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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

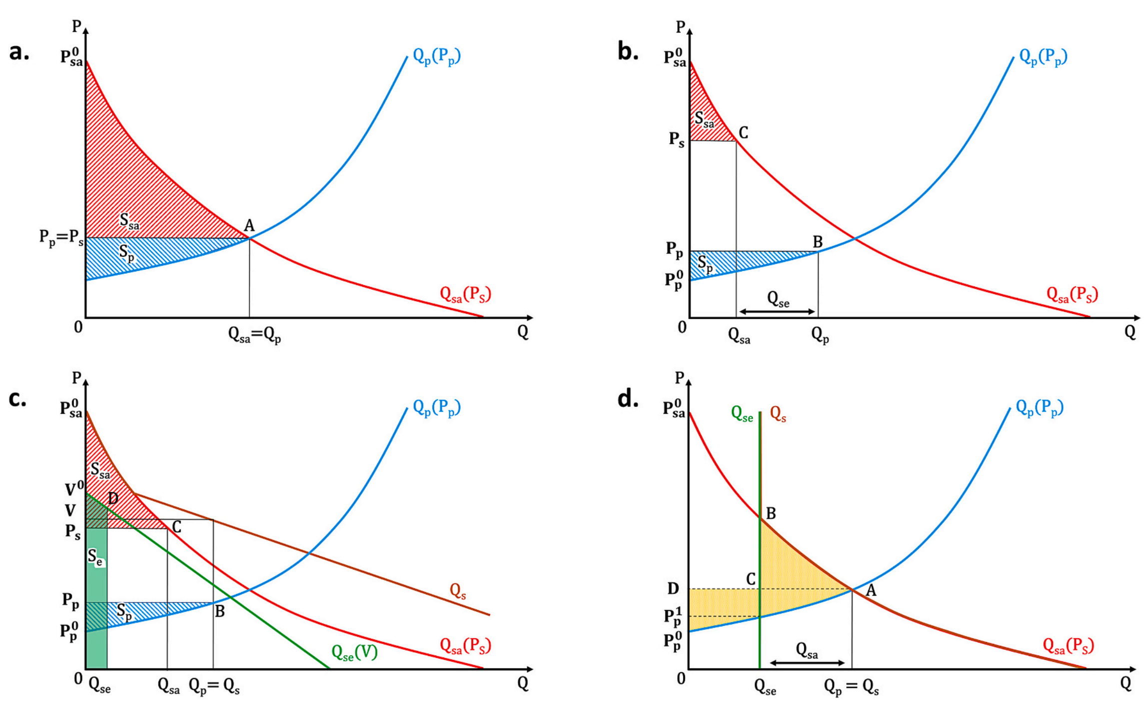

Fig. 1. Alternative water bank settings under a monopolistic-monopsonistic position. Source: a., b., c. Gutiérrez-Martín et al. (2020) and d. own elaboration.

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 $AP_{sa}^0P_s$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 $AP_p^0P_p$ 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 $BP_p^0P_p$ while the consumer surplus (S_{sa}) equals to $CP_{sa}^0P_s$, 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 ($S_p = BP_p^0P_p$), consumer ($S_{sa} = CP_{sa}^0P_s$), 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):

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

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

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

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

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

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:

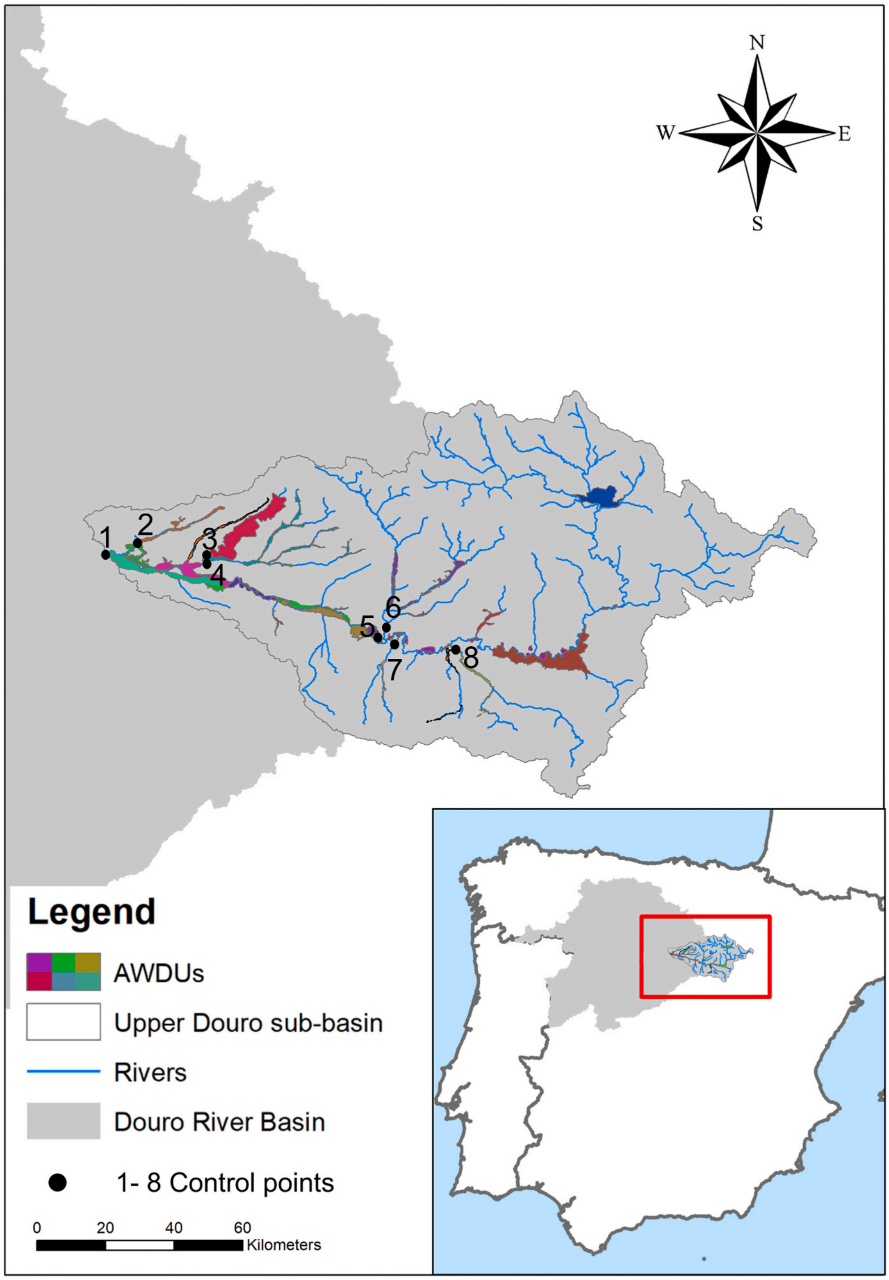

Fig. 2. Location of the Upper Douro Sub-basin in the Iberian Peninsula and detail of its AWDUs and the network of control points for environmental flows. Source: own elaboration.

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

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 ($g = 0, 1, 2, \ldots, G$), 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.

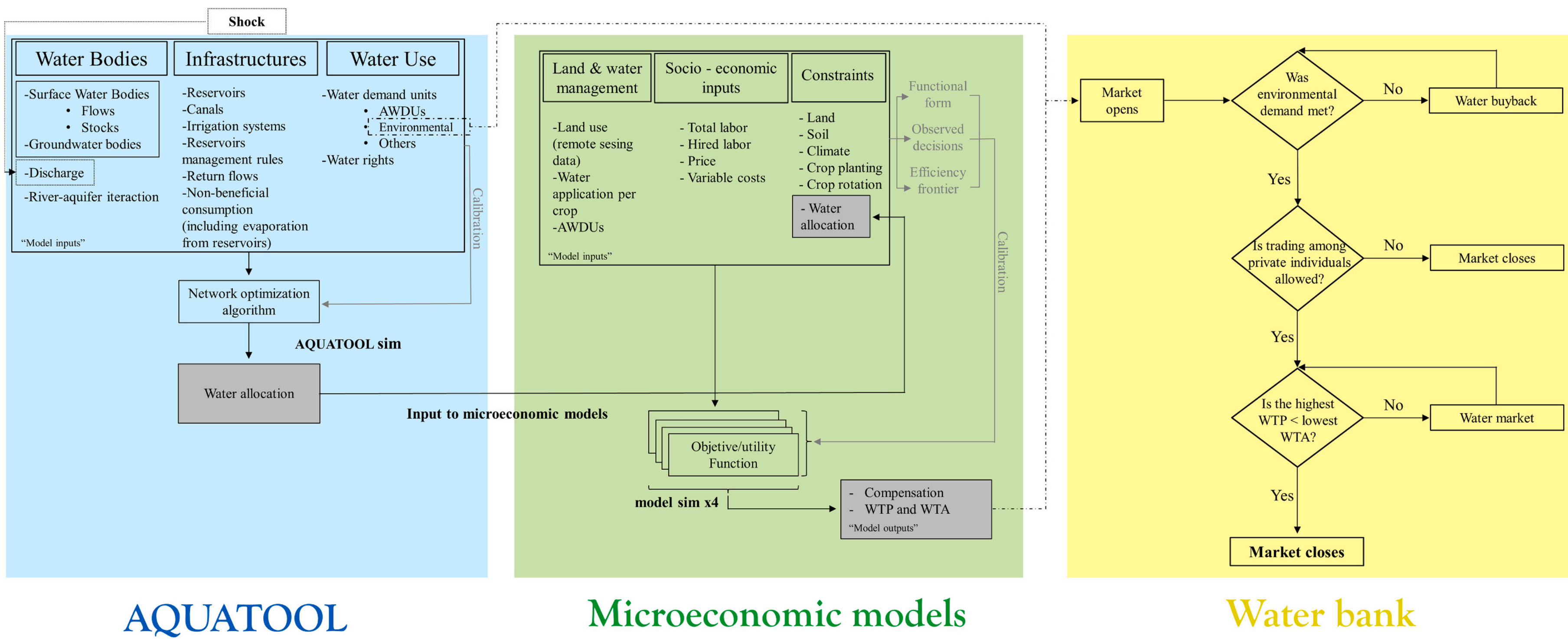

Fig. 3. Flowchart diagram of the modeling framework. Source: adapted from (Pérez-Blanco et al., 2021a).

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

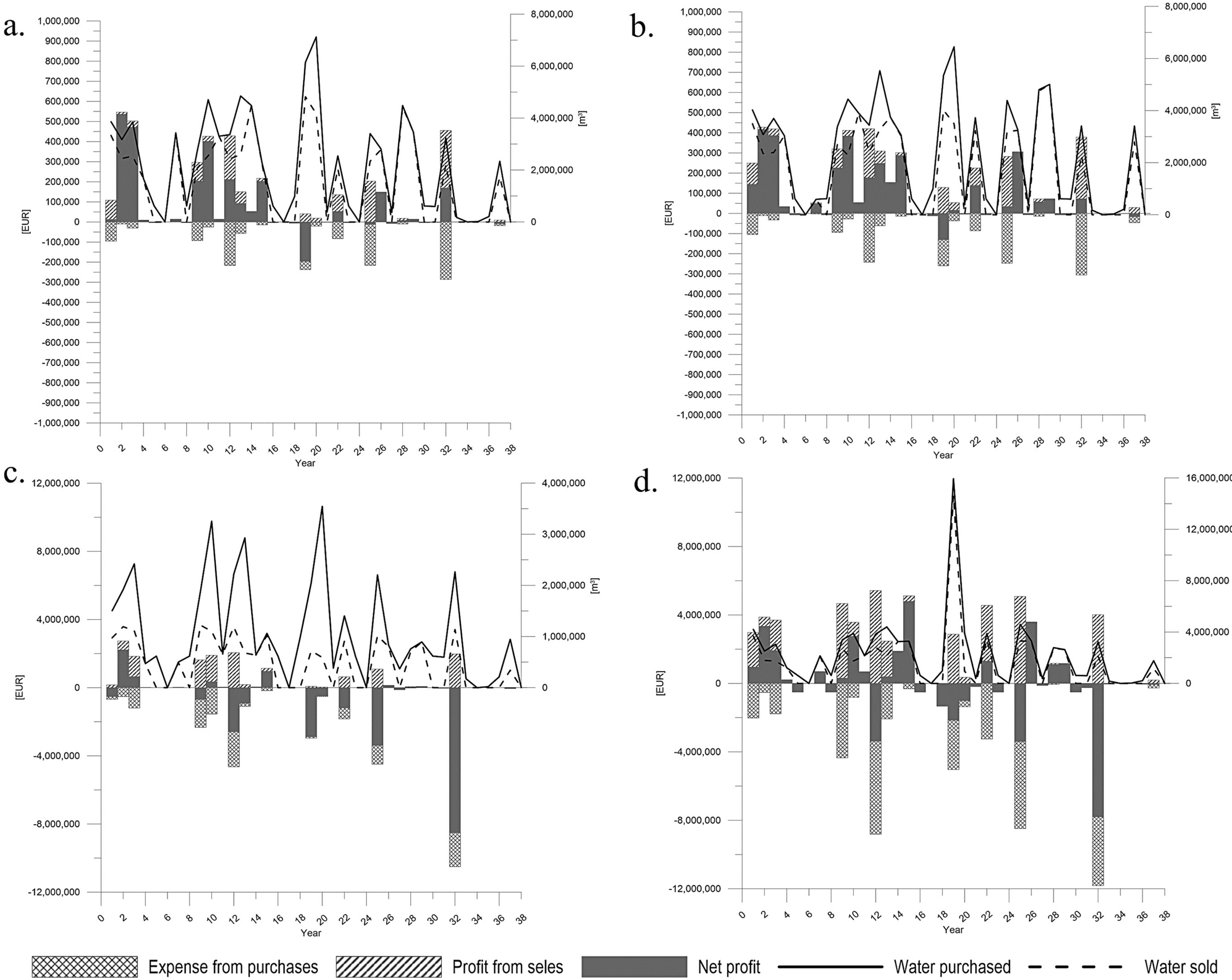

Fig. 4. Yearly water purchases and sales and net revenue over the 38y simulation period in the price discrimination water bank. a. PMAUP, b. WGP, c. PMP1 and d. PMP2.

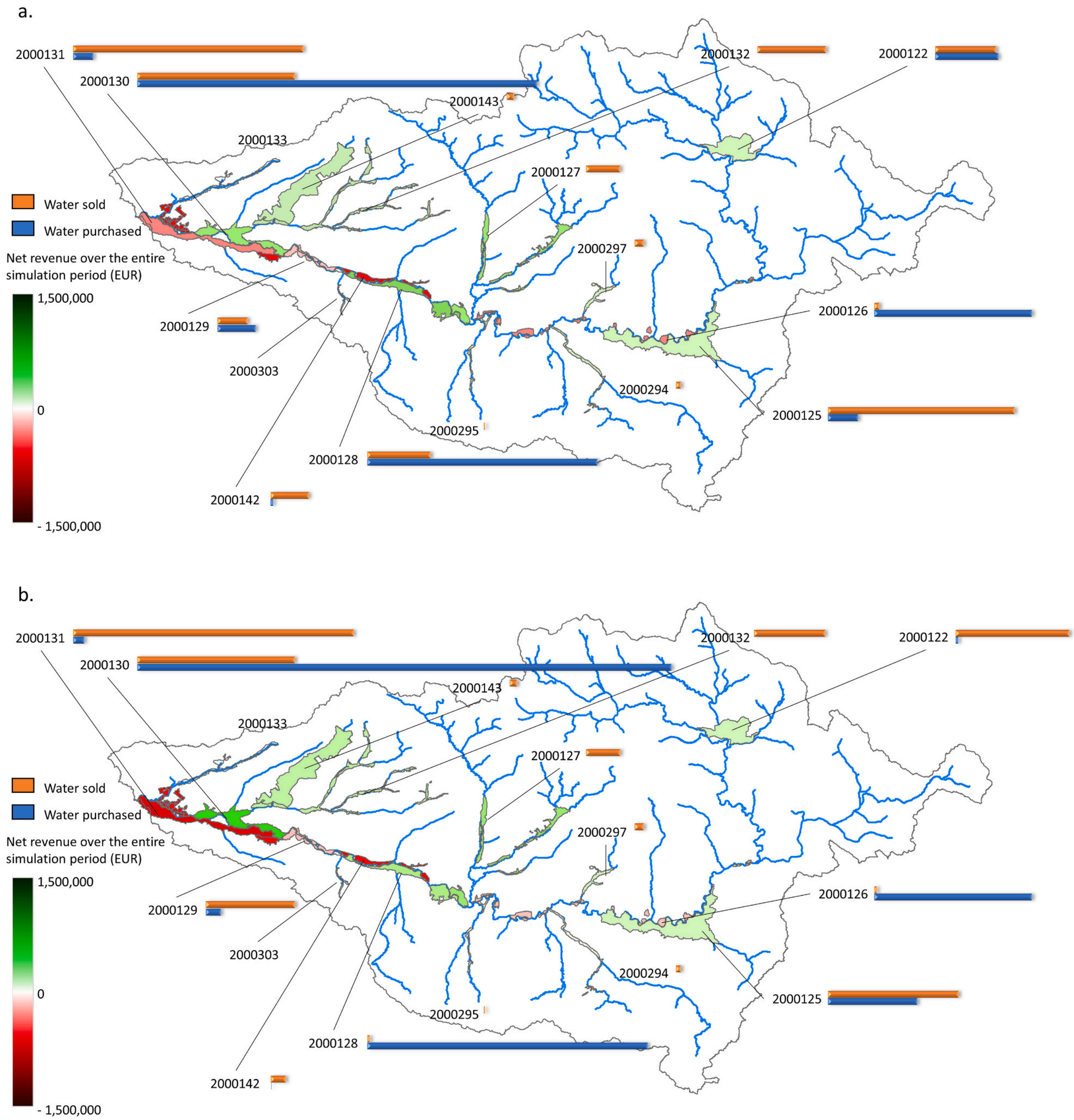

Fig. 5. Water purchased and sold and net revenue in the price discrimination water bank. a. PMAUP, b. WGP, c. PMP1, d. PMP2.

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

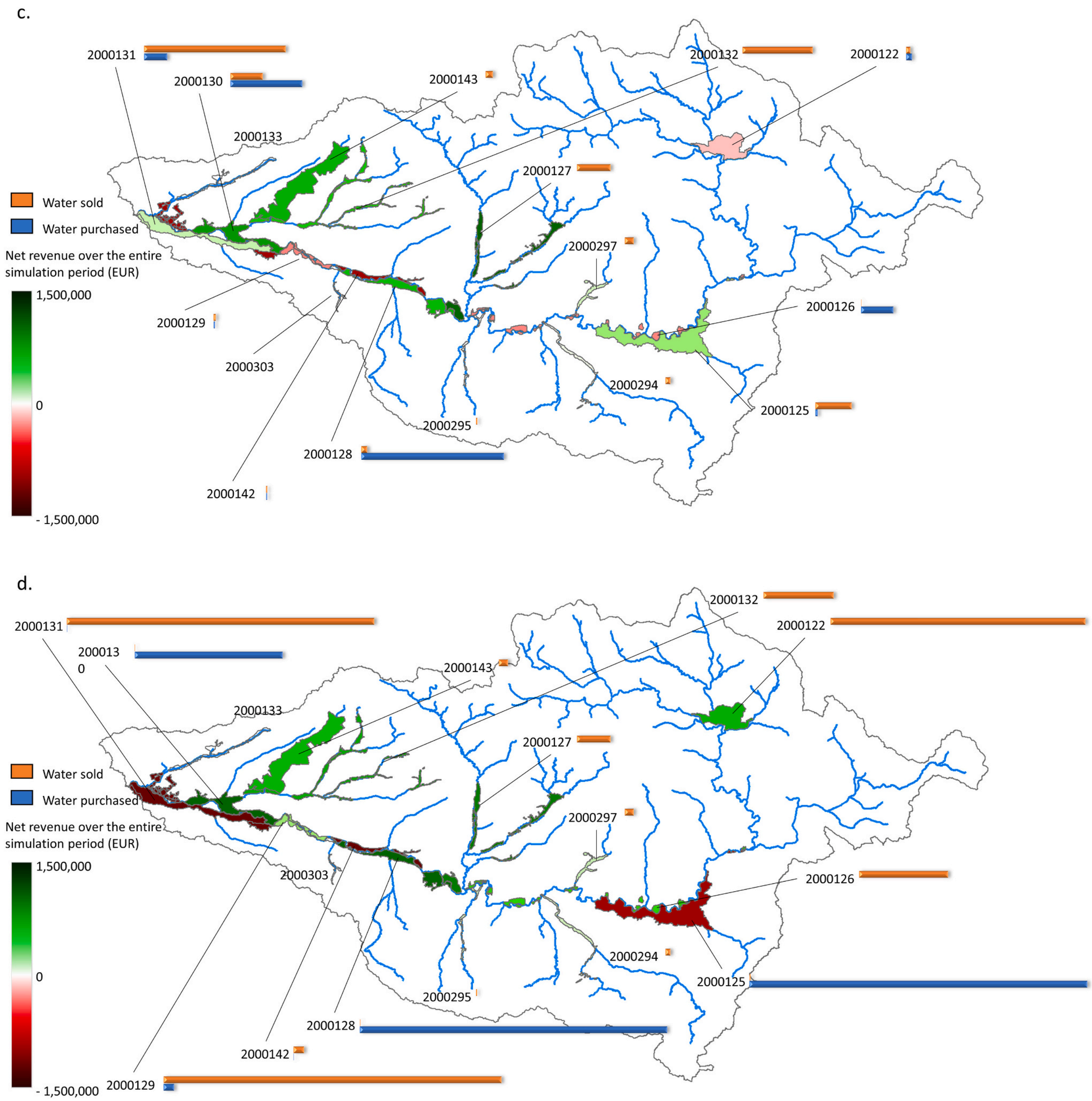

Fig. 5. (*continued*).

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.

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