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Recovering water for the environment during droughts through public water banks within a monopsony-monopoly setting

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ABSTRACT

Allocation trade is an instrument that has been widely used to recover water for the environment during periods of scarcity (droughts). This paper proposes a water bank operating within a monopsony-monopoly setting with the dual purpose of reallocating water among farmers and acquiring water for the environment during drought periods. The proposed water bank would be managed by a public agency seeking to maximize economic efficiency generated in purchases and sales of water for agriculture and the efficiency generated by the recovery of water allocations for the environment. An additional, innovative feature of the analysis performed is that it considers the inefficiencies in the economy as a whole caused by public spending on water allocation purchases, measured through the marginal cost of public funds. The potential performance of the proposed water bank is simulated by mathematical programming techniques, taking the Guadalquivir River Basin (Southern Spain) as an empirical case study. The results provide evidence that, in terms of economic efficiency, the proposed institutional arrangement outperforms the instruments currently in place to purchase water allocations.

Keywords: Allocation trade; Economic efficiency; Environmental flows; Marginal cost of public funds; Mathematical programming; Spain.

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1. Introduction

Economic development and growing demand from the population are the principal drivers of increasing water scarcity worldwide [1]. This is especially true in the Mediterranean and semi-arid regions, where the competitive advantage of irrigated agriculture has meant that water consumption for irrigation has multiplied over recent decades, leading to the closure of certain river basins [2]. In these river basins, strong competition for the use of water has led to the overexploitation of water resources without any reasonable engineering solution (supply-side measures) to further expand the water supply to meet new demands. In these cases, new demands can only be met by reducing the existing ones through demand-side policies. Among these demand-side measures, the use of economic instruments such as water markets and water banks has gradually become widespread [3-5].

In closed basins, scarcity problems become especially acute during drought periods, when water availability is far too low to meet the rigid demand from the different water users. Under such conditions of severe cyclical scarcity, the environment is often negatively affected since not enough resources are devoted to ensuring that water-related environmental services are sustained (e.g., instream flows are diminished, and may even drop below the minimum environmental flows). Although several demand-side policy instruments can be useful to cope with these situations [6,7], market instruments have been proven to be more suitable than other alternatives when it comes to recovering water for the environment, in terms of both cost-effectiveness and social acceptability [8,9]. This fact explains the increasingly common implementation of market instruments for the acquisition or lease of water rights from the lower value uses (mainly from the agricultural sector) for environmental purposes. Good examples of this type of policy action are the public purchase of water rights or entitlements (also known as 'buy-back') implemented in the Murray-Darling Basin in Australia through the Restoring the Balance Program [10] or the allocation trade (leasing or temporary reallocation of water rights) for environmental purposes during drought events used in California [11] and Spain [12].

In all the abovementioned cases, a public water agency operating in an existing water market has sought to mitigate the environmental effects of water scarcity by acquiring water rights (buy-back) or water allocations (allocation trade) from irrigators for the environment. These environmental purchases have been implemented through 'water banks', a kind of water market where the water agency centralizes purchase operations by organizing public tenders supported through public funding [13].

Although this market instrument (public water agency purchasing water rights or water allocations without subsequently reallocating them) has proven to be useful in real-world settings for the recovery of water for environmental purposes, a series of derivative instruments have been proposed in the literature to further improve drought management strategies. These include: a) counter-cyclical trade, where allocations are bought and stored when water availability is abundant and used later on for environmental purposes [14]; b) the purchase of water rights, allowing the agency to lease them through spot markets in years of abundance [15]; and c) the purchase of water option contracts, with the agency reserving the right to purchase water allocations in the event of seasonal scarcity [16].

The objective of this paper is to provide further insights into how the public purchase of water for environmental purposes during cyclical scarcity periods can be implemented more efficiently than with the water banks currently in place. In this regard, the alternative proposed in this paper is a water bank managed by a public agency that operates during drought events by purchasing and selling water allocations (allocation trade), with the dual purpose of reallocating water among productive users and acquiring water for environmental purposes. The key feature of the proposed water bank is that it will operate in a monopsony-monopoly setting (no other water markets will be allowed in the river basin at the same time), with the public agency using this market power to maximize economic efficiency. In this regard, this paper presents an empirical simulation of the potential performance of the proposed economic instrument, using the Guadalquivir River Basin (southern Spain) as a case study. The results of this analysis will enable an assessment of whether the implementation of this design of water bank could contribute to improving public water management during drought periods, reflected in social efficiency gains.

The idea of a water bank operating in a monopsony-monopoly setting managed by a public agency seeking to maximize economic efficiency has recently been suggested by Gutiérrez-Martín et al. [17]. These authors propose a design for this kind of water bank which would allow the public agency to balance its budget (i.e., expenditure on purchases is equal to revenues from sales). This self-financed water bank is appealing since it can use its market power to recover water for environmental purposes and increase efficiency gains without any public expenditure, but its potential performance is rather disappointing considering that only a small volume of water can be recovered for improving instream flows (Gutiérrez-Martín et al. [17] reported a maximum of 10.4% of the water used for irrigation could be recovered for the environment). In this sense, the main contribution of the present paper is to provide evidence that the former design can be improved by allowing the public agency to spend public funds to purchase additional water for the environment during drought periods. As will be demonstrated here, this alternative design combining public water purchases based on society's willingness to pay for enhanced instream flows and the market power stemming from the monopsony-monopoly setting achieves allocation arrangements that outperform the results obtained by the water banks currently in place or the one suggested by Gutiérrez-Martín et al. [17] in terms of efficiency gains.

A second innovative feature of this paper is the assessment carried out. To the best of our knowledge, this is the first time economic analysis has included the inefficiencies that inevitably result from the use of public funding to purchase environmental water through a water bank. These efficiency losses are due to the distortions generated by the tax system when collecting money from the whole economy; as in other analyses of public spending policies, they have been measured through the marginal cost of public funds.

To achieve the abovementioned objective, this paper is structured as follows. After this introduction, the second section explains the two alternative water banks assessed: first, the 'competitive water bank', which replicates the operation of the current instruments used to purchase environmental water, based on public tenders for the lease of water rights that are implemented within an already existing spot water market; and second, the 'monopsony-monopoly water bank', proposed as an alternative design of this economic instrument, aimed at improving economic efficiency. The analytical framework developed in this second section, based on an economic analysis accounting for efficiency gains and losses, is used to assess the performance of each of the water bank alternatives. Next, in the third section, the Guadalquivir River Basin is presented as the real case study considered for the empirical analysis. The fourth section develops the mathematical programming models used to simulate the performance of the two water banks considered. The fifth section summarizes the main results obtained for the two alternative economic instruments, which are further discussed in the sixth section. Finally, the last section concludes, providing the main insights derived from this study.

2. Alternative designs for a water bank aimed at recovering water for the environment during drought periods

2.1. Common settings for the two water banks analyzed

A water bank is a kind of water market where an institutional intermediary acts as a link between buyers and sellers to centralize and facilitate the purchases and sales of water rights or water allocations [18]. Although this economic instrument has been implemented through a wide variety of institutional arrangements, the review of the international experience provided by Montilla-López et al. [13] indicates that the most suitable design for a water bank aimed at recovering water for the environment during drought periods should be constituted as follows:

- Nature of the institution responsible for its implementation: a public authority (i.e., *public* water banks). In our case, we assume that the two banks analyzed are organized and managed by a public water agency.
- Type of rights exchanged: temporary transfer of water use rights or specific quantities of water (i.e., *allocation trade* water banks).
- Management strategy: the public water agency adopts a proactive strategy as a market-maker (i.e., *active* water banks), first purchasing temporary water rights from rights-holders (productive users) and then selling them (totally or partially) to other productive users. In our case, it is assumed that these operations are implemented through public purchase and sale tenders, respectively.
- Objective: *resource reallocation and environmental purposes*. The water banks are managed with the twofold objective of reallocating water among productive users, fostering the transfer of water from lower-value to higher-value uses, and recovering a share of the purchased water for the environment (purchasing allocations without subsequently reallocating them).

Under this bank design, the purchase and sale offers implemented by the public water agency are governed by the aggregate supply curve of water $Q_p(P_p)$ (Q_p being the allocations of water that can be purchased from rights-holders at a price P_p), and the aggregate demand curve of water $Q_s(P_s)$ (Q_s being the allocations of water that can be sold to other productive users at a price P_s), respectively (see Figs. 1 and 2). These aggregate curves represent, respectively, rights-holders' willingness to sell water and productive users' willingness to buy water based on the heterogeneous marginal values of water in their production functions. In fact, Q_p is the aggregate volume of water (among rights-holders) with a marginal value below P_p, while Q_s is the aggregate volume of water (among productive users) with marginal values above P_s.

Bank operations in the two water banks analyzed are also determined by the social demand for environmental water $Q_e(V_e)$. In this sense, it is assumed that society attributes a null value to environmental water (V_e=0) when the instream flows are equal to or greater than those in an average hydrological year (Q_e^{max}). This social value V_e increases as the deficit in instream flows (difference between the flows corresponding to an average hydrological year and the current actual flows) increases, reaching its highest value (V_e^0) when the instream flows equal those set as minimum environmental flows (Q_e=0) (see Figs. 1 and 2). Thus, Q_e is the aggregate volume of water that society is willing to recover for the environment with marginal social values above V_e.

Bank operations in the two water banks analyzed are also determined by the social demand for environmental water $Q_e(V_e)$, with this demand curve representing the population's willingness to pay for different levels of instream flows. As pointed out by Horne et al. [19], developing this curve entails the following three steps: 1) quantifying the impact of changes in instream flows on ecological condition, 2) quantifying how these changes in ecological condition affect the provision of environmental services, and 3) estimating the value of these changes in environmental service provision to society. The complexity involved in relating instream flows, ecology, and the value to society (e.g., non-linear relationships or tipping points) makes the task of estimating these social demand curves particularly challenging. Indeed, it lies beyond the scope of this paper. In any case, despite these methodological difficulties and the scarcity of empirical evidence (see [20,21] as examples of the limited existing literature in this regard), it is worth pointing out that these demand curves are needed to look for the optimum social water allocation arrangements (i.e., those that achieve the optimum trade-off between the volumes to be allocated to productive and environmental uses), as proposed here through the implementation of the two water banks analyzed [22].

For the reasons provided above, the shape of the social demand for environmental water to be used in this empirical research ($Q_e(V_e)$) should rely on several simplifying assumptions. First, a simple linear environmental demand curve approximation is used (see Figs. 1 and 2), which is consistent with previous studies [17,19,23]. Second, it is assumed that society attributes a null value to environmental water (V_e=0) when the instream flows are equal to or greater than those in an average hydrological year (Q_e^{max}). And third, this social value V_e linearly increases as the deficit in instream flows (difference between the flows corresponding to an average hydrological year and the current actual flows) increases, reaching its highest value (V_e^0) when the instream flows equal those set as minimum environmental flows (Q_e=0) (see Figs. 1 and 2). Thus, this social demand can be formulated as follows:

$$Q_e = Q_e^{max}\left(1 - \frac{V_e}{V_e^0}\right) \quad (1)$$

where Q_e is the aggregate volume of water that society is willing to recover for the environment with marginal social values above V_e, and the environmental benefits for society are measured as the area underneath the linear demand curve.

Introducing society's demand curve for environmental water into the analysis, this demand must be added to the productive water demand to give an aggregate total demand $Q_t(P_s, V_e)$ for productive and environmental water (see Figs. 1 and 2).

The first institutional arrangement for the proposed water bank is a dual-purpose public bank aimed at reallocating water (allocation trade) between productive users and recovering a share of the water purchased for the environment. Under this bank design, social demand for environmental water is considered similar to any other demand since public funds support the recovery of environmental water at the same price as the water bought by productive users. The bank is therefore managed neutrally by the water agency, with the aim of reproducing the equilibrium reached in a competitive market ('*competitive* water bank'). Indeed, a similar type of bank operates in many regions of the world, where public water agencies seek to mitigate the environmental effects of water scarcity by purchasing water allocations from irrigators in an existing spot water market (allocation trade) for environmental purposes. Thus, this first design of the water bank attempts to replicate the operation of the market instruments already implemented worldwide to cope with cyclical scarcity [13]. Probably the best-known example of this market approach to improve water management during cyclical scarcity is the Drought Emergency Water Bank developed in California in 1991 [24]. Similarly, the public water banks developed in Spain as instruments to cope with droughts are also worth mentioning [12].

As an alternative instrument aimed at improving public drought management, the second arrangement proposed for the water bank is based in a monopsony-monopoly setting ('*monopsony-monopoly* water bank'). In this case, any water trading operation outside the bank is forbidden, allowing the water agency to use its market power to create a gap between purchase and sale prices, the water agency managing the bank should first act as the sole buyer (monopsony market) of water allocations, then the agency acts as the sole seller (monopoly market) of those purchased allocations. Moreover, the agency can distinguish between water uses (productive and environmental ones) in the implementation of public sale offers, enabling the marginal value of the water devoted to environmental uses to diverge from the sale price.

Under this design of the water bank, it is assumed that the water agency operates by buying and selling water allocations to maximize overall economic efficiency in water use.

Both alternative designs for the water bank are explained below in more depth, indicating how to assess the improvement in economic efficiency that each alternative entails compared with the baseline scenario (no trade).

2.2. Competitive water bank

Under this institutional arrangement, the water agency managing the bank aims to maximize economic efficiency during drought or cyclical scarcity events in a similar way to a competitive spot market. To this end, the agency first makes a public offer to purchase water allocations at a fixed price (P_p), buying an amount of water Q_p, as shown in Fig. 1. Next, the agency makes a public offer to sell those purchased water allocations, treating both demands—those from productive users and those from society (environmental water)—equally. This means that any demand for environmental water must be supported through public funding, with society paying the bank the same price for every unit of water recovered for the environment that productive users pay for the water they bought. This means that the social value of environmental water is equal to the price paid by productive users (i.e., $V_e = P_s$). It also implies that the social demand and the aggregate total demand for water became functions of the sale price fixed by the agency ($Q_e(P_s)$ and $Q_t(P_s)$, respectively). Therefore, under this bank design, at the price fixed by the agency for the sale offers (P_s), productive users demand an amount of water Q_s for their private use and society demands Q_e units of water for environmental purposes, with the total water demand being $Q_t = Q_s + Q_e$.

According to economic theory [25], this water bank reaches maximum economic efficiency operating at the point where the supply ($Q_p(P_p)$) and total demand ($Q_t(P_s)$) curves intersect[1], as occurs at point A in Fig. 1. This efficient solution can be achieved if purchase and sale prices are equal ($P_p = P_s$), also allowing the quantities purchased and sold to become equal (i.e. when the water balance is verified: $Q_p = Q_s + Q_e$). Operating at point A also allows the water agency to balance the bank's cash flows, since expenditure on purchases ($P_p \cdot Q_p$, represented by the area $P_p A Q_p O$) equals the sum of revenues obtained through sales to productive users ($P_s \cdot Q_s$, represented by the area $P_s B Q_s O$) and to society for environmental

[1] This is the optimum solution in the case of zero transaction costs. We have applied this assumption for the sake of simplicity.

purposes ($P_s \cdot Q_e$, represented by the area $P_s C Q_e O$). The latter revenue is provided by the public sector through the budget assigned to the water agency ($Budget = P_s \cdot Q_e$).

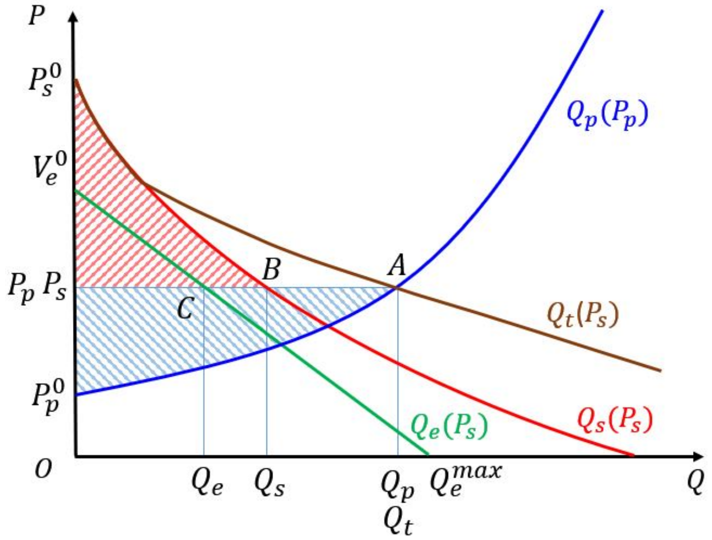

Fig. 1 Competitive water bank aimed at recovering water for the environment.

Intersection point A reproduces the equilibrium reached in a competitive market, providing the optimum solution by properly reallocating water resources between productive users and the environment. At this point economic efficiency is maximized since the sum of the efficiency gains generated by purchases and sales is also maximized. In this regard, the efficiency generated by the public purchase offer can be calculated through the *producer surplus* (E_p), measuring the profit obtained by water users when they sell their water allocations to the bank (the difference between the sales revenue and the income foregone due to lower water input). If the purchase price is fixed at P_p, as shown in Fig. 1, then $E_p = P_p A P_p^0$ area. The efficiency generated by the bank sales to productive users is calculated through the *consumer surplus* E_s, accounting for the profit that productive users gain when they purchase water from the bank (the difference between the additional income due to higher production and the purchase cost). If the sale price is fixed at P_s, then $E_s = P_s^0 B P_s$ area.

Water recovery for environmental purposes also generates efficiency gains, which can be accounted for using the consumer surplus concept as in the case of productive users. In this case, the *environmental surplus* $\tilde{E}_e$ measures the benefit that society gains from the water recovered for the environment as the difference between the additional economic efficiency due to higher instream flows and the budget needed to recover that water. If the sale price is fixed at P_s, then $\tilde{E}_e = V_e^0 C P_s$ area. However, it must be noted that the budgetary resources needed to implement any public spending policy (such as environmental water recovery through a water bank) must first be collected through the tax system, and this inevitably causes distortions that reduce economic efficiency [26]. Surprisingly, this fact has been largely ignored in the literature assessing the economic impact of policy instruments used to implement the public purchase or lease of water rights to reallocate water resources to environmental uses[2]. For this reason, as far as we are aware, all related empirical works to date have overestimated the efficiency gains generated. This paper thus contributes to the existing literature by carrying out an efficiency analysis that accounts for the efficiency losses introduced by the tax system. These losses can be quantified through the marginal cost of public funds (MCF), a parameter reflecting the shadow price that society pays for each monetary unit invested in public spending policies [29]. Thus, *environmental surplus* E_e can be more accurately measured as the difference between the efficiency generated by the amount of water diverted for environmental purposes (area below society's demand curve for environmental water, $V_e^0 C Q_e O$) and the social cost of this water in budgetary terms ($Budget \cdot MCF$):

$$E_e = \int_{P_s}^{V_e^0} Q_e(P_s)\, dP_s + P_s \cdot Q_e - Budget \cdot MCF \qquad (2)$$

In sum, the water agency operates this water bank in a similar way to a competitive market, achieving an optimum solution when the sum $E_p + E_s + E_e$ is maximized.

Finally, it is worth pointing out that under this theoretical framework, the efficiency assessment for the baseline scenario (no trade) is null, since $Q_p = Q_s = Q_e = 0$, and then $E_p = E_s = E_e = 0$. As this scenario is taken as a benchmark, it should be noted that the efficiency assessments for the two alternative designs proposed for the water bank should be considered as improvements over this baseline scenario.

2.3. Monopsony-monopoly water bank

Similar to the previous bank design, during a hydrological drought, the public water agency can mitigate any deficit in instream flows by recovering water allocations granted to productive users and reallocating them to the environment. However, the key feature of this innovative water bank arrangement is that the public agency operates in a monopsony-

[2] Within the water policy literature, efficiency losses due to distortive tax systems have been considered in only a few papers. In this regard, the only studies worth mentioning are those by Garcia and Reynaud [27] and van Heerden et al. [28], which both focus on water pricing.

monopoly setting (no other water markets are allowed). In this setting, the agency should first act as the sole buyer (monopsony market) of water allocations by organizing public purchase offers of temporary water rights. Subsequently, this agency also acts as the sole seller (monopoly market) of those purchased allocations, by organizing a public sale offer. Operating in this way, the agency managing the bank can use its market power to create a gap between the purchase price (P_p) and the sale price to productive users (P_s). Moreover, this market power allows the agency to distinguish between water uses (productive and environmental ones), enabling the marginal value of the water sold to productive users (P_s) to diverge from the marginal value of the water devoted to environmental uses (V_e). As a non-profit public institution, the water agency is assumed to use this monopsony-monopoly power to maximize economic efficiency, covering any deficit in the bank's cash flow (revenue obtained through public sale offers less expenditure on public purchase offers) with public funds.

Figure 2 shows how this design of the water bank should work. The water agency first makes a public purchase offer, fixing the purchase price P_p and buying a quantity of water Q_p (see point A), as determined by the supply curve of water $Q_p(P_p)$. Second, the agency implements a public sale offer, fixing a sale price P_s higher than P_p, and selling a smaller amount of water Q_s (see point B), as determined by the demand curve of water for productive users $Q_s(P_s)$. The difference between the amount of water purchased and subsequently sold is allocated to the environment ($Q_e = Q_p - Q_s$). Operating in this way, the deficit in the bank's cash flow equals the difference between the revenue obtained through sales ($P_s \cdot Q_s$, represented by the area P_sBQ_sO) and the expenditure on purchases ($P_p \cdot Q_p$, represented by the area P_pAQ_pO). This deficit is compensated for with public funds, through the budget assigned to the water agency ($Budget = P_p \cdot Q_p - P_s \cdot Q_s$).

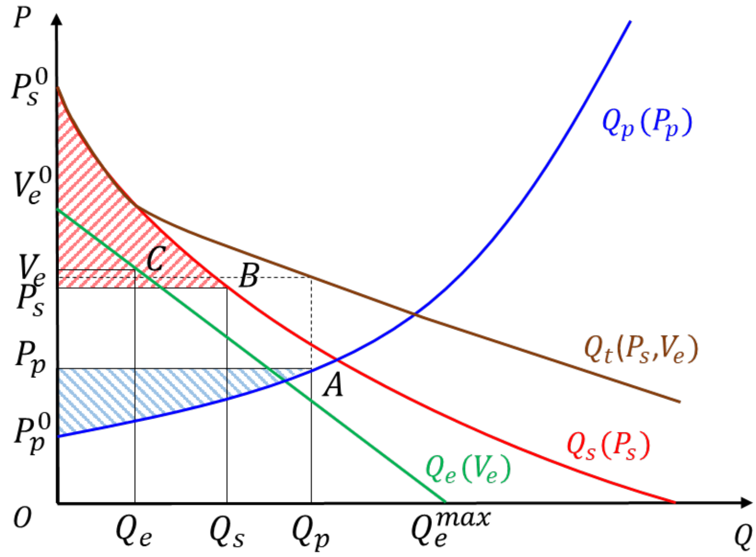

Fig. 2 Monopsony-monopoly water bank aimed at recovering water for the environment.

Among the infinite number of alternative operational strategies that fulfill the abovementioned water and financial balance constraints, the public water agency aims to maximize total economic efficiency. Thus, the single optimum solution to be achieved is the one that maximizes the sum of the efficiencies related to the water bank's purchases (*producer surplus* E_p, equivalent to the area $P_pAP_p^0$ in Fig. 2), the efficiency linked to the water bank's sales to productive users (*consumer surplus* E_s, measured by the $P_s^0BP_s$ area), and the efficiency derived from the amount of water the bank recovers for the environment (*environmental surplus* E_e). The latter efficiency is also calculated considering the efficiency losses introduced by the tax system, as in the previous bank setting; thus, it is measured as the difference between the economic efficiency generated by the amount of water diverted for environmental purposes (area $V_e^0CQ_eO$) and the social cost of the budget needed to recover this water:

$$E_e = \int_{V_e}^{V_e^0} Q_e(V_e)\, dV_e + V_e \cdot Q_e - Budget \cdot MCF \qquad (3)$$

Unlike with the competitive water bank, it can be easily seen that by lowering the purchase price and increasing the sale price in relation to the competitive equilibrium, both producer and consumer surpluses are reduced under this institutional arrangement. However, the environmental surplus increases when this monopsony-monopoly water bank is implemented since the social cost of environmental water recovery becomes smaller. In this sense, the main purpose of this paper is to empirically confirm that the overall variation of the three surpluses considered for this bank setting ($E_p + E_s + E_e$) outperforms the total economic efficiency gains generated by the competitive water bank, and to quantitatively assess the improvement in economic efficiency that this novel water bank design could provide.

3. Modeling the performance of the two alternative water banks

3.1. Competitive water bank

In order to simulate the potential performance of the first institutional arrangement proposed for the water bank, a mathematical programming model has been built. Considering that the competitive market's equilibrium is reached when the sum of the efficiencies derived from purchases, sales to productive users, and the recovery of water for environmental purposes is maximized, the objective function of this model is the maximization of the total economic efficiency ($E_{competitive}$). This efficiency can be operatively measured as the sum of the producer, consumer and environmental surpluses (E_p, E_s, and E_e, respectively), as shown in eq. 4.1. Moreover, based on the explanations provided in Section 2.2, the constraints needed for simulating the performance of this water bank design are as follows:

$$\max\ E_{competitive} = E_p + E_s + E_e \tag{4.1}$$

Subject to:

$$E_p = \int_{P_p^0}^{P_p} Q_p(P_p)\, dP_p \tag{4.2}$$

$$E_s = \int_{P_s}^{P_s^0} Q_s(P_s)\, dP_s \tag{4.3}$$

$$E_e = \int_{P_s}^{V_e^0} Q_e(P_s)\, dP_s + P_s \cdot Q_e - Budget \cdot MCF \tag{4.4}$$

$$Q_p = Q_p(P_p) \tag{4.5}$$

$$Q_s = Q_s(P_s) \tag{4.6}$$

$$Q_e = Q_e^{max}\left(1 - \frac{P_s}{V_e^0}\right) \tag{4.7}$$

$$Q_p = Q_s + Q_e \tag{4.8}$$

$$Budget = P_s \cdot Q_e \tag{4.9}$$

$$P_p = P_s \tag{4.10}$$

Equations 4.2, 4.3, and 4.4 account for the producer, consumer, and environmental surpluses, respectively. That is, they represent the efficiency gains from water bank purchases (E_p), from water bank sales to the agricultural sector (E_s), and from water recovery with environmental purposes. The first two are measured as the area under the $Q_p(P_p)$ and $Q_s(P_s)$ curves on the ordinate axis, while the third is computed as explained in eq. 2.

Equations 4.5 and 4.6 represent the volumes traded, on both the supply and the demand sides. The first represents the amount of water purchased by the bank from irrigators, which in turn is the aggregate supply curve $Q_p(P_p)$, while the second represents the amount of water sold to farmers, corresponding to the aggregate irrigators' demand curve $Q_s(P_s)$. Equation 4.7 denotes the social demand for environmental water, as expressed in eq. 1 but considering that $V_e = P_s$, making this demand a function of the sale price fixed by the agency ($Q_e(P_s)$).

Equations 4.8 and 4.9 ensure that the water and financial (funding needed) balances are verified. Finally, eq. 4.10 ensures that purchase and sale prices are equal.

3.2. Monopsony-monopoly water bank

Under the monopsony-monopoly institutional arrangement for the water bank (see explanations provided in Section 2.3), the public water agency aims to maximize total economic efficiency ($E_{monopoly}$). Thus, this agency manages the bank's operations in such a way as to maximize the sum of the surpluses generated in purchases (E_p), sales to productive users (E_s), and the recovery of water for environmental purposes (E_e), as shown in the objective function (eq. 5.1) of the mathematical programming model built to simulate the performance of this water bank design:

$$\max\ E_{monopoly} = E_p + E_s + E_e \tag{5.1}$$

Subject to:

$$E_p = \int_{P_p^0}^{P_p} Q_p(P_p)\, dP_p \tag{5.2}$$

$$E_s = \int_{P_s}^{P_s^0} Q_s(P_s)\, dP_s \tag{5.3}$$

$$E_e = \int_{V_e}^{V_e^0} Q_e(V_e)\, dV_e + V_e \cdot Q_e - Budget \cdot MCF \tag{5.4}$$

$$Q_p = Q_p(P_p) \tag{5.5}$$

$$Q_s = Q_s(P_s) \tag{5.6}$$

$$Q_e = Q_e^{max}\left(1 - \frac{V_e}{V_e^0}\right) \tag{5.7}$$

$$Q_p = Q_s + Q_e \tag{5.8}$$

$$Budget = P_p \cdot Q_p - P_s \cdot Q_s \tag{5.9}$$

In this case, the efficiency improvements generated by the public purchase offers (E_p in eq. 5.2) and by public sale offers to the agricultural sector (E_s in eq. 5.3) are calculated as in the previous institutional arrangement for the water bank. Similarly, the equations regarding the aggregate supply ($Q_p(P_p)$ in eq. 5.5), the aggregate agricultural demand ($Q_s(P_s)$ in eq. 5.6), and market balance for water quantities (eq. 5.8) remain the same as in the previous model.

The model simulating the monopsony-monopoly water bank diverges from the model (4.X) in three equations. First, since this bank setting allows the agency to distinguish between water uses, the marginal value of the water sold to productive users (P_s) can diverge from the marginal value of the water devoted to environmental uses (V_e). Thus, the efficiency derived from environmental water recovery (E_e in eq. 5.4) and the social demand for environmental water (Q_e in eq. 5.7) are calculated as functions of V_e, as shown in eqs. 1 and 2. Second, in this case, the public funding needed for the water bank operations (constraint 5.9 regarding the financial balance) is calculated as the amount of money required to cover the deficit in cash flow, considering the revenue obtained in sales ($P_s \cdot Q_s$) and the expenditure on purchases ($P_p \cdot Q_p$).

Finally, it is worth mentioning that under this monopsony-monopoly setting, the water agency managing the bank can create a gap between purchase prices (P_p) and sale prices to productive users (P_s). Thus, the strategy implemented by this agency (optimal solution) is not constrained by eq. 4.10 included in the previous model ($P_p = P_s$).

3.3. Comparing the performance of the two water banks proposed

Once the theoretical models for the two water banks have been established, we can compare their potential performance in terms of total economic efficiency. To that end, we can change eq. 4.10 to $P_p = P_s = V_e$, since in the optimum solutions of the competitive water bank, the marginal values of the water devoted to environmental uses (V_e) are the same as the equilibrium market price ($P_p = P_s$), whenever V_e^0 is higher than the clearing price[3]. Replacing P_s by V_e in eqs. 4.4 and 4.7, these constraints became identical to eqs. 5.4 and 5.7. Moreover, it can be proved that eqs. 4.9 and 5.9 are also equivalent[4]. Thus, it can be checked that the

[3] For values of V_e^0 lower than the clearing price, the two models perform identically as no water is recovered for the environment and only agricultural water is traded.

[4] As explained before, $Q_e = Q_p - Q_s$. Thus, eq. 4.9 can be written as $Budget = P_s \cdot (Q_p - Q_s) = P_s \cdot Q_p - P_s \cdot Q_s$. Furthermore, as in the case of the competitive water bank $P_p = P_s$, we can write this equation as eq. 5.9; $Budget = P_p \cdot Q_p - P_s \cdot Q_s$.

feasible region of the model 5.X (given by eqs. 5.2-5.9) is bigger than that of model 4.X (given by eqs. 4.2-4.9 −equivalent to 5.2-5.9− plus eq. 4.10) since the latter model has one additional constraint (eq. 4.10) that needs to be met. This means that the maximum values of the common objective function (total economic efficiency, E) reached in model 5.X are always equal to or higher than those obtained in model 4.X. In other words, the monopsony-monopoly design proposed for the water bank is able to reach higher levels of total economic efficiency than the competitive design.

Furthermore, it is worth comparing the objective function in the two designs of the water market ($E = E_p + E_s + E_e$). Considering the explanations provided above, the only difference regarding this mathematical expression in the two models is the inefficiency generated by the public budget used for water recovery (term $budget \cdot MCF$ in eqs. 4.3 and 5.3). To draw conclusions about the efficiency performance of these alternative designs, the inefficiency in the monopsony-monopoly setting is subtracted from the inefficiency in the competitive setting as follows:

$$\begin{aligned}
&[budget \cdot MCF]_{competitive} - [budget \cdot MCF]_{monopoly} = \\
&= P_s \cdot Q_e \cdot MCF - \left(P_p \cdot Q_p - P_s \cdot Q_s\right) \cdot MCF = \\
&= \left(P_s \cdot Q_e - P_p \cdot Q_p + P_s \cdot Q_s\right) \cdot MCF = \left(P_s \cdot (Q_e + Q_s) - P_p \cdot Q_p\right) \cdot MCF = \\
&= \left(P_s \cdot \left(Q_p\right) - P_p \cdot Q_p\right) \cdot MCF = Q_p\left(P_s - P_p\right) \cdot MCF
\end{aligned} \tag{6}$$

This proves that as long as the water agency can create a gap between purchase prices and sale prices to productive users, as proposed for the monopsony-monopoly setting (i.e., $P_s > P_p$), the inefficiency generated by the use of public funds in the competitive setting is higher (i.e., positive difference in eq. 6), and this difference increases with MCF. It also confirms that the total economic efficiency reached with the monopsony-monopoly design (model 5.X) is always equal to or higher than the one obtained in the competitive design (model 4.X).

The two proofs detailed above regarding the size of the solution space and the inefficiency in the use of public funds provide theoretical evidence of the superiority of the monopsony-monopoly setting in terms of total economic efficiency. The next sections of the paper seek to confirm this conclusion through an empirical application in a real-world setting.

4. Case study

4.1. The Guadalquivir River Basin

The Guadalquivir River Basin (GRB) in southern Spain covers an area of 58,000 km^2 and is home to a population of 4.1 million. The competitive advantage enjoyed by irrigated agriculture (vegetables, olive, cotton, citrus, and other fruits) in this territory has generated a steady increase in the demand for agricultural water in recent decades, eventually leading to the closure of the basin [30]. Nowadays, the average water use in the GRB is 3815 Mm^3/year, the agricultural sector being the primary water user, consuming 88% of the available water while only 10% is used by households and other urban demands [31]. Structural water scarcity in this closed basin is cyclically aggravated by drought episodes, recurrent in Mediterranean climate regions. Climate change predictions for the GRB suggest that drought events are expected to become more frequent and intense [32].

The basin is closed to new users because any further increase in the water supply is impossible. Thus, several demand-side policy instruments have been implemented to facilitate a more efficient reallocation of the scant water resources. In this sense, the 1999 reform of the Spanish Water Act allowed the implementation of water trading instruments, both spot water markets between productive users and the temporary establishment of public water banks (termed 'water exchange centers') for environmental purposes in the event of 'exceptional situations of water scarcity' (i.e., severe droughts).

The water rights priority system established by law in Spain guarantees that urban uses are served first in case of water scarcity. This means that only farmers are willing to participate in water trading as a drought management instrument since of all the users, they are the most negatively affected by shortages in the water supply. During this cyclical scarcity, only irrigators with valuable productions are willing to buy additional allocations, while only the farmers with low marginal value of water are willing to lease their own water rights. However, market activity has been limited to date because of the high transaction costs [12]. Moreover, even in the most severe drought episodes (e.g., 2007-2008), public water banks have not been implemented in the GRB due to the lack of political will (i.e., public budget) to implement them [33].

The GRB Management Plan [GRBMP, CHG 34] establishes a minimal ecological flow of 305 Mm^3/year to guarantee the resilience of environmental services. However, during drought periods, when instream flows are just above this minimum level, water-related ecosystems are negatively impacted, and society would be willing to recover water from agriculture to minimize environmental damages [35].

All the reasons provided above (the structural and cyclical water scarcity, the legal framework for water trading instruments, the heterogeneity in the marginal value of water between irrigators and the social demand for the recovery of environmental water) justify the use of the GRB as case study to assess the potential performance of the two water bank arrangements proposed as demand-side instruments aimed at improving drought management.

4.2. Feeding the models simulating the performance of the water banks in the GRB

The empirical analysis performed relies on the aggregate supply curves of water from agricultural users $Q_p(P_p)$ and the aggregate demand curves of water for irrigation $Q_s(P_s)$ in the GRB estimated by Gutiérrez-Martín et al. [17]. These authors based their estimations on the results of mathematical programming models built by Montilla-López et al. [33] to simulate irrigators' heterogenous trading behavior within a water bank framework (decisions regarding the sale and purchase of water allocations) in this basin.

Gutiérrez-Martín et al. [17] estimated aggregate supply and demand curves for three different scenarios of water availability (intensity of droughts): a) 'moderate' drought, where water allocations to irrigators are only 75% of those granted in an average hydrological year; b) 'severe' drought, where only 50% of the water granted in an average hydrological year is allocated to irrigators; and c) 'extreme' drought, where allocations to irrigators are only 25% of those granted in an average hydrological year. These scenarios are labeled as S75%, S50%, and S25%, respectively. The polynomial functions expressing the aggregate supply and the aggregate demand in each scenario are shown in Table 1. These functions are the ones used for simulating the purchase and sale offers implemented by the public water agency within the two alternative institutional arrangements proposed for the water bank.

Table 1 Aggregate supply and demand.

Drought scenario	Aggregate supply curve $Q_p(P_p)$	Aggregate demand curve $Q_s(P_s)$
Moderate drought (S75%)	$Q_p = 2273.9\ P_p - 30.6$	$Q_s = -2759.9\ P_s + 742.5$
Severe drought (S50%)	$Q_p = -294.9\ P_p^2 + 2214.9\ P_p - 191.6$	$Q_s = 407.4\ P_s^2 - 2755.8\ P_s + 1041.1$
Extreme drought (S25%)	$Q_p = 770.1\ P_p - 52.9$	$Q_s = 6905.9\ P_s^2 - 8216.6\ P_s + 2443.5$

Source: Gutiérrez-Martín et al. [17].

Another common setting for both institutional arrangements is related to the social demand for environmental water $Q_e(V_e)$. Unfortunately, there is no previous estimate of this demand in the GBR that can be used for the empirical analysis proposed. This justifies the use of the simplified linear demand defined in eq. 1. The parameter Q_e^{max} has been estimated considering that the average instream flow (the flow registered in an average hydrological year) in the GRB is 7092 Mm^3/year, while the minimum ecological flow established in the GRBMP is 305 Mm^3/year. Thus, it is reasonable to assume that the maximum quantity of water that society would demand for environmental purposes with a non-zero willingness to pay (Q_e^{max}) is the difference between average flow and minimum ecological flow (6787 Mm^3/year). However, there is no robust empirical estimate available for the maximum value that society would be willing to pay for environmental water (V_e^0). Given this lack of information, the potential performance of both water bank designs is simulated parameterizing this value from 0 to 1 euro per cubic meter. This range of hypothetical values for V_e^0 is considered wide enough to comprise all the unit costs for water supply estimated for Spain [36] and all water prices paid in the existing spot water markets [12].

Furthermore, the parameter MCF, used for modeling both water bank designs, is taken as equal to 1.2. This means that every 100 monetary units collected by the public sector through taxes involves efficiency losses of 20 monetary units, due to the distortion of the whole economic system caused by taxation. This reference value for this parameter is based on more recent estimates for Spain [37], which are in line with the values estimated for most OECD countries [38].

5. Results

5.1. Competitive water bank

Figure 3 shows the results obtained for the competitive water bank for the three water scarcity scenarios proposed (S75%, S50%, and S25%) in terms of volumes traded, clearing prices, public funding required, and efficiency improvements relative to the baseline scenario (no trading) achieved for every maximum marginal value of environmental water (V_e^0) ranging from 0 to 0.8 €/m^3. The three scenarios considered reveal net efficiency gains compared to the baseline scenario, although with some differences.

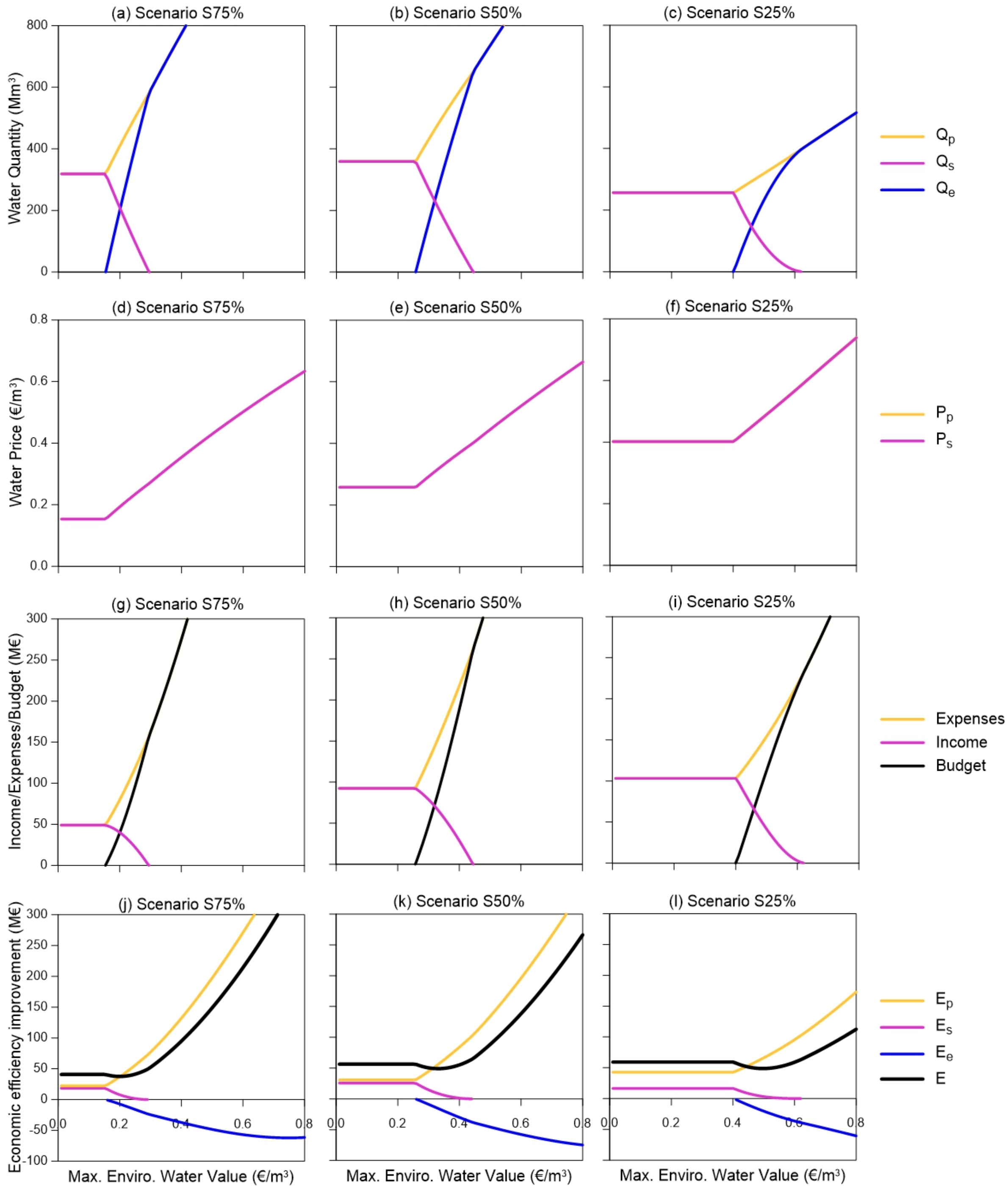

Fig. 3 Results for competitive water bank: volumes traded, prices, cash flows, and efficiency improvements.

The parametrization of V_e^0 shows three defined stages or phases in all the scenarios considered. The main features of every phase are summarized in Table 2. The *first phase* is found for the lowest values of V_e^0, with this phase characterized by the fact that no water is recovered for the environment (see Figs. 3a-c) as the marginal value of environmental water is below the market equilibrium price ($P_p = P_s$, see Figs. 3d-f). Thus, the water bank buys water allocations from farmers who are willing to sell and subsequently sells them all to other farmers who are willing to buy ($Q_p = Q_s$). Since expenditure on purchases equals revenue obtained in sales (i.e., the bank is self-financed), no public funding is needed for water bank operations (see Figs. 3g-i).

This first phase covers the values of V_e^0 ranging from 0 to the hypothetical clearing prices of a competitive market in each drought scenario. Since this hypothetical clearing price for each scenario is higher when the water shortage is more acute, ranging from 0.15 €/m^3 for scenario S75% to 0.40 €/m^3 for scenario S25% (see Figs. 3d-f), the more severe the drought, the longer the first phase.

Comparing the volume of water traded (see Figs. 3a-c), it can be observed that the largest transfers measured in absolute terms (Mm^3) are found in the scenario S50%. However, it is worth pointing out that the quantity of water traded measured in relative terms (water trade over total water availability) rises from 16.0% in scenario S75% to 27.0% in S50%, and to 38.6% in scenario S25%.

Relative to the no-trade baseline scenario, it is shown that this water bank design involves an improvement in economic efficiency for the three water availability scenarios, amounting to around 50 million euros (see Figs. 3j-l). During this first phase, this efficiency enhancement is due only to the surpluses generated in purchases (E_p) and sales (E_s), since no water is recovered for the environment (E_e=0).

Table 2 Ranges and main features of the competitive water bank model.

Phase	V_e^0 **range (€/m³)** *S75%*	*S50%*	*S25%*	**Operation description**	**Economic relations**
Phase 1	0.00-0.15	0.00-0.25	0.00-0.40	The bank buys water allocations from farmers and subsequently sells them all to other farmers. No water is recovered for the environment. The bank is self-financed.	$P_p = P_s$ $Q_p = Q_s$ $Q_e = 0$ $Budget = 0$
Phase 2	0.26-0.29	0.26-0.44	0.41-0.62	The bank buys from farmers; a share is reallocated to other farmers and the rest is recovered for the environment. The bank needs public funding to finance purchases for environmental water.	$P_p = P_s$ $Q_p = Q_s + Q_e$ $Q_e > 0$ $Budget > 0$
Phase 3	>0.30	>0.45	>0.63	The bank buys from farmers and all water is allocated to the environment. The bank needs public funding to finance purchases for environmental water.	$P_p = P_s$ $Q_p = Q_e$ $Q_s = 0$ $Budget > 0$

After the maximum environmental water value (V_e^0) reaches the corresponding clearing prices, the simulation results of the competitive water bank enter a *second phase,* where a share of the water allocations bought from farmers is recovered for the environment, reallocating the rest among other farmers ($Q_p = Q_s + Q_e$ and $Q_e > 0$). The higher the maximum environmental water value, the larger the quantities purchased and recovered for the environment, and the lesser the quantity reallocated among farmers (see Figs. 3a-c). This second phase ends when V_e^0 becomes so high that no water is reallocated for productive uses and all purchased allocations are recovered for the environment.

During this second phase, the water agency managing the bank needs public funding to support the purchases of environmental water (see Figs. 3g-i). The amount of public funding needed increases as V_e^0 and the amount of water bought (Q_p) rise.

Regarding the efficiency assessment (see Figs. 3j-l), it is worth noting that over this second phase, the surplus generated in sales (E_s) decreases as the water reallocated for irrigation declines, while the surplus generated in agricultural purchases (E_p) increases as the water bought from irrigators rises. Moreover, environmental water purchases also generate a surplus (E_e), although this environmental surplus takes negatives values. This negative sign for environmental efficiency can be explained by the fact that this spending policy involves an efficiency loss derived from the economic distortions caused by the tax system when raising the public funds needed. That is, the efficiency gains stemming from the improvement in instream flows (measured as $\int_{P_s}^{V_e^0} Q_e(P_s)\, dP_s + P_s \cdot Q_e$ in expression 4.4) do not offset the loss of economic efficiency caused by raising public funds through taxes, measured as the marginal cost of public funds ($Budget \cdot MCF$ in the same expression). This negative environmental efficiency means that total economic efficiency ($E_p + E_s + E_e$) at the beginning of the second phase is slightly lower than in the first phase since the increases in the producer surplus are outweighed by the sum of the decreases in consumer and environmental surpluses. However, once the minimum total efficiency has been reached for 0.20, 0.33, and 0.50 €/m^3 for the scenarios S75%, S50%, and S25%, respectively, this measure of economic efficiency increases in all scenarios. Indeed, by the end of the second phase, these values are higher than in the first phase.

The *third phase* begins when the maximum environmental water value is high enough that all water purchases are allocated to the environment ($Q_p = Q_e$) and consequently, no water is sold for productive uses ($Q_s = 0$). This is when V_e^0 is higher than 0.30, 0.45, and 0.63 €/m^3

for the scenarios S75%, S50%, and S25%, respectively. In every scenario, total economic efficiency continues to follow the rising trends observed at the end of the second phase, since when values of V_e^0 are high enough, the increases in the producer surplus outweigh the sum of the decreases in consumer and environmental surpluses. The higher the value of V_e^0, the higher the economic efficiency; in all cases, the total efficiency improvements are greater than those obtained in the first phase.

5.2. Monopsony-monopoly water bank

In the case of the monopsony-monopoly water bank, the parametrization of V_e^0 shows four phases instead of three, as can be seen in Fig. 4. The main features of these phases are summarized in Table 3.

The *first phase* is identical to that of the previous model, where V_e^0 is below the market equilibrium price ($P_p = P_s$). Thus, no water is recovered for the environment ($Q_p = Q_s$; $Q_e = 0$) and the water bank is self-financed.

A *second phase* can be observed for the following ranges of V_e^0: 0.16-0.20 €/m^3, 0.26-0.32 €/m^3 and 0.41-0.49 €/m^3 for S75%, S50%, and S25%, respectively. Over this relatively short phase, the water bank reduces its purchases of water allocations from farmers. A decreasing share of these purchases is reallocated to other farmers while an increasing portion is recovered for the environment ($Q_p = Q_s + Q_e$) (see Figs. 4a-c). Unlike with the competitive bank, it is worth pointing out that during this phase, the agency managing the monopsony-monopoly water bank exerts its market power to create a gap between the purchase and the sale prices ($P_p < P_s$ as shown in Figs. 4d-f), allowing the bank to be self-financed since no public funding is needed for the acquisition of environmental water (see Figs. 4g-i). Moreover, the water agency maximizes total efficiency for these ranges of V_e^0 by discriminating between the marginal value of environmental water (V_e) and the sale price ($P_s < V_e$ as shown in Fig. 4 d-f). Discriminating between these two water values leads to improvements (positive values) in environmental efficiency; in this case, the efficiency gains resulting from the improvement in instream flows (measured as $\int_{V_e}^{V_e^0} Q_e(V_e)\, dV_e + V_e \cdot Q_e$ in expression 5.4) outweigh the loss of economic efficiency caused by raising public funds through taxes, quantified through the marginal cost of public funds ($Budget \cdot MCF$ in the same expression). There is therefore a continuous increase in total economic efficiency ($E_p + E_s + E_e$) as shown in Figs. 4j-l;

consequently, the level of economic efficiency linked to water use at basin level from this phase onwards is higher than in the competitive water bank.

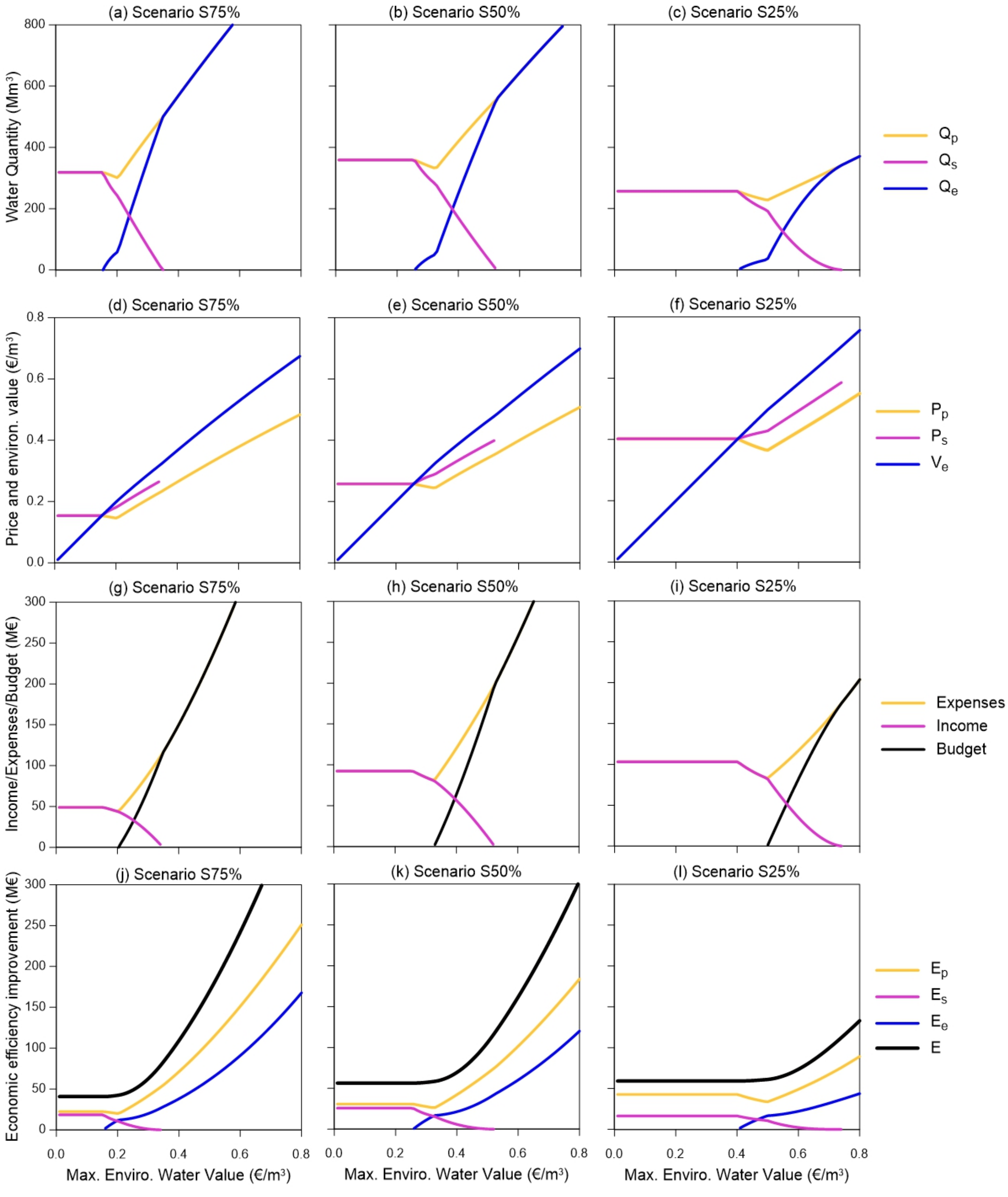

Fig. 4 Results for monopoly-monopsony water bank: volumes traded, prices, cash flows, and efficiency improvements.

Table 3 Ranges and main features of the monopsony-monopoly water bank model.

Phase	V_e^0 range (€/m³) *S75%*	*S50%*	*S25%*	Operation description	Economic relations
Phase 1	0.00-0.15	0.00-0.25	0.00-0.40	The bank buys water allocations from farmers and subsequently sells them all to other farmers. No water is recovered for the environment. The bank is self-financed.	$V_e < P_p = P_s$ $Q_p = Q_s$ $Q_e = 0$ $Budget = 0$
Phase 2	0.16-0.20	0.26-0.32	0.41-0.49	The bank buys from farmers; a share is reallocated to other farmers and the rest is recovered for the environment. The water bank is self-financed, exerting its monopsony-monopoly power.	$P_p < P_s < V_e$ $Q_p = Q_s + Q_e$ $Q_e > 0$ $Budget = 0$
Phase 3	0.21-0.34	0.33-0.52	0.50-0.74	The bank buys from farmers; a share is reallocated to other farmers and the rest is recovered for the environment. The water bank exerts monopsony-monopoly power but needs public funding.	$P_p < P_s < V_e$ $Q_p = Q_s + Q_e$ $Q_e > 0$ $Budget > 0$
Phase 4	>0.35	>0.53	>0.75	The bank buys from farmers and all water is allocated to the environment. The water bank exerts monopsony-monopoly power but needs public funding.	$P_p < V_e$ $Q_p = Q_e$ $Q_s = 0$ $Budget > 0$

The *third phase* begins when values of $V_e{}^0$ become high enough that the water bank has to spend public funds to further increase economic efficiency by recovering more water for the environment (see Figs. 4a-c). That is, over this third phase, the monopsony-monopoly power that allows the public agency to self-finance its acquisition of more environmental water is not enough to further increase economic efficiency, and an increasing amount of public funding is needed to purchase more water allocations for environmental purposes (see Figs. 4g-i). Nevertheless, unlike with the competitive water bank, it is worth pointing out that discriminating between water values ($P_p < P_s < V_e$) allows the water agency to maintain positive values (improvements) for environmental efficiency E_e, even at increasing positive marginal values (the higher the value of $V_e{}^0$, the higher the slope of E_e).

The gradual substitution of Q_s by Q_e ends at a point of $V_e{}^0$ where the agricultural sector cannot afford to compete for the purchase of water allocations with the social demand for environmental water ($Q_s = 0$). This point defines the beginning of the *fourth phase*, for values of V_e^0 above 0.35 €/m³, 0.53 €/m³and 0.75 €/m³ in the case of S75%, S50%, and S25%, respectively (see Figs. 4a-c). Over this phase, all the water allocations bought from farmers are devoted to the environment ($Q_p = Q_e$), supported by an increasing amount of public funding (see Figs. 4g-i). However, the monopsony-monopoly power of the water agency allows it to keep purchase prices below the marginal environmental value of water ($P_p < V_e$, as shown in Figs. 4d-f); thus we observe an increasing trend in total efficiency improvement (see Figs. 4j-l).

6. Discussion

The results have shown that both water bank settings can generate an increase in economic efficiency compared to the no-trade baseline scenario. However, the most relevant hypothesis to be tested is whether the monopsony-monopoly water bank improves drought management from a social point of view compared to current policy (competitive water bank). In order to do so, the results obtained for the competitive and the monopsony-monopoly water banks are compared using differential values of performance variables (volumes traded, prices, cash flows, and efficiency improvements). In this regard, Figs. 5 and 7 show the differences calculated by subtracting the values of the variable obtained in the simulations of the competitive water bank from the results found for the monopsony-monopoly water bank. For instance, a negative value for the volume of water recovered for the environment (Q_e as shown in Figs. 5a-c) indicates that the monopsony-monopoly bank secures less water for the environment than the competitive bank for the value of V_e^0 and drought scenario considered.

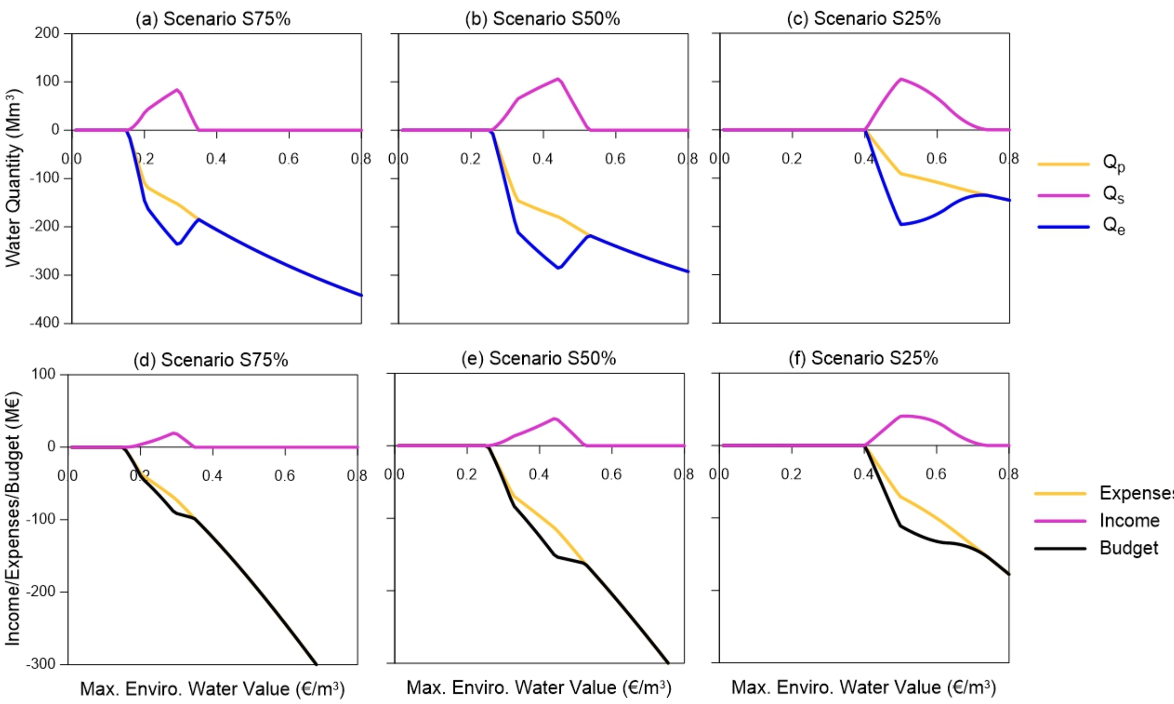

Fig. 5 Differences between monopsony-monopoly and competitive water banks. Volumes traded and cash flows.

The differential values observed in Fig. 5 show that the competitive water bank recovers more water for the environment, although more public funding is needed compared to the monopsony-monopoly water bank. The case of V_e^0=0.60 for scenario S50% can be taken as an example. In both water bank arrangements, this maximum value of environmental water means that no water allocations are sold to farmers (point located in the last phase). At this point, the competitive water bank recovers 240 Mm^3 more water for the environment than the monopsony-monopoly bank (see Fig. 5b). However, at this point, the competitive water bank needs 205 million euros more public funds to implement these environmental purchases (see Fig. 5e).

In order to provide a broader view of environmental purchases of water allocations, it is worth estimating the unit costs of environmental water recovered in terms of the public funding required, i.e., the ratio between the funds needed and the volume recovered expressed in euros per cubic meter. Figure 6 shows the differences between the two banks proposed in terms of the unit costs borne by the public agency managing the banks for any quantity of water recovered for the environment in each scarcity scenario. The lower costs for almost every quantity and drought scenario provide evidence of the advantage offered by the monopsony-monopoly water bank in terms of the public budget assigned (less public funding is needed per unit of water recovered for environmental purposes), especially in the phase where the monopsony-monopoly water bank is self-financed. It can be seen than the lower the quantity of water to recover, the greater the gap in the cost per cubic meter between the two bank designs. On the contrary, for large quantities of water to recover, the advantage of the monopsony-monopoly bank is diminished. Indeed, for V_e values greater than the maximum P_s, the agricultural sector does not participate in the market, meaning that the cost of environmental water is the same for both water bank designs, as can be seen in Fig. 6 for the scenario S25% when the volumes of water recovered are higher than 400 Mm^3.

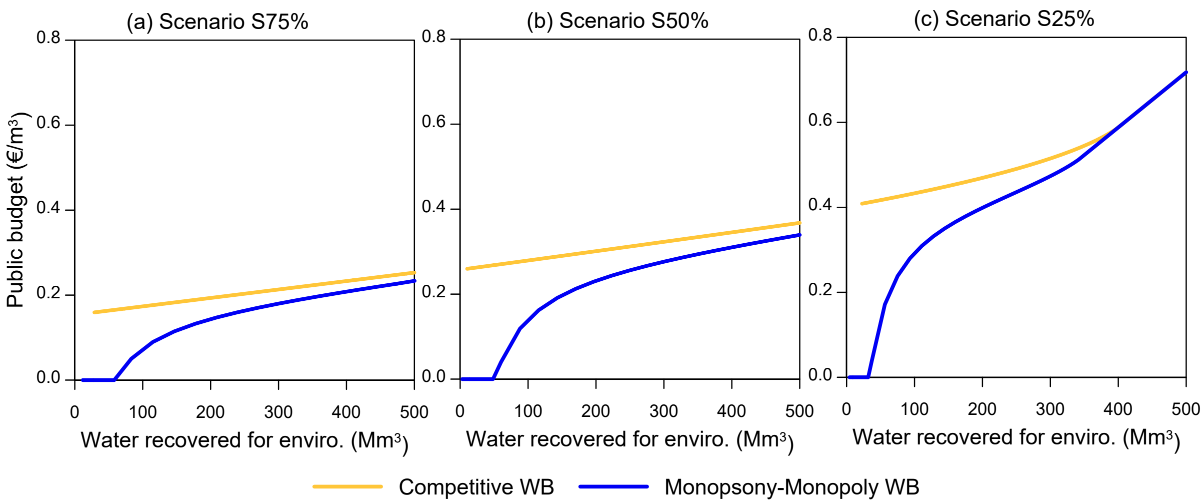

Fig. 6 Differences in public funding required to recover water for the environment between monopsony-monopoly and competitive water banks.

Figure 7 shows the differences between the two water bank settings considered regarding economic efficiency improvements. The most relevant outcome to highlight is the higher levels of total economic efficiency E achieved with the monopsony-monopoly water bank compared to the competitive one for every drought scenario when the values of V_e^0 exceed the competitive equilibrium prices. This confirms the potential benefits of the bank proposed here as an economic instrument to improve drought management.

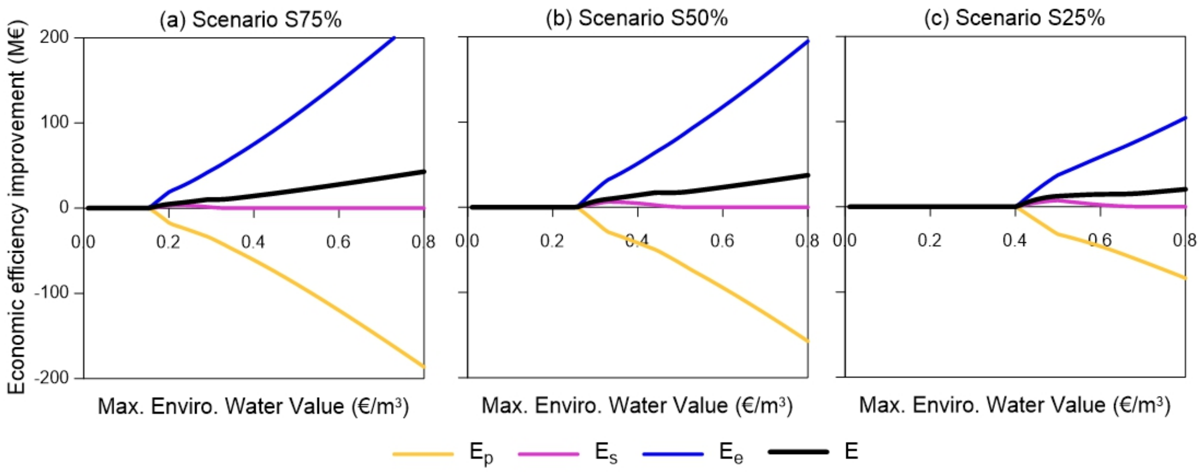

Fig. 7 Differences between monopsony-monopoly and competitive water banks. Economic efficiency improvements.

Finally, the role of the parameter measuring the marginal cost of public funds (MCF) in the empirical analyses also merits further analysis. As stated above, one of the contributions of this paper is that it accounts for this additional cost of using public funds, thus providing a more accurate estimation of efficiency improvements when implementing a water bank with environmental purposes. In this regard, it should be pointed out that the value for the MCF taken for the analysis performed (MCF=1.2) is a reasonable assumption considering a proportional tax rate increase (an equal marginal tax rate increase in all tax brackets) [38]. However, the estimates of this parameter vary significantly depending on the assumptions regarding how additional public funds are collected (i.e., taxation instruments implemented). In fact, more recent estimates for Spain [37] report values of MCF ranging from 0.82 to 1.88. In this sense, it is worth noting that this parameter has a marked effect on simulation results, since any change in the value considered would involve relatively large changes in the performance simulated for both water bank designs. Generally speaking, it can be stated that the higher the value of MCF, the more attractive the monopsony-monopoly water bank becomes, since differences in efficiency improvement between the two alternative banks grow (higher values of MCF lead to higher inefficiencies in the competitive water bank from phase 2 onwards). As an illustrative example, consider V_e^0=0.60 for the scenario S50%; if MCF increases from 1.2 to 1.4, the difference in the total economic efficiency improvement achieved between the monopsony-monopoly and the competitive water banks increases from 23.4 million euros to 74.9 million euros, further underlining the attractiveness of the monopsony-monopoly water bank proposed.

It is also worth pointing out that, for medium-to-high social values of environmental water, the economic efficiency achieved with the monopsony-monopoly water bank design proposed in this paper far exceeds the level generated by the self-financed monopsony-monopoly water bank proposed by Gutiérrez-Martín et al. [17]. Despite the appealing prospect of recovering water for the environment without any public expenditure, the amount of water that can be recovered for the environment by the latter is markedly lower than that achieved by the former, which is ultimately reflected in lower levels of economic efficiency. Indeed, the monopsony-monopoly water bank suggested in this paper shares some characteristics with the design proposed by Gutiérrez-Martín et al. [17], in that the two water banks operate identically until the start of phase 3, at which point the former makes use of public funds to purchase additional environmental water, thereby achieving higher levels of economic efficiency.

7. Conclusions

The public monopsony-monopoly water bank proposed is an interesting drought management policy option to recover water for the environment during scarcity events. This study demonstrates its potential to outperform the alternative design currently implemented (public purchases within an existing competitive spot water market), in terms of economic efficiency. The success of this water bank arrangement is based on its ability to partially self-finance public purchases of environmental water, with the public agency managing the bank using its market power to create a gap between the purchase and the sale prices, and also to set different prices at which the water bank can reallocate the water among end users (differentiating between prices to farmers willing to buy $-P_s-$ and prices to society for environmental water $-V_e$).

A key innovation of this study is that it accounts for the inefficiencies inherent in the use of public expenditure, measured through the marginal cost of public funds (MCF), when making public purchases of water allocations (or rights). It has been demonstrated that the net economic efficiency variations when purchasing water for the environment in an existing competitive water market are not always positive. In fact, for a range of environmental water values, this policy option leads to a decrease in total economic efficiency since the losses caused by the distortion effects of raising public funds on the whole economy are not fully offset by the efficiency increases achieved through market operations. Moreover, this finding suggests that previous assessments of public water recovery for the environment through market instruments could have overestimated efficiency gains because they ignored inefficiencies relating to public expenditure. It is suggested that, going forward, any policy assessment involving the public purchase of water allocations or rights should account for these inefficiencies through the marginal cost of public funds.

Even accounting for the inefficiencies regarding the use of public funds, it has been shown that the monopsony-monopoly water bank proposed can increase total economic efficiency for every value of environmental water, in all cases yielding equal or higher economic efficiency estimates than the competitive market option.

Although it would not change the conclusions presented above, it is worth commenting that the simulation results reported could be improved with more accurate estimates of some of the parameters considered in the models; specifically, society's demand for environmental water and the MCF parameter. Given the lack of accurate information about public willingness to pay for environmental water, we assumed linear demand and chose to parametrize the maximum value that society would be willing to pay for public water purchases (V_e^0). In order to better estimate society's demand for environmental water, there is a need for future research to quantify the impact of a change in instream flows on ecological condition, its effects on environmental service provision, and the value of the change in environmental service provision to society. In this sense, it is worth remarking that these demand curves are needed to enable policy-makers to look for optimum social water allocation arrangements. Regarding the MCF parameter, it should be noted that the estimate used in the model (MCF=1.2) is based on estimates for other developed countries, for a proportional increase in all taxes collected at national level. However, the actual value for this parameter should also be more accurately estimated for the case study analyzed, considering the inefficiencies caused by the Spanish tax system (the proposed bank would be financed by the Spanish national budget) and how the tax increase would be implemented. From a practical point of view, it is obvious that accurate estimates of both society's demand and MCF are a prerequisite for the proper implementation of the proposed economic instrument in a real-world setting.

Finally, it is also worth pointing out that, for the sake of simplicity, the assessment performed has been carried out assuming zero transaction costs. In this sense, it would be interesting for further research to estimate both institutional and operational transaction costs [39] for the case of the monopsony-monopoly water bank proposed, and to analyze their effect on its potential performance.

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