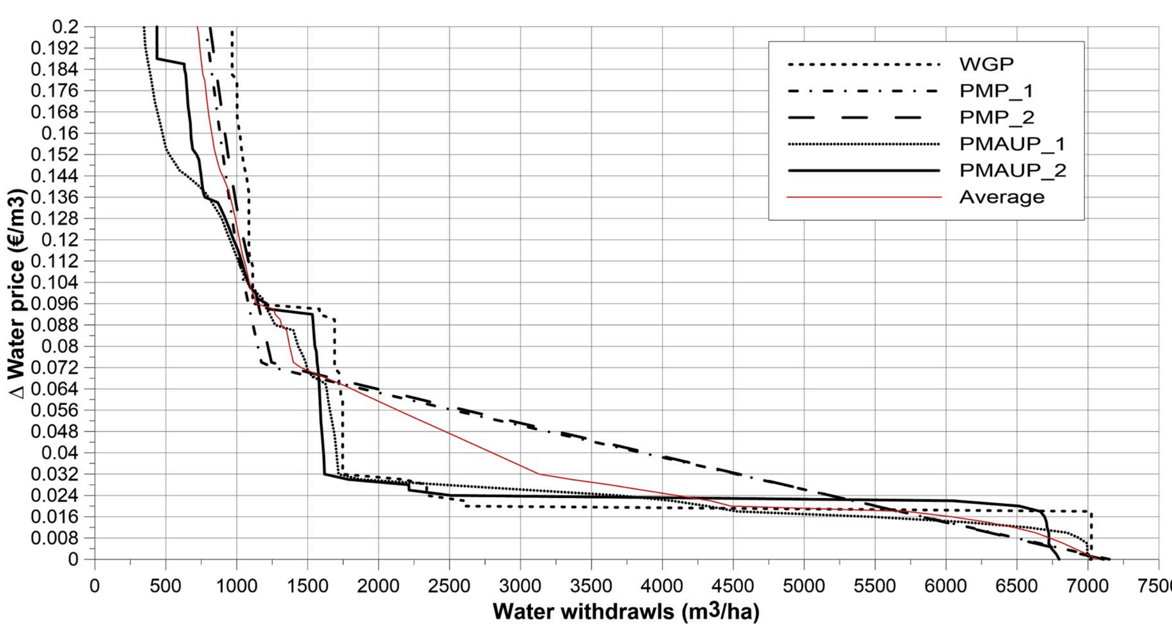

Fig. 4. Agricultural water demand curve.
Source: own elaboration.

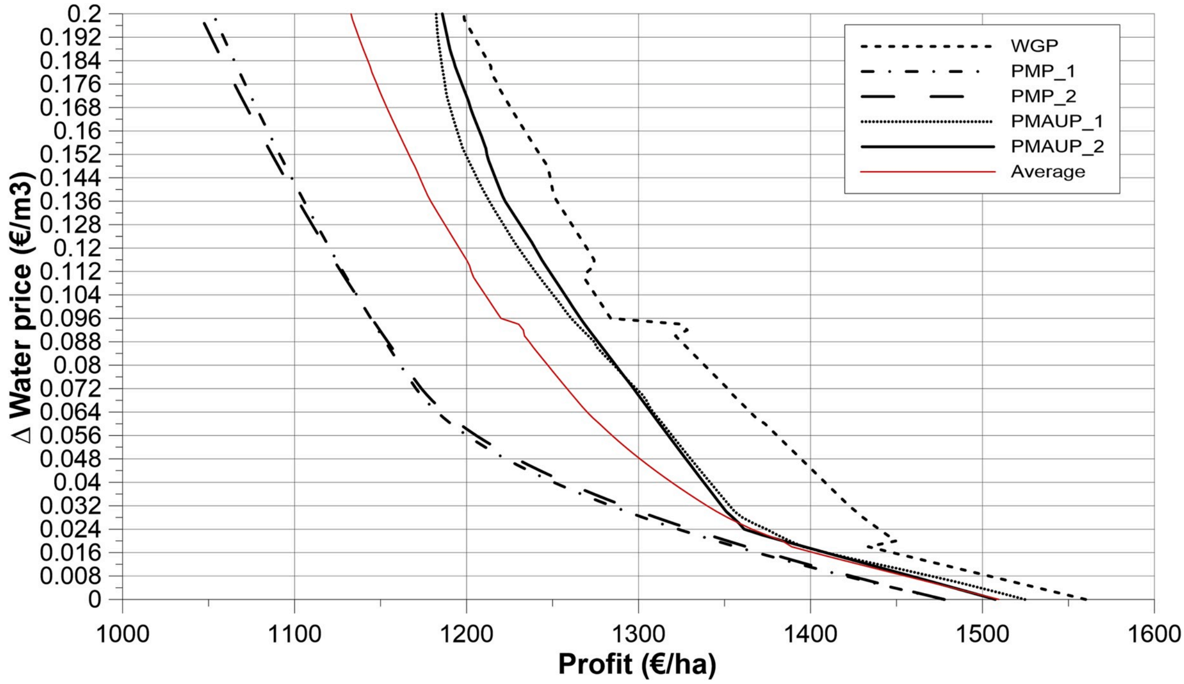

Fig. 5. Profit.
Source: own elaboration.

these instances, the water conservation effect overcomes the price increase effect and tariff revenue falls.

4.2. Robust decision making

Robust decision making was implemented following the steps described in Section 3.3. In *Step 1*, the two leading authors, policy makers (Regione Piemonte) and stakeholders (representatives from Land Reclamation and Irrigation Boards) gathered in a *kick-off* meeting to discuss and agree on the methodological approach to the research, and the initial set of pricing scenarios. In *Step 2,* preliminary results obtained using the approach agreed in Step 1 were presented and

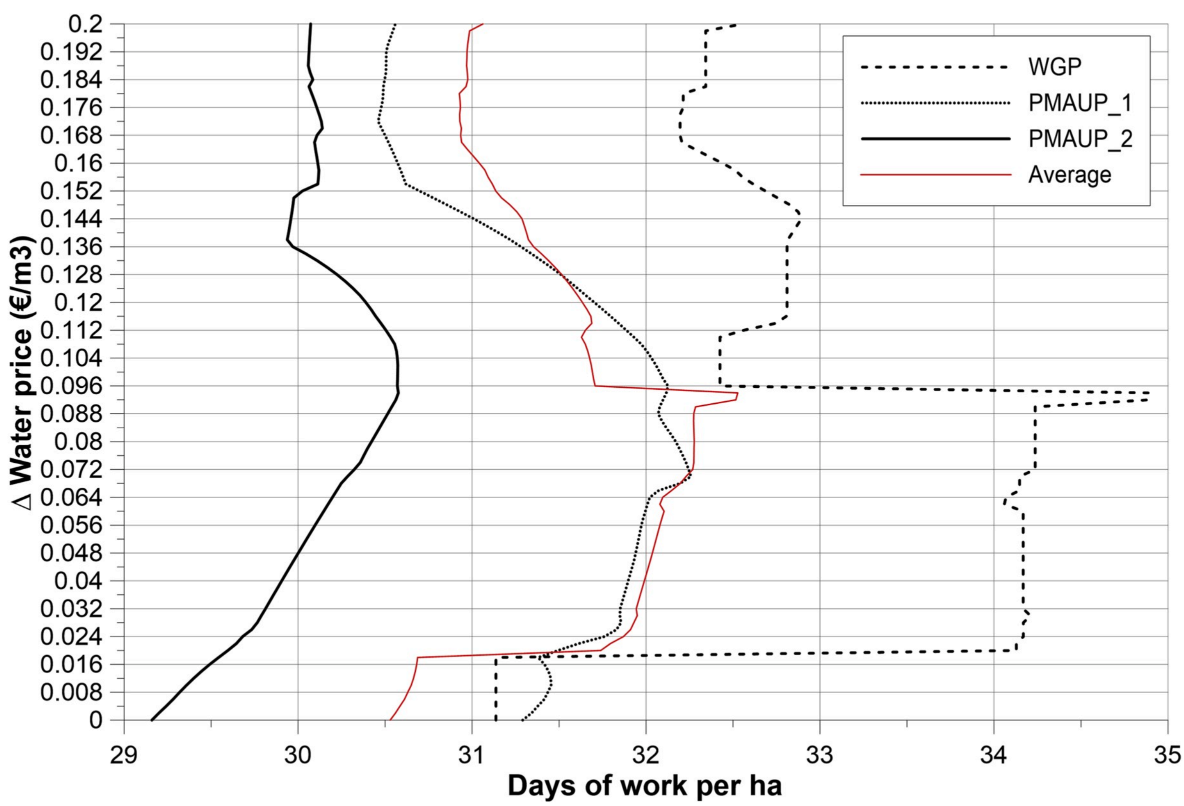

Fig. 6. Employment.
Source: own elaboration.

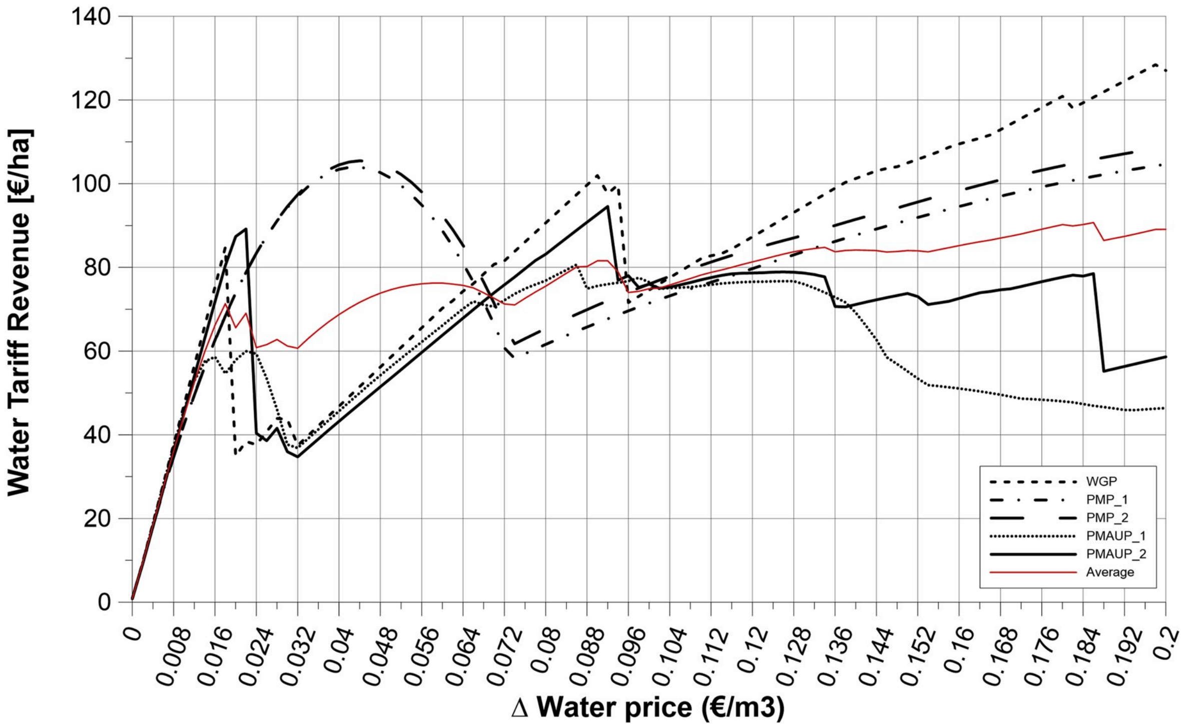

Fig. 7. Tariff revenue.
Source: own elaboration.

discussed in a second meeting. The range of pricing scenarios was revised and delimited to increase detail (from an initial range of 0–1 EUR/m^3 at 0.01 EUR/m^3 intervals to a range of 0–0.2 EUR/m^3 at 0.002 EUR/m^3 intervals, which remained unchanged), and the ensemble approach and its components were validated. In *Step 3*, the results obtained in Step 2 were used to assess vulnerabilities to pricing strategies.

At the beginning of the research, policy makers and stakeholders had already shown concern on the impact that water pricing policies may have on traditional rice fields and permanent crops, most notably vineyard. Simulations showed resilience of the latter crop to price increases; in contrast, rice systems were found to be highly vulnerable, with their area completely disappearing in *all* ensemble components in the price range 0.032–0.074 EUR/m^3. This was somewhat expected due to its intensive use of water (nearly 31,500 m^3/ha on average) and relatively low return as compared to alternative crops in the region (average, 1300 EUR/ha). As shown in the previous section, multi-attribute models suggest a rapid reduction in rice area in the interval 0.008–0.032 EUR/m^3, while single-attribute PMP models predict a less sharp, yet steady decrease in the interval 0–0.074 EUR/m^3. Among multi-attribute models, PMAUP_1 predicts a smooth decline of rice in the interval 0.008–0.012 EUR/m^3, which becomes more pronounced in the interval 0.012–0.018 EUR/m^3 (44% of rice area has already disappeared at this point) and again in the interval 0.02–0.032 EUR/m^3, after which rice systems disappear; PMAUP_2 predicts a faster decline of rice area, which goes from 18% to 0% of agricultural area in the interval 0.02–0.024 EUR/m^3; while WGP predicts the almost complete withdrawal of rice from the crop portfolio after prices increases of 0.02 EUR/m^3 (83% of the original area), followed by a progressive reduction of the remaining rice area in the interval 0.02–0.032 EUR/m^3. This suggests the existence of a tipping point for rice systems beyond a price increase of 0.012–0.02 EUR/m^3 for 3 of the 5 ensemble components.

Step 4 involves the decision on the policy to be adopted. We started by using constrained optimization methods through the utilization of the Minimization of Maximum regret algorithm (MinMax regret). MinMax regret measures regret as “the distance between the indicator for an instrument and the best indicator in a given scenario” (Graveline, 2019). In our case, we are looking for the pricing policy that yields the minimum maximum regret considering results from all models; in other words, the pricing policy that minimizes surprise/tipping points with potential disproportionate costs. The MinMax regret approach does not demand any additional information besides what is already available in the previous section; however, it tends to be highly conservative. Use of the MinMax regret approach suggested a maximum price increase of 0.008 EUR/m^3, i.e. the price at which the maximum regret is 0 (no loss of rice area in any model). At this price increase, water conservation is fairly small (0–650 m^3/ha), profit falls slightly (49–57 EUR/ha) and tariff revenue is quite significant (35–38 EUR/ha).

Constrained optimization methods were subsequently complemented with the use of heuristics through expert judgement, with the aim of exploring more ambitious and *feasible* water conservation-rice area tradeoffs. Through expert judgement, the feasible price increase was expanded to 0.012 EUR/m^3, right before the tipping point identified in Step 3. At a price increase of 0.012 EUR/m^3, the rice area diminishes but the impact is still moderate (16.7% reduction in PMP, 5.6% in PMAUP and 0% in WGP models), water conservation is limited (84–985 m^3/ha), foregone profit is small (73–85 EUR/ha) and tariff revenue is significant (49–56 EUR/ha). It should be noted that this price increase is slightly lower but still close to the 0.013 EUR/m^3 (Frontuto et al., 2020) price increase proposed by experts in a report prior to our analysis.

5. Discussion

Water crisis are among the greatest global societal threats – and Europe is not spared (WEF, 2019). The “total *cost of droughts* over the past thirty years amounts to *EUR 100 billion*”, with the yearly average cost *quadrupling* over the same period (EC, 2017). Structural water scarcity is an expanding phenomenon affecting at least 17% of the European territory. 55% of surface water bodies in the EU have failed to meet good ecological status, and although the first cycle River Basin Management Plans predicted a 10% improvement in this figure by 2015, “delays in implementing many of the improving measures” have caused deferrals, further disruptions and *irreversible* damage in the supply of valuable ecosystem services (EC, 2017). Against this backdrop, EU institutions have called on policy makers to find "the right price tag on water" (EC, 2012). Such price should: i) be volumetric to enhance incentive-pricing water conservation; ii) recover not only financial, but also resource and environmental costs to convey adequate price signals; and iii) avoid disproportionate costs through affordable prices for strategic sectors and related users.

Balancing these three aspects has proven to be challenging. While the seminal literature on water pricing substantiates the effectiveness of the instrument towards achieving water conservation (Dinar and Subramanian, 1997), full cost recovery in overallocated basins typically involves a significant increase in prices with non-negligible impacts on income and employment (Perry, 2005). Where water has been historically perceived as plentiful and irrigation techniques have remained essentially unchanged for decades or even centuries, responses to pricing generally involve reducing water use. This results in significant water conservation at low or medium price ranges (Rey et al., 2018), albeit (traditional) agricultural systems may suffer abrupt transformations with non-trivial impacts on the local economy, which can be further amplified economy-wide (Parrado et al., 2019). On the other hand, where autonomous adaptation to water scarcity has given rise to sophisticated and relatively profitable irrigation systems, we may observe high ability to pay and inelastic responses to prices at low or medium price ranges, which results in limited crop portfolio changes despite significant pricing-induced income losses (Zuo et al., 2015). This is e.g. the case of the absolute water scarce basins of southern Spain, where farmers have invested on greenhouses or irrigation modernization, among other techniques (Berbel and Mateos, 2014). The upshot is that farmers will shift to less-water intensive and less profitable crops at relatively high prices, thus increasing the economic costs of water conservation. This can be aggravated by non-virtuous adaptation strategies such as shifting from surface water to more loosely controlled groundwater, thus transferring the overallocation problem to water bodies where norms and regulations are more difficult to supervise and enforce (Gómez and Pérez-Blanco, 2012).

The non-trivial tradeoffs between economic efficiency and water conservation highlighted above raise barriers to the political acceptability of pricing (Rausser et al., 2011), which have *de facto* precluded the implementation of full cost recovery (Berbel and Mateos, 2014). Notwithstanding the difficulty to fully recover water use costs, pricing still represents a powerful incentive that can contribute towards achieving collectively agreed environmental goals if certain conditions are met. For example, where demand is inelastic, pricing can be "leveraged against the high willingness to pay of users" to raise revenues that contribute towards enhancing the environmental status of water bodies (e.g. payment for ecosystem services).

The challenges and opportunities above are observable in our case study area in the Piedmont Region. According to Piedmont Region estimates, achieving full cost recovery necessitates a 2500-fold water price increase (0.30 EUR/m^3 price increase equivalent) (Frontuto et al., 2020), which following our estimates would not only significantly reduce agricultural profit (−36% on average) but also be inconsequential in terms of water conservation beyond a price increase of 0.2 EUR/m^3 (ensemble average) due to increasingly inelastic response to higher prices. Perhaps not surprisingly, experts advising the water policy reform found the costs of such price increase disproportionate, also on the grounds of potential irreversible impacts on the structure of traditional irrigated agriculture, and suggested a (maximum) price increase of up to 0.014 EUR/m^3 (Frontuto et al., 2020). According to our modeling exercise, although the impacts in terms of foregone profit may not be regarded as disproportionate (5%–11% foregone profit in the price increase interval 0.012–0.032 EUR/m^3, up to 20% in PMP at 0.074 EUR/m^3), a price increase beyond 0.012 EUR/m^3 results in the rapid substitution of the traditional Piedmontese rice landscape by rainfed crops and corn, with rice completely disappearing from the crop portfolio following a price increase of 0.032 EUR/m^3 (0.074 EUR/m^3 for PMP). This is expected to have a critical impact on water retention capacity during the summer discharge peak. Furthermore, the forward and backward linkages of agriculture with related economic sectors (e.g. food industry) are likely to amplify the economic impact of rice systems removal, which may also affect historical water drainage and supply infrastructures.

The downside of setting a maximum 0.012 EUR/m^3 price increase is a modest water conservation potential: an ensemble average of 350 million m^3 of water conserved annually, or 6.82% of current withdrawals (between 1.7% and 9.5% depending on the ensemble component). On the other hand, tariff revenues increase consistently and almost peak for some models in the price interval 0–0.012 EUR/m^3 (up to 56 EUR/ha), while profit reduction is relatively low (4.9%–5.7% depending on the model). This suggests that if rice systems are to be preserved, water pricing is not an effective instrument to conserve water, but still retains some potential as a revenue raising tool.

6. Conclusions

This work develops a mathematical programming multi-model ensemble framework to sample uncertainty and underpin robust decision making. This is, to the best of our knowledge, the first ensemble experiment to assess the local impacts of agricultural water policy reform. Its development, implementation and subsequent iterative policy formulation along with stakeholders provided insights into modeling and scenario uncertainty that proved valuable towards the identification of robust pricing policy in the Piedmont Region. The ensemble could be improved through its connection to complementary ensemble experiments that sample uncertainty in physical systems, notably the water system (Cloke et al., 2013), and in other human systems (e.g. macroeconomics, which would allow us to model crops' demand and prices endogenously) (Parrado et al., 2019). Modularity and protocols could be used to connect such complex systems among them, in line with recent contributions to the area of socio-hydrology (Essenfelder et al., 2018). The ensemble could be also expanded through the inclusion of additional mathematical programming models (Graveline, 2016). The individual ensemble components could also benefit from further research through: i) exploration of new attributes in multi-attribute models, such as seasonal forecasts, where this data is available; and ii) expansion of the crop portfolio to explicitly differentiate management techniques in agriculture in general and irrigation in particular. Such improved ensemble could be used to assess the impact of adaptation policies other than pricing in the irrigation sector, to complement our assessment of agricultural water pricing with that of alternative/complementary policies; although the current institutional and legal context and policy agenda suggest the feasibility of such instruments in the Piedmont Region/Italy may be unclear, as previously discussed.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2020.110645.

References

Ajzen, I., 1991. The theory of planned behavior. Organizational behavior and human decision processes. Theories of Cognitive Self-Regulation 50, 179–211. https://doi.org/10.1016/0749-5978(91)90020-T.

Arpa Piemonte, 2018. Allegato 3 A. Bilancio Idrico Regionale Delle Acque Superficiali.

Augusti, E., Baglini, M., Bartolini, C., Cosimi, S., 2018. Prontuario Reda - Dati Tecnico-Economici Settore Agricoltura. Reda.

Banca D'Italia, 2018. Economie Regionali. L'economia del Piemonte.

Bartolini, F., Bazzani, G.M., Gallerani, V., Raggi, M., Viaggi, D., 2007. The impact of water and agriculture policy scenarios on irrigated farming systems in Italy: an analysis based on farm level multi-attribute linear programming models. Agric. Syst. 93, 90–114. https://doi.org/10.1016/j.agsy.2006.04.006.

Berbel, J., Mateos, L., 2014. Does investment in irrigation technology necessarily generate rebound effects? A simulation analysis based on an agro-economic model. Agric. Syst. 128, 25–34.

Charrad, M., Ghazzali, N., Boiteau, V., Niknafs, A., 2014. NbClust: an R package for determining the relevant number of clusters in a data set. J. Stat. Software 61, 1–36. https://doi.org/10.18637/jss.v061.i06.

Cloke, H.L., Pappenberger, F., van Andel, S.J., Schaake, J., Thielen, J., Ramos, M.H., 2013. Hydrological ensemble prediction systems. Hydrol. Process. 27, 1–4. https://doi.org/10.1002/hyp.9679.

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

Director of the Est Sesia Land Reclamation and Irrigation Board, 2019. Personal Communication, Meeting in the Est Sesia Land Reclamation and Irrigation Board. Personal Communication.

EC, 2012. EU Blueprint to Safeguard Europe's Water Resources (Communication No. COM(2012) 673 Final). European Commission, Brussels (Belgium).

EC, 2017. Commission Staff Working Document - Agriculture and Sustainable Water Management in the EU (Staff Working Document No. SWD(2017) 153 Final). European Commission, Brussels (Belgium).

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. http://www.eea.europa.eu/data-and-maps/indicators/water-exploitation-index.

EEA, 2018. CORINE Land Cover (CLC) [WWW Document]. https://land.copernicus.eu/pan-european/corine-land-cover/clc2018?tab=download.

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.

Eurostat, 2017. Eurostat Database [WWW Document]. Eurostat Database. https://ec.europa.eu/eurostat/web/regions/data/database. (Accessed 6 February 2019).

Frontuto, V., Dalmazzone, S., Mancin, P., Turrini, P., Pertile, M., Massarutto, A., de Carli, A., 2020. Abstraction charges based on the assessment of the environmental costs of water resource use: a simulation for the Piedmont region in Italy. In: Water Economics and Law in Italy. Springer. In press.

Gómez, C.M., Pérez-Blanco, C.D., 2012. Do drought management plans reduce drought risk? A risk assessment model for a Mediterranean river basin. Ecol. Econ. 76, 42–48. https://doi.org/10.1016/j.ecolecon.2012.01.008.

Gómez-Limón, J.A., Vera-Toscano, E., Rico-González, M., 2012. Measuring individual preferences for rural multifunctionality: the importance of demographic and residential heterogeneity. J. Agric. Econ. 63, 1–24. https://doi.org/10.1111/j.1477-9552.2011.00325.x.

Gómez-Limón, J.A., Gutiérrez-Martín, C., Riesgo, L., 2016. Modeling at farm level: positive multi-attribute utility programming. Omega 65, 17–27. 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., 2019. Combining flexible regulatory and economic instruments for agriculture water demand control under climate change in Beauce. Water Resources and Economics, 100143. https://doi.org/10.1016/j.wre.2019.100143.

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.

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.

Heckelei, T., Britz, W., 2005. Models based on positive mathematical programming: state of the art and futher extensions. In: Proceedings of the 89th EAAE Seminar. Presented at the 89th EAAE Seminar, Parma (Italy), pp. 48–73.

Heckelei, T., Britz, W., Zhang, Y., 2012. Positive mathematical programming approaches - recent developments in literature and applied modelling. Bio-based and Applied Economics Journal 1 (1), 109–124. https://doi.org/10.13128/BAE-10567.

Henry de Frahan, B., Buysse, J., Polome, P., Fernagut, B., Harmignie, O., Lauwers, L., Van Huylenbroeck, G., Van Meensel, J., 2007. Positive mathematical programming for agricultural and environmental policy analysis: review and practice. In: Handbook of Operations Research in Natural Resources, pp. 129–154. https://doi.org/10.1007/978-0-387-71815-6_8.

Hino, M., Hall, J.W., 2017. Real options analysis of adaptation to changing flood risk: structural and nonstructural measures. ASCE-ASME Journal of Risk and Uncertainty in Engineering Systems, Part A: Civ. Eng. 3, 04017005 https://doi.org/10.1061/AJRUA6.0000905.

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

INEA, 2016. Private Database RICA ([WWW Document]).

INEA, 2018. Private Database RICA (2008 - 2016): Economic Data at Province and Regional Level in Piedmont Region, Italy ([WWW Document]).

IPCC, 2014. IPCC Fifth Assessment Report (AR5) (No. WGII). Intergovernmental Panel on Climate Change, Geneva (Switzerland).

ISTAT, 2010. Aziende, superficie irrigata e volumi irrigui in Censimento Agricoltura 2010. http://dati-censimentoagricoltura.istat.it/Index.aspx. (Accessed 1 November 2018).

ISTAT, 2016. Database Istat. http://agri.istat.it/sag_is_pdwout/jsp/dawinci.jsp?q=plC020000010000012000&an=2017&ig=1&ct=244&id=15A|18A|25A. (Accessed 7 January 2019).

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

Júdez, L., de Miguel, J.M., Mas, J., Bru, R., 2002. Modeling crop regional production using positive mathematical programming. Math. Comput. Model. 35, 77–86. https://doi.org/10.1016/S0895-7177(01)00150-9.

Kampas, A., Rozakis, S., 2017. On the scarcity value of irrigation water: juxtaposing two market estimating approaches. Water Resour. Manag. 1–13. https://doi.org/10.1007/s11269-017-1574-0.

Marchau, V.A.W.J., Walker, W.E., Bloemen, P.J.T.M., Popper, S.W. (Eds.), 2019. Decision Making under Deep Uncertainty: from Theory to Practice. Springer International Publishing.

Murtagh, F., Legendre, P., 2014. Ward's hierarchical agglomerative clustering method: which algorithms implement Ward's criterion? J. Classif. 31, 274–295. https://doi.org/10.1007/s00357-014-9161-z.

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

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.

Pérez-Blanco, C.D., Koks, E.E., Calliari, E., Mysiak, J., 2017. Economic impacts of irrigation-constrained agriculture in the lower Po basin. Water Econs. Policy 4, 1750003. https://doi.org/10.1142/S2382624X17500035.

Perry, C., 2005. Irrigation reliability and the productivity of water: a proposed methodology using evapotranspiration mapping. Irrigat. Drain. Syst. 19, 211–221. https://doi.org/10.1007/s10795-005-8135-z.

Phillips, S.D., Estler, W.T., Doiron, T., Eberhardt, K.R., Levenson, M.S., 2001. A careful consideration of the calibration concept. J Res Natl Inst Stand Technol 106, 371–379. https://doi.org/10.6028/jres.106.014.

Rausser, G.C., Swinnen, J., Zusman, P., 2011. Political Power and Economic Policy: Theory, Analysis, and Empirical Applications. Cambridge University Press, New York.

Regione Piemonte, 2017. Condizionalità ex ante riferita al settore delle risorse idriche prevista dall'Accordo di Partenariato ai fini dell'accesso ai fondi europei relativi al Programma di Sviluppo Rurale 2014-2020. Attuazione della DGR n, 43-4410 del 19 dicembre 2016.

Regione Piemonte, 2018. Piano di Tutela delle Acque, Relazione Generale Dicembre 2018.

Rey, D., Pérez-Blanco, C.D., Escriva-Bou, A., Girard, C., Veldkamp, T.I.E., 2018. Role of economic instruments in water allocation reform: lessons from Europe. Int. J. Water Resour. Dev. 1–34. https://doi.org/10.1080/07900627.2017.1422702, 0.

Santato, S., Mysiak, J., Pérez-Blanco, C.D., 2016. The water abstraction license regime in Italy: a case for reform? Water 8, 103. https://doi.org/10.3390/w8030103.

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.

Sistemapiemonte, R.P., 2018. Anagrafe Agricola Unica - Data Warehouse e Open Data. http://www.sistemapiemonte.it/fedwanau/elenco.jsp. (Accessed 5 November 2018).

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.

Tebaldi, C., Knutti, R., 2007. The use of the multi-model ensemble in probabilistic climate projections. Phil. Trans. Math. Phys. Eng. Sci. 365, 2053–2075. https://doi.org/10.1098/rsta.2007.2076.

WEF, 2019. Global Risks 2019 (Report No. In: Global Risks, fourteenth ed. World Economic Forum.

Zuo, A., Wheeler, S.A., Adamowicz, W.L., Vic, Boxall, P.C., Hatton-MacDonald, D., 2015. Measuring price elasticities of demand and supply of water entitlements based on stated and revealed preference data. Am. J. Agric. Econ. https://doi.org/10.1093/ajae/aav022 aav022.

3.3. *Economic Sustainability of Irrigation-Dependent Ecosystem Services Under Growing Water Scarcity. Insights From the Reno River in Italy*

3.3.1. Resumen

A medida que aumenta la escasez de agua, los regantes están adoptando sistemas de riego modernos para aumentar la proporción de agua consumida por los cultivos y mitigar los efectos de la escasez sobre la producción. Este aumento del consumo beneficioso se debe fundamentalmente a los menores flujos de retorno – escorrentía y filtración – hacia el medio ambiente, que son fundamentales para mantener los humedales y otras infraestructuras verdes dependientes del riego que proporcionan valiosos servicios ecosistémicos. Adoptamos un innovador conjunto multimodelo de modelos de programación matemática para evaluar las respuestas de los regantes a las compensaciones pecuniarias diseñadas para sostener los servicios ecosistémicos dependientes del riego (en ingles *Payments for Watershed Services* – PWS), bajo múltiples escenarios. El resultado es una base de datos de pronósticos que representa el rango de estados futuros plausibles, que se utiliza para evaluar el rendimiento económico e identificar una estrategia robusta de adaptación a la creciente escasez. Nuestro análisis de la Confederación Hidrográfica del Río Reno en Italia muestra que, bajo la mayoría de los modelos y escenarios, la estrategia conservacionista tiene un rendimiento económico superior al de la estrategia de adaptación autónoma donde se adoptan sistemas de riego modernos. La estrategia conservacionista también se encuentra más robusta que la estrategia autónoma. Sin embargo, a menos que se establezcan incentivos razonables, los regantes pueden adoptar tecnologías de riego. En escenarios de cambio climático moderado, la eliminación de los subsidios existentes a los sistemas de riego modernos es suficiente para disuadir su adopción. En los escenarios de cambio climático severo, se necesitarán PWS adicionales para los regantes para asegurar la sostenibilidad de los servicios ecosistémicos dependientes del riego.

ADVANCING EARTH AND SPACE SCIENCE

Water Resources Research

RESEARCH ARTICLE

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Key Points:

- We assess economic performance of conservationist (sustain irrigation-dependent ecosystems) v. irrigation modernization strategy in NE Italy
- Under most models and scenarios, the conservationist strategy has a superior economic performance; and is also found more robust
- Unless sensible incentives are set (remove infrastructure subsidies, deploy PWS), irrigators may modernize nonetheless

Supporting Information:

Supporting Information may be found in the online version of this article.

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Economic Sustainability of Irrigation-Dependent Ecosystem Services Under Growing Water Scarcity. Insights From the Reno River in Italy

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Abstract As water scarcity grows, irrigators are adopting modern irrigation systems to increase the proportion of water use consumed by crops and mitigate impacts on production. This increased beneficial consumption is fundamentally sourced by lower return flows-runoff and percolation-back into the environment, which are critical to sustain wetlands and other irrigation-dependent green infrastructures that supply valuable ecosystem services. We adopt an innovative multi-model ensemble of mathematical programming models to assess irrigators' responses to pecuniary compensations designed to sustain irrigation-dependent ecosystem services (Payments for Watershed Services-PWS), under multiple scenarios. The upshot is a database of forecasts that represents the range of plausible future states, which is used to assess economic performance and identify a robust adaptation strategy to growing scarcity. Our analysis of the Reno River Land Reclamation and Irrigation Board in NE Italy shows that, under most models and scenarios, the conservationist strategy has a superior economic performance than the autonomous adaptation strategy where modern irrigation systems are adopted. The conservationist strategy is also found more robust than the autonomous strategy. However, unless sensible incentives are put in place, irrigators may adopt irrigation technologies nonetheless. Under mild to moderate climate change scenarios, removing existing subsidies to modern irrigation systems is sufficient to deter their adoption. Under severe climate change scenarios, additional PWS to irrigators will be necessary to ensure the sustainability of irrigation-dependent ecosystem services.

1. Introduction

Throughout the world, the surface and subsurface return flows that leave agricultural systems following water withdrawal and application have created and sustained wetlands, forested areas and other green infrastructures that supply valuable ecosystem services (Grafton et al., 2018). Examples of irrigation-dependent ecosystem services include habitat conservation (e.g., wetlands for migrating waterfowl), climate regulation (e.g., carbon sequestration), soil retention, cultural heritage (e.g., spiritual fulfillment, intellectual development) and amenity services (such as esthetic enjoyment or recreation), among others. Since these ecosystem services are typically outside of the market, their provision by irrigated systems is not included in the valuation of agricultural production, nor is their eventual loss where the underlying green infrastructure is degraded (TEEB, 2015).

Irrigation-dependent ecosystem services are under threat by rising water scarcity and adaptive responses by farmers through modern irrigation systems, such as sprinkler or drip irrigation systems, laser leveling of fields, piped delivery systems, canal lining, and other physical rehabilitation of irrigation and delivery systems (Perry & Steduto, 2017). Modern irrigation systems are designed to increase the proportion of beneficial consumption per unit of water use. As scarcity grows, this allows irrigators to negate reductions in biophysical production from diminished water supply. This increased beneficial consumption is partially sourced by lower non-beneficial consumption (e.g., evaporation from wet fields), but the predominant source is a reduction in return flows—runoff and percolation—back into the environment, which are often reused by irrigation-dependent ecosystem services downstream. The upshot is a sustained reduction of return flows due to increased beneficial consumption, degraded green infrastructures and a deterioration of irrigation-dependent ecosystem services (Figure 1; Pérez-Blanco et al., 2020).

Exceptions will arise in cases where irrigators can participate in incentive schemes that encourage environmental performance, such as payments for ecosystem services (PES). PES are a pecuniary compensation that internalizes

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the positive externalities generated by providers (e.g., irrigators) through the protection or enhancement of green infrastructure (Asbjornsen et al., 2015). PES include both private (where individuals and/or private and non-governmental organizations are the sole buyer) and public sector payments, although there are relatively few examples of the former in the case of watershed ecosystem services, where most funding comes from supranational, national or regional governments acting on behalf of their constituency. PES have received growing attention and funding due to their perceived capacity to enhance environmental and economic performance, including through the generation of relevant co-benefits such as increased and/or more stable incomes in rural areas (Bremer et al., 2016). The literature records over 400 water-related PES (also known as Payment for Watershed Services, PWS) in more than 60 countries, rehabilitating a surface one and a half times the size of India for a total financial value of $25 billion (Bennett & Franziksa, 2016). However, "few rigorous evaluations" on the economic performance of PWS programs exist (Bhaduri et al., 2021). Lack of economic rigor is attributed to two main factors: (a) the oversimplistic representation of human agency and (b) the treatment of uncertainty (or lack thereof).

1. Most economic assessments of PWS rely on oversimplistic representations of human agency (e.g., through linear objective functions) built entirely on the basis of relationships observed in historical data (e.g., projections from baseline conditions; Bhaduri et al., 2021; Harou et al., 2009; Pérez-Blanco et al., 2021). This contravenes the Lucas Critique, after the Nobel Laureate Robert Lucas, which states that it is inadequate to predict the effects of policy shocks on human behavior entirely on the basis of relationships observed in historical data (Lucas, 1976). This is because the parameters of models elicited this way are not structural, that is, not policy-invariant, and would necessarily change whenever the policy (e.g., PWS adoption) changes. Instead, historical data should be used to reveal the micro-foundations or deep parameters (utility, preferences, resource constraints) driving agent's responses, for example, through mathematical programming models (Graveline, 2016); and use these models for prediction.
2. Available PWS economic assessments typically disregard scenario and modeling uncertainty. Scenarios in PWS studies are typically built either using simplistic point predictions or through probabilistic descriptions of plausible future states; which are then fed to a single model that is used to produce a forecast. Yet, in deeply uncertain socio-ecological systems, where researchers and stakeholders typically do not know/cannot agree on the model that relates scenarios to outputs, or the probability of these scenarios, such consolidative approach risks providing more information than what we can reasonably claim to know (Marchau et al., 2019). Disregard of scenario and modeling uncertainty becomes problematic where PWS performance is highly sensitive to future states. For example, a recent assessment of PWS in Colorado (US) found that the potential financial returns to beneficiaries was expected to be positive, but also warned that these returns would vary considerably depending on the scenario and only hold under specific model assumptions (Jones et al., 2017). Unfavorable surprises, especially those resulting in abrupt change, can lead to tipping points that significantly and irreversibly deviate expected from realized policy performance (Anderies, 2015). This is closely related to issues of permanence, that is, whether PWS will lead to sustained restoration/conservation of water-related ecosystem services, especially when future conditions (e.g., climate) may abruptly change (Rode et al., 2015).

This paper builds on the concepts of micro-foundations (Lucas, 1976), exploratory modeling (Kwakkel & Pruyt, 2013) and multi-model ensemble (IPCC, 2014; Sapino et al., 2020) to develop an economic assessment framework for PWS that uses mathematical programming methods to elicit the deep parameters driving human behavior, while thoroughly sampling scenario and modeling uncertainty in the analysis of irrigators' responses. The ensemble includes 2 Positive Mathematical Programming (PMP) models, 2 Positive Multi-Attribute Utility Programming (PMAUP) models and 1 Linear Programming (LP) model. The assessment framework is used to simulate the economic performance (through utility) of an hypothetical conservationist strategy (i.e., no irrigation modernization and reallocation of water toward the environment) v. autonomous adaptation strategy (i.e., irrigation modernization and reallocation of water toward agriculture). By comparing the monetized foregone utility experienced by irrigators under the conservationist strategy v. the monetized foregone utility experienced by irrigators under the autonomous adaptation strategy we can obtain the minimum compensation irrigators would be willing to accept to sustain the PWS scheme. This information is subsequently compared with estimates of the economic value of ecosystem services to assess the economic performance of the proposed PWS. Repeating this process for multiple models and scenarios (climate change, irrigation modernization costs, economic values of ecosystem services) yields a database of forecasts that represents the range of plausible future states. Coupled with automated robust decision methods, this database is used to identify robust adaptation strategies that avoid

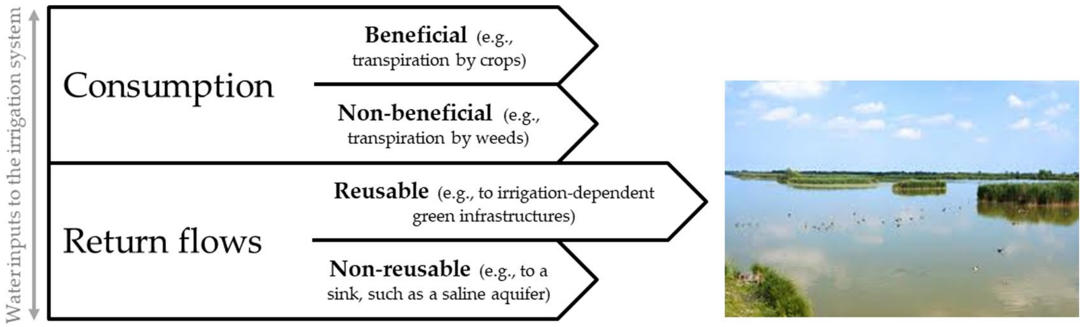

Figure 1. Water accounting balance. All water entering the irrigation system goes to either: (a) beneficial consumption, water that is purposefully converted to water vapor, primarily crop transpiration; (b) non-beneficial consumption, water that is not purposefully converted to vapor, such as through transpiration by weeds or evaporation from wet surfaces; (c) reusable return flows, water reaching a useable water body with downstream demand; and (d) non-reusable return flows, water flowing without benefit to a sink such as the sea, and therefore not useable.

unfavorable outcomes that can be identified ex-ante. Methods are flexible and replicable, and are illustrated with an application to the PWS program presently being discussed between the Regione Emilia Romagna and the Reno River Land Reclamation and Irrigation Board (Reno River LRIB, in Italian: Bonifica Renana) to conserve the ecosystem services provided by irrigation-dependent wetlands in NE Italy.

2. Case Study Background: The Reno River Land Reclamation and Irrigation Board

Since Roman times, irrigators in Northern Italy have built water retention, distribution and drainage infrastructures to expand the surface available for agricultural and urban developments. Canals, irrigation systems, detention basins, drainage systems and other water works have transformed the landscape and created a complex network of man-made gray and green infrastructures that is managed, monitored, maintained and modernized by LRIBs, a public-private partnership that brings together all the owners of land and buildings (public and private) within its area of influence.

The Reno River LRIB, located to the Southeast of the Po River Basin, manages an area of 341,953 ha across five provinces (Bologna, Firenze, Modena, Ferrara, Ravenna, Prato, and Pistoia). Of this surface, 56,067 ha are lowland alluvial plains that are drained using 1,667 km of canals, 26 detention basins (with the capacity to store over 42 million m^3 of rainfall water) and 26 pumping systems. The same network of infrastructures (plus 63 additional pumping systems for irrigation) is used to support the irrigation of 18,000 ha, which are divided into 5 irrigation districts (the agents in the mathematical programming model): C001 (Dep. Bologna-Po), C002 (Po), C003 (Reno, Reno-Po, Dep. Bologna), C004 (Quaderna, Sillaro, Dep. Ozzano Emilia, Dep. Castel S. Pietro), C005 (Ghironda, Lavino, Rii Pedecollinari, Dep. Anzola Emilia, Dep. Calcara, Dep. Calderara di Reno, Dep. Padulle di Sala). To this end, the Reno River LRIB distributes on average 68 million m^3/year of water for irrigation, which comes exclusively from surface water sources: 73% from the nearby Po River, via the Emiliano Romagnolo Canal (in Italian: Canale Emiliano-Romagnolo—CER); 16% from the Reno River; and the remainder from detention basins (see Figure 2; Nomisma, 2019a).

Transportation and application technical efficiencies vary across the LRIB, being estimated on average at 50% and 85%, respectively (Nomisma, 2019b). As a result, drainage and irrigation activities produce non-trivial surface return flows, which have created and sustain 160 ha of protected areas with

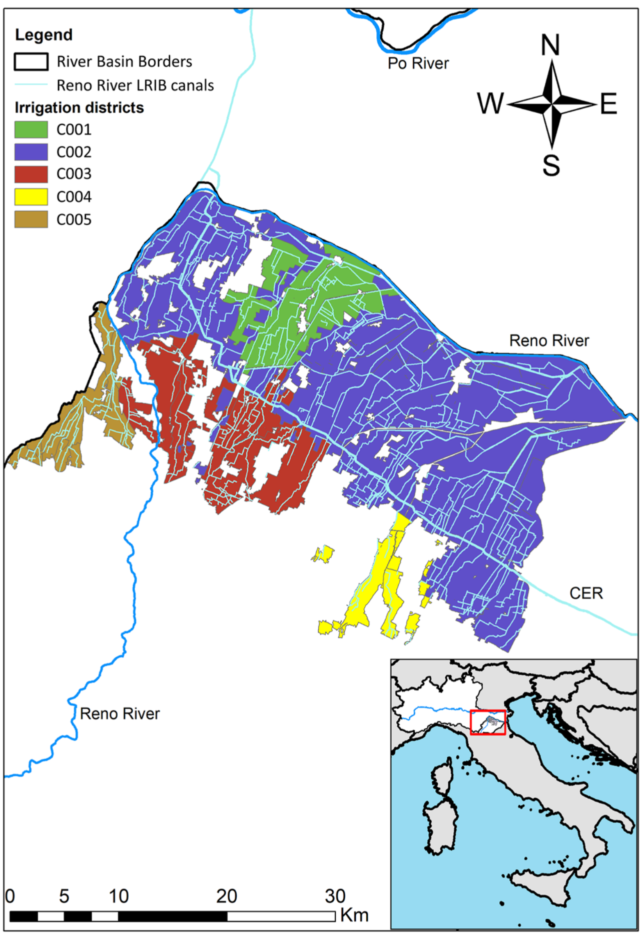

Figure 2. The Reno river LRIB.

wetlands across the LRIB. These wetlands provide valuable services beyond the conventional water supply and flood protection services typically attributed, and paid, to the Reno River LRIB, including: provisioning (e.g., food production, water storage), regulating (e.g., carbon sequestration, water purification), habitat (e.g., genetic diversity) and cultural services (e.g., esthetic, spiritual, educational and recreational; Nomisma, 2019a). Annex I in the Supporting Information S1 offers a comprehensive description of the ecosystem services provided by the Reno LRIB, which leverages on a review of the scientific literature complemented with the feedback provided by local stakeholders in a workshop held in Bologna (Italy) in April 2019.

On the other hand, in a context of diminishing water supply due to climate change, the same inefficiencies that maintain these valuable ecosystem services constrain water availability for irrigated agriculture. This has led agricultural landowners within the Reno River LRIB to call for investments toward the modernization of irrigation and drainage infrastructures, particularly the network of canals. Irrigation modernization plans have been received with caution by the government of the Emilia-Romagna Region (where most of the lowlands are located) and the Reno River LRIB itself, which are concerned of the impact this intervention would have on protected areas and their wetlands (Nomisma, 2019b). Following a series of exchanges between the regional government and the Reno River LRIB, a research project was commissioned to the authors to explore the economic feasibility and sustainability of a PWS between the regional government (buyer) and the irrigators of the Reno River LRIB (providers), whose methods, results and conclusions are reported below.

3. Methods

This paper develops an economic assessment framework for PWS that uses mathematical programming methods to elicit the deep parameters driving human behavior, while thoroughly sampling scenario and modeling uncertainty in the analysis of irrigators' responses. This mechanistic methodology is complemented with automated robust decision methods to identify a non-regret adaptation strategy.

3.1. Multi-Model Ensemble

While simplification through model conceptualization helps to effectively convey insights into how to better allocate resources in complex systems, it also leads to imperfections in the representation of the system and errors (Tebaldi & Knutti, 2007). Unawareness of these errors can result in misleading policy recommendations, significant deviations of expected from realized performance, and maladaptation (Hino & Hall, 2017), which can be aggravated by issues of non-convexity and irreversibility (Anderies, 2015). Despite these problems, economic performance evaluations of PES, and PWS specifically, typically rely on a single model and model setting to produce forecasts, which makes these schemes vulnerable to modeling uncertainty.

Ecological sciences have addressed modeling uncertainty through ensemble experiments that use multiple models to sample uncertainty (see e.g., Cloke et al., 2013; IPCC, 2014). Yet, ensemble experiments are under-researched in all disciplines of social sciences. In this paper, we sample uncertainty in human behavior and responses using a multi-model ensemble of mathematical programming models consisting of 2 PMP, 2 PMAUP and 1 LP models—all of which are widely used methods in the literature on economic models for agricultural water management. In these models, farmers decide on the crop portfolio so to maximize the utility provided by a set of utility-relevant variables, subject to a series of constraints (Graveline, 2016):

$$\underset{x}{\text{Max}}\ U(\boldsymbol{x}) = U\left(z_1(\boldsymbol{x});\ \ z_2(\boldsymbol{x}); z_3(\boldsymbol{x})\ldots\ \ z_m(\boldsymbol{x})\right) \tag{1}$$

$$s.t.:\ 0 \le x_i \le 1 \tag{2}$$

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

$$\boldsymbol{x} \in F(\boldsymbol{x}) \tag{4}$$

$$\boldsymbol{z}(\boldsymbol{x}) \in R^m \tag{5}$$

where $\boldsymbol{x}$ is the crop portfolio, a vector representing the share of land allotted to each crop i; $\boldsymbol{z}(\boldsymbol{x})$ is a vector of utility-relevant attributes defined so that "more-is-better" (i.e., increasing the provision of one attribute, *caeteris paribus*, increases utility); $U(\boldsymbol{x})$ is a parameterized objective function that relates inputs (the provision of utility-relevant attributes under a given crop portfolio) to outputs (utility); and $F(\boldsymbol{x})$ is the set of constraints conforming the domain, which are common to all models in the ensemble, and whose mathematical formulation is available in Annex II in the Supporting Information S1. Of particular relevance is the water allocation constraint:

$$\sum\nolimits_{c=1}^{n} \frac{w_i}{\mathrm{eff}_i}\ x_i \leq W_g \tag{6}$$

where w_i are the net water needs or evapotranspiration (i.e., excluding inefficiencies) of crop i (in m3/ha), eff_i is the irrigation efficiency (which ranges between 0 and 1), w_c/eff_c are the gross water needs or water applied to crop i (i.e., including inefficiencies), x_i is the share of land allotted to crop i and W_g is the total water allocation for the agent (m3/ha). Adopting modern irrigation systems increases eff_i and reduces w_i/eff_i.

Differences across the mathematical programming models considered in the ensemble stem from the form and calibration of the utility function (Graveline, 2016). Regarding the form, the utility functions used by mathematical programming models can be single- (the case of PMP) or multi-attribute (the case of LP and PMAUP). Single-attribute utility functions use expected profit as the sole utility-relevant attribute; while multi-attribute utility functions typically explore the relevance of expected profit, risk aversion, and management complexity aversion. A comprehensive description and mathematical formulation of the attributes explored in the ensemble (namely, expected profit, risk aversion and management complexity aversion), as well as the related data inputs, is available in Annex III in the Supporting Information S1. Utility functions can also adopt different functional forms across mathematical programming models, typically Cobb-Douglas (PMAUP), additive (LP) and quadratic (PMP).

Regarding the calibration, each mathematical programming model used in the ensemble (PMP, LP, PMAUP) has a unique calibration method, which are discussed in Annex IV in the Supporting Information S1. The calibration results for the five irrigation districts/agents in the Reno River LRIB using the five models above are presented in Annex V in the Supporting Information S1.

3.2. Exploratory Modeling and Scenarios

Exploratory modeling is a technique that uses computational experiments to study the behavior of complex systems over a set of plausible scenarios given a priori knowledge (Kwakkel & Pruyt, 2013). Exploratory modeling has been used to study structural transformations under uncertainty, and to inform the design of robust adaptation strategies (Bankes et al., 2013; Marchau et al., 2019). In this paper, exploratory modeling is used to create a set of plausible scenarios whose outcomes are subsequently tested, for each adaptation strategy (conservationist v. autonomous adaptation strategy), using the multi-model ensemble of mathematical programming models presented above. The following sets of scenarios are considered: (a) climate change, (b) irrigation modernization, and (c) environmental valuation scenarios.

Climate change scenarios. Climate change scenarios are based on the hydrologic projections for the Po River Basin in the Italian Climate Change Adaptation Plan (MITE, 2018), which are summarized in Annex VI in the Supporting Information S1. The Italian Climate Change Adaptation Plan foresees a reduction in runoff for the Po River Basin that ranges between 30% (RCP4.5) and 45% (RCP8.5) by 2,080, which will be coupled with an increase in upstream demand of up to 25% due to irrigation expansion. This will lead to increased agricultural water deficit, particularly downstream (up to 20%–40% reduction in agricultural water allocation). A total of 45 climate change scenarios were simulated using mathematical programming methods (agricultural water allocation reduction in Equation 6 from 0% to 45% at discrete intervals of 1%).

Irrigation modernization scenarios. The Reno River LRIB has designed a plan to implement canal lining and increase the average technical efficiency of transportation systems from 50% to 85%. The cost of canal lining for irrigators, based on available budget estimates, interviews with local experts (which were asked to account for overbudgeting in their responses) and subsidies (the modernization of collective irrigation infrastructures such as canals is eligible to receive direct payments from the Common Agricultural Policy (CAP) of 40%–90% of the investment cost, on the basis of a supposedly higher environmental performance (Official Journal of the European Union, 2013)), is estimated at 90,000–120,000 EUR/km (9,000–72,000 EUR/km with subsidies); which applying the standard amortization period (50 years) and interest rate (2%; Nomisma, 2019a) yields an annuity of 2,865–3,820 EUR/km (286.5–2,291 EUR/km with subsidy). This value is then multiplied by the total length of the canals in the lowlands, divided by the total number of irrigated hectares in the lowlands, and multiplied by the number of irrigated hectares in each irrigation district to obtain the irrigation modernization cost for each economic agent/irrigation district. Irrigation modernization costs are then charged to economic agents through a flat rate (EUR/ha). Local experts advised against using a volumetric charge (i.e., EUR/m^3) because metering devices are still unavailable for most irrigators in the area. A total of 112 irrigation modernization scenarios were simulated using mathematical programming methods (irrigation modernization costs from 9,000 to 120,000 EUR/km, at intervals of 1,000 EUR/km).

Environmental valuation scenarios. The two sets of scenarios above are used to estimate the utility perceived by irrigators under the two alternative strategies considered (conservationist v. autonomous adaptation), and the minimum compensation irrigators would be willing to accept to sustain the PWS scheme. This information is subsequently compared with the economic value provided by these ecosystems, to assess the economic performance of the PWS scheme.

The literature on ecosystem services does not prescribe a single technique to measure their economic value, and several methods can be used to this end (TEEB, 2015). With sufficient time and resources, original environmental valuation studies (such as contingent valuation or contingent ranking) are typically preferred (Arrow et al., 1993). However, original environmental valuation studies demand large research teams, and necessitate careful study design and data analysis before methods and results can be validated. Alternatively, benefit transfer methods can be used to approximate the economic value of ecosystem services through estimates obtained by other studies performed elsewhere. The benefit transfer approach, which is adopted here, transfers an "estimate from another study/studies to a different context", usually by multiplying the mean economic value for a person/family of the ecosystem service(s) X in location A by the population/number of families in location B, so to obtain the value of the ecosystem service(s) in B (Rosenberger & Loomis, 2003). Use of benefit transfer has the additional advantage of generating multiple plausible environmental valuation estimates for ecosystem services (one per study in the sample), instead of one point prediction as original valuation studies would do. This can be used to create multiple environmental valuation scenarios that more thoroughly sample scenario uncertainty.

Annex VII in the Supporting Information S1 presents the outcome of a literature survey from the Environmental Values Reference Inventory (www.evri.ca) on the economic benefits of the ecosystem services generated by (irrigated) agriculture. The relevant studies for our research were screened in three stages: (a) a review of the gray and academic literature concerned with the measurement of the total economic value of the ecosystem services provided by water-dependent ecosystems was performed, which led to 323 studies; (b) of this list, those studies that focused on at least 4 of the ecosystem services of relevance for the Reno River LRIB (see Annex I in the Supporting Information S1) were selected, which led to 47 studies; (c) the list was further reduced to account only for the most recent studies (last 15 years, 2007–2021), studies estimating annuity values (instead of lump sum values, to avoid discount rate uncertainties) and studies providing pecuniary values (i.e., qualitative studies were excluded), which led to 9 studies. Estimates in the original studies are reported in Annex VII in the Supporting Information S1 in current year's values in foreign currency, either per person, family or unit of surface, and were converted to 2,020 values using exchange rate and GDP deflator data from World Bank (2020). This resulted in an annuity value of ecosystem services in the Reno River LRIB that ranges between 57 and 372.4 EUR/ha/year, with a median of 126 EUR/ha/year.

3.3. Managing Uncertainty Through Robustness

Arguably, model selection techniques could be used to choose among candidates the model that performs better, for example, through minimization of calibration errors (see Annex V in the Supporting Information S1), instead of relying on an ensemble. Nonetheless, assessing model performance is controversial and goes beyond a straightforward comparison of calibration errors. Notably, models in our ensemble are designed as a substitute for direct experimentation, which means that we cannot evaluate the predictive performance of the models within the ensemble, a critical step in model selection (Konishi & Kitagawa, 2008). It may occur that a model with a relatively low calibration error performs poorly against non-observed data as compared to alternatives (poor predictive

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performance; Pindyck, 2015). Moreover, calibration errors are not directly comparable among different models, since modeling errors are independent (Cloke et al., 2013). Alternatively, multi-model ensemble modeling can be used to generate a probability distribution function that combines all models to generate a point prediction that avoids model selection bias. Yet, this is challenging due to the subjectivity involved in defining prior assumptions about the distribution and the accuracy and weight attributable to each model (Tebaldi & Knutti, 2007). Besides, a populated ensemble including several models is necessary to infer an accurate probability distribution function, and this requires a large amount of resources (computational, personnel, etc.) that may not be available. A similar argument could be made for consolidative v. multi-scenario analysis, since scenarios are typically the result of model predictions (e.g., climate models, environmental valuation models).

Therefore, rather than selecting those models/scenarios that better predict or using a weighting approach, which may artificially reduce uncertainty (Hino & Hall, 2017), this work considers multiple scenarios/models and an un-weighted approach. The result is a database that offers information on uncertainty regarding model design through the ensemble spread, as well as on scenario uncertainty through exploratory modeling. It has been argued that when "probabilistic information is not considered, each potential vulnerability is equally important on the overall robustness, which can also be interpreted as an implicitly equal weighting" (Taner et al., 2019). Yet, as noted above, in our case we cannot claim that each scenario/model has an equal weight, because these weights are essentially unknown. In this context, robustness is advised in decision making, so to minimize potential regret.

A robust decision can be informed through heuristic (i.e., inductive reasoning, building on the expertise of decision makers) and/or mechanistic methods. Since the design of the PWS in the Reno River LRIB is still in an exploratory stage, and a formal and structured discussion of the results that allows to articulate heuristic-based robust decision methods is ahead in time, this research adopts two widely used mechanistic robust decision algorithms to inform PWS performance: Minimization of maximum loss (Minimax) and Minimization of Maximum regret (MinMax regret; Aissi et al., 2009), two conservative decision making approaches that choose the strategy that minimizes the potential loss (Minmax) and regret (Minmax regret) under the models and scenarios considered.

4. Results

4.1. Simulation Results

The methods proposed above are used to assess the performance of the conservationist strategy (i.e., no irrigation modernization and reallocation of water toward the environment) v. autonomous adaptation strategy (i.e., irrigation modernization and reallocation of water toward agriculture) under multiple scenarios and models. The upshot is a database of simulations informing on the expected irrigators' choices and related economic performance of the two strategies (including profit, employment, Gross Value Added and, most notably, utility). Figure 3 informs on the crop portfolio choices of irrigators under alternative climate change scenarios/reductions in water allocations, for the conservationist strategy (Figure 3a) and the autonomous adaptation strategy (Figure 3b), in each model of the ensemble. Note that since irrigation modernization costs/PWS are charged/paid through a flat rate on a per hectare basis, they do not alter the relative position in terms of utility return among alternative crops, and do not affect crop portfolio responses.

Irrigation modernization under the autonomous adaptation strategy increases the proportion of beneficial consumption per unit of water allocated for irrigation, which increases efficiency (eff_i) and allows irrigators to reduce the gross water needs of each crop i, or w_i/eff_i (see Equation 6), and thus negate/reduce the impacts of climate change and water scarcity on yields. As a result, under the autonomous adaptation strategy, the crop portfolio remains largely stable (although marginal changes are observed) under all scenarios and models, until water allocation is reduced by >30%. When water allocation is reduced by 30% or more, those irrigated crops with a lower utility return (mostly corn) are partially replaced by rainfed cereals (mostly wheat or barley, depending on the model). On the other hand, under the conservationist strategy where no irrigation modernization plan is implemented, the reduction of water allocation constrains water availability and leads to a substitution of relatively low return irrigated crops (corn, sugar beet and pasture) by rainfed wheat or barley (depending on the model) from the onset (>0% water allocation reduction). The surface of high return irrigated crops (vegetables, fruit

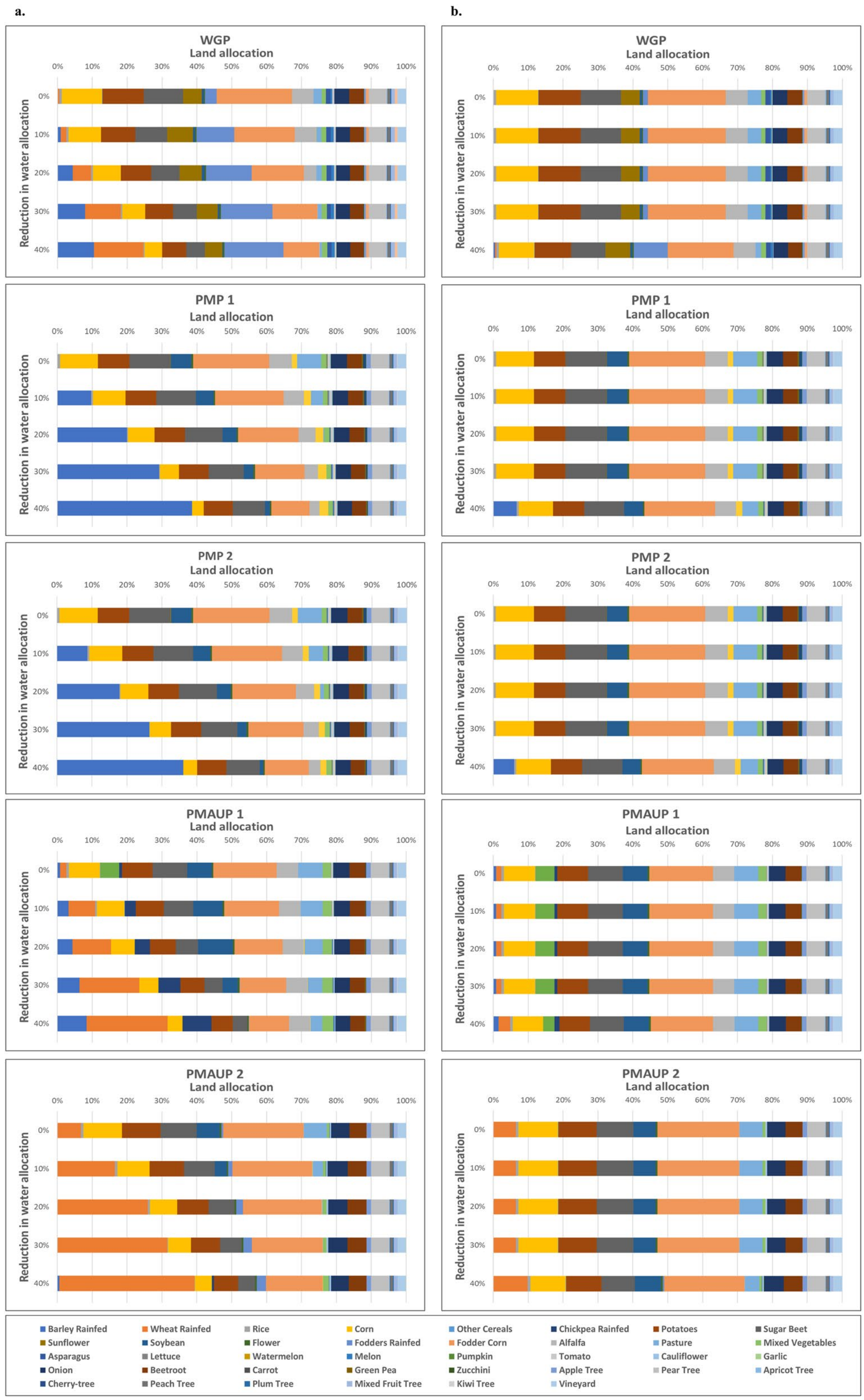

Figure 3. Crop portfolio choices: conservationist strategy (Figure 3a) v. autonomous adaptation strategy (Figure 3b).

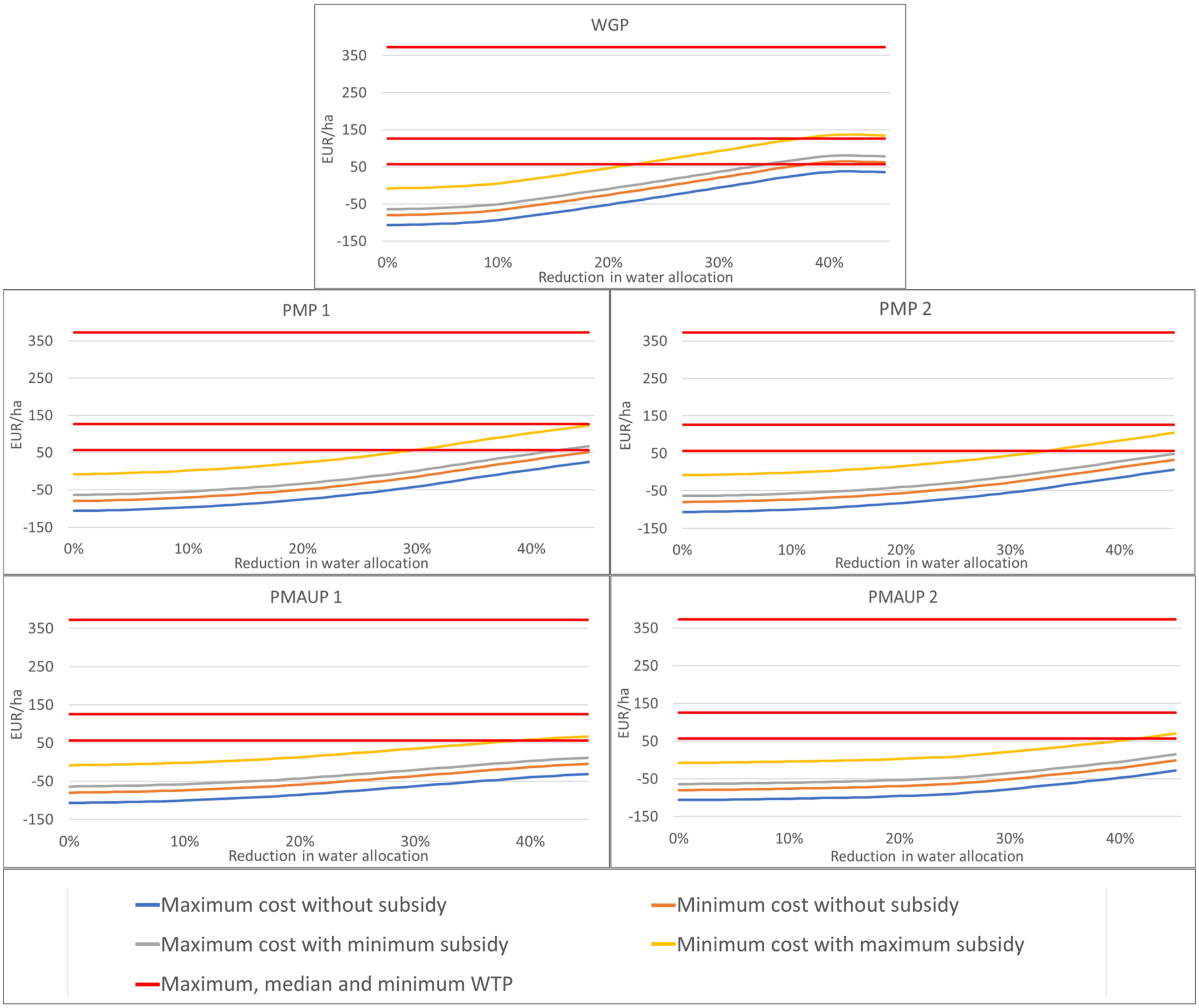

Figure 4. Willingness to accept v. Willingness to pay for PWS in the Reno LRIB for (a) each model of the ensemble and (b) all models of the ensemble.

trees) remains constant in the initial simulation steps (<20% water allocation reduction); and is progressively substituted by less water intensive irrigated and/or rainfed (wheat or barley) crops with a lower return when water allocation is reduced by >20%.

Figure 4 assesses the economic performance of the PWS scheme by comparing the willingness to accept (WTA) to the willingness to pay (WTP) for an hypothetical PWS in the Reno LRIB, under alternative combinations of scenarios and models. The WTP is obtained from a literature review using benefit transfer methods (see Annex VII in the Supporting Information S1). The WTA is obtained in two stages using mathematical programming methods. First, we use the utility function calibrated in Equations 1–6 to calculate the monetized foregone utility (through the compensating variation, CV) experienced by irrigators in every possible scenario g under each strategy s, as follows:

$$\mathrm{CV}_{g,s} = \frac{e\left(U_{g=0,s}, W_g\right)}{\mathrm{Surface}} \tag{7}$$

Where e is an expenditure function representing the minimum amount of money agents would need to attain the utility level they experience in the baseline scenario $g = 0$ $(U_{g=0})$, where there is neither climate change nor irrigation modernization, starting from an alternative scenario g.

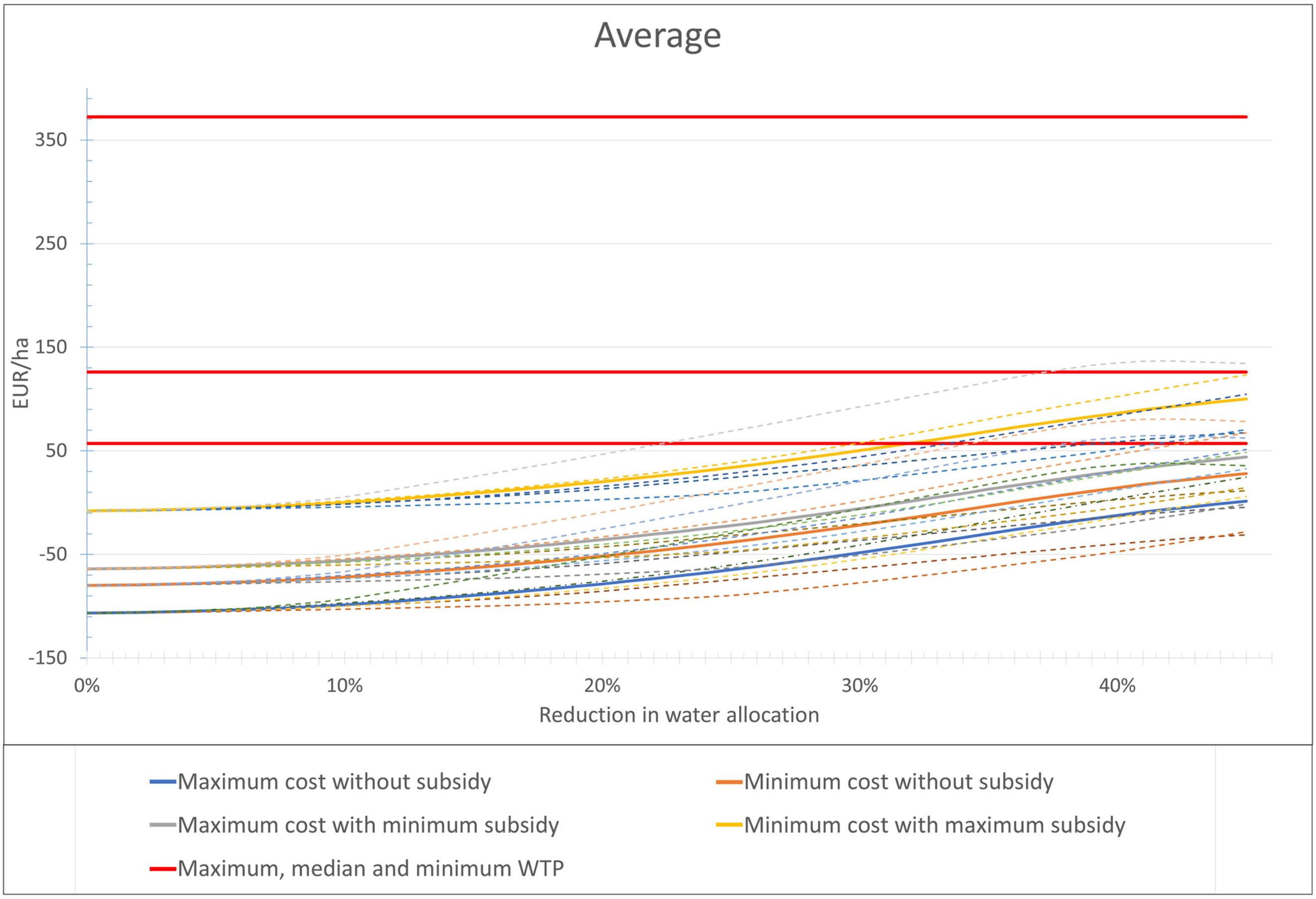

Figure 4. (Continued)

Second, by comparing the CV of the autonomous adaptation strategy ($s = A$) v. conservationist strategy ($s = C$), the WTA in every possible scenario g is obtained as follows:

$$\mathrm{WTA}_g = \mathrm{CV}_{g,C} - \mathrm{CV}_{g,A} \tag{8}$$

A positive WTA denotes a preference for the autonomous adaptation strategy; while a negative WTA indicates that irrigators would experience a higher utility loss from the adoption of irrigation modernization v. the conservationist strategy, and therefore the latter would be preferred even in the absence of pecuniary compensations to irrigators through PWS.

When reductions in water allocation are null or low (<9%), the WTA is negative for all models and scenarios considered. At this stage the costs of irrigation modernization exceed the benefits from additional water availability, which is mostly used to irrigate crops with a low utility return. This results in a lower utility and higher CV under the autonomous adaptation strategy relative to the conservationist strategy, and a negative WTA. As climate change strengthens the water allocation constraint and threatens the irrigation of crops with a higher return, utility (CV) under the autonomous adaptation strategy increases (decreases) relative to utility (CV) under the conservationist strategy, and the WTA for the PWS scheme increases. Beyond a water allocation reduction of 9%, the WTA starts exceeding a value of 0 EUR/ha for some models and scenarios—meaning that irrigators will demand a compensation for not adopting irrigation modernization and conserving the Reno LRIB green infrastructures and ecosystem services instead. Beyond a water allocation reduction of 22%, the WTA starts exceeding the lower threshold of the WTP for some models and scenarios, indicating that irrigators would prefer the autonomous adaptation to the conservationist strategy even in presence of a pecuniary compensation (minimum WTP) through PWS. Eventually, the median WTP is also surpassed, although this only happens for severe climate change

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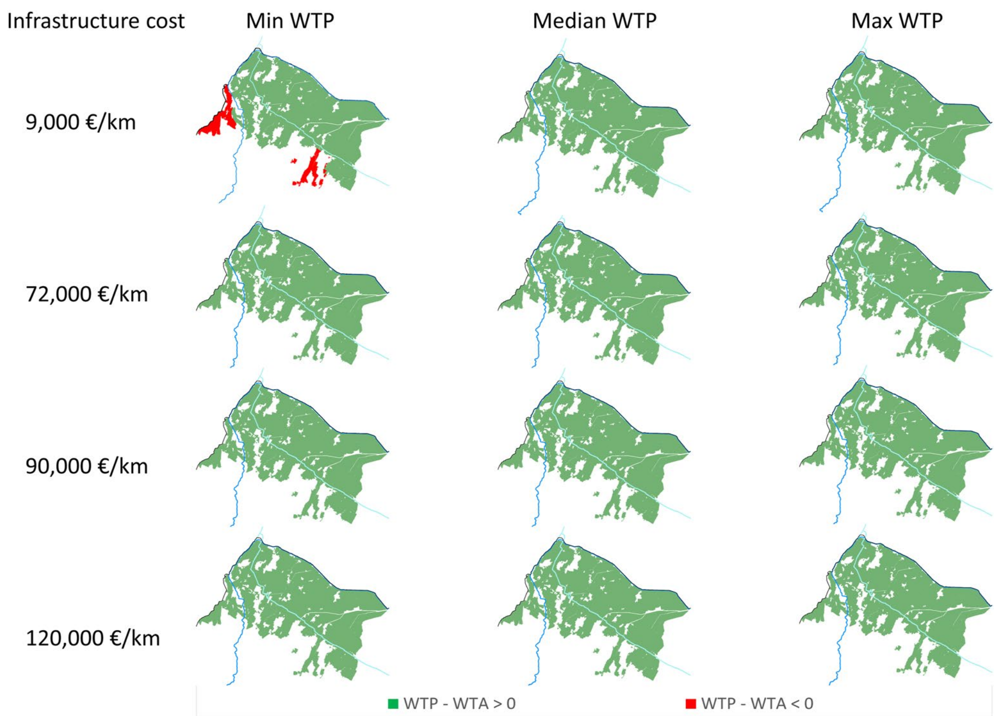

Figure 5. Economic performance (measured as WTP minus WTA, in EUR/ha) of PWS and spatial asymmetries. (a) 25% of water allocation reduction; (b) 40% of water allocation reduction.

scenarios (water allocation reduction >38%) and only in a few models and irrigation modernization scenarios. No combination of models and scenarios leads to a WTA that exceeds the maximum WTP.

Importantly, in the absence of irrigation modernization subsidies the WTA only becomes positive beyond a water allocation reduction of 26%; and exceeds the minimum WTP beyond a water allocation reduction of 35%. Neither the median nor the maximum WTP are surpassed by the WTA in those scenarios without irrigation modernization subsidies.

Figure 5 compares the WTP to the WTA for each agent/irrigation district under selected scenarios. This information is relevant to identify and redress potential asymmetries in the implementation of PWS (e.g., through direct payments to those who experience losses and/or water reallocations among farmers). The WTA is calculated here as a simple average of all the models in the ensemble—sometimes referred to as best estimate (IPCC, 2014).

Those irrigation districts with a larger share of high return crops (mostly vegetables) are more likely to experience losses from the adoption of PWS, especially under severe climate change scenarios. Under a water allocation reduction of 40%, the irrigation district C004 is better off adopting the autonomous adaptation strategy, provided infrastructures are subsidized and the minimum or median WTP is paid. The irrigation district C005 is better off adopting the autonomous adaptation strategy even if no subsidy toward the adoption of modern irrigation infrastructures is paid.

4.2. Robustness

Robustness is assessed using two mechanistic robust decision algorithms: Minimax and Minmax regret.

The performance indicator used in the case of the Minimax algorithm is the gain/loss experienced in each scenario as compared to the baseline scenario g = 0 (no climate change, no irrigation modernization). Under the autonomous adaptation strategy, the Minimax performance indicator equates $-\mathrm{CV}_{g,A}$; while under the conservationist strategy, the Minimax performance indicator equates WTP$-\mathrm{CV}_{g,C}$. The Minimax performance indicator is obtained for every model, scenario and strategy considered. Subsequently, for each of the strategies, the

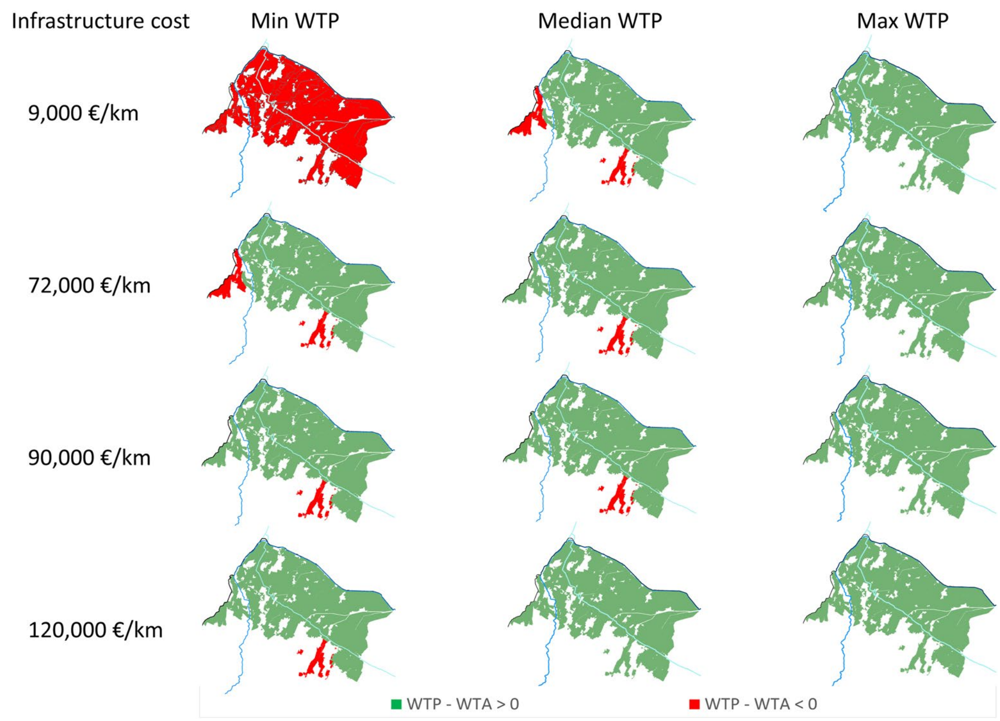

Figure 5. (Continued)

maximum loss is obtained across all models and scenarios. The strategy that minimizes the maximum potential loss (Minmax) is found to be more robust; which in our case is the conservationist strategy.

The performance indicator used in the case of the Minmax regret is obtained in two steps. First, the best performance indicator for each scenario among the Minmax performance indicators (see above) is obtained. Then, a regret indicator is obtained by subtracting actual gain/loss (again obtained as $-\mathrm{CV}_{g,A}$ for the autonomous adaptation strategy, and as $\mathrm{WTP}-\mathrm{CV}_{g,C}$ for the conservationist strategy) from the best performance indicator under a given scenario. The strategy that minimizes regret is then chosen; which is found to be, again, the conservationist strategy.

4.3. Discussion

Farmers are in charge of managing the land and are given the responsibility to protect it. An important part of this stewardship role involves the conservation of natural resources, such as water, and the environmental assets and ecosystem services that depend on them. Historically, landscape stewardship has included environmentally friendly adaptive strategies such as no-till, planting cover crops, collecting water runoff to reduce nutrient load to water bodies, integrating crop and pasture rotations, and others. However, without adequate rules and incentives, adaptation strategies may as well be unsustainable (e.g., water theft, aquifer overdraft).

We show that as climate and water resource allocations change, farmers may decide to deploy modern irrigation systems that increase agricultural water consumption to mitigate/negate production losses, while reducing water availability for other uses—including valuable ecosystems and their services. This autonomous adaptation strategy is being encouraged by ill-designed incentive schemes—most notably the subsidization of modern irrigation systems (Perry & Steduto, 2017). In our study in the Reno LRIB we find that, under mild to moderate climate change scenarios (water allocation reduction <26%), removing infrastructure subsidies is sufficient to prevent maladaptation through modern irrigation technologies that deplete environmental water allocations. This critical result aligns well with research at the global (Pérez-Blanco et al., 2020) and private irrigator level (Adamson & Loch, 2021), which has shown that when subsidies are removed, expected water savings are often insufficient on their own to motivate private irrigation modernization investments.

This is likely to change under moderate to severe climate change scenarios, when water scarcity will affect increasingly valuable crops and new schemes of incentives such as PWS may be necessary to prevent autonomous (mal)adaptation. In our study in the Reno LRIB we find that, if no infrastructure subsidies are applied, PWS would have a cost (measured through the WTA) that is below the minimum WTP for irrigation-dependent ecosystem services until a water allocation reduction of >35%; and below the median and maximum WTP for all scenarios considered. In the presence of subsidies, the cost of PWS can exceed the minimum WTP earlier (water allocation reduction >22%), while the median WTP can be also surpassed, albeit only at severe climate change scenarios (water allocation reduction >38%). This suggests a satisfactory economic performance of PWS for most scenarios and models considered. Applying automatic robust decision-making methods, the PWS is found to be a more robust strategy than the adoption of modern irrigation systems.

Our findings have relevant implications for water policy design in the EU and in other areas where modern irrigation systems are being adopted, often with the support of public subsidization programmes (Pérez-Blanco et al., 2020). In the EU, the Common Agricultural Policy (CAP) subsidizes up to 90% of the investment cost of modern irrigation systems, on the basis of a supposedly higher environmental performance of these technologies (Official Journal of the European Union, 2013). According to CAP reasoning, if an irrigation system *X* with 50% technical efficiency is substituted with an irrigation system Y with 75% technical efficiency, the original water needs can be satisfied with a fraction (50/75) of the original water applied (e.g., 100 v. 66.67 units of water), and therefore 33.33 units of water will be saved. This confuses water applied with water consumed and assumes that economic agents will behave the same way (planting exactly the same crop portfolio) after the modernization, as before—two widespread but incorrect assumptions among policymakers. In reality, unless water use is curtailed following the adoption of modern irrigation systems, we should expect the farmer to increase consumption, reduce return flows, and limit water availability for third party uses, including green infrastructures, so to increase farm income (Grafton et al., 2018; Pérez-Blanco et al., 2020; Perry & Steduto, 2017). Therefore, subsidies to modern irrigation technologies are not only ineffective to save water—they can also exacerbate water scarcity by increasing the consumed fraction of water applied.

In light of the overwhelming available scientific evidence showing that modern irrigation systems increase consumption and aggravate scarcity, why do policymakers continue to subsidize them to save water? First, despite the growing consensus among scientists that modern irrigation systems increase water consumption (Grafton et al., 2018), there is a widespread belief among non-experts that modern irrigation systems will save water. Once a belief has been established, individuals are more likely to accept (or even build) arguments that conform to that belief (Nickerson, 1998; Shermer, 2011), even when more recent information discredits it (Johnson & Seifert, 1994). This makes very challenging to debias and debunk the belief that modern irrigation systems will save water (Lewandowsky et al., 2012), particularly among policymakers that are not familiar with the behavioral drivers explaining farmer's responses to modern irrigation systems. Second, those who benefit from modern irrigation systems (e.g., adopting farmers, equipment suppliers) exert political pressure and other lobbying efforts to obtain public subsidies that develop new resources and increase farm income—often at a marginal cost that exceeds marginal value. This is visible in our study in the Reno River LRIB, where in the absence of public subsidies, the autonomous adaptation strategy is preferred to the conservationist strategy only under severe climate change scenarios (water allocation reduction of >35%); while with subsidies, modern irrigation technologies are preferred to PWS following a water allocation reduction of >22% (moderate climate change).

Since modern irrigation systems worsen rather than alleviate water scarcity, it is necessary that policymakers abandon the preconceived idea that these technologies will almost always save water and start adopting new frameworks that contribute to align individual farmer choices with collectively agreed policy goals, such as alleviating water scarcity while mitigating and potentially reverting income losses under climate change (i.e., sustainable growth). A prerequisite to achieve this goal is to conduct debiasing and debunking exercises among policymakers to put to rest the belief that modern irrigation systems will almost always save water (examples of debunking and debiasing exercises are available in Cook et al., 2018; Lewandowsky et al., 2012; Linden et al., 2017). Additionally, achieving sustainable growth under increasing water scarcity necessitates sensible water reallocations that conform to basic economic principles, including:

- The theory of economic policy, which argues that in order to meet a number of goals, an equal number of instruments are necessary (Tinbergen, 1952). Thus, if the objective is to save water (objective 1) while enhancing/protecting farm income (objective 2), two instruments will be necessary (e.g., decoupled subsidies to farmers to enhance/protect income and quotas to save water)

- The Assignment Principle, which argues that each instrument should be assigned to the objective to which it is best suited, and that this instrument should not be used to pursue another objective (Mundell, 1962). The Assignement Principle complements Tinbergen's (1952) work and can be interpreted as a warning against water panaceas or "win-win" solutions, where a single instrument is adopted to pursue two (often conflicting) objectives
- A framework for the effective design of interventions, where the objectives and the instruments set by policymakers do not directly affect behavioral responses (in our case, the decision of whether to adopt or not modern irrigation systems; Ciriacy-Wantrup & Bishop, 1975). For example, instead of subsidizing modern irrigation systems (which directly affects behavioral responses by promoting the adoption of modern irrigation technologies by farmers), policymakers should set the objectives to be met (e.g., ecological flows) and the instruments to achieve them (e.g., quotas, pricing), and let farmers respond to these new conditions through changes in inputs and technology (e.g., reduced water use, modern irrigation technologies)

These basic economic principles suggest that the failure of modern irrigation systems to save water is the consequence of flawed policy design. Policymakers promoting subsidies to modern irrigation systems "talk" about saving water but "dream" about increasing production (Connell, 2007), and thus violate the Tinbergen Principle (two objectives, one instrument). Moreover, scientific evidence shows that there are "much more cost-effective" alternatives to save water than modern irrigation systems (Qureshi et al., 2011), such as quotas or pricing, meaning modern irrigation systems also violate the Assignement Principle. Finally, coupled subsidies such as subsidies to modern irrigation systems directly affect the operational decisions by farmers, instead of setting objectives and instruments farmers have to comply/deal with. These basic economic principles further underpin the findings obtained using our quantitative framework, namely, that the conservationist strategy (sometimes complemented with PWS) has a superior economic performance than the (subsidized) autonomous adaptation strategy where modern irrigation systems are adopted under most models and scenarios, and is also more robust.

Does all the above mean that modern irrigation systems are always ineffective toward saving water? No. Modern irrigation systems can yield savings while protecting and/or enhancing agricultural income if they are complemented with effective water saving policies (such as quotas or pricing) that strengthen water allocations to farmers, so to ensure that any additional agricultural consumption following the adoption of modern irrigation systems is equal or lower than the foregone non-beneficial consumption and non-beneficial return flows (see Figure 1). Under conventional return flow regimes where return flows are beneficial, the water savings and/or additional farm income that can be achieved this way are rather marginal, and typically do not justify investments in modern irrigation systems (Adamson & Loch, 2021). This is not the case under (infrequent) escape flow regimes where return flows are non-beneficial and can be appropriated by farmers at no economic cost (i.e., higher farm income without reducing water availability to third party users; Huffaker, 2008).

5. Conclusions

This paper develops a multi-model and multi-scenario method to assess irrigators' responses to, and the economic performance of, pecuniary compensations designed to sustain irrigation-dependent ecosystem services through PWS. We find that, under most models and scenarios, the conservationist strategy (sometimes complemented with PWS) has a superior economic performance than the autonomous adaptation strategy where modern irrigation systems are adopted. The conservationist strategy is also found more robust than the autonomous strategy.

We envision several ways in which our model and research could be improved. First, the ensemble of mathematical programming models used in this paper could be expanded by including other models available in the literature, so to more thoroughly sample uncertainty. Second, a sensitivity analysis is also warranted to further sample uncertainty. This could be done exploring additional attributes in the multi-attribute models, PMAUP 1 and PMAUP 2; or considering alternative crops and adaptation strategies in the portfolio, for example, allowing for continuous yield functions and deficit irrigation instead of fully irrigated v. rainfed crops (Graveline & Mérel, 2014). Third, our ensemble focuses on the microeconomic aspect of a human system, which is one of the two components of complex human-water systems. Future research should study the interconnection of our human system multi-model ensemble with existing multi-model ensembles that represent the water system (see e.g., Cloke et al., 2013). This would make possible to model the two-way feedbacks between human and water systems and their consequences (e.g., how the value of ecosystem services increases/decreases during periods of scarcity/abundance; Sivapalan et al., 2014), and sample the cascading uncertainties across them. Fourth, the representation of the human system could be also enhanced by modeling the interconnection between the micro level explored in this paper and the macro level, for example, through price feedbacks, which can be modeled endogenously in macroeconomic models (Parrado et al., 2020). Finally, efforts to produce more comprehensive and robust models should be complemented with debiasing and debunking exercises—a prerequisite to overcome locked-in policy failures that subsidize ineffective water saving policies (Cook et al., 2018; Lewandowsky et al., 2012).

Data Availability Statement

The data used for the calibration of the ensemble of mathematical programming models and for the benefit transfer exercise is available free of charge at the online supplementary material and in an online repository at the following link: https://doi.org/10.5281/zenodo.5578968.

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References

Adamson, D., & Loch, A. (2021). Incorporating uncertainty in the economic evaluation of capital investments for water use efficiency improvements. *Land Economics*, *97*(3), 655–671. 10.3368/wple.97.3.100119-0143R

Aissi, H., Bazgan, C., & Vanderpooten, D. (2009). Min–max and min–max regret versions of combinatorial optimization problems: A survey. *European Journal of Operational Research*, *197*, 427–438. https://doi.org/10.1016/j.ejor.2008.09.012

Anderies, J. M. (2015). Managing variance: Key policy challenges for the Anthropocene. *Proceedings of the National Academy of Sciences*, 112, 14402–14403. https://doi.org/10.1073/pnas.1519071112

Arrow, K., Solow, R., Portney, P., Leamer, E., Radner, R., & Schuman, H. (1993). *Report of the NOAA panel on contingent valuation (report)*. National Oceanic and Atmospheric Administration.

Asbjornsen, H., Mayer, A. S., Jones, K. W., Selfa, T., Saenz, L., Kolka, R. K., & Halvorsen, K. E. (2015). Assessing impacts of payments for watershed services on sustainability in coupled human and natural systems. *BioScience*, *65*, 579–591. https://doi.org/10.1093/biosci/biv051

Bankes, S., Walker, W. E., & Kwakkel, J. H. (2013). Exploratory modeling and analysis. In S. I. Gass & M. C. Fu (Eds.), *Encyclopedia of operations research and management science* (pp. 532–537). Springer US. https://doi.org/10.1007/978-1-4419-1153-7_314

Bennett, G., & Franziksa, R., (2016). *Alliances for green infrastructure: State of watershed investment 2016*, PreventionWeb.net. Ecosystem Marketplace.

Bhaduri, A., Pérez-Blanco, C. D., Rey, D., Iftekhar, M. S., Kaushik, A., Escriva-Bou, A., et al. (2021). Economics of water security. In *Handbook of water resources management: Discourses, concepts and examples, handbook of water resouces*. Springer International Publishing.

Bremer, L. L., Auerbach, D. A., Goldstein, J. H., Vogl, A. L., Shemie, D., Kroeger, T., et al. (2016). One size does not fit all: Natural infrastructure investments within the Latin American Water Funds Partnership. *Ecosystem Services*, *17*, 217–236. https://doi.org/10.1016/j.ecoser.2015.12.006

Ciriacy-Wantrup, S., & Bishop, R. (1975). Common property as a concept in natural resources policy. *Natural Resources Journal*, *15*, 713–727.

Cloke, H. L., Pappenberger, F., van Andel, S. J., Schaake, J., Thielen, J., & Ramos, M. H. (2013). Hydrological ensemble prediction systems. *Hydrological Processes*, *27*, 1–4. https://doi.org/10.1002/hyp.9679

Connell, D. (2007). *Water politics in the Murray-darling basin*. Federation Press.

Cook, J., Ellerton, P., & Kinkead, D. (2018). Deconstructing climate misinformation to identify reasoning errors. *Environmental Research Letters*, *13*, 024018. https://doi.org/10.1088/1748-9326/aaa49f

Grafton, R. Q., Williams, J., Perry, C. J., Molle, F., Ringler, C., Steduto, P., et al. (2018). The paradox of irrigation efficiency. *Science*, *361*, 748–750. https://doi.org/10.1126/science.aat9314

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., & Mérel, P. (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

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. *Journal of Hydrology*, *375*, 627–643. https://doi.org/10.1016/j.jhydrol.2009.06.037

Hino, M., & Hall, J. W. (2017). Real options analysis of adaptation to changing flood risk: Structural and nonstructural measures. *ASCE-ASME Journal of Risk and Uncertainty in Engineering Systems, Part A: Civil Engineering*, *3*, 04017005. https://doi.org/10.1061/AJRUA6.0000905

Huffaker, R. (2008). Conservation potential of agricultural water conservation subsidies. *Water Resources Research*, *44*, W00E01. https://doi.org/10.1029/2007WR006183

IPCC. (2014). *IPCC fifth assessment report (AR5) (No. WGII)*. Intergovernmental Panel on Climate Change.

Johnson, H. M., & Seifert, C. M. (1994). Sources of the continued influence effect: When misinformation in memory affects later inferences. *Journal of Experimental Psychology: Learning, Memory, and Cognition*, *20*, 1420–1436. https://doi.org/10.1037/0278-7393.20.6.1420

Jones, K. W., Cannon, J. B., Saavedra, F. A., Kampf, S. K., Addington, R. N., Cheng, A. S., et al. (2017). Return on investment from fuel treatments to reduce severe wildfire and erosion in a watershed investment program in Colorado. *Journal of Environmental Management*, *198*, 66–77. https://doi.org/10.1016/j.jenvman.2017.05.023

Konishi, S., & Kitagawa, G. (2008). *Information criteria and statistical modeling*. In *Springer Series in Statistics*. Springer-Verlag.

Kwakkel, J. H., & Pruyt, E. (2013). Exploratory modeling and analysis, an approach for model-based foresight under deep uncertainty. Technological forecasting and social change. *Future-Oriented Technology Analysis*, *80*, 419–431. https://doi.org/10.1016/j.techfore.2012.10.005

Lewandowsky, S., Ecker, U. K. H., Seifert, C. M., Schwarz, N., & Cook, J. (2012). Misinformation and its correction: Continued influence and successful debiasing. *Psychological Science in the Public Interest*, *13*, 106–131. https://doi.org/10.1177/1529100612451018

Linden, S. van der, Maibach, E., Cook, J., Leiserowitz, A., & Lewandowsky, S. (2017). Inoculating against misinformation. *Science*, *358*, 1141–1142. https://doi.org/10.1126/science.aar4533

Lucas, R. E. (1976). Econometric policy evaluation: A critique. *Carnegie-Rochester Conference Series on Public Policy*, *1*, 19–46. https://doi.org/10.1016/S0167-2231(76)80003-6

Marchau, V. A. W. J., Walker, W. E., Bloemen, P., & Popper, S. W. (2019). *Decision making under deep uncertainty: From theory to practice* (2019th ed.). Springer.

MITE. (2018). *Piano Nazionale di Adattamento ai Cambiamenti Climatici*. Ministero della Transizione Ecologica. https://www.mite.gov.it/pagina/piano-nazionale-di-adattamento-ai-cambiamenti-climatici

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

Nickerson, R. S. (1998). Confirmation bias: A ubiquitous phenomenon in many guises. *Review of General Psychology*, *2*, 175–220. https://doi.org/10.1037/1089-2680.2.2.175

Nomisma. (2019a). *Pagamento dei Servizi Ecosistemici generati dalle attività irrigue (Report No. 2)*. Nomisma.

Nomisma. (2019b). *Valutazione economica degli effetti ecosistemici generati dalle attività irrigue (Report)*. Nomisma.

Official Journal of the European Union (2013). *Regulation (EU) No 1305/2013 of the European parliament and of the council of 17 december 2013 on support for rural development by the European agricultural fund for rural development (EAFRD) and repealing council regulation (EC) No 1698/2005*. Official Journal of the European Union - Regulation.

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. *The Science of the Total Environment*, *742*, 140526. https://doi.org/10.1016/j.scitotenv.2020.140526

Pérez-Blanco, C. D., González-López, H., & Hrast-Essenfelder, A. (2021). Beyond piecewise methods: Modular integrated hydroeconomic modeling to assess the impacts of adaptation policies in irrigated agriculture. *Environmental Modelling & Software*, *136*, 104943. https://doi.org/10.1016/j.envsoft.2020.104943

Pérez-Blanco, C. D., Hrast-Essenfelder, A., & Perry, C. (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

Perry, C., & Steduto, P. (2017). *Does improved irrigation technology save water? A review of the evidence (discussion paper on irrigation and sustainable water resources management in the near East and north Africa), regional initiative on water scarcity for the near East and north Africa*. FAO.

Pindyck, R. S. (2015). *The use and Misuse of models for climate policy (working paper No. 21097)*. National Bureau of Economic Research.

Qureshi, M. E., Grafton, R. Q., Kirby, M., & Hanjra, M. A. (2011). Understanding irrigation water use efficiency at different scales for better policy reform: A case study of the murray-darling basin, Australia. *Water Policy*, *13*, 1–17. https://doi.org/10.2166/wp.2010.063

Rode, J., Gómez-Baggethun, E., & Krause, T. (2015). Motivation crowding by economic incentives in conservation policy: A review of the empirical evidence. *Ecological Economics*, *117*, 270–282. https://doi.org/10.1016/j.ecolecon.2014.11.019

Rosenberger, R. S., & Loomis, J. B. (2003). Benefit transfer. In P. Champ, K. Boyle, & T. Brown (Eds.), *A primer on nonmarket valuation*. Springer.

Sapino, F., Pérez-Blanco, C. D., Gutiérrez-Martín, C., & Frontuto, V. (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

Shermer, M. (2011). *The believing brain: From ghosts and gods to politics and conspiracies---how we construct beliefs and reinforce them as truths*.

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

Taner, M. Ü., Ray, P., & Brown, C. (2019). Incorporating multidimensional probabilistic information into robustness-based water systems planning. *Water Resources Research*, *55*, 3659–3679. https://doi.org/10.1029/2018WR022909

Tebaldi, C., & Knutti, R. (2007). The use of the multi-model ensemble in probabilistic climate projections. *Philosophical Transactions of the Royal Society A: Mathematical, Physical & Engineering Sciences*, *365*, 2053–2075. https://doi.org/10.1098/rsta.2007.2076

TEEB. (2015). *TEEB for agriculture & food interim report (TEEB document)*. United Nations Environment Programme.

Tinbergen, J. (1952). *On the theory of economic policy*. North-Holland Pub. Co.

Chapter 4. Multi-system ensemble

"The man of system, ..., seems to imagine that he can arrange the different members of a great society with as much ease as the hand arranges the different pieces upon a chess-board. He does not consider that the pieces upon the chess-board have no other principle of motion besides that which the hand impresses upon them; but that, in the great chess-board of human society, every single piece has a principle of motion of its own, altogether different from that which the legislature might chuse [sic] to impress upon it. If those two principles coincide and act in the same direction, the game of human society will go on easily and harmoniously, and is very likely to be happy and successful. If they are opposite or different, the game will go on miserably, and the society must be at all times in the highest degree of disorder."

Adam Smith, Theory of Moral Sentiments, 1759.

4.1. Introduction

In the previous chapter, we presented the multi-model ensemble of MPMs used to advise robust decisions. Even if this methodology reduces the uncertainty and allows us to advice robust decision, it still fails to address a critical issue of conventional modeling tools, i.e., it does not consider the feedback between the natural and the socioeconomic systems. In fact, when farmers allocate water differently, the natural system reacts as well, so affecting water availability in other parts of the basin. Following the socio-hydrology literature, which aims to "observe, understand and predict the co-evolution of coupled human-water systems" (Sivapalan et al., 2012), we develop a multi-system ensemble that simultaneously considers both the hydrologic and socioeconomic systems. Our approach follows the TAMAL approach and is therefore based on previous models that are combined in a modular hierarchy. The modularity provides the foreseen outcome of simulating the co-evolution of the two systems, but in an easier-to-understand way (than, e.g., presenting a completely new model). Moreover, it also allows easier changes and updates to adapt the modeling framework to other study cases.

In the following sections, we present two applications of our multi-model and multi-system ensemble: in 4.2 we calculate the resource cost of agricultural water, while in 4.3 we propose an innovative water bank system used to enhance environmental and economic performance.

4.2. A hydro-economic model to calculate the resource costs of agricultural water use and the economic and environmental impacts of their recovery

4.2.1. Resumen

Hemos estimado los costes del recurso del uso del agua agrícola y hemos simulado los impactos ambientales y económicos de su recuperación. Con este fin, desarrollamos un método inspirado en la socio-hidrología modular, dinámico y basado en protocolos que interconecta los módulos económico e hidrológico a través de protocolos bidireccionales. El módulo hidrológico se lleva a cabo con el modelo AQUATOOL, el *Decision Support System* utilizado por las confederaciones hidrográficas españolas; mientras que el módulo económico se lleva a cabo con un conjunto de cuatro Modelos de Programación Matemática (MPM) que capturan el comportamiento de los agricultores y sus reacciones. Esto nos permite evaluar la incertidumbre y estimar un rango de costes del recurso y los impactos ambientales y económicos de su recuperación, en lugar de una estimación puntual. El método se ilustra con una aplicación a la cuenca hidrográfica del Órbigo, una subcuenca de la cuenca del río Duero en España. Nuestros resultados sugieren costes del recurso significativos (que causarían un aumento de entre el 34% y el 62% en los cargos existentes, dependiendo del modelo) con impactos significativos sobre los ingresos (reducción de entre el 2% y el 27%) y el medio ambiente (el ahorro de agua oscila entre el 6% y el 69%), mientras que el impacto sobre la recaudación fiscal es ambiguo, pero potencialmente significativo (entre -2,3 millones de euros/año y 5 millones de euros/año).

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

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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 (UN 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.

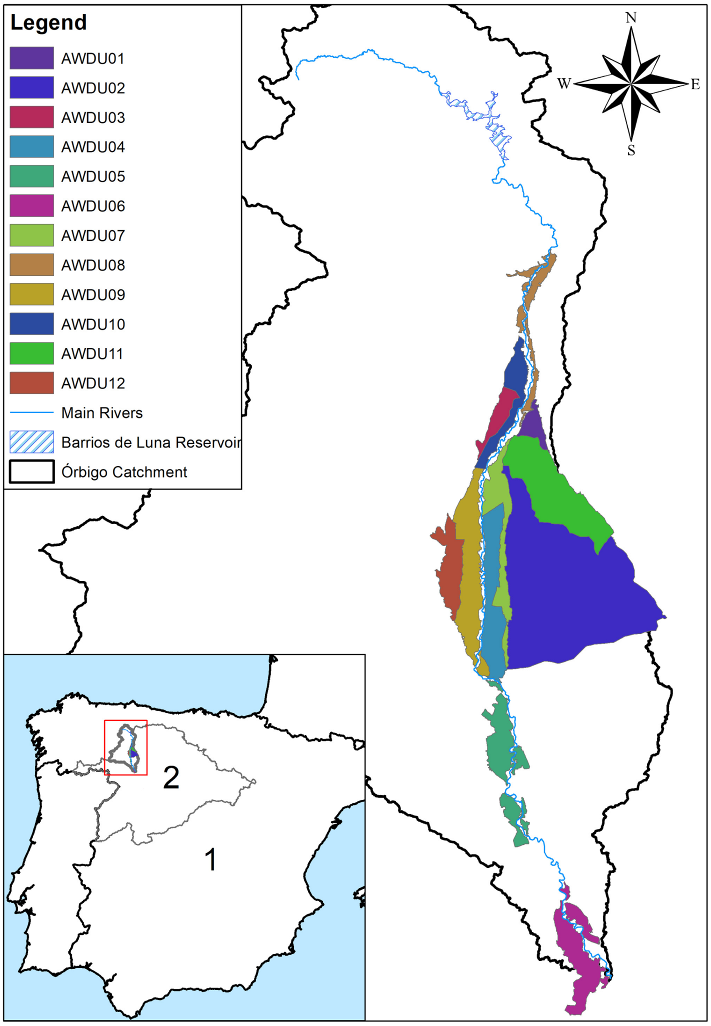

Figure 1. Study Area: The Órbigo Catchment

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

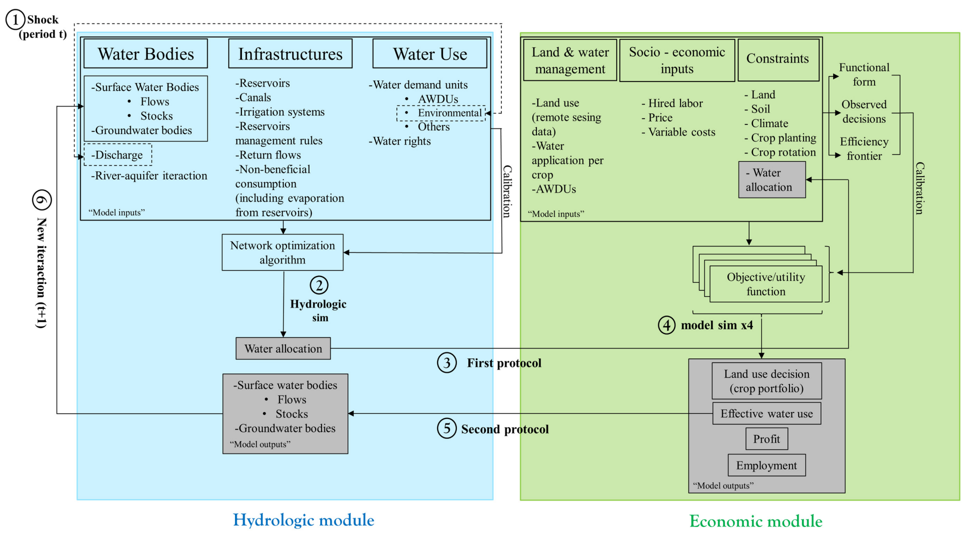

Figure 2. Flowchart of the Coupled Human and Water Systems

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

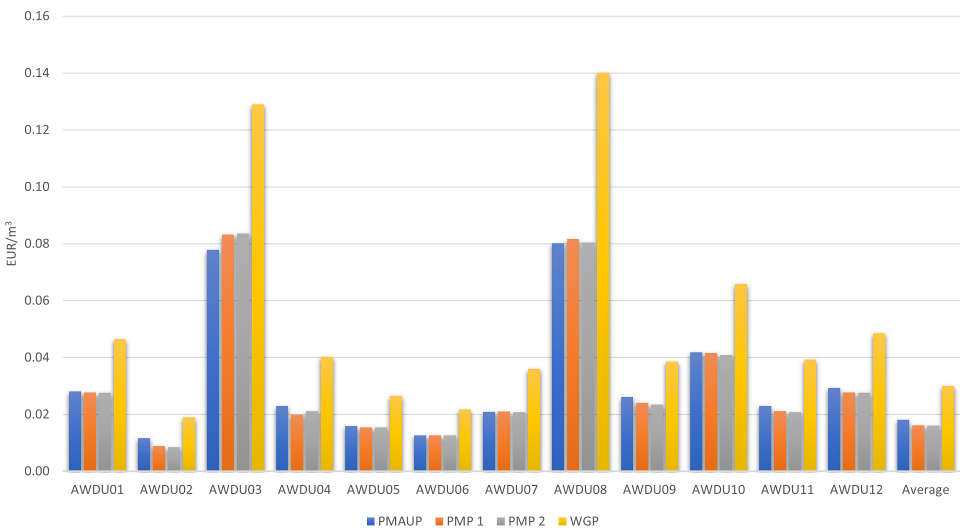

Figure 3. Resource Cost for AWDUs

Source: Own elaboration.

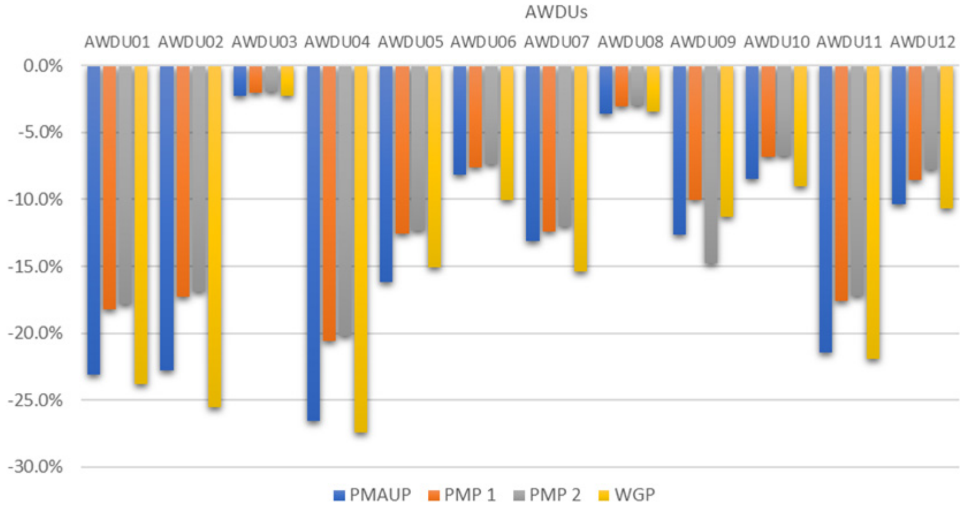

Figure 4. GVA Reduction Per AWDU Following Resource Costs Recovery

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

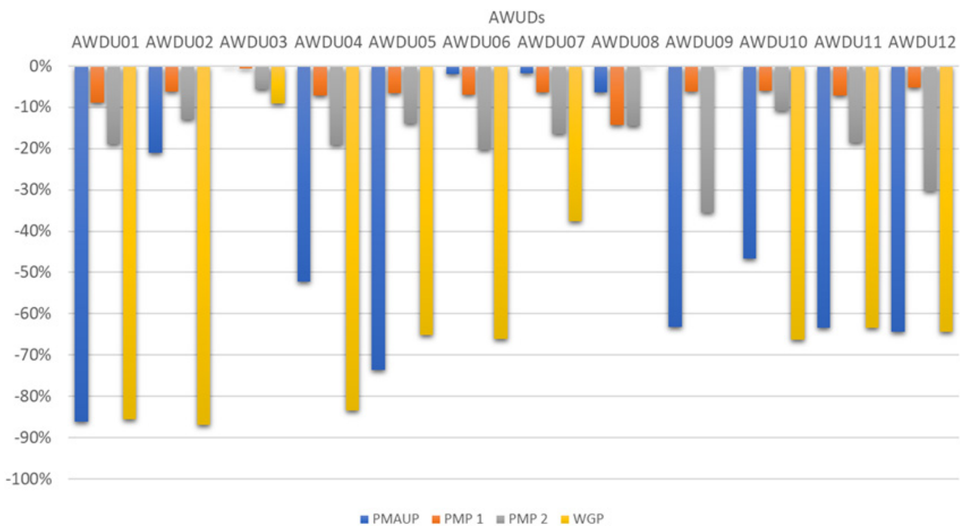

Figure 5. Water Savings Per AWDU Following Resource Costs Recovery

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

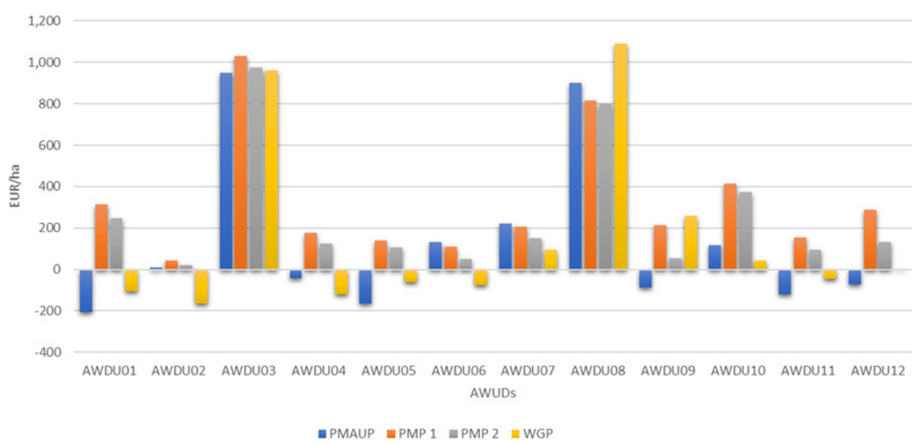

Figure 6. Changes in Tax Revenue Per AWDU Following Resource Costs Recovery
Source: Own elaboration.

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

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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.

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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.

4.3. First-degree price discrimination water bank to reduce reacquisition costs and enhance economic efficiency in agricultural water buyback. Insightful results from the Douro River Basin in Spain.

4.3.1. Resumen

En los programas de recompra de agua, una institución pública compra una cantidad predeterminada de agua a vendedores dispuestos; parte de esa agua puede reasignarse a los usuarios en una fase posterior de arrendamiento. Esto hace posible comprar agua a (bajos) precios monopsonistas y vender una fracción de esta agua a (altos) precios monopolistas, donde el agua readquirida en exceso de las ventas se utiliza para restaurar los bienes naturales. Proponemos un banco de agua con discriminación de precios donde la institución pública aprovecha su posición monopsonista (monopolista) para pagar (pedir) un precio por cada unidad de agua vendida (comprada) que coincida con el precio de reserva de cada comprador (vendedor) en el mercado. Tanto los excedentes de los consumidores como de los productores se transforman totalmente en ingresos públicos, lo que reduce la carga presupuestaria de la restauración medioambiental sin afectar negativamente a la eficiencia económica. Ilustramos el desempeño del banco de aguas con discriminación de precios bajo incertidumbre a través de un conjunto hidroeconómico multimodelo que se aplica a la subcuenca del Alto Duero (España). Nuestros resultados muestran que el banco de agua con discriminación de precios puede lograr el mismo objetivo de readquisición de agua que un banco de agua convencional (sin discriminación de precios, sin fase de arrendamiento) a un costo significativamente más bajo (59.5% - 288.8% de reducción) mientras se logra un superávit productivo significativamente mayor (aumento del 331% al 570%).

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First-degree price discrimination water bank to reduce reacquisition costs and enhance economic efficiency in agricultural water buyback

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ABSTRACT

In water buyback programs a public institution (the *water bank*) purchases predetermined amount of water from willing sellers, part of which can be reallocated to users in a subsequent lease phase. This makes possible to buy water at low monopsonistic prices and sell a fraction of this water at high monopolistic prices, where the water reacquired in excess of sales is used to restore natural assets. We propose a price discrimination water bank where the public institution leverages its monopsonistic (monopolistic) position to pay (ask) a price for every unit of water sold (bought) that matches the reserve price of every willing buyer (seller) in the market. Thus, both the consumer and producer surpluses are wholly transformed into public revenues, which reduces the budgetary burden of the environmental restoration without negatively impacting economic efficiency. We illustrate the performance of the price discrimination water bank under uncertainty through an hydroeconomic multi-model ensemble that is applied to the Upper Douro sub-basin (Spain). Our results show that the price discrimination water bank can achieve the same water reacquisition target as a conventional water bank (no price discrimination, no lease phase) at a significantly lower cost (59.5%–288.8% reduction) while achieving a significantly higher productive surplus (331%–570% increase).

1. Introduction

Water supply is unevenly distributed across time and space, resulting in markedly wet and markedly dry periods and regions, which are becoming increasingly polarized due to climate change (IPCC, 2019). In addition, population growth, improving living standards, changing consumption patterns and irrigation expansion are causing water demand to rise sharply (UN, 2020). The resultant temporal and geographical supply-demand imbalances have been traditionally addressed through the construction of waterworks to expand the supply base (Hassan, 2010). Yet, as the limits to total water supply are reached, and surpassed, a growing number of basins are entering a contraction phase where total water demand must be decreased to reach a new sustainable level (Loch et al., 2020a). To this end, the scientific community and policymakers have advised the adoption of demand-side policies that reallocate available supply among existing uses, including environmental ones, to address the economic and environmental impacts of growing scarcity and droughts (OECD, 2015; World Bank, 2017). One such policy is the public reacquisition of water, or *buyback*, which is gaining momentum in areas like Australia's Murray-Darling Basin (AUD 3.1 billion for the period 2009–2024), SE Spain (EUR 829.9 million for the period 2007–2027) and the US, notably California (USD 547 million during 1987–2011, 55% of which after 2003) (Adamson and Loch, 2018; Hanak and Stryjewski, 2012; Rey et al., 2019).

In conventional water buyback programs, a public institution (the *water bank*) issues purchase tenders to reacquire a predetermined amount of water from willing sellers—usually irrigators, who concentrate the least value-added uses of the resource. Water reacquisitions are subsequently used to preserve or restore natural assets (Adamson and Loch, 2018). While adequately designed water markets can generate Pareto improvements and enhance economic efficiency (Mendelsohn, 2016), buyback typically comes along with non-trivial value-added losses in the agricultural sector and a significant burden on the public budget, with non-negligible opportunity costs for highly indebted water scarce economies (Pérez-Blanco and Standardi, 2019). This is compounded by information asymmetries and agency costs, which may inflate market prices and the extent of the compensation, thus

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hampering the ability of buyback programs to restore natural assets (Iftekhar et al., 2013). Accordingly, buyback research has largely focused on developing models and techniques that produce information on the environmental outcomes and costs of the program, including the reserve price of potential sellers. Table 1 summarizes the literature on water banks in agricultural water buyback programs. A more detailed account of this literature is available in the online supplementary material (Annex I).

Critically, buyback literature assumes that the water bank will elicit and pay the equilibrium price that would emerge from a hypothetical competitive market, although the water bank is often the sole buyer (and subsequently the sole water allocator and potential seller) in the market. Most recently, Gutiérrez-Martín et al. (2020) and Gómez-Limón et al. (2020) have researched buyback programs where water banks use their monopolistic-monopsonistic position to set monopoly-monopsony prices during the buyback (monopsony) and a subsequent lease phase (monopoly). This makes possible to buy water at low monopsonistic prices and sell a fraction of this water at high monopolistic prices, where the quantity of water not sold is used to restore natural assets. In their research, authors make the critical assumption that society's demand function for water is known. Under this assumption, the sale price for productive uses can differ from the marginal value of environmental uses, and this difference can be measured. Thus, while lowering purchase prices and increasing sale prices with respect to equilibrium prices in a competitive market reduces the consumer and producer surpluses of productive users, it also increases the environmental surplus through water reallocations towards environmental uses (see Fig. 1.c). Authors show that, if optimally managed, this water bank outperforms conventional buyback programs in competitive markets both in terms of economic and environmental outcomes.

However, there are several limitations that obstruct the implementation of the monopolistic-monopsonistic water banks proposed by Gutiérrez-Martín et al. (2020) and Gómez-Limón et al. (2020) in practice. **First**, although the inclusion of a society's demand curve for environmental water offers an optimal and elegant solution to the problem of how much water to reallocate towards the environment, eliciting such a curve is challenging. On the one hand, there is a large and growing literature that estimates the demand for environmental goods in general and water-related ecosystem services in particular. This literature provides a wide range of methods and tools to estimate societal demand for environmental water within or across sectors, including, inter alia, contingent valuation (Loomis, 2002; Loomis et al., 2000), choice experiments (Carlsson et al., 2003), production function (Gutiérrez-Martín et al., 2014; Sapino et al., 2020), hedonic pricing (Moore et al., 2020; van Dijk et al., 2016), defensive costs (Cooley et al., 2019), travel costs (US EPA, 2019) and benefit transfer (Honey-Rosés, 2008; Pérez-Blanco and Sapino, 2022). On the other hand though, there are few standardized approaches to the economic valuation of environmental water, and there are often "large differences between values obtained through different methods" (UN, 2021). As Gómez-Limón et al. (2020) duly noted, "there is no robust empirical estimate available for the maximum value that society would be willing to pay for environmental water". For example, the world's "most advanced" (Seidl et al., 2020a) water market in Australia's Murray-Darling Basin lacks standardized approaches to valuation, which leads to significant differences in water values reported (Seidl et al., 2020b). More importantly, there are many different

Table 1
– Literature review on water banks in agricultural buyback programs.

Author(s)	Model	Region	Market	Traders	Settlement
Burke et al. (2004)	Integrated hydrologic and economic model	Klamath Project (US)	Competitive	Farmers and public agency	Spot market
Hollinshead and Lund (2006)	Multistage linear programming model	San Francisco Bay/ Sacramento-San Joaquin Delta (US)	Competitive	Farmers and public agency	Long-term trade, spot market, and option contract
Kirby et al. (2006)	Hydrology model combined with economic information that drives land and water use (ARISCtrade)	Murray-Darling Basin (Australia)	Competitive	Farmers and Environmental Steward (public agency)	Counter-cyclical trading
Qureshi et al. (2007)	Mathematical programming model with a hydrologic and an agronomic component	Murray River Basin (Australia)	Competitive	Irrigation regions, public agency	Spot market
Dixon et al. (2011)	Computable General Equilibrium model	Southern Murray-Darling Basin	Competitive	Farmers and public agency	Spot market
Dixon et al. (2012)	Computable General Equilibrium model	Southern Murray-Darling Basin	Competitive	Farmers and public agency	Spot market
Connor et al. (2013)	Dynamic integrated hydrologic, economic, and environmental model	Murrumbidgee Catchment (Australia)	Competitive	Farmers and public or not-for-profit environmental water holders	Spot market
Iftekhar et al. (2013)	Agent-Based Model	Murray-Darling Basin (Australia)	Competitive	Farmers and public or not-for-profit environmental water holders	Spot market
Lane-Miller et al. (2013)	Review of buyback programs	Australia & US	Competitive	Farmers and public agency	Spot market and derivatives
Loch et al. (2014)	Irrigators survey	Murray-Darling Basin (Australia)	Competitive	NA	NA
Rey (2014)	Mathematical programming model	Tagus-Segura Water Transfer (Spain)	Competitive	Farmers and public agency	Option contract
Kahil et al. (2015)	Reduced form hydrological model combined with mathematical programming economic model and institutional and environmental variables	Júcar Basin (Spain)	Competitive	Public agency, irrigators, and municipalities	Spot market
Adamson and Loch (2017)	State-Contingent Approach	Murray-Darling Basin (Australia)	Competitive	Farmers and public agency	Spot market
Pérez-Blanco and Gutiérrez-Martín (2017)	Positive Multi-Attribute Programming	Segura Basin (Spain)	Competitive	Farmers and public agency	Spot market
Pérez-Blanco and Standardi (2019)	Coupled mathematical programming and Computable General Equilibrium model	Region of Murcia (Spain)	Competitive	Farmers and public agency	Spot market
Gómez-Limón et al. (2020)	Mathematical programming model	Guadalquivir Basin (Spain)	Monopoly-monopsony	Farmers and public agency	Spot market
Gutiérrez-Martín et al. (2020)	Mathematical programming model	Guadalquivir Basin (Spain)	Monopoly-monopsony	Farmers and public agency	Spot market

perspectives of what 'value' specifically means to various decision makers and water users, which makes challenging to quantitatively compare the value of water for economic uses such as agriculture v. the value of environmental water or the human right to water (UN, 2021). This leads to unresolved tradeoffs that complicate decision making. When confronting this dilemma, decision-makers typically opt for cost-effectiveness methods to inform their decisions, where the objective is to achieve a predefined target (e.g., minimum environmental flows) at the least cost—thus disregarding the economic benefits of reallocations. This is notably the case of the EU Water Framework Directive (OJ, 2000). Since a key objective of our research is that of producing an actionable method that conveys information of value towards the implementation of first-degree price discrimination water banks in real life, we also adopt a cost-effectiveness approach.

Second, the research on agency costs and water banks above uses a single system model to generate point predictions of human behavior and supply and demand functions (i.e., consolidative modeling), which artificially reduces uncertainty. This may lead to "surprises" arising from "the non-mechanistic dynamics of complex adaptive socio-ecological systems" (Anderies et al., 2006, p. 867). For example, if the willingness to pay for productive water uses is overestimated, or the willingness to pay for environmental water underestimated, the optimal water allocation in equilibrium may lead to an insufficient provision of water-dependent ecosystems with non-trivial and potentially irreversible environmental and economic impacts. Accordingly, some legislations have made the application of the precautionary principle a statutory requirement, as is the case in European law (OJ, 2012, chap. 191.2). This is illustrated by the EU Water Framework Directive, whose main objective is to achieve the "good ecological status" of water bodies (OJ, 2000).

Third, buyback and water markets research often relies on economic models that do not observe hydrological integrity. As a result, large-scale water markets that consider the basin as a single homogeneous entity are often proposed, which may lead to large reacquisitions in areas where water is cheap but also abundant, and small and insufficient reacquisitions in areas where water is expensive but scarce—which are often disconnected from one another (Young, 2014). Moreover, water markets and buyback programs are designed to trade withdrawal rights, and ignore the return flow externalities that occur where buyers consume a higher fraction of the water withdrawn than the previous user and thus reduce water supply for downstream users not directly involved in the trading (Pérez-Blanco et al., 2020). Bridging the hydrological integrity gap calls for "limiting trading to the seller's consumptive water use" (Huffaker and Whittlesey, 2000), as well as market segmentation to consistently address supply-demand imbalances across the basin (Delacámara et al., 2015), which can only be addressed through a comprehensive understanding of water system dynamics (e.g., through hydrologic modeling).

The objective of this research is to assess how water banks operating under a monopoly-monopsony position can outperform existing buyback programs while addressing the key limitations above. To this end, we propose a water bank that operates under a monopsony-monopoly setting à-là-Gutiérrez-Martín et al. (2020), with the purpose of 1) restoring the balance between supply and demand during droughts and 2) identifying and realizing Pareto improvements through water reallocations among users, in this order (i.e., in line with the precautionary principle, first supply-demand gaps are addressed, and only at that point are reallocations towards productive uses allowed). The defining characteristic of the proposed water bank is that it uses *first-degree price discrimination*. Under first-degree price discrimination, the water bank pays/asks a different price for every unit of water sold/bought, which matches the maximum/minimum price that every buyer/seller in the market is willing to pay/accept (reserve price). Thus, both the consumer and producer surpluses are wholly transformed into public revenues, which reduces the budgetary burden of the environmental restoration without negatively impacting economic efficiency (see Fig. 1). Examples of first-degree price discrimination are typically observed in markets that are operated through tenders (e.g., online ads bids such as Google Ads, art auctions, etc.), which is also the case of buyback programs. Use of price discrimination in buyback programs has been previously theorized in the literature (Pérez-Blanco and Gutiérrez-Martín, 2017). In fact, the mechanics of public purchase tenders in existing water buyback programs already offer water banks information to price-discriminate potential sellers according to their willingness to accept: bidders make *m* bids to meet the public demand of *n* units of water; next, the $m > n$ bids received are ranked in ascending order, and

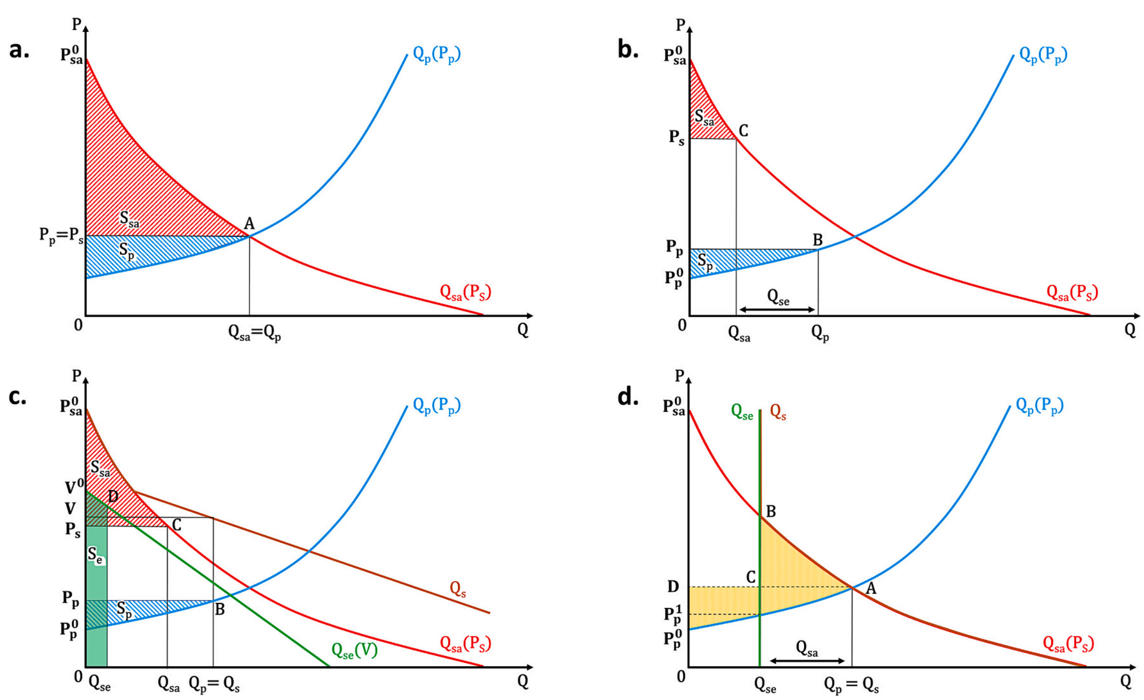

Fig. 1. Alternative water bank settings under a monopolistic-monopsonistic position. Source: a., b., c. Gutiérrez-Martín et al. (2020) and d. own elaboration.

the less costly n bids are accepted; finally, a clearing price that is equal to that of the last accepted (higher) bid is established, and all successful bidders receive that compensation. Under first-degree price discrimination, this last step is removed, and sellers would receive their reserve price.

To assess the potential of the proposed water bank we use a modular and time-variant hydroeconomic model (Essenfelder et al., 2018) that integrates the hydrological Decision Support System (DSS) AQUATOOL (Andreu et al., 1991) and an ensemble of 4 mathematical programming models: 1 Positive Multi-Attribute Utility Programming (PMAUP) model (Gutiérrez-Martín and Gómez, 2011), 2 Positive Mathematical Programming (PMP(models (Cortignani and Severini, 2009; Howitt, 1995) and 1 linear Weighted Goals Programming (WGP) model (Sumpsi et al., 1997). AQUATOOL is the DSS adopted by Spanish river basin authorities for water resources planning and management, and it is used to generate estimates of water allocation to productive (including irrigation, but also households, industry, fish farming, hydropower, cooling) and environmental uses over a time period, which are subsequently used as inputs to the ensemble of microeconomic models, so to elicit reserve prices under the water bank. By using AQUATOOL we aim to mimic actual decision making and ensure replicability of our methods, to deliver actionable science. The rationale for the use of an ensemble approach is that of sampling uncertainty in the estimation of the willingness to pay/accept of economic agents in the model, to obtain a robust estimation of the reserve price and a reliable range for the economic performance of the proposed water bank. Markets are segmented based on the basin's hydrological features: in each monitoring station in the hydrological model, the minimum environmental flow threshold set by the basin authority must be achieved; and reallocations among productive users are only possible where they are hydrologically connected. In addition, trading is limited to the original seller's consumptive use to ensure no harm to third parties not directly involved in the reallocation. Methods are illustrated with an application to the Upper Douro in Spain, an otherwise water-abundant basin increasingly affected by drought events.

2. Water banks and the role of price discrimination

We present below four alternative water bank settings that can be adopted under a monopolistic-monopsonistic position, to provide the background and rationale for our proposed water bank that uses first-degree price discrimination. The first setting presents a conventional competitive water bank where the objective is that of maximizing productive efficiency (i.e., all water reacquired is allocated to productive uses such as irrigation). The second and third settings briefly summarize the research by Gutiérrez-Martín et al. (2020) and Gómez-Limón et al. (2020). The second setting refers to a self-financed water bank (i.e., buyback expenditures equal revenues from the lease) à-là-Gutiérrez-Martín et al. (2020) that aims to maximize the volume of water that is reacquired for environmental uses, irrespectively of the economic efficiency achieved. The third setting presents a water bank that aims to maximize economic efficiency considering the social demand for environmental water, as proposed by Gutiérrez-Martín et al. (2020) and Gómez-Limón et al. (2020). Finally, the fourth setting presents the main innovation of this paper, namely a water bank that uses first-degree price discrimination to minimize the budgetary burden and the deadweight efficiency loss where the social demand for environmental water is unknown.

2.1. Competitive water bank

Under this setting, the water bank aims to temporarily reallocate water among productive users during drought events to maximize productive efficiency (i.e., the efficiency derived from productive uses of water, such as irrigation). Productive efficiency is measured as the aggregation of the consumer (S_{sa}) and producer surplus (S_p) (see Fig. 1.a). The consumer surplus (S_{sa}) is the difference between the buyer's willingness to pay/reserve price of water and the equilibrium price in the market ($P_p = P_s$), and equals the area $AP^0_{sa}P_s$under the demand curve $Q_{sa}(P_s)$ in Fig. 1.a; while the producer surplus is the difference between the seller's willingness to accept/reserve price of water and the price observed in the market, and equals the area $AP^0_pP_p$ above the supply curve $Q_p(P_p)$ in Fig. 1.a.[1]

Productive efficiency maximization entails a reallocation of the water traded in the bank entirely to productive uses, meaning that no water is devoted to environmental uses. Therefore, the water bank has no buyback purpose and cannot remedy overallocation problems. This limitation is addressed in the next sub-sections.

2.2. Self-financed water bank for the maximization of water reacquired for the environment

In this setting proposed by Gutiérrez-Martín et al. (2020), the objective of the water bank is shifted from maximizing productive efficiency to maximizing the volume of water reacquired for the environment (Fig. 1.b). The water bank is self-financed: it buys water at a monopsonistic price (P_p) below the equilibrium price of the competitive water bank, and sells a fraction of this water at a high monopolistic price (P_s) above the equilibrium price of the competitive water bank, while ensuring that expenditures from purchases equal revenues from sales. The water purchased (Q_p) in excess of sales (Q_{sa}) is used to restore natural assets (Q_{se}). On the other hand, although efficiency is increased as compared to the situation without a water bank, it is lower than the efficiency achieved under the competitive water bank: the producer surplus (S_p) now equals to $BP^0_pP_p$ while the consumer surplus (S_{sa}) equals to $CP^0_{sa}P_s$, which as can be seen in Fig. 1.b, are both significantly below the surplus achieved under the competitive water bank due to the deadweight efficiency loss incurred. However, this ignores the economic surplus that is generated by environmental water. Like the consumer surplus obtained from productive uses, there is another type of consumer surplus derived from the provision of water to water-dependent ecosystem services, also known as environmental surplus (S_e). Such surplus can only be assessed if we incorporate society's demand for environmental water into the demand function of the market.

2.3. Water bank considering the social demand for environmental water

Under this water bank setting proposed by Gutiérrez-Martín et al. (2020) and Gómez-Limón et al. (2020), the aggregate water demand Q_s includes a productive demand function $Q_{sa}(P_s)$ and a society's demand function for environmental uses $Q_{se}(V)$ (Fig. 1.c). The productive demand function $Q_{sa}(P_s)$ and the supply function $Q_p(P_p)$ are the same as in the competitive and self-financed water banks. By means of accounting for the society's demand for environmental water, it is now possible for the water bank to target the maximization of the total economic efficiency, which includes the producer ($S_p = BP^0_pP_p$), consumer ($S_{sa} = CP^0_{sa}P_s$), and environmental surplus ($S_e = DV^0Q_{se}O$). As it is shown in Fig. 1.c, this setting involves lower purchase prices (P_p) and higher sale prices (P_s) as compared to the equilibrium price in the competitive market ($P_p = P_s$), which leads to a reduction in the producer and consumer surplus; but also creates an environmental surplus due to the water reallocation towards environmental uses. The water bank can be self-financed, where the revenues from sales ($P_s * Q_s$) equal the expenditures on purchases ($P_p * Q_p$) (Gutiérrez-Martín et al., 2020); alternatively, this constraint can be loosened, allowing for a deficit in the cash flow of the water bank that is covered through an ad-hoc budget ($P_p * Q_p$

[1] Note that throughout all water bank settings, we assume that transaction costs are zero. Transaction costs larger than zero would displace the supply function leftwards, increase equilibrium prices and contract the volume of water traded for all settings considered.

$- P_s * Q_s$) (Gómez-Limón et al., 2020). Irrespectively of the budgetary constraint adopted, authors demonstrate theoretically and empirically that when the social demand for environmental water is known, their proposed water bank outperforms a competitive water market in terms of overall economic efficiency (i.e., $S_p + S_{sa} + S_e$).

2.4. Water bank under the precautionary principle and the role of first-degree price discrimination

Since society's demand function for environmental uses is unknown (UN, 2021), the water bank setting proposed by Gutiérrez-Martín et al. (2020) and Gómez-Limón et al. (2020) and described in Section 2.3 cannot be implemented under real-life conditions. The other water banks considered above (sections 2.1 and 2.2) offer no satisfactory alternative: the competitive water bank entirely reallocates water to productive uses, meaning that no water is devoted towards environmental uses; while the self-financed water bank for the maximization of water reacquired for the environment has no clear environmental (it is not informed by ecological and hydrologic criteria, it does not observe hydrological integrity) or economic rationale (it is suboptimal in terms of efficiency and cost-effectiveness). Yet, as we will show below, it is still possible to find a realistic water bank setting that exploits its monopolistic-monopsonistic position to address overallocation problems in a way that is consistent with basic economic and hydrologic principles.

Environmental water reacquisitions in real-life buyback programs are not based on a hypothetical society's demand function for environmental uses. Rather, the decision of how much water to reallocate towards the environment is based on hydrological and ecological criteria leveraging the precautionary principle, where the objective is that of achieving a minimum performance in a series of environmental indicators, for which a water reacquisition target is set (e.g., the volume of water needed to achieve minimum environmental flows as is the case in the EU) (EC, 2015; Tonkin et al., 2018). Accordingly, the demand function for environmental uses Q_{se} now adopts an inelastic form as shown in Fig. 1.d, where the volume of water to be reacquired is fixed. The productive demand function $Q_{sa}(P_s)$ and the supply function $Q_p(P_p)$ are the same as in the water bank settings above. If the sole objective of the water bank is that of reacquiring the volume Q_{se} for the environment, as happens in conventional buyback programs, it can do so at a market price P_p^1, for a total cost of $P_p^1 * Q_{se}$.

It is also possible to establish a competitive water bank that reallocates water among productive users rightwards of Q_{se}. This would ensure that safe minimum environmental standards are met, while productive efficiency is maximized within the constraints imposed by the precautionary principle. Thus, the producer surplus would be $S_p = DP_p^0A$, while the consumer surplus would be $S_{sa} = ABC$. However, maximizing efficiency does not alleviate the budgetary burden that is faced by public agencies operating water banks; quite the contrary, such burden is aggravated. By including productive water users into the aggregate water demand Q_s, the purchase price would grow from P_p^1 to the equilibrium price in a competitive water bank ($P_p = P_s$), and the cost of the water reacquisition for the environment would increase from $P_p^1 * Q_{se}$ to $ODCQ_{se}$. Thus, increased productive efficiency would come at the expense of higher costs for the agency operating the water bank, which worsens the cost-effectiveness of the buyback program.

We propose the use of first-degree price discrimination to address overallocation problems in a cost-effective and (productively) efficient way while observing the precautionary principle. Under first-degree price discrimination, the water bank pays/asks a different price for every unit of water sold/bought, which matches the reserve price of every buyer/seller in the market. The reserve price of potential buyers, or willingness to pay, is signaled by the productive demand function $Q_{sa}(P_s)$ rightwards of Q_{se}; while the reserve price of potential sellers, or willingness to accept, is signaled by the supply function $Q_p(P_p)$. Under price discrimination, both the consumer ($S_{sa} = ABC$) and producer ($S_p = DP_p^0A$) surpluses are wholly transferred to the public sector (either through revenues, in the case of the consumer surplus, or through foregone costs, in the case of the producer surplus), while productive efficiency is maximized within the constraints imposed by the precautionary principle and equals $S_p + S_{sa} = DP_p^0A + ABC$. This minimizes both the cost of the program/budgetary burden (cost-effectiveness) and the deadweight loss (efficiency) (see Fig. 1.d).

3. Background to the case study: water markets in Spain and the Upper Douro Basin

3.1. Spanish water markets in the EU context

In compliance with the Water Framework Directive (WFD) that aims to achieve the "good ecological status" of water bodies (OJ, 2000), overallocated EU basins are identifying and adopting measures to reallocate scarce water resources towards environmental uses. At minimum, basin authorities should define and achieve safe minimum standards through minimum environmental flows; although more ambitious environmental targets are also pursued, e.g., through pulse flows that use water infrastructures to mimic natural flow regimes and restore water bodies. To underpin and coordinate the work by river basin authorities, the Commission published in 2015 a guidance document for the implementation of environmental flows in all EU basins (Bussettini et al., 2015). Both the WFD and the guidance document recommend the adoption of Polluter-Pays Principle (PPP)-based instruments, notably caps and charges, to implement environmental flows. PPP-based instruments fit well in the Spanish concessional model, where water rights are awarded for a fixed term (a maximum of 75 years), charged under the principle of full cost recovery (including environmental costs), and subject to forfeiture, expropriation, and waiving (BOE, 2001, chaps. 52, 53, 59). *De iure*, river basin authorities are entitled to limit (e.g., through higher charges or caps) or even terminate a water concession that harms the environment, without any compensation (BOE, 2001, chaps. 3, 14, 65). De facto, the relevant transaction costs of capping granted rights and/or applying incremental charges (Loch and Gregg, 2018), and concerns over the negative economic impact this may have on rural areas (Rey et al., 2019), result in caps and charges only being partially implemented, and also in recurrent infringements of minimum environmental flows. In this context, water buyback programs have emerged as a pragmatic response to achieve environmental flows targets while overcoming resistance from farmers through financial compensations, and compensating other possible negative feedbacks[2] (Pérez-Blanco and Gutiérrez-Martín, 2017).

Spain is the first and only EU country where water markets and buyback programs have been made legally feasible (Rey et al., 2019). The 1999 reform of the Water Law created the so-called exchange centers (in Spanish: *centros de intercambio*), a water bank managed by the public sector that operates under a monopolistic-monopsonistic position (BOE, 1999). The Royal Decree 9/2006 allowed public institutions to use the *centros de intercambio* to purchase water rights from productive uses and reallocate them to environmental uses (BOE, 2006). Since then, buyback programs have been used to achieve environmental flows targets in overallocated Spanish basins, including the Júcar River Basin, the Segura River Basin, and the Guadiana River Basin (Gómez et al., 2017). Importantly, water trading in Spain has been conducted only during emergency droughts and requires formal approval from the central government to take place, which has limited thus far its scope.

[2] Other possible negative feedbacks from water buyback are typically balanced out through complementary policies including, inter alia, subsidies for economic diversification, water efficiency improvements, and new transportation, communication and energy infrastructures (GRBA, 2008; MDBA, 2012).

3.2. The Upper Douro River Basin

The Upper Douro in NW Spain is one of the 13 sub-basins of the Douro River Basin, the largest basin in the Iberian Peninsula. The Upper Douro Sub-basin occupies an area of 8905 km^2 that runs through the easternmost part of the Douro River Basin. Historically considered a water-abundant sub-basin, climate change, and demand growth are aggravating scarcity, and the frequency and intensity of drought events are increasing. Annual water supply since 1980 has been, on average, 817.9 million m^3, a reduction of −23.4% as compared to the average supply of 1068.2 million m^3 per year since 1940 (DRBA, 2016a; MAGRAMA, 2017). Annual water demand in the basin is estimated at 187.9 million m^3, of which 83.7% originate from agriculture (DRBA, 2016a). Accordingly, the ratio of freshwater withdrawals to the renewable resources available in the basin is estimated at 23%. Under climate change, when water resources are projected to decrease to 768.8 million m^3 per year already by 2030, this ratio will increase to 24.4% (DRBA, 2020). This means that the basin is already beyond the water scarcity threshold (set at 20%) and moving towards the severe water scarcity threshold (40%) as per the EU's Water Exploitation Index + (EEA, 2021).

In compliance with the WFD, since 2009 the Douro River Basin Authority has designed and progressively implemented minimum environmental flows (defined on a monthly basis) for all rivers in the basin, including those in the Upper Douro.[3] Environmental flows are assessed through a network of control points (see Fig. 2), which in the case of the Upper Douro report an acceptable performance with limited monthly infringements. Such performance has been achieved thus far at a relatively low economic cost through moderate caps on agricultural withdrawals defined in the Drought Management Plan (DRBA, 2017); albeit occasionally, where droughts have been more extreme, caps have not been fully enforced and environmental demands have not been met (DRBA, 2020). Future climate is expected to aggravate scarcity and droughts in the Upper Douro (MAGRAMA, 2017), and meeting minimum environmental flows in this context will call for reduced supplies for productive uses, notably irrigated agriculture, a low priority use—i.e., the first use to suffer supply restrictions when water availability is reduced (DRBA, 2017). Decreasing water allocations to a new sustainable level will constrain farmers to choose new crop *portfolios* that use less water and have a lower return (Parrado et al., 2020), which will likely increase opposition to PPP-based instruments such as caps. In this context, several Spanish basin authorities are exploring the use of water markets to implement buyback programs that compensate irrigators.

>46,900 ha of the Upper Douro is farmland, of which 40% is irrigated (Table 2). The most relevant irrigated crops in the area are barley (34.3% of the irrigated surface), sunflower (11.8%), wheat (11.2%), sugar beet (7.8%), and maize (7.6%) (see Table 2). Irrigated farmland is divided into 24 Agricultural Water Demand Units (AWDUs), the basic irrigation unit in Spain, which are defined as "groups of irrigators sharing a common source of water, territorial, administrative, and hydrological characteristics" (see Fig. 2) (DRBA, 2016a). AWDUs are the economic agents in the multi-model microeconomic ensemble. The aggregation of individual farmers into representative economic agents is well documented in the literature, and in the case of Spain is typically done through Water User Associations WUAs (García-Mollá et al., 2013), agricultural districts (Gutiérrez-Martín and Gómez, 2011), or AWDUs (Calatrava and Martínez-Granados, 2012). Here we work with AWDUs due to two key reasons: 1) AWDUs are the relevant aggregation unit for water allocation decisions, including buyback programs and definition of environmental flows; and 2) AWDUs are the aggregation unit used by AQUATOOL. Fig. 2 represents the case study area and its AWDUs, as well as the location of the control points for the assessment of environmental flows in AQUATOOL. Control points are also used to segment the market and ensure hydrological connectivity across AWDUs (i.e., reallocations that would comply with hydrological integrity).

[3] Pilot pulse flows have been designed and tested for a number of sub-basins across the Douro River Basin, although their development is still in a preliminary phase that will not be concluded until the next planning cycle (2021-2027).

4. Methods

We build on previous work by Essenfelder et al. (2018) and Pérez-Blanco et al. (2021b) to develop a time-variant hydroeconomic model that integrates a hydrologic module and a microeconomic module through a protocol that conveys information on water allocation decisions from the hydrologic to the microeconomic module. The hydroeconomic module is populated with the hydrological DSS AQUATOOL (Andreu et al., 1991), the DSS used by Spanish river basin authorities to inform water allocation to economic and environmental uses at a basin level. The microeconomic module is populated with an ensemble of 4 mathematical programming models that elicit the behavior of irrigators and simulate their responses to changes in water allocations. In this way, uncertainties regarding parameter calibration and model development/design in the microeconomic module are sampled considering at once (i.e., for each input from the hydrological module) 4 representations of the economic system that use the same dataset and rely on alternative calibration methods, instead of using a single model to produce a point prediction.

4.1. Microeconomic multi-model ensemble

The behavior of agricultural agents and their responses to changes in water or agricultural policies are typically assessed through structural microeconomic models that incorporate the "deep parameters" or microfoundations (relating to preferences, technology, and resource constraints) driving human responses to change. Structural microeconomic models are mathematically stated representations of human agency that are calibrated to mimic the observed behavior of economic agents and can be used to understand key behavioral drivers and predict responses to exogenous shocks. The range of plausible responses is limited by a domain F(x) conformed by a set of physical and socioeconomic restrictions (Graveline, 2016). Agents in the models are assumed to be rational, i.e., they allocate available production inputs (in our case, through a decision on land use, x) to maximize the economic return within the domain, where the economic return is measured through a utility function U(x) conformed by one (single-attribute) or multiple (multi-attribute) utility-relevant attributes z(x) (e.g., profit, risk avoidance):

$$\underset{x}{Max}\, U(x) = U(z_1(x); z_2(x); z_3(x) \ldots z_m(x)) \tag{1}$$

$$s.t.: 0 \leq x_i \leq 1 \tag{2}$$

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

$$x \in F(x) \tag{4}$$

$$z(x) \in R^m \tag{5}$$

where x is the decision variable or crop portfolio, a vector indicating the share of land used by each crop x_i, which is revised yearly (irrigation campaign). Each crop x_i delivers a unique combination of the *j* utility-relevant attributes $z_j(x_i)z(x_i)$. F(x) represents the set of constraints that conform the domain, including the water availability constraint, of particular relevance for our research:

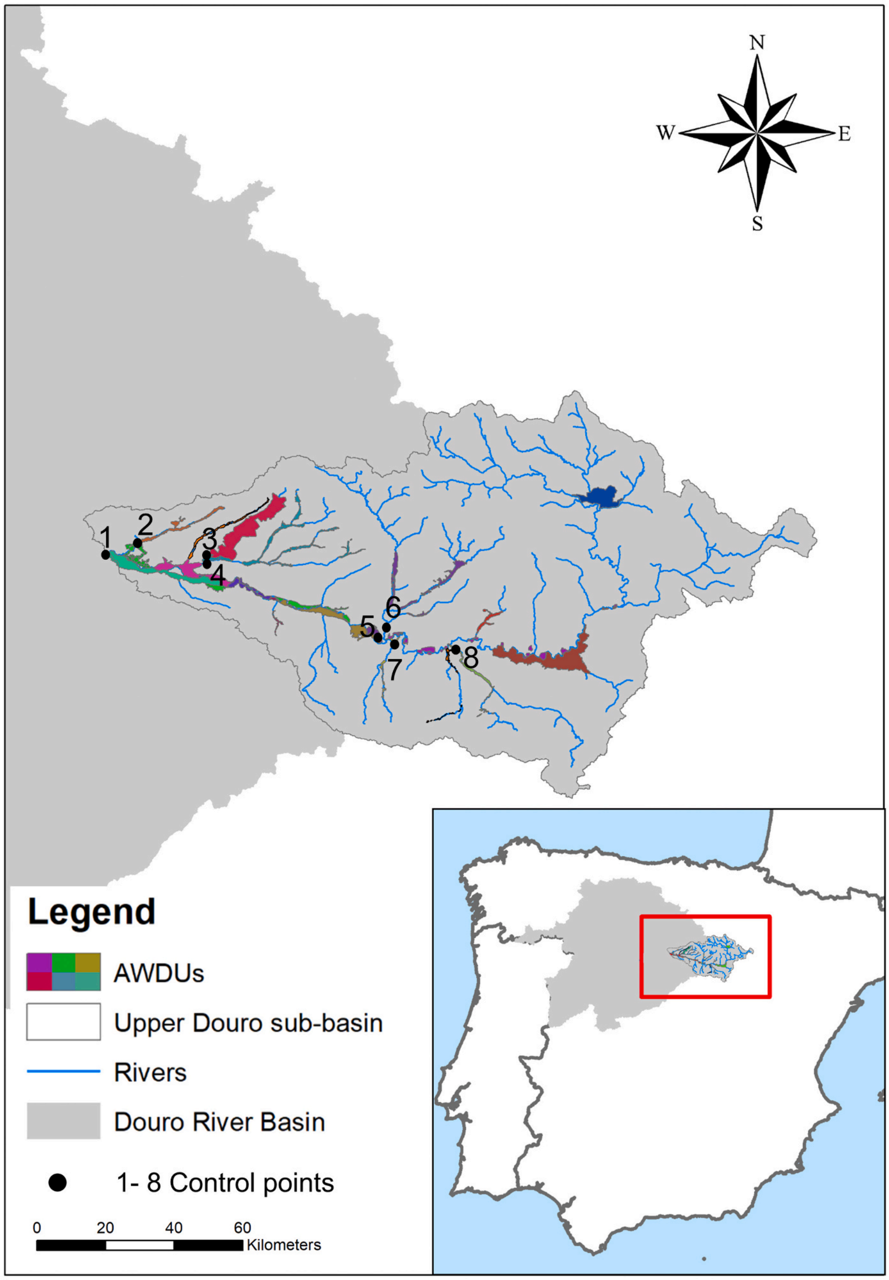

Fig. 2. Location of the Upper Douro Sub-basin in the Iberian Peninsula and detail of its AWDUs and the network of control points for environmental flows. Source: own elaboration.

$$\sum_{i=1}^{n} \frac{w_i}{eff} x_i \leq W_g \tag{6}$$

where w_i represents water consumption by crop *i*; eff is a parameter capturing technical efficiency in the irrigation system, which is obtained using data on return flows, conveyance, distribution, and application inefficiencies per AWDU from the hydrologic model. $\frac{w_i}{eff_i}$ is, therefore, the water use per crop (i.e. including water consumption and return flows); $\sum_{i=1}^{n} \frac{w_i}{eff} x_i$ represents the total water use by the economic agent (again, water consumed plus return flows); and W_g represents the water allocation constraint (per hectare), i.e. the amount of water that is assigned to each AWDU in a given irrigation campaign, which is determined by the hydrologic model. Note that to avoid return flow externalities (i.e., reduced water availability to downstream users not directly involved in trading), reallocations among productive users through the water bank are limited to the seller's consumptive water use, namely $\sum_{i=1}^{n} w_i x_i$.

The attributes in the utility function and their parameter values can be elicited using normative methods based on value judgments by experts (e.g., the agent aims to maximize total profit); or positive methods that use mathematical programming models to identify the utility-relevant attributes and *calibrate* the parameters that more accurately reproduce observed decisions. Positive methods are typically preferred by researchers due to their ability to more accurately reproduce observed behavior (Graveline, 2016). Most frequently used positive models include Linear Programming (both single- and multi-attribute), Positive Mathematical Programming – PMP (single-attribute), and Positive Multi-Attribute Utility Programming – PMAUP (multi-attribute). This paper uses an ensemble of 4 positive microeconomic models to model agent behavior, including a multi-attribute linear programming model, termed Weighted Goals Programming (WGP) following Sumpsi et al. (1997) calibration; two PMP models that follow the calibration

Table 2
Irrigated surface in the Upper Douro. Source: own elaboration from ITACyL (2019).

Crop	Irrigated area (ha)	Crop	Irrigated area (ha)
Barley	6437.21	Onion	203.10
Sunflower	2216.70	Oats	166.30
Wheat	2095.70	Escarole	118.20
Sugar Beet	1486.00	Pumpkin & Zucchini	115.50
Maize	1427.00	Walnut	63.30
Alfalfa	1236.50	Green chicory	36.00
Potato	1121.00	Forage vetch	34.20
Apple tree	510.54	Forage corn	19.00
Carrot	291.00	Pepper	11.00
Green peas	286.20	Cauliflower	10.00
Garlic	273.60	Peas	8.40
Lettuce	257.70	Other irrigated	81.31
Vineyard	243.74	**Total**	**18,749.2**

procedure of Howitt (1995) (PMP1) and Cortignani and Severini (2009) (PMP2); and a PMAUP model calibrated á-la-Gutiérrez-Martín and Gómez (2011). A comprehensive description of the domain, attributes, and database used by the models in the microeconomic ensemble is available in the literature above and in the online supplementary material, which includes the mathematical formulation of the domain F(x) (Annex II); the attributes explored, which include expected profit (the only relevant attribute for single-attribute models), risk avoidance, and management complexity avoidance (of which the latter is measured through three proxy attributes: total labor avoidance, hired labor avoidance, and direct costs avoidance), as well as the related data inputs (Annex III); and the calibration methods and results (Annex IV). Rather than running the water bank simulations with those models that better reproduce observed behavior, or generating a probability distribution function that combines all models to generate a point prediction, which may artificially reduce modeling uncertainty (Hino and Hall, 2017), this work adopts an un-weighted multi-model ensemble approach (IPCC, 2014). The result is a database that offers information on uncertainty regarding model design through the ensemble spread.[4]

4.2. AQUATOOL

AQUATOOL is a DSS for the edition, operation, review, and analysis of hydrologic models for river basin management that produces information on the quantitative and qualitative status of water bodies. The AQUATOOL DSS features several modular blocks, each with its own software/model suitable for alternative tasks: AQUATOOL is the general interface for editing data and managing the other blocks; SIMGES is the block for simulating watershed management, including conjunctive use; GESCAL is the block for simulating water quality at the basin scale; OPTIGES is the block for optimizing watershed management; SIMRISK is the block for risk assessment and management; EGRAF is the block for the graphical visualization of the results obtained through SIMGES, OPTIGES, GESCAL and SIMRISK; and EXTOPO is the block for exporting spatial data to vector format (PUV, 2020). Our study in the Douro River Basin uses the AQUATOOL (setup) and SIMGES (simulation) blocks to conduct a longitudinal and spatial assessment of water allocations under climate change conditions.

The different elements of the water system that are incorporated into the AQUATOOL block include surface water bodies, groundwater bodies, discharge series under natural conditions, river-aquifer interaction, infrastructures (reservoirs, canals, irrigation systems), water demand units (including AWDUs—the agent in the microeconomic model, but also other agricultural uses, households, industry, fish farming, hydropower, cooling, and other minor uses), return flows, conveyance, distribution and application inefficiencies (a key input to obtain water consumption by economic agents), evaporation from reservoirs, environmental flows, water rights, and water operation rules. All the necessary data for the setup of AQUATOOL in the Douro River Basin is accessible from online databases made available by the Douro River Basin Authority (DRBA, 2017; 2016a; 2016b), except for the discharge series under natural conditions, which need to be produced. Discharge series under natural conditions are derived by processing daily series of precipitation for the 1950–2015 period using the EVALHID tool, which integrates several rainfall-runoff models (Lerma et al., 2017). The resultant 1950–2015 series is further expanded using data from the SIMPA (*Sistema Integrado para la Modelación del proceso Precipitación Aportación*) rainfall-runoff model for the 1940–1950 and 2015–2018 periods (CEDEX, 2020). Data records from reservoirs and monitoring stations representative of the natural regime were used to address discrepant values. Given the applied policy focus of our paper, for all modeling exercises in this paper, we adopt the latest version of AQUATOOL that was set up and calibrated by the Douro River Basin Authority to inform its 2021 Douro River Basin Management Plan (DRBA, 2020).

Once the AQUATOOL block has been set up, the SIMGES block can be used to run longitudinal simulations that offer spatial information on the impacts of several exogenous shocks (e.g., climate change) on surface and subsurface water flows on a monthly basis. For surface water bodies, water flows are obtained by continuity or balance, while for groundwater bodies this is obtained through unicellular and multicellular models. Next, the management of the water system by the river basin authority that determines *water allocations* among alternative uses (including irrigators and the environment through environmental flows, but also other productive uses such as households or industry) is simulated using a network optimization algorithm. This algorithm determines water allocations across the basin conditional to the achievement of several objectives, including: i) meeting environmental flows targets but also ii) minimizing water deficits among uses, iii) achieving a certain water stock in reservoirs, and iv) achieving hydropower generation targets. The management algorithm is calibrated using up-to-date data on water rights and observed water allocations among uses, to match simulation outputs with the historical discharge and water stock in reservoirs (PUV, 2020). Thus, although one key objective in AQUATOOL is that of enforcing environmental flows, during periods of scarcity where there are relevant tradeoffs between this and other objective(s), environmental flow targets may not be fully achieved. For example, the basin authority will not deplete the water stock in reservoirs beyond a minimum threshold to achieve environmental flows; nor will reduce water allocations to productive uses below historical allocations in historical drought events. This can lead to infringements of environmental flows, particularly where water discharge is reduced due to climate change, which can be addressed through water banks. Importantly, the network optimization algorithm also provides

[4] Arguably, model selection techniques could be used to choose among candidates the model that performs better, for example through minimization of calibration errors, instead of relying on an ensemble. Nonetheless, assessing model performance is controversial and goes beyond a straightforward comparison of calibration errors. Notably, models in our ensemble are designed as a substitute for direct experimentation (there is no sufficient data on the performance of water markets in Europe or Spain), which means that we cannot *evaluate* the predictive performance of the models within the ensemble, a critical step in model selection (Konishi and Kitagawa, 2008). It may occur that a model with a relatively low calibration error performs poorly against non-observed data as compared to alternatives (poor predictive performance) (Pindyck, 2015). Alternatively, multi-model ensemble modeling can be used to generate a probability distribution function that combines all models to generate a point prediction that avoids model selection bias. Yet, this is challenging due to the subjectivity involved in defining prior assumptions about the distribution and the accuracy and weight attributable to each model (Tebaldi and Knutti, 2007). Besides, a populated ensemble including several models is necessary to infer an accurate probability distribution function, and this requires a large amount of resources (computational, personnel, etc.) that may not be available.

information on water allocation to other productive uses, albeit this information is not considered in our microeconomic analysis of the supply costs of water banks. This is because in the Douro and elsewhere in Spain agriculture is a low priority use—i.e., the first use to suffer supply restrictions when water availability is reduced (DRBA, 2017). Moreover, agriculture concentrates the least valuable uses—i.e., it is the most cost-effective alternative to reacquire water. Therefore, any restrictions or reacquisitions to meet environmental demand will focus on this sector, which is also the largest water user (83.7% of total demand) in our case study site and elsewhere in Spain and Southern Europe (around 80% of total demand) (EEA, 2021).

4.3. Coupling and simulation

In our simulations, a climatic shock (modified discharge under climate change) forces the hydrologic module, which simulates the water allocations to environmental and productive uses (including water allocation to irrigators/AWDUs, W_{Aq}) in each year t using the network optimization algorithm (see Fig. 3). Following the river basin management plan (DRBA, 2020), the climate shock is produced by adjusting the historical discharge series produced with EVALHID and SIMPA with inputs from MAGRAMA's (2017) report on the impact of climate change on water resources. Although relevant hydrological data is available for the period 1940–2018 (78 hydrological years), in our simulations we use a short series starting from 1980 (being 2018 the last year with available data as of April 2022, i.e., 38 hydrological years). The use of a short hydrological series is common practice in Spain and is the approach adopted by river basin authorities, including the Douro River Basin Authority, to inform their river basin management plans (see e.g., DRBA, 2020; 2016a). This is because short series are considered more representative of the current water regime, which has been significantly affected by anthropogenic activities (MAGRAMA, 2017).

Next, information on water allocations to environmental uses and irrigators/AWDU (W_{Aq}) under climate change obtained through AQUATOOL's network optimization algorithm is conveyed to the multi-model ensemble of mathematical programming models. If the amount of water allocated to the environment is insufficient to meet the minimum environmental flows established in the basin plan in one or more of the control points (see Fig. 2), the water bank intervenes and purchases water from the AWDUs located upstream of the relevant control point until the environmental flow deficit Q_{se} (the environmental demand in the water bank, see Fig. 1.d) is addressed. During the water reacquisition, the water bank leverages its monopsonistic position to target the lower bids and pay the reserve price. To elicit the reserve price, we quantify utility (U_g) under a series of simulations in which the water allocation W_g in eq. 6 is progressively reduced in each AWDU at equal intervals (g = 0, 1, 2, …, G), where the scenario without water restrictions corresponds to g = 0, and the scenario where the water allocation is fully relinquished corresponds to g = G. Then, the reserve price is obtained as the compensating variation, i.e., the monetized utility loss derived from a marginal change in the water allocation constraint (in EUR/m^3):

$$CV_g = \frac{e\left(U^*_{g-1}, W_g\right)}{W_g - W_{g-1}} \tag{7}$$

Where e is an expenditure function representing the minimum amount of money agents would need to attain the utility level in $g-1$ given a water constraint W_g. Using information from AQUATOOL's network optimization algorithm on the initial water allocation to each AWDU before the water bank is activated (g = Aq), it is possible to delimit the range of relevant reserve prices for each AWDU (agents can only sell the water they have been allocated, i.e., $g \leq Aq$), and rank reserve prices within that range from lower to higher to obtain the supply function ($Q_p(P_p)$ in Fig. 1). Note that since the water bank aims to restore minimum environmental flows in all control points, some of which are not hydrologically connected, markets can be segmented (i.e., multiple supply functions).

Once the environmental demand Q_{se} has been met, the water bank starts reallocating water among irrigators. Water reallocations among productive users take place until the economic surplus is fully captured by the water bank, i.e., until the marginal reserve price of potential sellers (supply function) exceeds that of potential buyers (productive demand function—corresponding to $Q_{sa}(P_s)$ in Fig. 1). The productive demand function is obtained by ranking AWDUs reserve prices from higher to lower in the range from g = 0 to g = G.[5] Again, markets can be segmented to ensure hydrological connectivity; while trading is limited to the sellers' consumptive use to avoid return flow externalities.

The process above is repeated for every year in the short series (1980–2018), and for each model in the microeconomic ensemble.

5. Results

Table 3 below reports the economic performance of the proposed water bank (precautionary principle, first-degree price discrimination, surplus maximization through reallocations among productive uses) over the entire time series modeled (1980–2018, total values). The performance of the proposed water bank is compared to that of a conventional water bank that aims to reacquire the same amount of water for the environment, without price discrimination and reallocations among productive uses.

Our results show that the price discrimination water bank increases economic efficiency (higher productive surplus) and reduces the costs of water reacquisition for the public sector (higher net public revenue) as compared to the conventional water bank. Note that in both water banks the precautionary principle is applied and thus the quantity of water reacquired for the environment (which equals the volume of water needed to achieve minimum environmental flows in the Upper Douro) is the same. In terms of efficiency, implementing price discrimination increases the performance of the water bank between 331% (PMP1) and 570% (WGP), depending on the model. Price discrimination also reduces the public costs of water reacquisition, for all models considered: while the conventional water bank yields a negative net public revenue ranging between EUR -37,908.6 and − 1,453,864.9 (i.e., the public sector experiences losses), the price discrimination water bank yields a positive net public revenue for all models except PMP1 (between EUR 62,105.8 and 162,201.8). Moreover, although in the PMP1 model the net public revenue is negative under both the conventional and price discrimination water bank, the costs of water reacquisition are considerably lower (−59.4%) in the latter. Figs. 4 and 5 show, respectively, the time (Fig. 4) and spatial (Fig. 5) distribution of water sales and purchases and their economic value, under the price discrimination water bank.

Both the PMP1 and PMP2 models yield significantly larger public expenditures and public revenues from the water bank as compared to the PMAUP and WGP. This is owed to the calibration methods used by PMP models (see Annex IV). To calibrate a PMP model, the researcher must introduce an ad-hoc area constraint to the domain that bounds the calibration results to the observed crop portfolio and thus obtain the dual values associated with this constraint for each crop. Next, these dual values are used to add a non-linear component to the objective function (a quadratic cost function in this case), to “specify a non-linear objective function such that observed activity levels are reproduced by the optimal solution of the new programming problem without bounds” (Heckelei and Britz, 2005). This procedure penalizes the shift towards rainfed or less water-intensive crops that represent a minor share in the original crop portfolio because they have been assigned a high quadratic cost. Accordingly, the compensating variation in PMP models increases

[5] Note that this implies that agents can only purchase water up to W_0—i.e., they cannot purchase more water than the formal right they have been granted.

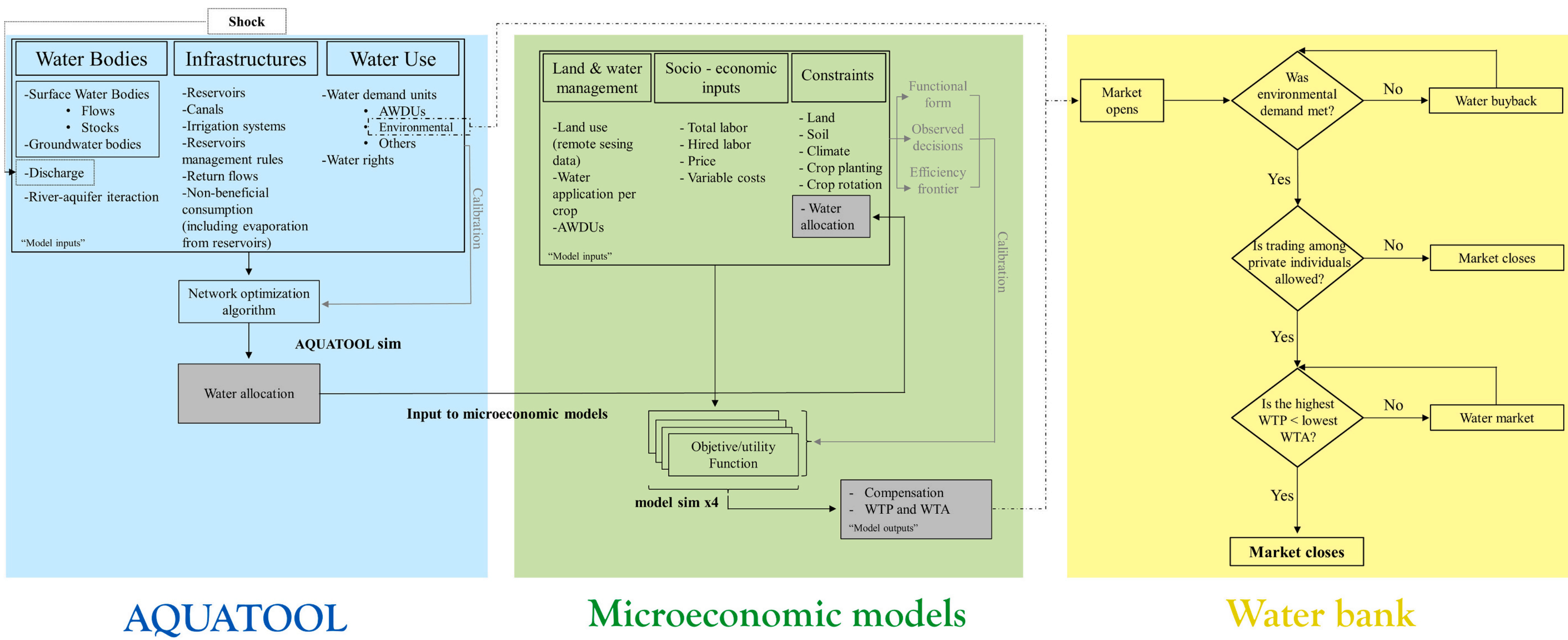

Fig. 3. Flowchart diagram of the modeling framework. Source: adapted from (Pérez-Blanco et al., 2021a).

Table 3
Economic performance of a conventional water bank and a water bank with price discrimination over the simulated period (38 hydrological years).

Conventional water bank	Water purchased [M m^3]	Water sold [M m^3]	Public revenues [EUR]	Public expenditure [EUR]	Net public revenue [EUR]	Water reacquired for environment [M m^3]	Productive surplus [EUR]
PMAUP	0.58	–	–	37,908.62	−37,908.62	0.58	41,120.75
WGP	0.58	–	–	43,292.80	−43,292.80	0.58	35,533.46
PMP1	0.58	–	–	858,899.30	−858,899.30	0.58	545,252.23
PMP2	0.58	–	–	1,453,864.93	−1,453,864.93	0.58	780,719.79
Price discrimination water bank	*Water purchased [M m^3]*	*Water sold [M m^3]*	*Public revenues [EUR]*	*Public expenditure [EUR]*	*Net public revenue [EUR]*	*Water reacquired for environment [M m^3]*	*Productive surplus [EUR]*
PMAUP	2.14	1.56	147,199.95	85,094.13	62,105.82	0.58	178,108.14
WGP	2.35	1.77	186,583.57	104,814.78	81,768.79	0.58	202,655.51
PMP1	1.06	0.48	836,016.98	1,184,126.23	−348,109.25	0.58	1,807,481.11
PMP2	1.93	1.35	3,598,793.31	3,436,591.51	162,201.80	0.58	3,507,712.57

at a significantly faster pace than in PMAUP/WGP models, particularly during severe droughts where large areas of conventional crops are substituted by rainfed and less water-intensive crops with high quadratic costs.

6. Discussion

Two broad categories of demand-side water policies are available to decisionmakers to restore the balance in overallocated basins: those based on PPP, such as charges or caps, where the costs of demand contraction fall on the water user; and those based on the beneficiary-pays principle (BPP), such as water buyback, where the costs of demand contraction fall on society as a whole, which pays a compensation to the water user for relinquishing an agreed share of its water allocation (OECD, 2015). The PPP pervades several environmental legislations, including that of EU countries (OJ, 2000), and over-abstraction is often interpreted as pollution due to the environmental cost it generates (Lindhout et al., 2014). Accordingly, instruments such as caps and water charges have been repeatedly endorsed by EU institutions as a means to restore the balance in overallocated basins in compliance with the PPP (EC, 2012; EEA, 2013). Caps and charges are effective in protecting and restoring the environment, albeit costs for water users are often non-trivial—particularly in water-scarce areas where demand is inelastic (Rey et al., 2019). This wealth transfer from water users to society has been typically met with resistance from those negatively affected, notably irrigators (the largest water user worldwide), and has raised significant institutional transaction costs—i.e., the costs of arranging a resource reallocation ex-ante, and then monitoring and enforcing it ex-post (Matthews, 1986). The high institutional transaction costs of PPP demand-side policies are quoted as a major cause for the delay or obstruction of much-needed water reallocations (Gómez et al., 2017).

BPP-based instruments such as buyback can help unblock transition by setting a bidirectional wealth transfer, where the restoration of natural assets for the society comes at the expense of financial compensations to water users who relinquish (part of) their water allocation. On top of that, adequately designed buyback programs can generate Pareto improvements and enhance economic efficiency (Mendelsohn, 2016). Yet, a key limitation to buyback and other BPP programs is that they can create a significant budgetary burden for the public budget with non-negligible opportunity costs, which can be aggravated by information asymmetries and agency costs. Our paper shows how this budgetary burden can be mitigated (if not surmounted) through price discrimination water banks, which can significantly reduce the cost of water reacquisitions for the water bank, and even generate net revenues while enhancing the productive surplus of the market.

Despite the promising performance of price discrimination water banks for agricultural water buyback, several barriers and caveats exist that should be carefully considered when assessing the design and implementation of this instrument in practice. We discuss these barriers below.

First, there are **legal barriers**. In most regions of the world, including the EU, water is a public good, managed by the government on behalf of its people (Lane-Miller et al., 2013). This poses the question of whether a government should pay private users for a resource that already belongs to the public, even if the transaction costs of implementing PPP-based instruments exceed the reacquisition costs of BPP-based instruments. In the EU context, the European Commission has issued lawsuits against nine member states for their (allegedly) incorrect application of Article 9 of the Water Framework Directive, which states that member states "shall take account of the principle of recovery of the costs of water services" through PPP-based water charges (OJ, 2000). However, legal ruling in this regard has been dichotomic. In 2014, in Case C-525/12 European Commission v Federal Republic of Germany, the European Court of Justice concluded that member states "may decide which economic instruments and design are to be implemented, as long as they meet WFD objectives", and that charges "are not the central and definitive instrument for addressing the problems facing Europe in terms of water resources, but rather a specific measure which should be applied in connection with [other measures]" (Jääskinen, 2014). This paved the way for the implementation of BPP-based instruments that, as the price discrimination water bank proposed here, restore the ecological status of water bodies while compensating users. However, a more recent ruling from European legal bodies has contradicted this interpretation. Notably, the Supreme Court of Spain (the first and only EU member state that has implemented water markets) recently ruled against BPP-based reallocations included in the Júcar River Basin Management Plan (BOE, 2017). Moreover, water markets and buyback in Spain are only temporary allowed, and conditional to the formal approval by the government (which has been limited to extreme droughts).

Second, there are **distributive and economic barriers**. By contracting agricultural production, buyback programs can induce non-trivial economy-wide impacts that can significantly affect third parties not directly involved in the trading, through forward and backward linkages across sectors (Dixon et al., 2011). This pecuniary externality affects more strongly those economic sectors that are heavily dependent on agricultural commodities for their production, such as the food industry, or those supplying agricultural inputs such as fertilizers. Pecuniary externalities can be offset at a regional level by the increase of agricultural commodity prices and the reallocation of agricultural production factors, albeit at a national level there is typically a net loss (Dixon et al., 2012). Noteworthy, water banks that include a partial lease of water reacquisitions towards high value-added agricultural uses, such as the mechanism proposed here, can contribute to mitigating this impact (Lane-Miller et al., 2013).

Third, there are **institutional barriers.** Some of the preconditions to ensure hydrological integrity in water buyback and trading adopted in this paper may be difficult to implement in practice. Notably, water allocations in Spain and most places worldwide are currently based on

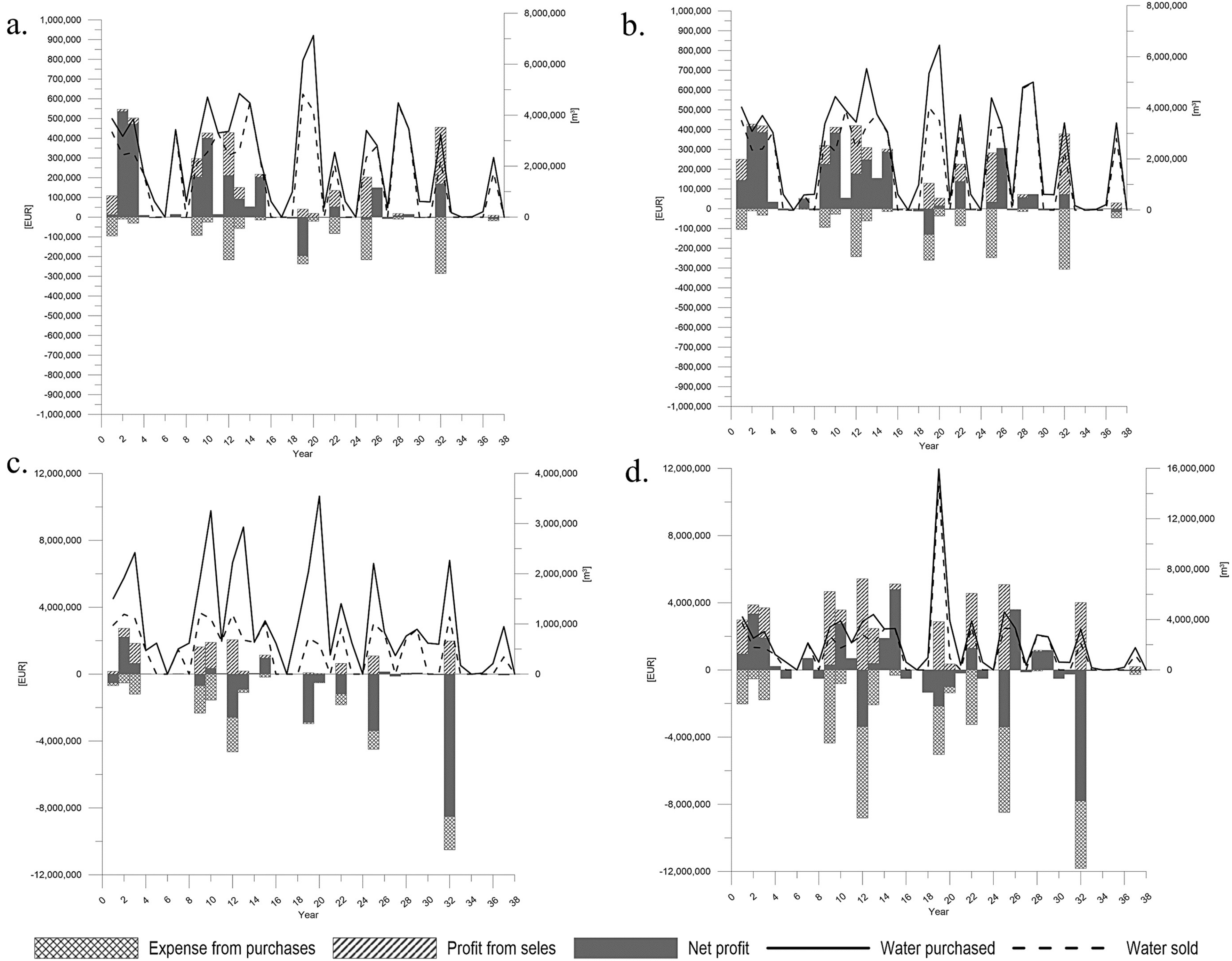

Fig. 4. Yearly water purchases and sales and net revenue over the 38y simulation period in the price discrimination water bank. a. PMAUP, b. WGP, c. PMP1 and d. PMP2.

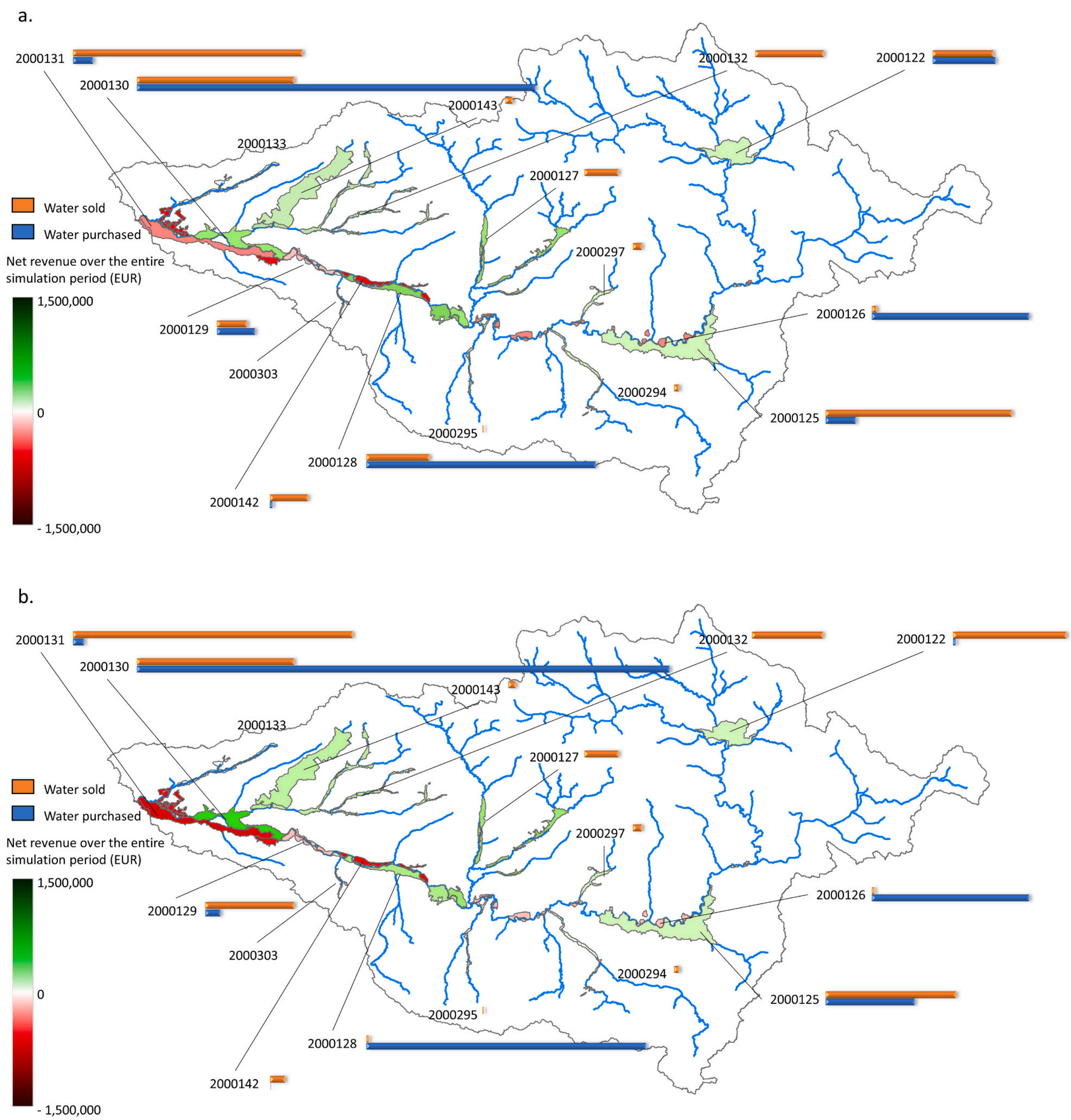

Fig. 5. Water purchased and sold and net revenue in the price discrimination water bank. a. PMAUP, b. WGP, c. PMP1, d. PMP2.

the right to withdraw, and not on the right to consume, which complicates limiting trading to the seller's consumptive water use (exceptions include the Western US doctrine of prior appropriation, where water rights are based on withdrawals, but trading generally transfers only the consumptive fraction of the right to avoid harm to a third party). Additionally, research has revealed significant incentives towards non-compliance in water resources allocation through water theft, which can reduce the effectiveness of buyback programs (Loch et al., 2020b). Ensuring hydrological integrity in this context necessitates a far-reaching institutional reform that is likely to involve significant *institutional* transaction costs. Moreover, implementing first-degree price discrimination tenders in real life is likely to demand higher search and information, bargaining, and policy and enforcement costs than conventional water banks, which will comparatively inflate also the non-trivial *private* transaction costs of the reform. For example, particularly in segmented markets, irrigators can resort to collusive tendering to profiteers, which can significantly reduce the performance of price discrimination. Scientific research has monetized the transaction costs of water market reallocations in the US and Australia, showing that private transaction costs can represent up to 35% of the total costs of policy reform (i.e., transaction costs plus abatement costs) (Loch and Gregg, 2018); and institutional transaction costs up to 30% of total costs (Njiraini et al., 2017). Accordingly, our analysis of abatement costs of first-degree price discrimination water banks should be complemented with transaction costs analysis before any concluding information on the cost-effectiveness of this instrument relative to others can be provided (e.g., v. charges, which is often assumed to involve higher transaction costs). Future research should bridge this gap and explore private and

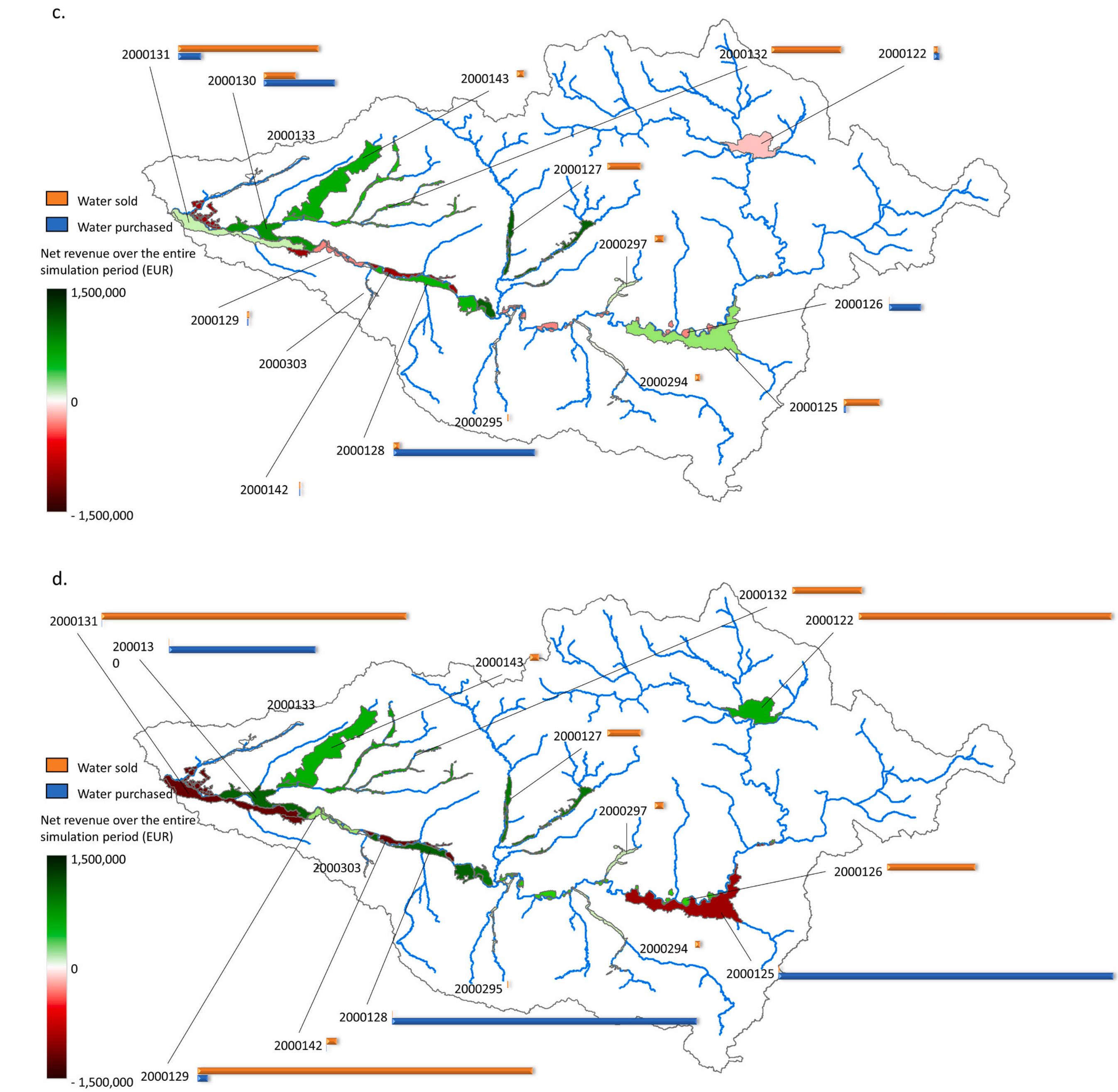

Fig. 5. (*continued*).

institutional transaction costs in water banks under a monopsony-monopoly setting to analyze their impact on the performance of the water bank.

Fourth, there are **unresolved policy design issues**. Restoring the balance in severely overallocated basins may require large reacquisitions of water in areas where the resource is most expensive, which can lead to disproportionate costs that are unaffordable to the public sector even in the presence of price discrimination (e.g., coastal basins with greenhouses). Moreover, even in situations where water banks can increase economic efficiency, this does not guarantee the achievement of an equitable outcome. This calls for the assessment of policy mix and sequencing strategies that address multiple objectives in water policy reform (Gómez et al., 2017).

7. Conclusions

This paper proposes and assesses the performance of a price discrimination water bank to minimize reacquisition costs and enhance economic efficiency. To this end, we develop a hydroeconomic model that couples the hydrological DSS AQUATOOL used by Spanish river basin authorities to inform water allocation decisions with an ensemble of 4 mathematical programming models that elicit the behavior of irrigators and simulate their responses to changes in water allocations, to assess parameter and structural uncertainties. Methods are designed to be replicable and flexible, and are capable of including additional mathematical programming models, as well as other DSS used by basin authorities elsewhere, such as WEAP (Yates et al., 2005), MIKE (Refsgaard and Storm, 1995), RIBASIM (Deltares, 2022), LISFLOOD (JRC, 2022). The model is applied to the case of the Upper Douro sub-basin (Spain), where we assess and compare the performance of the proposed price discrimination water bank v. a conventional water bank (no price discrimination, no water reallocations among productive uses) in achieving the same environmental water reacquisition target. Our results show that price discrimination enhances the productive surplus of the water bank between 331% and 570% and reduces the costs of water reacquisition between 59.5% and 288.8%, depending on the model. In three out of the four models considered, the reduction in reacquisition costs exceeds 100%, meaning negative net revenues under the conventional water bank are turned into positive net revenues under the price discrimination water bank.

We envision several ways in which the proposed methodology and price discrimination mechanism could be improved.

First, although our model is designed to be flexible and replicable and could be adopted in alternative basins using a DSS other than AQUATOOL, the framework presented in this paper applies to the AQUATOOL DSS. Simple, yet ad-hoc transformations are needed to adapt the proposed framework to alternative DSS (alternative mathematical programming models can be added without changes to the model setting). More importantly, while the proposed hydroeconomic model samples uncertainty in the economic module, the quantification of modeling uncertainty in the hydrologic module requires the incorporation of alternative DSS, which are often not available since basin authorities typically rely on a single DSS for their decisions.

Second, beyond adding new models to the hydroeconomic model, the models already in use could be improved. For example, the classical mathematical programming models used in the microeconomic ensemble only allow for adaptation at the extensive (land reallocations towards less water-intensive crops) and superextensive (land reallocations from irrigated to rainfed agriculture) margin, but do not allow for intensive margin adaptation (i.e., supplementary or deficit irrigation). Similarly, AQUATOOL is presently being expanded to increase its granularity by incorporating a larger number of control points to measure key hydrogeologic variables.

Third, additional scenarios should be explored to better quantify uncertainty. For example, river basin authorities in Spain use a single climate change scenario to force the hydrologic DSS and inform their decisions (AQUATOOL) (see e.g. DRBA, 2020). This artificially reduces uncertainty and can lead to surprises. Similarly, DSS models typically reproduce the currently existing set of infrastructures, which may change in the future (e.g., new dams, canals or water treatment and reuse infrastructures, irrigation modernization, irrigation expansion, etc.) and affect the impacts of water banks.

Fourth, the water bank design proposed here relies on a spot market and does not explore the impact of derivatives such as option contracts, leasing, or counter-cyclical trading—all of which can affect its environmental and economic performance. The focus on spot markets is justified because water market experiences in Spain (where our case study area is located) and elsewhere in the EU have thus far relied exclusively on spot markets, while derivatives are still treated in an incipient form, with major barriers (including of legal nature) persisting. This is unlike water markets in Australia or the US, where derivatives are more frequently found. Particularly in these regions, future research should explore the impact of derivatives on the environmental and economic performance of first-degree price discrimination water banks.

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.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolecon.2022.107694.

References

Adamson, D., Loch, A., 2017. Achieving environmental flows where buyback is constrained. Aust. J. Agric. Resour. Econ. https://doi.org/10.1111/1467-8489.12231 n/a-n/a.

Adamson, D., Loch, A., 2018. Achieving environmental flows where buyback is constrained. Aust. J. Agric. Resour. Econ. 62, 83–102. https://doi.org/10.1111/1467-8489.12231.

Anderies, J.M., Ryan, P., Walker, B.H., 2006. Loss of resilience, crisis, and institutional change: lessons from an intensive agricultural system in southeastern Australia. Ecosystems 9, 865–878. https://doi.org/10.1007/s10021-006-0017-1.

Andreu, J., Capilla, J., Sanchis, E., 1991. AQUATOOL: acomputer-assisted support system for water resources research management including conjunctive use. In: Loucks, D.P., da Costa, J.R. (Eds.), Decision Support Systems, NATO ASI Series. Springer, Berlin Heidelberg, pp. 333–355.

BOE, 1999. Ley 46/1999 de 13 de diciembre, de modificación de la Ley 29/1985, de 2 de agosto, de Aguas, Law.

BOE, 2001. Real Decreto Legislativo 1/01 del 20 de julio, por el que se aprueba el Texto Refundido de la Ley de Aguas. Royal Decree.

BOE, 2006. Real Decreto-Ley 9/2006, de 15 de septiembre, por el que se adoptan medidas urgentes para paliar los efectos producidos por la sequía en las poblaciones y en las explotaciones agrarias de regadío en determinadas cuencas hidrográficas. Royal Decree.

BOE, 2017. Sentencia de 23 de marzo de 2017, de la Sala Tercera del Tribunal Supremo, que estima en parte el recurso contencioso-administrativo interpuesto contra el Plan Hidrológico de la Demarcación Hidrográfica del Júcar aprobado por el Real Decreto 595/2014, de 11 de julio, y anula su artículo 28.B,d, Sentence.

Burke, S.M., Adams, R.M., Wallender, W.W., 2004. Water banks and environmental water demands: case of the Klamath project. Water Resour. Res. 40 https://doi.org/10.1029/2003WR002832.

Bussettini, M., Linsen, M., Petitguyot, T., Schmidt, G., Smolar-Žvanut, N., Alves, M., Theodoropoulos, C., Skoulikidis, N., Herrero, E., Navarro, R., Tanner, K., Maeso, J., 2015. Ecological Flows in the Implementation of the Water Framework Directive.

Calatrava, J., Martínez-Granados, D., 2012. El valor del uso del agua en el regadío de la cuenca del Segura y en las zonas regables del trasvase Tajo-Segura. Economía agraria y recursos naturales 12, 5–32.

Carlsson, F., Frykblom, P., Liljenstolpe, C., 2003. Valuing wetland attributes: an application of choice experiments. In: Ecological Economics, Management of Eutrophicated Coastal Ecosystems, 47, pp. 95–103. https://doi.org/10.1016/j.ecolecon.2002.09.003.

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.

Connor, J.D., Franklin, B., Loch, A., Kirby, M., Wheeler, S.A., 2013. Trading water to improve environmental flow outcomes. Water Resour. Res. 49, 4265–4276. https://doi.org/10.1002/wrcr.20323.

Cooley, H., Phurisamban, R., Gleick, P., 2019. The cost of alternative urban water supply and efficiency options in California. Environ. Res. Commun. 1, 042001 https://doi.org/10.1088/2515-7620/ab22ca.

Cortignani, R., Severini, S., 2009. Modeling farm-level adoption of deficit irrigation using positive mathematical programming. Agric. Water Manag. 96, 1785–1791. https://doi.org/10.1016/j.agwat.2009.07.016.

Delacámara, G., Gómez, C.M., Maestu, J., 2015. Water trading opportunities and challenges in Europe. In: Burnett, K., Howitt, R., Roumasset, J., Wada, C. (Eds.), Routledge Handbook of Water Economics and Institutions. Routledge, pp. 281–294.

Deltares, 2022. RIBASIM - River Basin Planning and Management [WWW Document]. RIBASIM. URL. https://www.deltares.nl/en/software/ribasim/ (accessed 7.21.22).

Dixon, P., Rimmer, M.T., Wittwer, G., 2011. Saving the southern Murray-Darling basin: the economic effects of a buyback of irrigation water*. Econ. Rec. 87, 153–168. https://doi.org/10.1111/j.1475-4932.2010.00691.x.

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, 2016a. Plan Hidrológico de la Cuenca del Duero 2015–2021 (River Basin Management Plan). Duero River Basin Authority, Valladolid (Spain).
DRBA, 2016b. Plan de Gestión del Riesgo de Inundación (Flood Risk and 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).
EC, 2012. EU Blueprint to Safeguard Europe's Water Resources (Communication No. COM(2012) 673 final). European Commission, Brussels (Belgium).
EC, 2015. 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, 2013. Assessment of Cost Recovery through Water Pricing (Report No. 16/2013). European Environment Agency, Copenhagen (Denmark).
EEA, 2021. Use of freshwater resources in Europe — European Environment Agency [WWW Document]. In: Use of Freshwater Resources in Europe. European Environment Agency. URL. https://www.eea.europa.eu/data-and-maps/indicators/use-of-freshwater-resources-3/assessment-4 (accessed 12.14.21).
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.
García-Mollá, M., Sanchis-Ibor, C., Ortega-Reig, M.V., Avellá-Reus, L., 2013. Irrigation associations coping with drought: The case of four irrigation districts in Eastern Spain. In: Schwabe, K., Albiac, J., Connor, J.D., Hassan, R.M., González, L.M. (Eds.), Drought in Arid and Semi-Arid Regions. Springer, Netherlands, pp. 101–122. https://doi.org/10.1007/978-94-007-6636-5_6.
Gómez, C.M., Pérez-Blanco, C.D., Adamson, D., Loch, A., 2017. Managing water scarcity at a river basin scale with economic instruments. Water Econs. Pol. 04, 1750004. https://doi.org/10.1142/S2382624X17500047.
Gómez-Limón, J.A., Gutiérrez-Martín, C., Montilla-López, N.M., 2020. Recovering water for the environment during droughts through public water banks within a monopsony-monopoly setting. Water Res. Econom. 32, 100163 https://doi.org/10.1016/j.wre.2020.100163.
Graveline, N., 2016. Economic calibrated models for water allocation in agricultural production: a review. Environ. Model. Softw. 81, 12–25. https://doi.org/10.1016/j.envsoft.2016.03.004.
GRBA, 2008. Plan Especial del Alto Guadiana. Guadiana River Basin Authority.
Gutiérrez-Martín, C., Gómez, C.M., 2011. Assessing irrigation efficiency improvements by using a preference revelation model. Span. 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: Bournaris, T., Berbel, J., Manos, B., Viaggi, D. (Eds.), Economics of Water Management in Agriculture, pp. 319–348. Boca Raton (US).
Gutiérrez-Martín, C., Gómez-Limón, J.A., Montilla-López, N.M., 2020. Self-financed water bank for resource reallocation to the environment and within the agricultural sector. Ecol. Econ. 169, 106493 https://doi.org/10.1016/j.ecolecon.2019.106493.
Hanak, E., Stryjewski, E., 2012. California's Water Market, by the Numbers, Update 2012 - Technical Appendix (Report - Appendix No. Update November 2012). Public Policy Institute of California.
Hassan, F., 2010. Water History for Our Times, IHP Essays on Water History. UNESCO International Hydrological Programme, Paris (France).
Heckelei, T., Britz, W., 2005. Models based on positive mathematical programming: state of the art and futher extensions. In: Proceedings of the 89th EAAE Seminar. Parma (Italy), pp. 48–73.
Hino, M., Hall, J.W., 2017. Real options analysis of adaptation to changing flood risk: structural and nonstructural measures. ASCE-ASME J. Risk Uncertain. Eng. Syst. Part A: Civil Eng. 3, 04017005. https://doi.org/10.1061/AJRUA6.0000905.
Hollinshead, S.P., Lund, J.R., 2006. Optimization of environmental water purchases with uncertainty. Water Resour. Res. 42 https://doi.org/10.1029/2005WR004228.
Honey-Rosés, J., 2008. Estimating the Social Benefits of Protecting Minimum Instream Flows in a Mediterranean Watershed - the Case of the Ter River in Catalonia (NE Spain) (Report). Generalitat de Catalunya- Departament de Medi Ambienti Habitatge, Barcelona (Spain).
Howitt, R.E., 1995. A calibration method for agricultural economic production models. J. Agric. Econ. 46, 147–159. https://doi.org/10.1111/j.1477-9552.1995.tb00762.x.
Huffaker, R., Whittlesey, N., 2000. The allocative efficiency and conservation potential of water laws encouraging investments in on-farm irrigation technology. Agric. Econ. 24, 47–60. https://doi.org/10.1016/S0169-5150(00)00114-6.
Iftekhar, M.S., Tisdell, J.G., Connor, J.D., 2013. Effects of competition on environmental water buyback auctions. Agric. Water Manag. 127, 59–73. https://doi.org/10.1016/j.agwat.2013.05.015.
IPCC, 2014. IPCC fifth assessment report (AR5) (No. WGII). In: Intergovernmental Panel on Climate Change, Geneva (Switzerland).
IPCC, 2019. Special Report on Climate Change and Land. Intergovernmental Panel on Climate Change, Geneva (Switzerland).
ITACyL, 2019. Mapa de Cultivos y Superficies Naturales de Castilla y León [WWW Document]. Instituto Tecnológico Agrario de Castilla y León. URL. http://mcsncyl.itacyl.es/es/inicio.
Jääskinen, N., 2014. Case C-525/12 European Commission V Federal Republic of Germany.
JRC, 2022. LISFLOOD [WWW Document]. LISFLOOD. URL. https://web.jrc.ec.europa.eu/policy-model-inventory/explore/models/model-lisflood (accessed 7.21.22).
Kahil, M.T., Dinar, A., Albiac, J., 2015. Modeling water scarcity and droughts for policy adaptation to climate change in arid and semiarid regions. J. Hydrol. 522, 95–109. https://doi.org/10.1016/j.jhydrol.2014.12.042.
Kirby, M., Qureshi, M.E., Mainuddin, M., Dyack, B., 2006. Catchment behavior and counter-cyclical water trade: an integrated model. Nat. Resour. Model. 19, 483–510. https://doi.org/10.1111/j.1939-7445.2006.tb00191.x.
Konishi, S., Kitagawa, G., 2008. Information Criteria and Statistical Modeling, Springer Series in Statistics. Springer-Verlag, New York.
Lane-Miller, C.C., Wheeler, S., Bjornlund, H., Connor, J., 2013. Acquiring water for the environment: lessons from natural resources management. J. Environ. Policy Plan. 15, 513–532. https://doi.org/10.1080/1523908X.2013.807210.
Lerma, N., Paredes, J., Solera, A., Andreu, J., 2017. Herramienta EvalHid para la evaluación de recursos hídricos (Manual No. 1.1). Polytechnic University of Valencia, Valencia, Spain.
Lindhout, P.E., Broek, Den Berthy, V., 2014. The Polluter Pays Principle: Guidelines for Cost Recovery and Burden Sharing in the Case Law of the European Court of Justice (SSRN Scholarly Paper No. ID 2436984). Social Science Research Network, Rochester, NY.
Loch, A., Gregg, D., 2018. Salinity management in the Murray-Darling basin: a transaction cost study. Water Resour. Res. 54, 8813–8827. https://doi.org/10.1029/2018WR022912.
Loch, A., Wheeler, S., Boxall, P., Hatton-Macdonald, D., Adamowicz, W.L., Bjornlund, H., 2014. Irrigator preferences for water recovery budget expenditure in the Murray-Darling Basin, Australia. Land Use Policy 36, 396–404. https://doi.org/10.1016/j.landusepol.2013.09.007.
Loch, A., Adamson, D., Dumbrell, N.P., 2020a. The fifth stage in water management: policy lessons for water governance. Water Resour. Res. 56 https://doi.org/10.1029/2019WR026714 e2019WR026714.
Loch, A., Pérez-Blanco, C.D., Carmody, E., Felbab-Brown, V., Adamson, D., Seidl, C., 2020b. Grand theft water and the calculus of compliance. Nat. Sustainab. 3, 1012–1018. https://doi.org/10.1038/s41893-020-0589-3.
Loomis, J., 2002. Quantifying recreation use values from removing dams and restoring free-flowing rivers: a contingent behavior travel cost demand model for the Lower Snake River. Water Resour. Res. 38 https://doi.org/10.1029/2000WR000136, 2-1-2–8.
Loomis, J., Kent, P., Strange, L., Fausch, K., Covich, A., 2000. Measuring the total economic value of restoring ecosystem services in an impaired river basin: results from a contingent valuation survey. Ecol. Econ. 33, 103–117. https://doi.org/10.1016/S0921-8009(99)00131-7.
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).
Matthews, R.C.O., 1986. The economics of institutions and the sources of growth. Econ. J. 96, 903–918. https://doi.org/10.2307/2233164.
MDBA, 2012. Water Act 2007 – Basin Plan (River Basin Management Plan). Murray-Darling Basin Authority.
Mendelsohn, R.O., 2016. Adaptation, climate change, agriculture, and water. Choices Quarter 3.
Moore, M.R., Doubek, J.P., Xu, H., Cardinale, B.J., 2020. Hedonic Price estimates of Lake water quality: valued attribute, instrumental variables, and ecological-economic benefits. Ecol. Econ. 176, 106692 https://doi.org/10.1016/j.ecolecon.2020.106692.
Njiraini, G.W., Thiam, D.R., Coggan, A., 2017. The analysis of transaction costs in water policy implementation in South Africa: trends, determinants and economic implications. Water Econs. Pol. 03, 1650020. https://doi.org/10.1142/S2382624X1650020X.
OECD, 2015. Water Resources Allocation: Sharing Risks and Opportunities, OECD Studies on Water. OECD Publishing.
OJ, 2000. Water Framework Directive 2000/60/EC. Council Directive.
OJ, 2012. Consolidated version of the treaty on the functioning of the European Union. Off. J. Eur. Union C 326, 47–199.
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.
Pérez-Blanco, C.D., Gutiérrez-Martín, C., 2017. Buy me a river: use of multi-attribute non-linear utility functions to address overcompensation in agricultural water buyback. Agric. Water Manag. 190, 6–20. https://doi.org/10.1016/j.agwat.2017.05.006.
Pérez-Blanco, C.D., Sapino, F., 2022. Economic sustainability of irrigation-dependent ecosystem services under growing water scarcity. Insights from the Reno River in Italy. Water Resour. Res. 58 https://doi.org/10.1029/2021WR030478 e2021WR030478.
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., Essenfelder, A.H., Gutiérrez-Martín, C., 2020. A tale of two rivers: integrated hydro-economic modeling for the evaluation of trading opportunities and return flow externalities in inter-basin agricultural water markets. J. Hydrol. 584, 124676 https://doi.org/10.1016/j.jhydrol.2020.124676.
Pérez-Blanco, C.D., Gil-García, L., Saiz-Santiago, P., 2021a. An actionable hydroeconomic decision support system for the assessment of water reallocations in irrigated agriculture. A study of minimum environmental flows in the Douro River Basin, Spain. J. Environ. Manag. 298, 113432 https://doi.org/10.1016/j.jenvman.2021.113432.

Pérez-Blanco, C.D., González-López, H., Hrast-Essenfelder, A., 2021b. Beyond piecewise methods: modular integrated hydroeconomic modeling to assess the impacts of adaptation policies in irrigated agriculture. Environ. Model. Softw. 136, 104943 https://doi.org/10.1016/j.envsoft.2020.104943.

Pindyck, R.S., 2015. The Use and Misuse of Models for Climate Policy (Working Paper No. 21097). National Bureau of Economic Research.

PUV, 2020. Manuals – AquaTool [WWW Document]. Manuals – AquaTool. URL. https://aquatool.webs.upv.es/aqt/en/manuals/ (accessed 1.19.21).

Qureshi, M.E., Connor, J., Kirby, M., Mainuddin, M., 2007. Economic assessment of acquiring water for environmental flows in the Murray Basin*. Aust. J. Agric. Resour. Econ. 51, 283–303. https://doi.org/10.1111/j.1467-8489.2007.00383.x.

Refsgaard, J.C., Storm, B., 1995. MIKE SHE. Computer Models of Watershed Hydrology, pp. 809–846.

Rey, D., 2014. Water Option Contracts for Reducing Water Supply Risks: an Application to the Tagus-Segura Transfer (phd). E.T.S.I. Agrónomos (UPM).

Rey, D., Pérez-Blanco, C.D., Escriva-Bou, A., Girard, C., Veldkamp, T.I.E., 2019. Role of economic instruments in water allocation reform: lessons from Europe. Int. J. Water Res. Develop. 35, 206–239. https://doi.org/10.1080/07900627.2017.1422702.

Sapino, F., Pérez-Blanco, C.D., Gutiérrez-Martín, C., Frontuto, V., 2020. An ensemble experiment of mathematical programming models to assess socio-economic effects of agricultural water pricing reform in the Piedmont Region, Italy. J. Environ. Manag. 267, 110645 https://doi.org/10.1016/j.jenvman.2020.110645.

Seidl, C., Wheeler, S.A., Zuo, A., 2020a. Treating water markets like stock markets: key water market reform lessons in the Murray-Darling Basin. J. Hydrol. 581, 124399 https://doi.org/10.1016/j.jhydrol.2019.124399.

Seidl, C., Wheeler, S.A., Zuo, A., 2020b. High turbidity: water valuation and accounting in the Murray-Darling Basin. Agric. Water Manag. 230, 105929 https://doi.org/10.1016/j.agwat.2019.105929.

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.

Tebaldi, C., Knutti, R., 2007. The use of the multi-model ensemble in probabilistic climate projections. Philos. Trans. R. Soc. A Math. Phys. Eng. Sci. 365, 2053–2075. https://doi.org/10.1098/rsta.2007.2076.

Tonkin, J.D., Merritt, D.M., Olden, J.D., Reynolds, L.V., Lytle, D.A., 2018. Flow regime alteration degrades ecological networks in riparian ecosystems. Nat. Ecol. Evolut. 2, 86–93. https://doi.org/10.1038/s41559-017-0379-0.

UN, 2020. The United Nations World Water Development Report 2020 (Report).. United Nations.

UN, 2021. The United Nations World Water Development Report 2021 (Report).. United Nations.

US EPA, 2019. Estimating the Value of Water: A Literature Review (Overviews and Factsheets). Environment Protection Agency, Washington D.C. (US).

van Dijk, D., Siber, R., Brouwer, R., Logar, I., Sanadgol, D., 2016. Valuing water resources in Switzerland using a hedonic price model. Water Resour. Res. 52, 3510–3526. https://doi.org/10.1002/2015WR017534.

World Bank, 2017. Uncharted Waters: The New Economics of Water Scarcity and Variability. World Bank Publications, S.l.

Yates, D., Sieber, J., Purkey, D., Huber-Lee, A., 2005. WEAP21–A Demand-, Priority-, and Preference-Driven Water Planning Model: Part 1, Model Characteristics.

Young, M.D., 2014. Designing water abstraction regimes for an ever-changing and ever-varying future. In: Agricultural Water Management, Exploring Some of the socio-Economic Realities of Sustainable Water Management in Irrigation, 145, pp. 32–38. https://doi.org/10.1016/j.agwat.2013.12.002.

Chapter 5. Conclusions and recommendations

"Scarcity and misuse of fresh water pose a serious and growing threat to sustainable development and protection of the environment. Human health and welfare, food security, industrial development and the ecosystems on which they depend, are all at risk, unless water and land resources are managed more effectively in the present decade and beyond than they have been in the past"
Eirah Gorre-Dale, The Dublin Statement on Water and Sustainable Development, 1992.

Water is one of the most complex goods humans need to manage: it is essential for life and economic activity, has many public and private uses, is vulnerable to freeriding, and is a heterogeneous and finite good in increasingly complex-to-manage hydrological systems. As water was historically abundant in many parts of the world, humans have approached water resources management through engineering solutions (i.e., increasing storage and/or transporting the resource from water-abundant to water-scarce places). This is no longer possible in mature water economies, where the growing costs of water provision due to inelastic water supply increasingly exceed the benefits of its use. Moreover, conventional water management policies have systematically failed to reach their objectives due to poor design and/or unaccounted feedback between ecological and socioeconomic systems. This unsustainable situation calls for additional modeling efforts that can support the design and implementation of robust adaptation strategies that address water scarcity to preserve both human wellbeing and the good ecological status of water bodies.

To this end, this thesis provides a methodology that quantifies and assesses modeling and scenario uncertainties to inform robust adaptation strategies to water scarcity in the agricultural sector. This chapter encompasses a summary of the findings and depicts the general conclusions of the author, highlighting the achievements and limitations of his work. In the first chapter, we showed how water scarcity is rapidly growing due to climate change in many parts of the world and how this threatens water bodies and human wellbeing. Then we introduced the concept of deep uncertainty and argued that under deeply uncertain conditions, we should fundamentally reassess our approach to the design and implementation of policies. The non-consideration of uncertainty is indeed one of the main reasons water policies consistently fail to achieve their objectives (Marchau et al., 2019). From chapters 2 to 4, we led the reader in a step-by-step construction of a modular multi-system modeling framework that can inform and assess the performance of robust adaptation strategies in coupled ecological and socioeconomic systems. These chapters include the five publications produced during the Ph.D. by the author. In each of these publications, the modeling framework is progressively developed and applied to a real policy case thanks to the collaborations existing with river basin authorities. In chapter 2, we presented the microeconomic methodology and briefly reviewed the large literature that supports it; then, with a new model enclosed in the first publication (2.2), we expanded the literature on multi-attribute MPMs. In the following chapter, we presented a methodology to address deep uncertainty through an innovative multi-model ensemble of MPMs and its application with two different policy cases (3.2 and 3.3). Finally, we strengthened our approach by explicitly including the water system in our modeling framework that became a multi-system (human-water) ensemble, and we applied this framework to two other policy cases (4.2 and 4.3). The modeling complexity grows through the chapters, but the modularity of the ensembles allows to understand it more smoothly. Moreover, a modeling framework composed of different modules results in easier changes and updates to adapt to the different case studies. Note that adaptability and updatability are key recommendations of RDM theory (see Box 1).

The main **findings** of this thesis are as follows. *First,* we included a water-production function in the PMAUP model, which allowed us to consider and explore the repercussions of intensive margin adaptation (i.e., deficit irrigation) under water scarcity, commonly not considered by conventional microeconomic models. We found that excluding deficit irrigation can lead to underestimating water savings and overestimating profit loss (up to 8% in our application of a water pricing policy). *Second,* to advise robust policies we must consider modeling uncertainty in the socioeconomic system. This is achieved by developing a multi-model ensemble of MPMs and assessing its results with the mechanistic algorithms Minmax and Minmax regret. Using different MPMs is critical to avoid biases emerging from model choice. Our multi-model ensemble provides insights to advise policy and highlights tipping points (3.2) and other non-trivial outcomes (e.g., that full cost recovery is not recommended to reach the WDF objectives in our case study in 4.2). *Third,* our coupled human-water modeling framework, inspired by the socio-hydrology and hydroeconomics literature (Harou et al., 2009; Heinz et al., 2007; Sivapalan et al., 2014), provides critical information to assess cascading impacts across coupled human and water systems while quantifying the associated uncertainty. The interactions between the two systems generate new outcomes that could not be predicted by each system in isolation.

The key **policy findings** of our research are as follow. *First*, following previous research by Adamson and Loch (2021) and Pérez-Blanco et al. (2020b), our multi-model ensemble results suggest that “subsidies to modern irrigation technologies are not only ineffective to save water—they can also exacerbate water scarcity by increasing the consumed fraction of water applied”(Pérez-Blanco and Sapino, 2022). This finding dispraises the effort of many countries and international organizations to subsidize modern irrigation techniques as an effective policy to reduce water consumption. Instead, we encourage complementing subsidies with other policies (e.g., quotas or pricing) to reach the same objective at a lower cost and with higher economic efficiency. *Second*, the multi-system modeling framework allowed us to evaluate the resource cost of agricultural water, a key part of the total cost of water as defined by the EU WFD, usually not recovered with the price. Our results indicate that the recovery of this cost can have a disproportionate impact on farmers' income; thus, we suggest that, at least for our case study, other policies should be used to reach the environmental objectives of the WFD. *Third*, we used our multi-system ensemble to assess the economic performance of a water bank, another relevant policy option to reallocate water to the environment and to the most productive users in water-scarce basins. We found that water banks can achieve the same environmental objectives as a buyback program but with a significantly lower cost for the regulatory authority.

Despite the abovementioned achievements, we **recommend** addressing some issues through further research to improve the framework presented in this thesis. *First*, the MPMs ensemble can be expanded with other models, notably, PMP with a CES (Constant Elasticity of Substitution) water-production function (Graveline and Mérel, 2014; Mérel et al., 2011) or with the inclusion of risk (Arata et al., 2017; Jansson et al., 2014; Petsakos and Rozakis, 2015). *Second*, another possible improvement concerning the multi-attribute models (both PMAUP and the linear MPM), regards the exploration of other relevant attributes that can be included in the decision process of the economic agent (the farmer). *Third*, the multi-system ensemble can be expanded by adding multiple hydrologic models rather than one (i.e., an ensemble of hydrologic models). This would allow us to take advantage of the different specificities of the models to sample structural and parameter uncertainties within models. *Fourth*, other human systems can be added to our modular framework to explore their feedback with the water and human systems presently incorporated, e.g., an ABM (Agent Based Model) model to explore water market performance. This is currently being addressed by the author (Sapino et al., under review). *Fifth*, more effort should be made to estimate the economic value of environmental uses of water with a modeling framework such as the one presented in this thesis to have comparable estimates of the value of water for alternative uses, although a major constraint is the lack of data on demand, which requires ad hoc valuation with significant uncertainty bounds. Noteworthy, in the presence of mature water markets in which irrigators and actors who act on behalf of the environment are free to operate (e.g., in Australia), these exercises may be redundant since exchange prices and quantities can be observed in the market and provide an observable measure for valuation. *Finally*, it should be noted that changes at the microeconomic level (e.g., a reduction/increase in the production of a crop) affect the regional/global economy with possible impacts on commodity prices. We can assess changes in prices with a macroeconomic model, which could be coupled with our microeconomic models to assess feedback, e.g., exploring how adaptation responses by irrigators can affect the prices of agricultural commodities traded in regional, national and international markets, depending on the granularity of the macroeconomic model adopted (Britz, 2014; Parrado et al., 2019). Although the coupling of micro- and macro-economic systems has already been explored in the literature, e.g., in EU CAPRI (Common Agricultural Policy Regionalised Impact Modelling System) model (Britz, 2014), there is no ensemble available that couples multiple microeconomic (e.g., MPMs) models with one or multiple macroeconomic models to sample uncertainty. Therefore, we recommend expanding our multi-system modeling framework with a macroeconomic system populated with one or multiple models.

Capítulo 5. Conclusiones y recomendaciones

El agua es uno de los bienes más complejos que los seres humanos necesitan gestionar: es esencial para la vida y la actividad económica, tiene muchos usos públicos y privados, es vulnerable al *freeriding* y es un bien heterogéneo y finito en sistemas hidrológicos cada vez más complejos de gestionar. Dado que, históricamente, el agua era abundante en muchas partes del mundo, los seres humanos han abordado la gestión de los recursos hídricos a través de soluciones de ingeniería (es decir, el aumento del almacenamiento y/o el transporte desde lugares abundantes a lugares escasos en agua). Esto ya no es posible en economías hídricas maduras, donde los costes crecientes del suministro de agua debido a la inelasticidad de la provisión de agua superan cada vez más los beneficios de su uso. Además, las políticas convencionales de gestión de los recursos hídricos han fracasado sistemáticamente en alcanzar sus objetivos debido a un diseño mediocre y/o a una retroalimentación no contabilizada entre los sistemas ecológicos y socioeconómicos. Esta situación insostenible requiere esfuerzos adicionales en desarrollar marcos de modelos que puedan apoyar el diseño y la implementación de estrategias de adaptación robustas que aborden la escasez de agua para preservar tanto el bienestar humano como el buen estado ecológico de las masas de agua.

Con este fin, esta tesis proporciona una metodología que cuantifica y evalúa la modelización y las incertidumbres de los escenarios para fundamentar estrategias de adaptación robustas a la escasez de agua en el sector agrícola. Este capítulo contiene un resumen de los hallazgos y describe las conclusiones generales del autor, destacando los logros y limitaciones de su trabajo.

En el primer capítulo, mostramos cómo la escasez de agua está creciendo rápidamente debido al cambio climático en muchas partes del mundo y cómo esto amenaza los recursos hídricos y el bienestar humano. Luego introducimos el concepto de incertidumbre profunda y sostenemos que, en esa condición, deberíamos reevaluar fundamentalmente nuestro enfoque de diseño y aplicación de políticas. La no consideración de la incertidumbre es de hecho una de las principales razones por las que las políticas de agua fallan consistentemente en lograr sus objetivos (Marchau et al., 2019). De los capítulos 2 a 4, guiamos al lector en una construcción paso a paso de un marco modular de modelos que consideran múltiples sistemas con el fin de informar y evaluar el desempeño de estrategias de adaptación robustas en sistemas ecológicos y socioeconómicos acoplados. Estos capítulos incluyen las 5 publicaciones producidas durante mi doctorado. En cada una de estas publicaciones, el marco de modelización se desarrolla progresivamente y se aplica a un verdadero caso político gracias a las colaboraciones existentes con las autoridades de cuenca. En el capítulo 2, presentamos la metodología microeconómica (Modelos de Programación Matemática – MPMs) y repasamos brevemente la gran literatura que la apoya; luego, con un nuevo modelo incluido en la primera publicación (2.2), ampliamos la literatura sobre los MPMs multiatributo. En el siguiente capítulo, presentamos una metodología para abordar la incertidumbre profunda a través de un innovador conjunto multimodelo de MPMs (3.2, 3.3) y su aplicación con dos casos de políticas diferentes. Finalmente, reforzamos nuestro enfoque incluyendo explícitamente el sistema hidrológico en nuestro marco de modelos que se convirtió en un conjunto multisistémico (humano-hidrológico), y aplicamos este marco a otros dos casos de políticas (4.2 y 4.3). La complejidad del marco de modelos crece a través de los capítulos, pero la modularidad del conjunto permite entenderlo más fácilmente. Además, un marco de modelos compuesto por diferentes módulos permite cambios y actualizaciones más fáciles para adaptarse a los diferentes casos de estudios. Obsérvese que la adaptabilidad y la posibilidad de actualizar son recomendaciones clave de la teoría de la RDM (véase el Box 1).

Las principales conclusiones de esta tesis son las siguientes. *Primero*, incluimos una función de producción de agua en el modelo PMAUP, que nos permitió considerar y explorar las repercusiones de la adaptación intensiva de márgenes (es decir, riego deficitario) a la escasez de agua, comúnmente no considerada por los modelos microeconómicos convencionales. Encontramos que la exclusión del riego deficitario puede llevar a subestimar el ahorro de agua y sobreestimar la pérdida de beneficios (hasta un 8% en nuestra aplicación de una política de precios del agua). *Segundo*, para aconsejar políticas robustas debemos considerar y modelar la incertidumbre en el sistema socioeconómico. Esto se logra mediante el desarrollo de un conjunto multimodelo de MPMs y la evaluación de sus resultados con los algoritmos mecanicistas Minmax y Minmax regret. El uso de diferentes MPMs es fundamental para evitar sesgos que surgen de la elección del modelo. Nuestro conjunto de modelos múltiples proporciona información para asesorar sobre políticas y destaca puntos de inflexión (3.2) y otros resultados interesantes (por ejemplo, que no se recomienda la recuperación total de costos para alcanzar los objetivos del WDF en nuestro estudio de caso en 4.2.). *Tercero*, nuestro marco de modelos hidrológicos y económicos, inspirado en la literatura sociohidrológica e hidroeconómica (Harou et al., 2009; Heinz et al., 2007a; Sivapalan et al., 2014), proporciona información crítica para evaluar los impactos en cascada a través de los sistemas hidrológicos y socioeconómicos acoplados mientras cuantifica la incertidumbre asociada. Las interacciones entre los dos sistemas generan nuevos resultados que no podían ser predichos por cada sistema de forma aislada. *Cuarto*, utilizamos nuestro conjunto multisistema para evaluar el desempeño económico de un banco de agua, otra opción política relevante para reasignar el agua al medio ambiente y a los usuarios más productivos en cuencas con escasez de agua.

Los principales resultados a nivel político de nuestra investigación son los siguientes. En primer lugar, siguiendo investigaciones anteriores de Adamson y Loch (2021) y Pérez-Blanco et al. (2020b), nuestros resultados de conjunto multimodelo sugieren que los subsidios a las modernas tecnologías de riego no solo son ineficaces para ahorrar agua, sino que también pueden aumentar la escasez de agua al aumentar la fracción consumida de agua aplicada (Pérez-Blanco y Sapino, 2022). Este hallazgo desmiente el esfuerzo de muchos países y organizaciones internacionales para subsidiar las técnicas modernas de riego como una política eficaz para reducir el consumo de agua. En su lugar, alentamos a complementar los subsidios con otras políticas (por ejemplo, cuotas o precios) para alcanzar el mismo objetivo a un menor costo y con una mayor eficiencia económica. En segundo lugar, el marco de modelos multisistema nos permitió evaluar el coste de los recursos de agua agrícola, una parte clave del coste total del agua definido por la DMA (Directiva Marco del Agua) de la UE, que por lo general no se recupera con el precio. Nuestros resultados indican que la recuperación de este coste puede tener un impacto desproporcionado en la renta de los agricultores; por lo tanto, sugerimos que, al menos para nuestro caso de estudio, se utilicen otras políticas para alcanzar los objetivos medioambientales de la DMA. En tercer lugar, encontramos que los bancos de agua pueden lograr los mismos objetivos ambientales que un programa de recompra, pero con un costo significativamente menor para la autoridad reguladora.

A pesar de los logros antes mencionados, recomendamos abordar algunos temas a través de investigaciones adicionales para mejorar el marco presentado en esta tesis. En primer lugar, el ensemble de MPMs puede ampliarse con otros modelos, en particular, PMP con una función de producción de agua CES (Graveline y Mérel, 2014; Mérel et al., 2011) o con la inclusión del riesgo (Arata et al., 2017; Jansson et al., 2014; Petsakos y Rozakis, 2015). En segundo lugar, otra posible mejora relativa a los modelos de múltiples atributos (tanto PMAUP como el MPM lineal), se refiere a la exploración de otros atributos relevantes que pueden incluirse en el proceso de decisión del agente económico (el agricultor). En tercer lugar, el conjunto multisistémico puede ampliarse añadiendo múltiples modelos hidrológicos en lugar de uno (es decir, un conjunto de modelos hidrológicos). Esto nos permitiría aprovechar las diferentes especificidades de los modelos para muestrear incertidumbres estructurales y de parámetros dentro de los modelos. Cuarto, se pueden agregar otros sistemas socioeconómicos a nuestro marco de modelos para explorar su retroalimentación con los sistemas de agua y humanos actualmente incorporados, por ejemplo, un modelo ABM para explorar el rendimiento del mercado de agua. Esto está siendo abordado por el autor (Sapino et al., en revisión). Quinto, se debe hacer más esfuerzo para estimar el valor económico de los usos ambientales del agua con un marco de modelos, como el presentado en esta tesis, para tener estimaciones comparables del valor del agua para usos alternativos, aunque una limitación importante es la falta de datos sobre la demanda, que requiere una valoración ad hoc con importantes límites de incertidumbre. Cabe destacar que, en presencia de mercados de agua maduros, donde los regantes y los agentes que actúan en nombre del medio ambiente son libres de operar (por ejemplo, en Australia), estos ejercicios pueden ser redundantes ya que los precios de cambio y las cantidades se pueden observar en el mercado y proporcionan una medida observable para la valoración. Por último, cabe señalar que los cambios a nivel microeconómico (por ejemplo, una reducción/aumento de la producción de un cultivo) afectan a la economía mundial con posibles efectos en los precios de los productos básicos. Podemos evaluar los cambios en los precios con un modelo macroeconómico, que podría combinarse con nuestros modelos microeconómicos para evaluar la retroalimentación, por ejemplo, explorar cómo las respuestas de adaptación de los regantes pueden afectar los precios de los productos agrícolas comercializados en la región, mercados nacionales e internacionales, dependiendo de la granularidad del modelo macroeconómico adoptado (Britz, 2014; Parrado et al., 2019). Aunque el acoplamiento de los sistemas micro y macroeconómicos ya se ha explorado en la literatura, por ejemplo, en el modelo CAPRI de la UE (Britz, 2014), no hay ningún conjunto disponible que combine múltiples modelos microeconómicos (por ejemplo, MPM) con uno o varios modelos macroeconómicos para considerar la incertidumbre. Por lo tanto, recomendamos ampliar nuestro marco de modelos multisistema con un sistema macroeconómico poblado con uno o varios modelos.

References

Adamson, D., Loch, A., 2021. Incorporating uncertainty in the economic evaluation of capital investments for water use efficiency improvements. Land Economics 0, 1–15.

AdBPo (Autorità di Bacino distrettuale del fiume Po), 2022. Osservatorio permanente sugli utilizzi idrici nel distretto idrografico del fiume Po. Bollettino n 17/2022 del 12/10/2022.

AgMIP, 2023. Agricultural Model Intercomparison and Improvement Project [WWW Document]. agmiporg. URL https://agmip.org/ (accessed 1.9.23).

Ajzen, I., 1991. The theory of planned behavior. Organizational Behavior and Human Decision Processes, Theories of Cognitive Self-Regulation 50, 179–211. https://doi.org/10.1016/0749-5978(91)90020-T

Arata, L., Donati, M., Sckokai, P., Arfini, F., 2017. Incorporating risk in a positive mathematical programming framework: a dual approach. Australian Journal of Agricultural and Resource Economics 61, 265–284. https://doi.org/10.1111/1467-8489.12199

Bartolini, F., Bazzani, G.M., Gallerani, V., Raggi, M., Viaggi, D., 2007. The impact of water and agriculture policy scenarios on irrigated farming systems in Italy: An analysis based on farm level multi-attribute linear programming models. Agricultural Systems 93, 90–114. https://doi.org/10.1016/j.agsy.2006.04.006

Box, G.E.P., 1979. Robustness in the Strategy of Scientific Model Building, in: Launer, R.L., Wilkinson, G.N. (Eds.), Robustness in Statistics. Academic Press, pp. 201–236. https://doi.org/10.1016/B978-0-12-438150-6.50018-2

Britz, W., 2014. The Graphical User Interface for CAPRI version 2014.

Buysse, J., Van Huylenbroeck, G., Lauwers, L., 2007. Normative, positive and econometric mathematical programming as tools for incorporation of multifunctionality in agricultural policy modelling. Agriculture, Ecosystems & Environment, Multifunctionality of Agriculture: Tools and Methods for Impact Assessment and Valuation 120, 70–81. https://doi.org/10.1016/j.agee.2006.03.035

Caretta, M.A., Mukherji, A., Arfanuzzaman, M., Betts, R.A., Gelfan, A., Hirabayashi, Y., Lissner, T.K., Liu, J., Lopez Gunn, E., Morgan, R., Mwanga, S., Supratid, S., 2022. Water, in: Climate Change 2022: Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [H.-O. Pörtner, D.C. Roberts, M. Tignor, E.S. Poloczanska, K. Mintenbeck, A. Alegría, M. Craig, S. Langsdorf, S. Löschke, V. Möller, A. Okem, B. Rama (Eds.)]. Cambridge University Press.

Cherlet, M., Hutchinson, C., Reynolds, J., Hill, J., Sommer, S., von Maltitz, G., 2018. World atlas of desertification: rethinking land degradation and sustainable land management. Publications Office, LU.

CIMIP, 2022. WCRP (World Climate Research Programme) Coupled Model Intercomparison Project - CMIP [WWW Document]. URL https://www.wcrp-climate.org/wgcm-cmip (accessed 7.5.22).

Cloke, H.L., Pappenberger, F., van Andel, S.J., Schaake, J., Thielen, J., Ramos, M.H., 2013. Hydrological ensemble prediction systems. Hydrol. Process. 27, 1–4. https://doi.org/10.1002/hyp.9679

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.

Dinar, A., Schwabe, K., 2015. Handbook of Water Economics, Handbook of Water Economics. Edward Elgar Publishing.

Erfani, T., Binions, O., Harou, J.J., 2014. Simulating water markets with transaction costs. Water Resources Research 50, 4726–4745. https://doi.org/10.1002/2013WR014493

FAO (Food and Agriculture Organization), 2021. The State of the World's Land and Water Resources for Food and Agriculture – Systems at breaking point (SOLAW 2021): Synthesis report 2021. FAO, Rome, Italy. https://doi.org/10.4060/cb7654en

FAO, UN Water (United Nations Water), 2021. Progress on the level of water stress: Global status and acceleration needs for SDG indicator 6.4.2, 2021. FAO, United Nations Water (UN Water), Rome, Italy. https://doi.org/10.4060/cb6241en

Frenken, K., 2006. Technological innovation and complexity theory. Economics of Innovation and New Technology 15, 137–155. https://doi.org/10.1080/10438590500141453

Gómez-Limón, J.A., Gutiérrez-Martín, C., Riesgo, L., 2016. Modeling at farm level: Positive Multi-Attribute Utility Programming. Omega 65, 17–27. https://doi.org/10.1016/j.omega.2015.12.004

Gorre-Dale, E., 1992. The Dublin Statement on Water and Sustainable Development. Environmental Conservation 19, 181–181. https://doi.org/10.1017/S0376892900030733

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., Loubier, S., Gleyses, G., Rinaudo, J.-D., 2012. Impact of farming on water resources: Assessing uncertainty with Monte Carlo simulations in a global change context. Agricultural Systems 108, 29–41. https://doi.org/10.1016/j.agsy.2012.01.002

Graveline, N., Majone, B., Van Duinen, R., Ansink, E., 2014. Hydro-economic modeling of water scarcity under global change: an application to the Gállego river basin (Spain). Reg Environ Change 14, 119–132. https://doi.org/10.1007/s10113-013-0472-0

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 Journal of Agricultural Research 9, 1009–1020. https://doi.org/10.5424/sjar/20110904-514-10

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
Heckelei, T., Britz, W., Zhang, Y., 2012. Positive Mathematical Programming Approaches - Recent Developments in Literature and Applied Modelling. Bio-based and Applied Economics Journal 1, 109–124.
Heinz, I., Pulido-Velazquez, M., Lund, J.R., Andreu, J., 2007. Hydro-economic Modeling in River Basin Management: Implications and Applications for the European Water Framework Directive. Water Resour Manage 21, 1103–1125. https://doi.org/10.1007/s11269-006-9101-8
HEPEX, 2023. Hydrologic Ensemble Prediction EXperiment - HEPEX [WWW Document]. URL https://hepex.inrae.fr/ (accessed 1.9.23).
IPCC (Intergovernmental Panel on Climate Change), 2021. IPCC Sixth Assessment Report (AR6) (No. WGII). Intergovernmental Panel on Climate Change, Geneva (Switzerland).
ISIMIP, 2023. The Inter-Sectoral Impact Model Intercomparison Project [WWW Document]. The Inter-Sectoral Impact Model Intercomparison Project. URL https://www.isimip.org/gettingstarted/data-access/ (accessed 1.9.23).
Jansson, T., Heckelei, T., Gocht, A., Basnet, S.K., Zhang, Y., Neuenfeldt, S., 2014. Analysing impacts of changing price variability with estimated farm risk-programming models [WWW Document]. AgEcon Search. https://doi.org/10.22004/ag.econ.182665
Jiménez Cisneros, B.E., Oki, T., Arnell, N.W., Benito, G., Cogley, J.G., Döll, P., Jiang, T., Mwakalila, S.S., 2014. Freshwater resources., in: Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Field, C.B., V.R. Barros, D.J. Dokken, K.J. Mach, M.D. Mastrandrea, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White (Eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 229–269.
Knight, F.H., 1921. Risk, uncertainty and profit. Houghton Mifflin Company, Boston, New York.
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
Lempert, R.J., Groves, D.G., Popper, S.W., Bankes, S.C., 2006. A General, Analytic Method for Generating Robust Strategies and Narrative Scenarios. Management Science 52, 514–528. https://doi.org/10.1287/mnsc.1050.0472
Loch, A., Adamson, D., Dumbrell, N.P., 2020. The Fifth Stage in Water Management: Policy Lessons for Water Governance. Water Resources Research 56, e2019WR026714. https://doi.org/10.1029/2019WR026714
Marchau, V.A.W.J., Walker, W.E., Bloemen, P.J.T.M., Popper, S.W. (Eds.), 2019. Decision Making under Deep Uncertainty: From Theory to Practice. Springer International Publishing.
Mérel, P., Simon, L.K., Yi, F., 2011. A Fully Calibrated Generalized Constant-Elasticity-of-Substitution Programming Model of Agricultural Supply. Am J Agric Econ 93, 936–948. https://doi.org/10.1093/ajae/aar029

Mills, G., 1984. Optimisation in Economic Analysis. Routledge. https://doi.org/10.4324/9781315824024

Mills, R., 2022. The heat is on. URL https://aheadoftheherd.com/the-heat-is-on/ (accessed 1.11.23).

Msangi, S., 2019. Introduction, in: Msangi, S., MacEwan, D. (Eds.), Applied Methods for Agriculture and Natural Resource Management: A Festschrift in Honor of Richard E. Howitt, Natural Resource Management and Policy. Springer International Publishing, Cham, pp. 1–7. https://doi.org/10.1007/978-3-030-13487-7_1

OECD (Organisation for Economic Co-operation and Development), 2021. Toolkit for Water Policies and Governance: Converging Towards the OECD Council Recommendation on Water (Report). Paris (France).

OECD, 2015. Water Resources Allocation: Sharing risks and opportunities, OECD Studies on Water. OECD Publishing.

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. Journal of Hydrology 569, 291–309. https://doi.org/10.1016/j.jhydrol.2018.12.009

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 Policy 14, 216–239. https://doi.org/10.1093/reep/reaa004

Pérez-Blanco, C.D., Koks, E.E., Calliari, E., Mysiak, J., 2017. Economic Impacts of Irrigation-Constrained Agriculture in the Lower Po Basin. Water Econs. Policy 04, 1750003. https://doi.org/10.1142/S2382624X17500035

Pérez-Blanco, C.D., Sapino, F., 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

Petsakos, A., Rozakis, S., 2015. Calibration of agricultural risk programming models. European Journal of Operational Research 242, 536–545. https://doi.org/10.1016/j.ejor.2014.10.018

Polade, S.D., Pierce, D.W., Cayan, D.R., Gershunov, A., Dettinger, M.D., 2014. The key role of dry days in changing regional climate and precipitation regimes. Sci Rep 4, 4364. https://doi.org/10.1038/srep04364

Rausser, G.C., Yassour, J., 1981. Multiattribute Utility Analysis: The Case of Filipino Rice Policy. Am. J. Agric. Econ. 63, 484–494. https://doi.org/10.2307/1240539

Sapino, F., Haer, T., Saiz-Santiago, P., Pérez Blanco, C.D., under review. A multi-agent cellular automata model to explore water trading potential under transaction costs.

Schewe, J., Heinke, J., Gerten, D., Haddeland, I., Arnell, N.W., Clark, D.B., Dankers, R., Eisner, S., Fekete, B.M., Colón-González, F.J., Gosling, S.N., Kim, H., Liu, X., Masaki, Y., Portmann, F.T., Satoh, Y., Stacke, T., Tang, Q., Wada, Y., Wisser, D., Albrecht, T., Frieler, K., Piontek, F., Warszawski, L., Kabat, P., 2014. Multimodel assessment of water scarcity under climate change. Proceedings of the National Academy of Sciences 111, 3245–3250. https://doi.org/10.1073/pnas.1222460110

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
Sivapalan, M., Savenije, H.H.G., Blöschl, G., 2012. Socio-hydrology: A new science of people and water. Hydrological Processes 26, 1270–1276. https://doi.org/10.1002/hyp.8426
Smith, A., 1759. The Theory of Moral Sentiments. A. Millar, London.
UN (United Nations), 2022. World Economic Situation Prospects. United Nations Department of Economic and Social Affairs, New York.
UNDRR (United Nations Office for Disaster Risk Reduction), 2021. GAR Special Report on Drought 2021 - World (Special Report on Drought). United Nations Office for Disaster Risk Reduction, Geneva (Switzerland).
UNEP (United Nations Environmental Programme), 2021. Adaptation Gap Report 2021 (No. 2021). United Nations Environment Programme, New York (US).
UNESCO (United Nations Educational, Scientific and Cultural Organization), 2021. The United Nations World Water Development Report 2021: Valuing Water. UNESCO, Paris.
UNESCO, 2020. The United Nations World Water Development Report 2020: Water and Climate Change. UNESCO, Paris.
Walker, W.E., Harremoës, P., Rotmans, J., Sluijs, J.P. van der, Asselt, M.B.A. van, Janssen, P., Krauss, M.P.K. von, 2003. Defining Uncertainty: A Conceptual Basis for Uncertainty Management in Model-Based Decision Support. Integrated Assessment 4, 5–17. https://doi.org/10.1076/iaij.4.1.5.16466
WCD (World Commission on Dams), 2000. Dams and development: a new framework for decision-making. Earthscan, London.
World Bank, 2022. World Bank DataBank [WWW Document]. DataBank. URL https://databank.worldbank.org/databases/exchange-rates (accessed 12.16.22).
World Bank, 2020. The 2030 Water Resources Group Annual Report (Report No. 2020). World Bank, Washington D.C. (US).
WRI (World Resources Institute), 2019. Baseline Water Stress, Aqueduct 3.0. [WWW Document]. URL https://www.wri.org/data/aqueduct-global-maps-30-data (accessed 12.30.22).