Abatement and transaction costs of water reallocation

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ABSTRACT

Water reallocations have costs to the users of water, or abatement costs (e.g., charges designed to marginally increase environmental water flows), but also nontrivial institutional transaction costs (e.g., costs incurred to develop institutions and organizations to support and enforce environmental reallocations). However, institutional transaction costs studies are very limited and those available do not integrate abatement costs measurements, which constrains our ability to assess the performance of water reallocation. This paper presents the first integrated analysis of abatement and transaction costs of water reallocation. The analysis is illustrated with an application to the Douro River Basin, an agricultural basin in central Spain that has recently finished its second planning cycle (2015–2021). First, we use a hydroeconomic model that accounts for the two-way feedback responses between human and water systems to estimate the abatement costs of water reallocations, as well as their effectiveness in achieving the good ecological status of water bodies. Second, we measure and monetize realized institutional transaction costs of river basin planning over time and build on this cutting-edge longitudinal dataset to assess future directions and magnitude of transaction costs. We use this information to assess and rank the performance (through cost-effectiveness) of the water reallocations considered in the latest Douro River Basin Plan under alternative climate change scenarios. We find that under the hypothesis of stationary transaction costs, these can represent between 5.7% and 8.3% of the total reallocation costs (abatement plus transaction costs). This non-trivial magnitude highlights the need to account for both abatement *and transaction* costs when assessing the performance of water reallocations, and environmental policy overall.

1. Introduction

We are facing a global water crisis—one that is getting worse (UN, 2022). Population growth and changes in the standards of living are driving higher water demand (UNDRR, 2021) and climate change is reducing the available supply in arid and semi-arid regions like the Mediterranean Basin (IPCC, 2022), leading to overallocation and water scarcity. Conventional supply-based solutions to water scarcity such as dams, while historically effective, are now characterized by rapidly growing costs of new supplies that often exceed the ability to pay of users (Loch et al., 2020a,b). Moreover, where left unchecked, demand growth of equal or larger magnitude typically follows supply growth since the latter generates unrealistic expectations on the ability of water systems to accommodate new uses (Di Baldassarre et al., 2018). To achieve sustainable and equitable growth a redistribution of water resources (or reallocations) is required to shift all economic uses towards or to sustainable levels (OECD, 2015; World Bank, 2021).

Water reallocations via marginal reduction of prior uses will have costs to all water users, or *abatement costs* (e.g., charges designed to marginally increase environmental water flows through reallocation away from agricultural users). Water reallocation will also require nontrivial *institutional transaction costs* incurred in developing institutions and organizations to support and enforce reallocations (e.g., costs incurred to design and develop institutions and organizations to support, monitor, and enforce environmental reallocations) (Krutilla and Krause, 2011). Since both abatement and transaction costs determine the total cost of water reallocations their quantification would seem a prerequisite for any policy performance assessment. However, most performance assessments to date focus on quantifying abatement costs and ignore institutional transaction costs, and the limited

transaction costs studies available do not measure abatement costs. This may be constraining our ability to assess the true performance of water reallocation and other environmental policies. The remainder of this section presents the two (hydro)economic traditions from which abatement (neoclassical economics) and institutional transaction costs (institutional economics) concepts and methods emerge, provides an overview of the existing literature, and summarizes how this work combines both approaches into the first integrated analysis of abatement and transaction costs of water reallocation.

Most water reallocation assessments have focused on quantifying **abatement costs**, which can also be defined as the opportunity cost of relinquishing economic water uses, typically towards meeting fixed or variable environmental flow requirements (see e.g., Medellín-Azuara et al., 2007). Abatement costs are usually quantified by means of combining hydrology and neoclassical microeconomics techniques and models that can be used to estimate the economic repercussions of water shifts (demand and/or supply) over the system (Harou et al., 2009). The combination of economics and hydrology is often implemented by incorporating economic demand curves into full-fledged hydrologic models, which can be subsequently used to assess benefits conditional to environmental constraints, and thus identify those reallocations that perform better. Early uses of demand functions in hydrologic modeling can be found in Bear et al. (1964) and Bear and Levin (1966), whose pioneering research kickstarted hydroeconomics (Noel and Howitt, 1982). Non-economic hydrologic models, treat water demand as an exogenous and static variable, while hydroeconomic models endogenize human behavior into the water system thus allowing for adaptive and dynamic decisions from economic agents (e.g., irrigators) that are driven by opportunity costs.

In-depth reviews on hydroeconomic research can be found elsewhere in the literature (Harou et al., 2009; Ortiz-Partida et al., 2023). These reviews characterize hydroeconomic models around three key axes: problem resolution (optimization, simulation), time representation (deterministic time series, stochastic and multi-stage stochastic, dynamic optimization), and model integration (modular, holistic). A recent review of 198 hydroeconomic models (González-López et al., 2023) revealed that most hydroeconomic models use a dynamic (85.4 %) and holistic (69.2 %) setup, while the proportion of models using simulation and optimization was similar—and often both approaches are used simultaneously (e.g., hydrologic simulations that feed economic optimization models using a modular setup). The review also revealed that most holistic models rely on (potentially over-) simplified equations rather than full-fledged representations of the system to represent some key processes, notably the behavior of economic agents, with parts of the hydrology being often (over)simplified as well. This approach has the advantage of more effectively representing causal relationships and interdependencies but reduces detail in the simplified sub-model. Examples of holistic models include the integration of the Water Evaluation And Planning System (WEAP) with various microeconomic models, where the latter are used as an external sub-model to produce demand functions that are incorporated into WEAP (Forni et al., 2016; Lempert and Groves, 2010). The reduction in the detail of key processes and entire systems (notably the human system) can be problematic where feedbacks between hydrology and the economy can lead to emergent phenomena and behaviors that cannot be explained by the simplified sub-model(s) (Graveline and Mérel, 2014; Konar et al., 2019). The problem of oversimplification can be addressed using modular approaches that integrate full-fledged hydrologic and economic models, as in Noel and Howitt's (1982) Linear quadratic control model (LQCM), Draper et al.'s (2003) CALVIN model, or Esteve et al.'s (2015) WEAP and Mathematical Programming Model (WEAP-MPM) *inter alia*. Modularity means that water and human systems are represented through specialized and self-contained hydrologic and economic mathematical models, which process information and generate outputs and are subsequently interconnected using bidirectional protocols, i.e. "rules designed to manage [inter-]relationships and processes between modules" (Csete

and Doyle, 2002). Modularity comes at the expense of larger computational costs (Harou et al., 2009), increased uncertainties (Franzke et al., 2022), and higher complexity, which may render models "close to useless" for decision making (Pindyck, 2017, p. 1). Building on this research, the sub-field of sociohydrology has highlighted since 2012 the need to develop novel modeling approaches that can represent the complexity of human and water systems and their two-way interactions, while addressing the tradeoffs with computational costs, uncertainty, and complexity (Sivapalan et al., 2012; Sivapalan and Blöschl, 2015). Examples in this regard include the use of novel coupling mechanisms to reduce computational costs, such as the introduction of HydroEconomic Response Units (HERU) that replace the conventional Hydrologic Response Units (HRU) in the Soil and Water Assessment Tool (SWAT) (Essenfelder et al., 2018); the adoption of ensemble forecasting techniques to quantify and address uncertainty (Sapino et al., 2023); or the incorporation of Decision Support Systems (DSS) into the modeling framework to provide a user-friendly interface that facilitates understanding and adoption by stakeholders, as is the case of WEAP (Blanco-Gutiérrez et al., 2013; Esteve et al., 2015; Sridharan et al., 2019).

Despite the nontrivial scientific advances described above and the adoption of cutting-edge hydroeconomic models for policymaking hydroeconomists often complain about the lack of willingness, or incapacity, of decision makers to implement the demand-side water reallocations their models recommend (Méndez et al., 2019; Nhim and Richter, 2022). In fact, the transition towards sustainable water use and economic development remains elusive in all arid and semi-arid basins around the world where supply-side policies and overallocation are the norm (FAO, 2023). A major result of these approaches is that the world faces a "monumental challenge" in overcoming institutional rigidities going forward (Barbier, 2011). While traditional neoclassical economics argues that marginal efficiency improvements in economic exchanges are sufficient to drive the adoption of superior allocations, applied studies show that the path-dependent increasing returns to adopted institutions and related technologies may require "up to an order-of-magnitude improvement" in economic performance to induce transition (Unruh, 2002). This reveals additional costs to policy implementation in the form of institutional transaction costs, which add up to conventional neoclassical abatement costs.

We define **institutional transaction costs** as the institutional development costs (rules and regulation capacity) and organizational investments (people and knowledge capacity) required to arrange a resource reallocation ex-ante, and then the monitoring and enforcing costs ex-post, as opposed to the more conventional abatement costs (also referred to as production costs) that are defined as the costs of executing the reallocation (Matthews, 1986). The influence of New Institutional Economics has progressively led mainstream neoclassical economics to accept the relevance of institutional transaction costs (Arthur, 1994; North, 1990), albeit their relevance is still subsidiary to that of abatement costs studies. While conceptual and theoretical research has called for more empirical work, the applied studies that quantify institutional transaction costs are still limited, and in the case of water reallocations focus on water markets in the US and Australia (Garrick et al., 2013; Loch and Gregg, 2018; McCann and Easter, 1999), with other reallocation mechanisms (which are significantly more widespread than markets) receiving much less attention (Loch et al., 2020a,b). When combined with loosely comparable abatement costs studies, these studies reveal that transaction costs can represent up to 5–20 % of total reallocation costs (i.e. transaction costs plus abatement costs) (Garrick et al., 2013). These nontrivial magnitudes suggest that transaction costs can significantly affect the optimal choice and design of reallocations and other policy instruments and effect their *ex-ante* and/or *ex-post* effectiveness assessment. Thus, these estimates need to be validated with new studies that expand the analysis of transaction costs both temporally and spatially. As per the spatial dimension, the empirical base on transaction costs of water reallocations outside the US and Australia is virtually non-existent (Njiraini et al., 2017). As per the

temporal dimension, in those areas where transaction costs data is available, studies usually do not quantify them over time. Longitudinal time-variant data is critical for the analysis of transaction costs, since conceptual models have hypothesized that these costs follow a sinusoidal trend along time (Garrick, 2015)—an hypothesis that has been experimentally validated recently for specific contexts (Loch et al., 2020a,b; Loch and Gregg, 2018).

This paper presents the first integrated analysis of abatement and transaction costs of water reallocations. The analysis is illustrated with an application to the Douro River Basin, an agricultural basin in central Spain that has recently finished its second planning cycle (2015–2021) and is presently conducting a performance assessment of the draft river basin management plan for 2021–2027 (DRBA, 2020). The analysis is implemented in two stages. First, we use a hydroeconomic modeling framework (Pérez-Blanco et al., 2021a) that accounts for the two-way feedback responses between human and water systems to estimate compliance with environmental flow regulations (*effectiveness*) and the related *abatement costs*. Next, we measure and monetize realized institutional *transaction costs* of river basin planning over time and build on this cutting-edge longitudinal dataset to assess future trends and evaluate the presence of institutional adaptive efficiencies (i.e., institutional capacity to adapt to more-likely unpredictable and surprising events in future). Insights gleaned from this process may be able to inform water reallocations elsewhere, where the relevance of transaction costs is often neglected in policy assessments.

2. Background to the case study

2.1. EU's Water Framework Directive

The Water Framework Directive (WFD) is an EU directive that asks Member States to achieve the good quantitative and qualitative status of water bodies (OJ, 2000). To this end, each river basin authority in the EU must produce a river basin management plan that provides a detailed account of the measures that will be adopted to achieve the environmental targets (including minimum environmental flows) set for the different water bodies within their basin. Measures in river basin management have historically focused on hard engineering policies (water works) that can reallocate resources across space (distribution infrastructures) and time (storage infrastructures); albeit authorities in water scarce river basins are increasingly constrained to adopt transformational reallocation policies that limit the demand of economic uses, such as water charges and caps—both of which feature prominently in the EU legal *acquis* (OJ, 2000).

A major innovation of the WFD was the integration of economic analyses into performance assessments to inform a rational discussion on the cost-effectiveness of any proposed plan and its measures (OJ, 2000). In fact, cost-effectiveness analyses of basin management plans and their measures are mandatory for all river basin authorities under the WFD. The implementation and assessment of the WFD across EU river basins, including the economic aspects thereof, is guided by the Common Implementation Strategy (CIS), which produces guidance documents on technical aspects and organizes key events among member states related to different aspects of the implementation. The guidance on water economics (WATECO, 2003) is a part of the CIS that aims at strengthening and homogenizing the knowledge and application of water economics across EU river basins and river basin management plans. WATECO establishes a common framework for economic assessments, including cost-effectiveness assessments, and provides detail on the categories and indicators of costs that should be considered, and how they should be compared to effectiveness indicators such as compliance with minimum environmental flows (which is typically used as an indicator of the quantitative status). The framework defined by WATECO is in principle adaptable, and qualitative and quantitative indicators are allowed in the cost-effectiveness analysis—albeit the latter are used in Spanish basins. Importantly, the categories and indicators of costs that

are considered are limited in scope and refer to abatement costs only (CIS Working Group 2B, 2004), where institutional transaction costs are key to affecting reallocation contracts, monitoring and enforcement. An additional problem is the insufficient integration between economic and hydrologic methods in cost-effectiveness and other performance assessments, since most river basin authorities rely on stand-alone hydrologic models (sometimes complemented with exogenous economic estimates) for their assessments, ignoring the two-way feedbacks between human-water systems (EC, 2015).

2.2. The Douro River basin in Spain

To achieve its objectives, the WFD constrains member states to develop and periodically update programs of measures in river basin management plans. River basin management plans must be updated every six years through successive planning cycles, the second of which finalized in 2021 (2015–2021 cycle, which follows the 2009–2015 first cycle). In this context, EU river basin authorities are conducting performance assessments of the resultant river basin management plans. One such river basin is the Spanish part of the Douro River Basin. The Douro River Basin has a total surface of 78,889 km^2 that approximately corresponds to the Castile and León region (NUTS2[1]) in Spain. Water supply has been decreasing over the past decades, from a yearly average of 12,892 million m^3 for the period 1940–2017 to 11,934 million m^3 for the period 1980–2017, or a −7.5 % decrease in supply (DRBA, 2020). Demand from economic uses, on the other hand, totals on average 4,330.24 million m^3 per year (DRBA, 2020), leading to a ratio of water withdrawals to available resources higher than 20 %, which is the threshold used by the European Environment Agency to identify a basin as water scarce (EEA, 2024).

The response of the Douro River Basin Authority to the challenges posed by growing water scarcity and more stringent environmental standards involves a combination of: hard reallocation policies (water works), such as new dams and canals; and soft reallocation policies, notably caps (during droughts) and charges to recover the costs of water works (including environmental costs) and caps. The complete set of water works considered in the draft of the Douro River Basin Management Plan (DRBA, 2020), along with their construction and environmental costs, are listed in Annex I in the online supplementary material and include 21 new dams, 7 new canals and 24 irrigation modernization projects. The estimation of the environmental costs of water works is based on the methodology adopted by the Douro River Basin Authority for the economic assessment of dam construction projects (explained more in detail e.g. in Pérez-Blanco et al., (2021b) and Gil-García et al., (2023)); while the annuity payment and water charge that is levied on water users is calculated following the Spanish Water Law (BOE, 2003, 1985). Importantly, water reallocation policies are complemented with other measures aiming at enhancing economic outcomes e.g., through the reallocation of part of the expected water savings towards economic sectors via the expansion of irrigated areas.

All the water reallocation policies above target the agricultural sector, the largest consumptive water use in the basin. Agriculture represents 89.4 % of total water withdrawals for consumptive uses within the Douro River Basin, produces 5.5 % of the Castile and León's GDP (Spanish average: 2.7 %) and employs 10 % of its workforce (Spanish average: 5 %) (INE, 2022). More than a half of the basin's surface area is covered by agriculture (5 783 831 ha), 45.6 % of which is irrigable (of which 20.9 % are effectively irrigated) (see Table 1). The irrigated area is divided into 150 Agricultural Water Demand Units (AWDUs), the basic irrigation unit in Spain, which comprises "groups of agricultural areas sharing a common source of water, territorial,

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[1] The Nomenclature des Unités Territoriales Statistiques (NUTS), is "a hierarchical system for dividing up the economic territory of the EU" (Eurostat, 2020). In Spain, NUTS 2 refers to regions.

Table 1

Agricultural land use in the Douro River Basin – irrigated and rainfed crops, including crop categories as shown in Results Section. Own elaboration from ITACyL (2019).

Category (as shown in results section)	Crop	Irrigated area (ha)	Rainfed area (ha)
Wheat	Wheat	**39 577**	765 960
Other cereals	Barley	**23 361**	677 675
	Oats	**6 461**	42 699
	Rye	**2 491**	57 075
	Triticale	**1 302**	17 992
	Corn	**116 667**	
	Forage corn	**7 323**	72
Vegetables and fruits	Apple tree	**1 525**	
	Green pea	**2 534**	
	Potato	**17 614**	
	Carrot	**6 003**	
	Vineyard	**3 296**	59 703
	Walnut	**446**	310
	Almond	**600**	3 300
	Lettuce	**710**	
	Spinach	**576**	
	Green chicory	**468**	
	Pumpkin	**630**	
	Garlic	**2 581**	152
	Onion	**2 931**	
	Leek	**1 837**	
Oilseeds	Sunflower	**16 300**	184 649
	Olive tree	**943**	2 777
Sugar beet	Sugar Beet	**24 744**	
Other crops	Alfalfa	**39 737**	43 889
	Vetch	**1 836**	37 354
	Forage vetch	**3 578**	49 556
	Pea	**1 475**	35 239
	Lentils		3 936
	Chickpeas	**516**	5 968
	Lupine		3 106
	Carob		1 417
	Hop	**942**	
	Rapeseed	**5 276**	14 218
	Total	**334 280**	1 969 621

administrative, and hydrological characteristics" (DRBA, 2016) (Fig. 1). AWDUs are the economic agents in the human system module used for the hydroeconomic modeling and assessment of abatement costs. Since

agriculture is a low priority water use in Spain (i.e. the first one to experience water restrictions under drought/scarcity conditions), growing scarcity and the more restrictive environmental standards introduced through the WFD increasingly constrain agricultural water use. This will be compounded by future climate change, which will aggravate scarcity and increase the magnitude and frequency of drought events as well as their economic and environmental costs (IPCC, 2022; MAGRAMA, 2017). Accordingly, economic and hydroeconomic assessments on the costs and effectiveness of adaptation policies to water scarcity under climate change in the Douro and elsewhere in Spain have focused on the agricultural sector (DRBA, 2020).

3. Methods

Building on previous contributions (Challen, 2000; Marshall, 2005; North, 1990), Marshall (2013) theorizes a conceptual framework to quantify the cost-effectiveness of environmental policies that accounts for both abatement and transaction costs. According to Marshall (2013), accounting for transaction costs is important because the abatement costs advantages of environmental policy can be outweighed by its transaction costs disadvantages, and vice versa. Transaction costs also influence policy choices, which in turn affect technological choices and agents' decisions thereby affecting lock-in and related abatement costs. Accordingly, evaluating the performance of water reallocations in meeting the good ecological status of water bodies requires a "comparison of the total cost impacts [i.e., transaction plus abatement costs] of the policy relative to a consistent counterfactual scenario". The categories of abatement and transaction costs considered in Marshall's (2013) conceptual framework are detailed in Table 2.

According to Marshall's (2013) conceptual framework, "the appropriate criterion for evaluating cost-effectiveness in achieving a given adaptation target" (i.e., good ecological status of water bodies) involves "identifying the option that minimizes the sum of the cost impacts (measured monetarily and discounted for time preferences) incurred" by the reallocation (Marshall, 2013).

Importantly, the frameworks adopted in Marshall (2013), Challen (2000), Marshall (2005) and North (1990) for the quantification and measurement of transaction and/or abatement costs are only applicable when we can assume that costs categories arise mechanistically and have a single possible equilibrium outcome. This is not the case of institutional and technological lock-in costs, whose non-mechanistic and
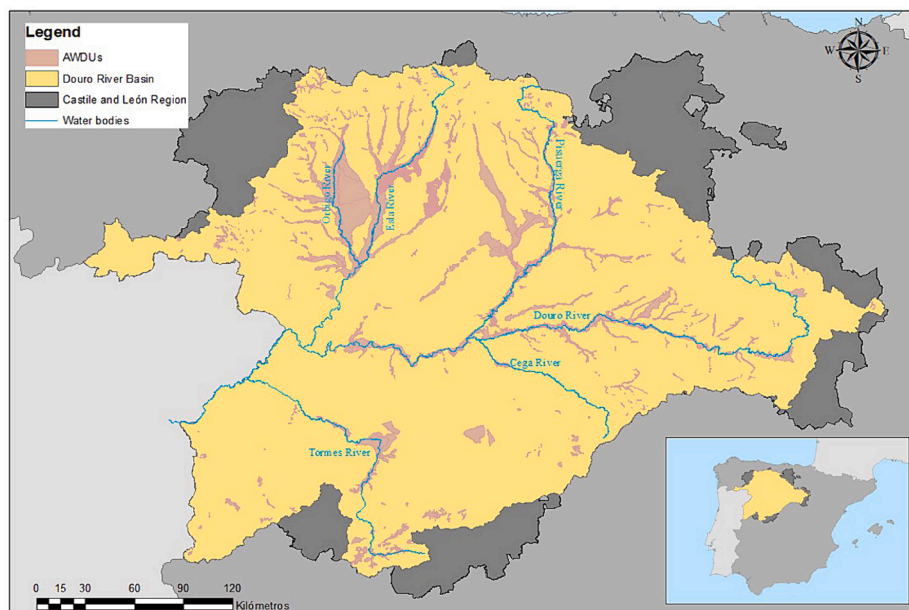
Fig. 1. Location of the Douro River Basin in the Iberian Peninsula and detail of its AWDUs.

Table 2

Institutional transaction costs and abatement costs categories. Adapted from Marshall (2013).

Category	Typology		Water reallocation examples from the Douro River Basin
.Institutional transaction costs	Institutional transition costs	Research and information	River Basin development planning and closure (cap).Hydrologic and socio-economic studies
		Enactment or litigation	Water rights reform (adjudication, charging, conflict resolution, rules).
		Design and implementation	Licensing system, reallocation rules and registries.Water accounting system setup.
	Static transaction costs	Support and administration	Discharge and stock management, environmental management, data management.
		Contracting	Water rights due diligence.
		Monitoring and detection	Water use and consumption accounting.
		Prosecution and enforcement	Compliance monitoring and enforcement.Dispute resolution.
	Institutional lock-in costs	Adaptation or replacement	Revise cap, charges. Adapt water rights and water user associations rules.
Abatement costs	Static transformation costs	Construction, maintenance and operation of water infrastructures	Added costs of building, operating, and maintaining the new and upgraded infrastructure, including storage and distribution infrastructures and irrigation modernization.
	Technological transition costs	Foregone income due to caps and cost recovery (charges) of water infrastructures	Caps are implemented during droughts.Charges for the recovery of environmental and financial cost of water works.
	Technological lock-in costs	Adaptation or replacement	Costs associated with path dependencies arising from existing infrastructure and associated technology.

path-dependent dynamics lead to multiple plausible equilibria that are highly sensitive to random contingencies that cannot be predicted (Arthur, 1999) and thus are typically excluded from the estimation of transaction (Garrick et al., 2013; Loch et al., 2020a,b; Loch and Gregg, 2018; Njiraini et al., 2017) and abatement costs (Marchau et al., 2019; Marshall, 2013).

Recent research has attempted to account for technological lock-in costs by combining heuristics and mechanistic approaches to hypothesize and assess the impact of multiple alternative futures in which incumbent policies are adapted and/or replaced, notably through Dynamic Adaptive Policy Pathways research (Haasnoot et al., 2013; Kwakkel and Pruyt, 2013). In our research we simulate the impacts of

the alternative policy pathways and climate scenarios described by the Douro River Basin Authority in its plan (DRBA, 2020), where differences in the abatement costs between alternative policy choices also reflect on the path dependencies of adopted infrastructures (e.g. irrigation modernization and/or irrigation expansion, once adopted, are assumed to be irreversible in the pathways considered in the basin plan) and provide some measure of technological lock-in costs. Admittedly, this initial analysis could be refined through a more comprehensive exploratory analysis of policy pathways and levers using heuristics and working alongside stakeholders (Groves et al., 2015; Kwakkel et al., 2015), which is out of the scope of the current research. On the other hand, information on the costs incurred in adapting the river basin plan to adopted policy arrangements provides some measure of lock-in transaction costs, albeit projecting these into the future remains a major research challenge (Loch and Gregg, 2018).

3.1. Abatement costs

Abatement costs in Marshall's (2013) conceptual framework include three components:

- *transformation costs*, which are the abatement costs incurred in effecting change from previous technologies or practices to those newly adopted, and in our case mostly refer to the costs of the new water works;
- *technological transition costs*, which are the abatement costs incurred by economic agents operating under the newly adopted technologies or practices; and
- *technological lock-in costs*, which are the costs of path dependencies arising from adopted infrastructures and technologies.

These three cost categories are not independent from one another, and empirical studies typically estimate them together (e.g., measuring changes in income). For example, transformation costs involve the costs of projected infrastructures (available in the Douro River Basin *Mírame* online database (DRBA, 2023) and summarized in Annex I), which as per the WFD are to be fully recovered via charges to water users giving rise to an opportunity cost manifested through technological transition costs. Accordingly, if we were to add transformation and technological transition costs estimates to calculate abatement costs, we would engage in double accounting. To avoid this, transformation costs are converted into charges and their impact on irrigators' income simulated to calculate transition costs. On the other hand, technological lock-in costs are obtained through scenario analysis, i.e., using mechanistic models to quantify transition costs under alternative plausible future scenarios developed by stakeholders using heuristics (Kwakkel and Pruyt, 2013).

To empirically measure abatement costs this paper proposes a socio-hydrology-inspired hydroeconomic model (Pérez-Blanco et al., 2021a) that interconnects economic and hydrologic modules using bidirectional protocols (i.e. two-way feedbacks) to assess the coevolution of complex human-water systems at a basin scale. The framework is populated with a microeconomic Positive Multi-Attribute Mathematical Programming (PMAMP) model that elicits irrigators' preferences towards their behavior and simulates their adaptive responses (microeconomic module) (Gutiérrez-Martín and Gómez, 2011); and the water management model AQUATOOL, the DSS used by Spanish river basin authorities to inform decision-making at a basin level (hydrologic module) (Andreu et al., 1991). The choice of the PMAMP and AQUATOOL models is pragmatic and aims to produce actionable science, since both models are currently used (albeit for separate applications) by the Douro River Basin Authority. In the remainder of this subsection we present the PMAMP model that populates the human module (Section 3.1.1), the AQUATOOL model that populates the water module (Section 3.1.2), and the coupling protocols (Section 3.1.3).

3.1.1. Human module

We represent and simulate the behavior of AWDUs (the economic agent) through a PMAMP model, a mathematical programming model that incorporates multiple utility-relevant attributes in its objective or utility function. In the PMAMP model, economic agents are rational individuals that maximize their utility U(x) subject to a domain F(x), as follows:

$$\underset{\mathbf{x}}{\text{Max U}(\mathbf{x})} = U(z_1(\mathbf{x}); z_2(\mathbf{x}); z_3(\mathbf{x}) \cdots z_m(x)) \tag{1}$$

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

$$\sum_{c=1}^{n} x_c = 1 \tag{3}$$

$$\mathbf{x} \in \mathbf{F} \tag{4}$$

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

where $\mathbf{x}$ is a crop portfolio vector indicating the share of land allocated to each individual crop x_c. The crop portfolio is defined on a yearly basis. Each individual crop x_c represents a unique combination of crops, management options and capital endowment that yields a unique combination of attributes $z(x_c) z(x_i)$. $\mathbf{F}$ represents the set of constraints that conform the domain, including the water constraint per hectare or W_g. W_g is a key variable in the coupling between the human and water system module, which is originally determined by the hydrologic model and then incorporated as a constraint to the microeconomic model as follows:

$$\sum_{i=1}^{n} w_i x_i \le W \sum_{i=1}^{n} w_i x_i \le W \sum_{c=1}^{n} \frac{w_c}{\text{eff}} x_c \le W_g \tag{6}$$

where $w_i w_c$ is the evapotranspiration for crop *c*, and eff represents the irrigation efficiency present in the AWDU (obtained from the hydrologic model).

A detailed description of the microeconomic model and the PMAMP calibration method is available elsewhere in the literature (see e.g. Essenfelder et al., 2018; Gómez-Limón et al., 2016; Pérez-Blanco and Gutiérrez-Martín, 2017). The relevant literature is compiled for the convenience of the reader in the online supplementary material, which includes a detailed description of the model constraints that conform the domain $\mathbf{F}$ (Annex II); the mathematical formulation of the PMAMP calibration procedure (Annex III); the attributes explored and the related data inputs (Annex IV); and the calibration results (Annex V).

3.1.2. Water module

The water module is populated with the AQUATOOL model, the DSS used by the Douro River Basin to inform the current and previous river basin plans (DRBA, 2020). AQUATOOL allows to edit, operate, review, and analyze hydrological models for the management of river basins. It comprises several components, each equipped with its own software suitable for specific tasks (PUV, 2020). In our research within the Douro River Basin, we employ the AQUATOOL (setup) and SIMGES (simulation) modules to conduct a comprehensive assessment, longitudinally and spatially, of the impacts of the water reallocations and infrastructures (reservoirs, canals, irrigation systems) outlined in the new river basin plan on the quantitative and qualitative status of surface (via continuity or balance) and groundwater bodies (using unicellular and multicellular models). Based on the status of water bodies, and other objectives, AQUATOOL uses its network optimization algorithm to allocate water to AWDUs (W_g), the economic agent in the human module.

AQUATOOL's network optimization algorithm allocates available water resources across uses conditional to achieving several objectives, including reallocation targets (e.g., environmental flows), minimizing water deficits among uses, maintaining a specific water stock in reservoirs, and achieving hydropower generation targets. The management algorithm is calibrated using current data on water rights and observed water allocation among uses, aiming to align simulation results with observed discharge and water stock in reservoirs (PUV, 2020). Thus, while one key objective in AQUATOOL is enforcing environmental flows, during periods of acute scarcity where tradeoffs exist between this and other objectives, minimum environmental flows may not be achieved (for instance, to keep water stock in reservoirs above a minimum threshold). A more in-depth description of the AQUATOOL model, its methods and its key outputs is available, is available in Annex VI in the online supplementary material.

3.1.3. Coupling protocols

Our modeling framework follows a protocol-based modular approach à-là-(Pérez-Blanco et al., 2021a, p. 6) that "connects, in a sequential and recursive fashion, a human and a water system module using bidirectional protocols" and a time-variant approach. This approach allows us to simulate the bidirectional feedbacks between the hydrologic and assess the dynamic co-evolution of coupled human and water systems. The time-scale adopted for the dynamic coupling is annual, which is the minimum disaggregation possible (although AQUATOOL works at monthly time steps, microeconomic models typically predict crop portfolios for the entire irrigation season—i.e. annual time-scale). The modeling framework is resolved via simulation rather than optimization, given that the objective is to reproduce the complexity and rules of the real system so as to quantify the consequences of water reallocation choices described in the Douro River Basin Management Plan (DRBA, 2020).

Two bidirectional protocols are developed. In the *first protocol*, information on the water constraint W_g for each agent/AWDU is transferred from the hydrologic to the microeconomic module. In the *second protocol*, information on the water allocated to each AWDU provided by AQUATOOL's network optimization algorithm is conveyed to the PMAMP model; while in the *second protocol*, the PMAMP model conveys information on the effective amount of water used by every AWDU based on their crop portfolio choices *x*. The complete sequence of the integrated modeling framework goes as follows:

i) the shocks directly affecting the water system, including water inputs to the system under climate change (precipitation, runoff) and any construction of new infrastructure, are simulated using AQUATOOL to assess water supply and the water availability constraint for every AWDU in the year *t*.

ii) Information on the water constraint for each agent in year *t*, W_g, alongside information on water charges for the cost recovery of the planned intervention is conveyed to the microeconomic module, which runs a series of simulations (one per agent) to assess AWDUs' crop portfolio responses and related water use, foregone profit, and foregone employment in year *t*. Importantly, by aggregating the foregone profit and the monetized foregone employment (labor income) we obtain the foregone Gross Value Added (GVA) or foregone income—a measure of abatement costs.

iii) The information on effective water use during year *t* is transferred back to the hydrologic module, and combined with hydrological input data for the following period *t + 1* to run a new simulation that assesses the status of the water system in *t + 1*, including water constraints for economic agents (Fig. 2) as well as the environment—a measure of policy effectiveness (see Section 3.3).

Following the standard procedure of the Douro River Basin Authority, as well as other Spanish river basin authorities, simulations are run for a period of 38 hydrological years that correspond to the "short hydrological series" 1980–2018, which is regarded as more representative of the current hydrological cycle than longer data series.
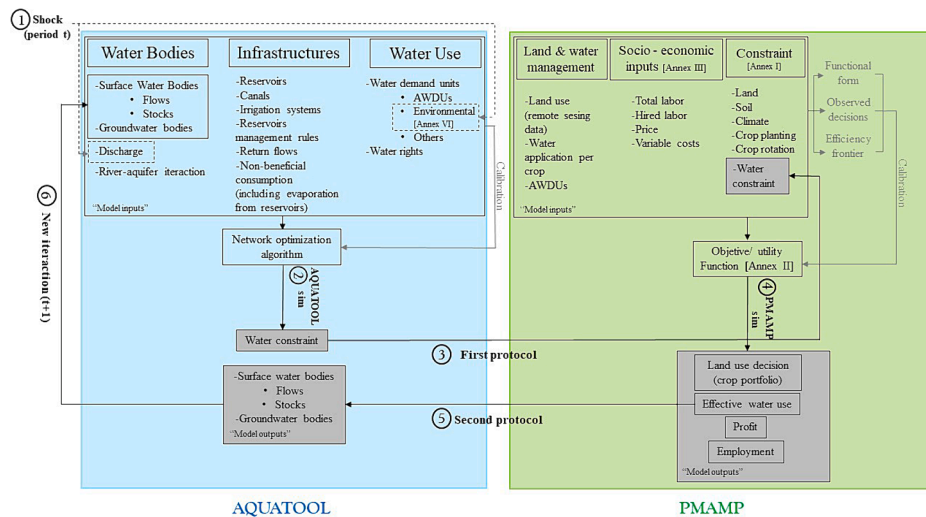
Fig. 2. Conceptual design of the coupled PMAMP – AQUATOOL modeling framework (adapted from Pérez-Blanco et al., 2021a). Step 1 introduces a shock (new minimum environmental flows under climate change), which forces the AQUATOOL model and yields the new water allocations for each AWDU (Step 2). In Step 3, the new water allocation constraints are conveyed to each economic agent/AWDU in the PMAMP model through the first protocol, and a simulation is run to determine land use and water application decisions (Step 4). In Step 5, information on effective water use is conveyed from the PMAMP to the AQUATOOL model through the second protocol, and this information is used to estimate the status of the water system (stock and flows of surface and groundwater bodies). Steps 1 to 5 occur over the same period t. Finally, in Step 6, the status of the water system in t is used as an input to start a new iteration by simulating a new period t + 1.

3.2. Transaction costs

3.2.1. Data gathering

Our study collected and analyzed the transaction costs of the water reallocations implemented in the Douro River Basin during the adoption and implementation of the EU WFD (period 2004–2021). We actively engaged with public authorities and other relevant stakeholders to collect information from the following sources: i) records kept by public institutions including staff salaries, travel costs, fees, cost of studies assigned to third parties, etc. (Njiraini et al., 2017); and ii) personal interviews to collect data not included in records from key personnel in relevant institutions (e.g. unit directors in the planning office), such as time used or number of staff employed in a given task (see e.g. the questionnaires developed by McCann and Easter (1999) and Ofei-Mensah and Bennett (2013)).

Records from institutions were largely collected from *Mírame*, a major database containing information on the past and ongoing activities of the Douro River Basin Authority (DRBA, 2023). Of the > 3 000 entries in the *Mírame* database, 196 contained information on transaction costs, including institutional transition costs (e.g., studies and consultancies, hydrological planning), static transaction costs (e.g., monitoring of runoff and stock) some measure of lock-in costs (notably meetings to revise incumbent management rules and adapt new measures). We also reviewed other records from the Douro River Basin Authority (notably meeting minutes) and other stakeholders involved in river basin planning, including relevant ministries (e.g., Ministry of Finance for staff salaries), the Technical Office for Drought Management (in Spanish: *Oficina Técnica de sequía*), Government Council (*Junta de Gobierno*), Water Council (*Consejo del Agua*), Users' Assembly (*Asamblea de Usuarios*), Committee for Planning and Citizen Engagement (*Comité de Planificación y Participación Ciudadana*), the Commission for the Management of Reservoir Discharge (*Comisión de Desembalse*) and private stakeholders (e.g., consultancy companies supporting river basin planning). All relevant records were processed, analyzed, and incorporated into our transaction cost database which often required gathering additional validation and verification information via interviews. To this end, we conducted 11 personal interviews with high-level officers working at the Douro River Basin Authority.

All this data collection effort yielded 506 entries to our database over 18 years (2004–2021). Each data entry to our database was homogenized to discount for the effect of inflation using the GDP deflator to prices of 2018 (which is also the calibration year of the hydroeconomic model). We next used this database to look for temporal

Table 3

Institutional transaction costs categories and typologies used in the analysis and data sources.

Category		Typology	Data source
Institutional transaction costs	Institutional transition costs	Research and information Enactment or litigation	*Mírame* (DRBA, 2023), other records kept by institutions including meeting minutes (stakeholders involved, duration of the meeting), personal interviews (salary cost rates, physical or virtual participation, participation in meetings, travel distances, duration of meeting).
		Design and implementation	*Mírame* (financial records and other publicly available information/reports), interviews.
	Static transaction costs	Support and administration	*Mírame* (financial records) and other records kept by public institutions (cross-checked using other primary sources including budgets from private consultancy companies), interviews.
		Contracting	Not present
		Monitoring and detection	As in research and information & Enactment or litigation typology.
		Prosecution and enforcement	
	Institutional lock-in costs	Adaptation or replacement	As in research and information & Enactment or litigation typology.

trends in transaction costs, as explained in the following section. Table 3 summarizes the categories and typologies of transaction costs adopted in our study (where data could not be disaggregated, categories in Table 3 were merged), providing the sources used for data collection.

3.2.2. Time series analysis

According to Garrick et al. (2013), the capacity of institutions to adapt and change efficiently (or adaptive efficiency) can be assessed by determining whether we are investing in (i) trajectories with large transaction costs in which past decisions constrain future policy options (transaction costs increase) or (ii) trajectories with high institutional flexibility that allow for relatively lower transaction cost-intensive policy adoption (transaction costs plateau and/or decrease). Where transaction costs plateau and present no clear upward or downward trend they are said to be stationary (Loch and Gregg, 2018). Garrick (2015) further states that, "under flexible institutional arrangements and stable policy conditions, transaction costs should tend to be declining over the long-term, and stable with respect to shocks". Thus, determining the trajectory of transaction costs is relevant to understanding institutional adaptive efficiency and for determining where the system is headed and if that is an adaptively efficient or adaptively inflexible trajectory. Noteworthy, future institutional transaction costs may still evolve in unanticipated ways due to unpredictable shocks (e.g., COVID-19).

To determine the trajectory of transaction costs, this paper follows the approach developed by Loch and Gregg (2018), who apply the cointegration framework developed by Engle and Granger (1987) to identify and assess trajectories in transaction costs in three steps: (i) they apply unit-root tests to test for autocorrelation and detect the presence of a trend in transaction costs (either towards adaptive efficiency—decreasing trend—or not); (ii) they check for cointegrating relationships between transaction costs and structural changes (in the case of the EU WFD, a key shock would be the initiation of each 6-year planning cycle); and (iii) they study whether increasing transaction costs during structural change revert to a declining trajectory over the longer run. Thus, by applying Loch and Gregg's (2018) proposed time series analysis to our unique database of institutional transaction costs of water reallocations, we can determine whether the series is nonstationary, so as to develop assessments of transaction cost movements, magnitude and possible directions under future policy change.

3.3. Effectiveness

The guidance on water economics (WATECO, 2003) assesses the quantitative status of water bodies via the compliance with environmental flow regulations. Following this guidance, the DRBA (2020) assesses the effectiveness of water reallocations using as an indicator the number of monthly-infringements of environmental flows over 31 control points across the basin. This effectiveness indicator is simulated by the AQUATOOL DSS integrated in our modeling framework in Section 3.2.

4. Results

4.1. Abatement costs and effectiveness

We use the hydroeconomic model in Section 3.2 to assess i) the abatement costs and ii) effectiveness of the Douro River Basin Management Plan and its measures, including caps, pricing and water works. To this end we assess, under alternative scenarios, i) the abatement costs of the plan, measured through the foregone GVA or foregone income; and ii) the effectiveness of the policy, which is measured by the river basin authority through the number of months with environmental flows infringements.

4.1.1. Scenarios

Table 3 summarizes the scenarios run in the socio-hydrology

simulations. Each scenario considers different degrees of implementation of water reallocation policies, with (RCP4.5, equivalent to an 11 % average reduction in discharge basin-wide, as per MAGRAMA (2017)) and without climate change. All scenarios are simulated as described in the Douro River Basin Management Plan (DRBA, 2020).

Note that the Table 3 above reports two counterfactual scenarios: with (S_00) and without (S_00cc) climate change. This is because considering a single counterfactual or baseline scenario (S_00) would incorrectly attribute the costs of climate change in S_01cc, S_02cc and S_03cc to the reallocation policies implemented (in other words, climate change costs would be counted as abatement costs). Also, the effectiveness of reallocation policies under different climatic scenarios are not comparable since further climate change makes minimum environmental flows significantly more challenging to achieve.

4.1.2. Simulation results

We use our hydroeconomic model in Section 3.2 to assess, for the scenarios above, the environmental and economic performance of the new river basin management plan over the 38 hydrological years in the short hydrological series. Simulations run as follows. First, we reproduce the scenarios in Table 3 in the hydrologic module to assess impacts on the water system, including the water availability for each AWDU based on the AQUATOOL network optimization algorithm. Next, the water allotment in Eq. 6 in the microeconomic module is revised for each agent/AWDU in accordance with the water constraint prediction in the hydrologic module, and the water charges from new water works are recovered (this is done through the attribute expected profit—see Annex IV for a description of the attributes in the model). Each agent/AWDU in the human system then solves the optimization problem in Eqs. 1–6 to maximize their utility within the new domain and with the new prices. In those years and for those AWDUs where the water constraint is reduced (and is binding) and/or water charges increase, the agent will be constrained to revise their crop portfolio choices, leading to a drop in utility and changes in employment and profit. The process above is repeated for all scenarios, all years in the hydrologic series, and all AWDUs in the basin, which results in a database of simulations including longitudinal (38 years in the short hydrologic series) and cross-section data (150 AWDUs in the basin) for each scenario considered. To calculate abatement costs, we obtain for each scenario and AWDU the foregone profit (gross variable margin) and foregone employment (hours of work) over the 38-year period, as compared to the relevant counterfactual (see Table 3). Next, we aggregate foregone profit and the monetized foregone employment (labor income) to obtain the abatement costs as the foregone GVA.

Changes in aggregate water availability in the Douro River Basin (measured as the fraction of the existing water allocation rights that are delivered for its use by farmers) and changes in the aggregate crop portfolio (obtained aggregating the individual crop portfolios across all AWDUs) for each year in the series are shown in Fig. 3 (see Table 1 for the baseline crop portfolio).

Instead of plotting the 35 individual crops in our model, the aggregate changes in the crop portfolio are represented in six macrocategories, including irrigated and rainfed agricultural land: wheat, other cereals (barley, oats, rye, triticale and corn), vegetables and fruits (potato, lettuce, spinach, green chicory, pumpkin, garlic, onion, leek, carrot, peas, apple tree, pear, almond, walnut and vineyard), oilseeds (sunflower and olive tree), sugar beet and other crops (termed "cropnec" including beans, lentils, chickpeas, dried pea, pea, lupine, carob, hops, rapeseed, fodder corn, alfalfa and forage vetch). During normal and wet hydrological years irrigated agricultural lands predominate in AWDUs, while in dry years rainfed farmland increases as water availability decreases. The results shown in Fig. 3 represent the changes in the water allocation and the crop portfolio at the basin level. Note these results may (significantly) differ from the trends observed in each of the AWDUs individually. A detailed description of the attribute values per crop and AWDU (including expected profit) driving agents' behavior, as well as of
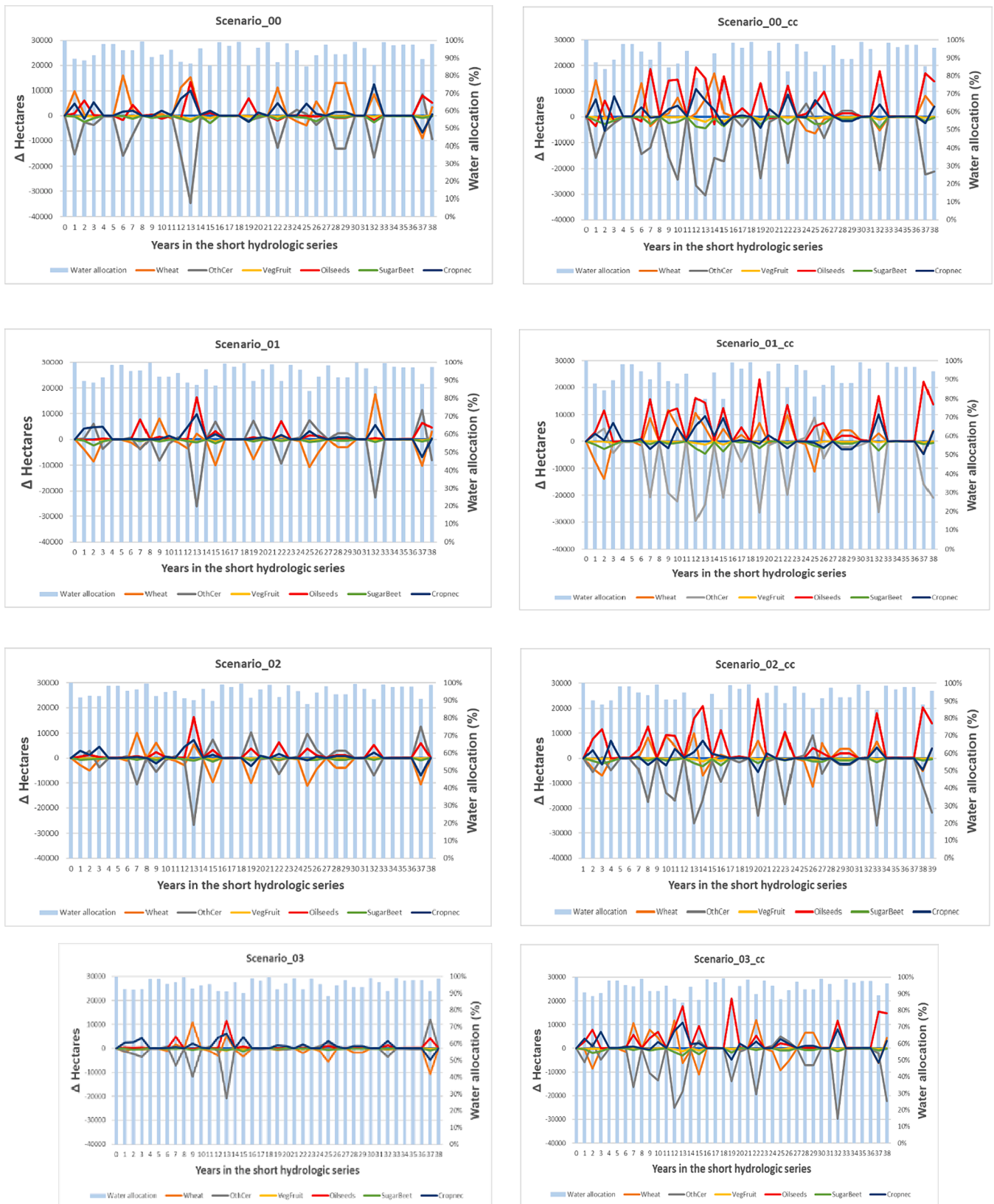
Fig. 3. Water availability and changes in the aggregate crop portfolio in the Douro River Basin. The blue bars "Water reallocation (%)" represent the percentage of the water allocation rights that are satisfied. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

the simulation results, is available in Annex IV.

When water availability decreases, the area of high value-added crops such as sugar beets, vegetables and fruits is reduced in all AWDUs. The area of maize (other cereals) is consistently reduced as well and replaced by a combination of less water intensive or rainfed crops (mostly sunflower, wheat and other cereals). The area of wheat and sunflower can also experience reductions due to less water availability in some AWDUs and years. The final impact on the aggregate crop portfolio is disparate from year to year due to the asymmetric impacts on AWDUs, which can experience water constraints of different intensity (e.g., extreme drought in one sub-basin and normality in another), as well as due to the heterogeneous preferences shown by agents that condition responses to shocks. Overall, scenario S_03 shows the lowest crop area volatility due to the maximum development of storage and distribution infrastructure and the adoption of irrigation modernization. In climate change scenarios (S_00_cc, S_01_cc, S_02_cc, and S_03_cc) there is a significantly lower water availability than in scenarios without climate change (S_00, S_01, S_02, and S_03), and thus the reduction in the area of corn (other cereals), vegetables and fruits, and sugar beets, as well as the increase in the area of rainfed sunflower (oilseeds), are more pronounced.

The impacts of water reallocations on annual profit, employment and GVA for each AWDU and scenario, as compared to the relevant counterfactual scenario (S_00 for scenarios without climate change, S_00cc for scenarios with climate change), are shown in Fig. 4.

AWDUs average yearly profit, employment, and GVA in S_00 (counterfactual for S_01, S_02 and S_03) are estimated at 1728.3 EUR/ha, 3.07 h/ha, and 1755.9 EUR/ha, respectively; while in S_00_cc (counterfactual for S_01_cc, S_02_cc and S_03_cc) they are estimated at 1683.6 EUR/ha, 3.01 h/ha, and 1710.7 EUR/ha, respectively. Transitioning from S_00 to S_01 implies a basin-wide reduction in GVA of 7.8

%, 7.1 % in the transition from S_00 to S_02, and 5.9 % in the transition from S_00 to S_03, as compared to the counterfactual. On the other hand, transitioning from S_00_cc to S_01_cc implies a basin-wide reduction in GVA of 9.1 %, 7.2 % in the transition from S_00_cc to S_02_cc, and 6.2 % in the transition from S_00_cc to S_03_cc, as compared to the counterfactual.

The highest economic losses are observed in those AWDUs with higher value-added crops (maize, vegetables and fruits, and sugar beet), while AWDUs where cereals predominate have lower economic losses as the income gap between irrigated cereals and the best rainfed alternative is usually smaller. Also, in some AWDUs the abatement costs are negative, showing net gains from water reallocations. This happens in those (limited) cases where the construction of water infrastructures and irrigation expansion favor the development of a more productive agriculture, without impairing the supply to environmental uses. Fig. 5 shows the impacts of water reallocations on the ecological status of water bodies, measured through the compliance with environmental flows.

To assess effectiveness, we follow DRBA (2020) and quantify the number of monthly-infringements of environmental flows in each scenario over 31 control points across the basin (i.e., 38 years x 12 = 456 months, and 456 x 31 control points = 14136 outputs), and compare results to the relevant counterfactual. The red circles signal the control points where the number of monthly infringements increase as compared to the counterfactual, while the green circles signal improvements in environmental performance. Overall, the number of total infringements in the scenarios considered ranges from 159 to 388 (i.e., between 1.1 % and 2.7 % of the 14,136 simulated outputs). Climate change increases the number of monthly infringements, with S_00cc (counterfactual) being the worst performing scenario, and S_01cc, S_02cc and S_03cc showing a higher number of monthly infringements
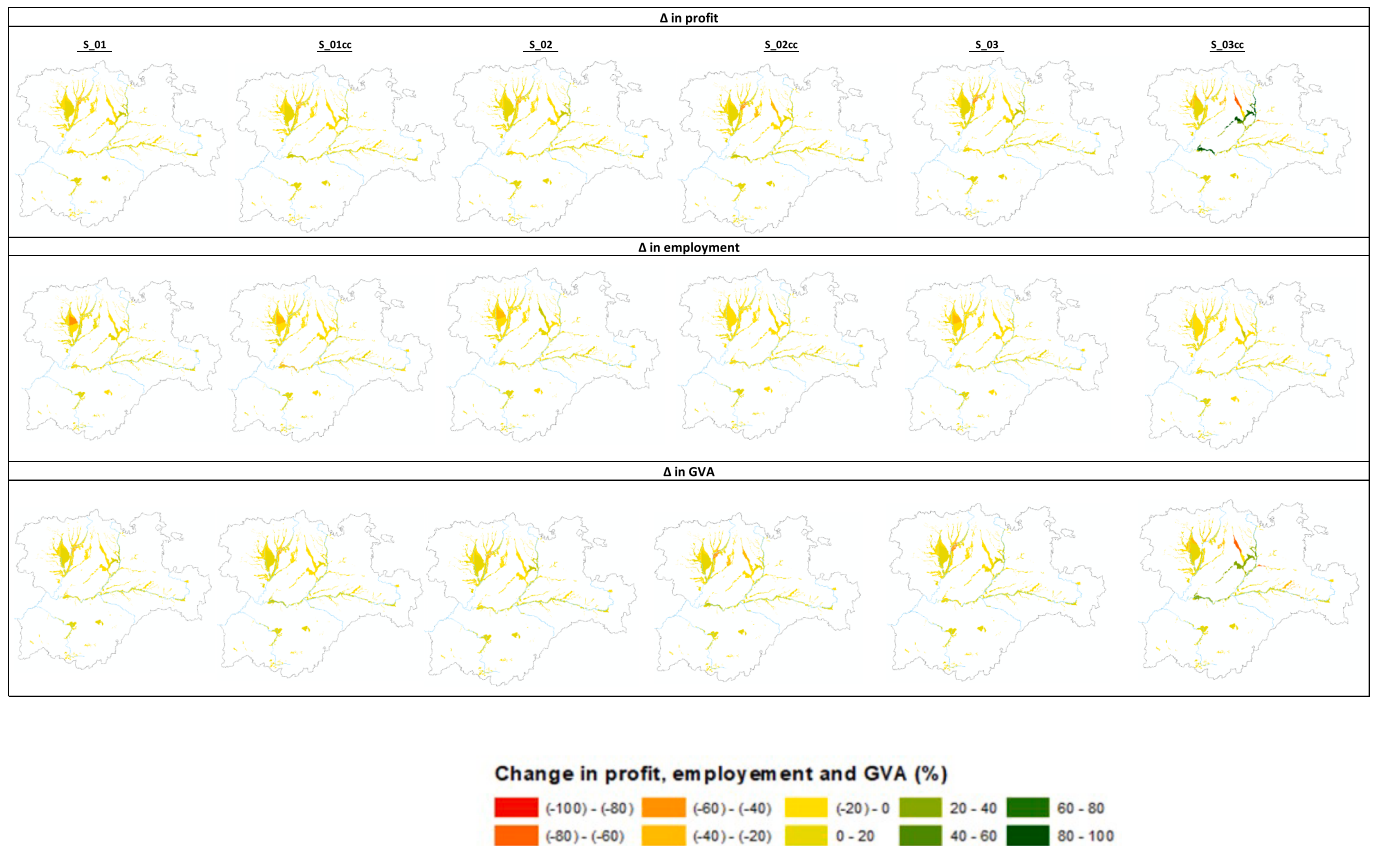

Fig. 4. Changes in (a) profit (as a % of profit in the counterfactual), (b) employment (as a % of employment in the counterfactual) and (c) GVA (as a % of GVA in the counterfactual).

Fig. 5. Effectiveness of water reallocation measured through monthly infringements of environmental flows in each scenario as compared to the relevant counterfactual. A green circle denotes a decrease in the number of monthly infringements as compared to the relevant counterfactual, while a red circle denotes an increase in the number of infringements. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

than S_01, S_02 and S_03, respectively. Scenarios S_01 and S_01cc reduce the number of monthly infringements as compared to S_00 (by −18.4 %) and S_00cc (–23.2 %), respectively, albeit their performance is considerably improved by scenarios S_02 (-39 %) and S_03 (-39 %), and S_02cc (–32.2 %) and S_03cc (-34.5 %), respectively. These results suggest that adopting caps, charges and new infrastructure *while*

Table 4
Simulation scenarios.

Scenario	Storage and distribution infrastructures	Irrigation infrastructures	Soft policies	Climate change	Brief description
S_00	Business as usual (BAU)	BAU	BAU	Not considered	**Counterfactual for S_01, S_02 & S_03**
S_01	Maximum development	Irrigation modernization & irrigation expansion	Caps & charges	Not considered	Maximum development of water works
S_02	Maximum development	BAU	Caps & charges	Not considered	Assesses the performance of new storage and distribution infrastructures with the agricultural water demand of the counterfactual (S_00)
S_03	Maximum development	Irrigation modernization	Caps & charges	Not considered	Assesses the performance of irrigation modernization when irrigated land is capped (v. S_01)
S_00cc	BAU	BAU	BAU	RCP 4.5 (11 % discharge reduction)	**Counterfactual for S_01cc, S_02cc & S_03cc**
S_01cc	Maximum development	Irrigation modernization & irrigation expansion	Caps & charges	RCP 4.5 (11 % discharge reduction)	Adds climate change to S_01
S_02cc	Maximum development	BAU	Caps & charges	RCP 4.5 (11 % discharge reduction)	Adds climate change to S_02
S_03cc	Maximum development	Irrigation modernization	Caps & charges	RCP 4.5 (11 % discharge reduction)	Adds climate change to S_03

avoiding irrigation expansion (S_02 and S_03 avoid irrigation expansion, while S_01 does not—see Table 4) delivers a significantly better performance in terms of reduction in monthly infringements.

4.2. Transaction costs

Time series results for river basin planning institutional transition (research and information, enactment or litigation, design, and implementation), static transition (support and administration, monitoring and detection, prosecution, and enforcement) and institutional lock-in (adaptation and replacement) costs are presented as the shaded bars in Fig. 6. The peaks and throughs in the data correspond with the initial/final and intermediate periods within the 6-year river basin planning cycles. Monitoring and detection costs represent the most relevant transaction cost (up to 80 % of total transaction costs during 2004–2008) due to the setup of the Automated Hydrologic System (*Sistema Automático de Información Hidrológica*—SAIH) and other monitoring and surveillance activities (e.g., control points such as those used to assess monthly-infringements of environmental flows in Fig. 5). Research and information costs peak around the year 2009 (2007–2012), when the river basin plan of 1998 that predated the WFD (OJ, 2000) was adapted to the EU legal *acquis*, and represent up to 20 % of transaction costs in 2011. Design and implementation costs peak from 2017 due to the initiation and management of a series of large river restoration and re-naturalization projects, representing about 25 % of the total transaction costs in 2017–2021. Support and administration largely involve the recurrent meetings between the river basin authority and other relevant institutions to assess discharge and water stocks, as well as data management and operations support, which tend to peak during dry periods. Finally, adaptation and replacement costs are relevant from 2014 and largely refer to the update and adoption of a public electronic water accounting census.

The upward linear trend over the data series in Fig. 6 may indicate inflexible institutional arrangements as per Garrick (2015). On the other hand, a downward trend during the last years of the series is also observed (2015–2021). To determine the presence of non-stationarity in transaction costs and its direction we follow Loch and Gregg (2018) and test for autocorrelation in the data series using unit-root tests (Table 5) by using the Augmented Dickey Fuller test. The non-stationarity auto-correlation test is used to see if, following any shock to the system, the trajectory returns to its original path (or close to it). Results using support rejection of the null hypothesis of unitary root/non-stationarity under the trend-only and random walk with drift/trend models.

Table 5
Augmented Dickey-Fuller test.

Augmented Dickey-Fuller Test alternative: stationary		
Type 1: no drift no trend		
Lag	ADF	P-value
0	−0.1924	0.58
1	0.0614	0.653
2	0.6516	0.823
3	0.935	0.902
Type 2: with drift no trend		
Lag	ADF	P-value
0	−3.81	0.01
1	−2.52	0.142
2	−2.42	0.18
3	−2.19	0.263
Type 3: with drift and trend		
Lag	ADF	P-value
0	−4.38	0.0101
1	−3.45	0.0705
2	−2.66	0.3114
3	−2.26	0.4576

Note: in fact, P-value = 0.01 means P-value <= 0.01.

Accordingly, time series analysis using our database does not provide support for an increasing or decreasing trajectory in transaction costs, and we cannot discard stationarity in the time series (plateaued trend). This result is not surprising given the relatively 'immature' stage at which this reallocation is in terms of implementation (18 years, 2004–2021). Over time, more conclusive trends may emerge (Loch et al., 2020a,b).

Our pioneering transaction costs study offers a first database that can be further developed into the future to obtain more conclusive results. It also points towards a stable pattern in data series that may be indicative of a plateaued trend that aligns with Garrick (2015), providing some indication of: i) a flexible/adaptively efficient trajectory over a larger time frame, ii) that transactions costs related to arranging the reallocation ex-ante that are typically larger will decrease, while iii) the transaction costs related to ex-post monitoring and enforcement will likely stabilize. Yet, additional monitoring efforts are necessary to determine how transaction costs develop and to inform future predictions on how transaction costs directions and their magnitude may change/shift into the future.
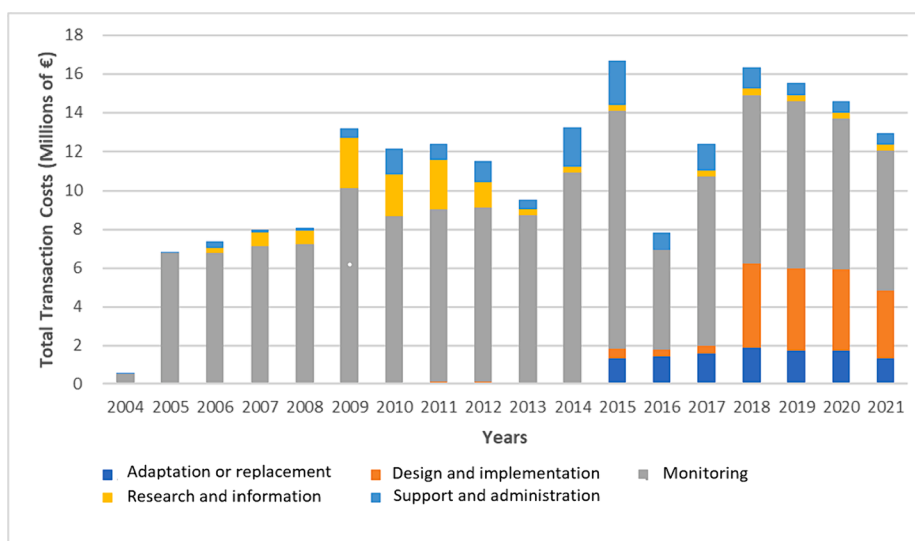
Fig. 6. River basin planning transaction costs by category, 2004–2021.

4.3. Cost-effectiveness

Governments have limited budgets and increasing opportunity costs associated with those budgets. If policy makers do not provide sufficient budgets to achieve policy objectives of water reallocation programs to ensure value for money over time we will face decreased social welfare. Our assessment of the Douro River Basin Management Plan provides evidence for increasing objectives/effectiveness (Fig. 5); evidence of plateauing transaction costs (Table 5) and for some relevant typologies such as monitoring and research and information costs evidence of a decreasing trend (see Fig. 6); and evidence of a sufficient budget given the link between the public expenditure and outcomes.

The total transaction costs of water reallocations in the Douro River Basin up to the end of the second planning cycle (2004–2021) amount to EUR 204.6 million (2018 constant prices) or an annuity equivalent of EUR 11.4 million. On the other hand, the average abatement costs of water reallocations is projected to generate from 2021 onwards amount to EUR 165.6 million/year (S01), EUR 150.7 million/year (S02) and EUR 125.2 million/year (S03) without climate change; and to EUR 188.2 million/year (S01_cc), EUR 148.9 million/year (S02_cc) and EUR 128.2 million/year (S03_cc) under climate change (RCP4.5). Accordingly, under the assumption of stationarity, transaction costs can represent between 6.4 % and 8.3 % (scenarios without climate change) or between 5.7 % and 8.2 % (scenarios with climate change) of the total (abatement plus transaction) costs of water reallocation. This is a non-trivial magnitude that highlights the need to account for both abatement *and transaction* costs when assessing the performance of water reallocations, and environmental policy cost-effectiveness overall.

In terms of overall cost-effectiveness (i.e. ratio of abatement *plus* transaction costs to effectiveness) and under the assumption of stationarity/plateauing transaction costs, the policy scenarios S_02, S_02cc, S_03 and S_03cc where irrigation expansions are avoided have lower reallocation costs and higher effectiveness (and therefore a superior cost-effectiveness) than the policy scenarios S_01 and S_01cc where irrigated surface is expanded, while S_03 and S_03cc show a slightly superior performance/cost-effectiveness than S_02 and S_02cc. This suggests that contrary to the common belief that irrigation expansion can mitigate the economic costs of water reallocations towards the environment, it can increase them through higher water consumption (and income) by adopters and aggravated scarcity (and reduced income) elsewhere that reduce overall income (thus increasing costs) and effectiveness and undermine cost-effectiveness. On the other hand, implementing irrigation modernization in S_03 and S_03cc reduces the economic cost but has a marginal impact on effectiveness as compared to S_02 and S_02cc, respectively. This is because, where possible, farmers will withdraw the same amount of water as they did before the adoption of modern irrigation systems (to which they hold a right), while consuming a larger fraction of it, which will reduce water availability for downstream uses—including the environment (Pérez-Blanco et al., 2020). In the case of the Douro River Basin Management Plan, this trend towards aggravated water depletion is mitigated by caps and the higher prices that are simultaneously applied to recover the cost of new water works (including irrigation modernization programs), which tend to reduce water use and offset the increase in water demand induced by irrigation modernization technologies. Overall, S_03 and S_03cc show a superior cost-effectiveness than S_02 and S_02cc mostly due to reduced reallocation costs, albeit the improvement is lower than one may initially expect.

5. Conclusions

This paper presents the first integrated analysis of abatement and transaction costs of water reallocations. The analysis is illustrated with an application to the river basin management plan of the Douro River Basin in central Spain. Our results provide evidence for increasing policy effectiveness, and we cannot discard plateauing/stationary transaction

costs—a sign of adaptive efficiency (i.e., institutional capacity to adapt to surprising events in future). Under the assumption of stationarity, transaction costs in the Douro River Basin represent between 5.7 % and 8.3 % of total reallocation costs (abatement plus transaction costs). This non-trivial magnitude highlights the need to account for both abatement *and transaction* costs when assessing the performance of water reallocations and effectiveness assessments of environmental policy overall.

The methods for the measurement of abatement and transaction costs developed in this paper are therefore designed to be replicable and flexible. The data collection and time series analysis used for the measurement and analysis of transaction costs build on standard methods available in the literature (Garrick, 2015; Loch and Gregg, 2018). On the other hand, the socio-hydrology-inspired protocol-based modular framework used to estimate the abatement costs of water reallocations is flexible and capable of including alternative DSS used by basin authorities elsewhere, as well as any other standard agricultural microeconomic model such as Expected Utility (von Neumann and Morgenstern, 1953), Linear Programming (Paris, 2015), Positive Mathematical Programming (Howitt, 1995), Multi-criteria Decision Models (Pereira et al., 2003; Sumpsi et al., 1997) and Positive Multi-Attribute Utility Programming models (Gutiérrez-Martín and Gómez, 2011; Sapino et al., 2020).

This research could be improved in several ways. First, given that the flexible socio-hydrology framework developed in this paper can be populated with alternative models that can be used to simulate multiple scenarios, this feature could be employed to produce multi-scenario (which could be developed in turn using heuristic methods such as DAPP) and multi-model ensembles that more thoroughly assess uncertainties in forecasts, including in estimates of technological lock-in costs. Second, the individual models included in the human and water modules for the analysis of abatement costs can be also improved, notably by incorporating the possibility to adapt via intensive margin adaptation, namely supplementary and/or deficit irrigation (Sapino et al., 2022). Third, measures other than the GVA can be used to measure abatement and transaction costs which are potentially more precise. For example the monetized foregone utility of farmers could be used to measure abatement costs—albeit utility monetization is less feasible in transaction costs measurement (Pérez-Blanco and Gutiérrez-Martín, 2017). Moreover, the estimate of foregone GVA is incomplete since it is focused on the impacts on microeconomic agents, and ignores potentially relevant forward and backward linkages that can amplify/mitigate the abatement costs over the wider economy, notably via price feedback (i.e., in dry years commodity prices will increase, mitigating the negative impact of commodity shortages on farm income). Fourth, additional modules could be added to the socio-hydrology framework, including through the inclusion of a climate module that thoroughly assesses climate scenarios or a macroeconomic module that allows for the simulation of commodity price feedbacks. Fifth, our longitudinal transaction costs database needs to be expanded with new data gathered over the incoming planning cycle (2021–2027) to obtain more conclusive results on the direction and magnitude of transaction costs in the future.

CRediT authorship contribution statement

C. Dionisio Pérez-Blanco: Conceptualization, Funding acquisition, Investigation, Methodology, Project administration, Resources, Supervision, Writing – original draft, Writing – review & editing. **Adam Loch:** Conceptualization, Investigation, Methodology, Writing – original draft. **Juan Mejino-López:** Data curation, Investigation, Visualization, Writing – original draft. **Laura Gil-García:** Data curation, Investigation, Software, Visualization, Writing – original draft, Writing – review & editing. **David Adamson:** Conceptualization, Methodology, Writing – review & editing. **Pablo Saiz-Santiago:** Data curation, Investigation, Software. **José Antonio Ortega:** Conceptualization, Investigation, Methodology, Supervision.

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.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jhydrol.2024.131119.

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