

A taxonomy of economic water policy instruments and their dynamic effects for sustainable water management

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ABSTRACT: Economic instruments are widely used in environmental policy; however, their application in water governance often lacks contextual clarity. This paper presents an integrated taxonomy of economic instruments for sustainable water management and proposes a complementary, trajectory-based framework for assessing time-dependent performance. Interventions are characterised by implementation profile (single, discontinuous, or continuous) and by response type (static or dynamic, with reinforcing or eroding effects). Combining these dimensions clarifies how behavioural and institutional adjustment shapes outcomes over time, supporting the design and evaluation of robust, adaptable, and context-sensitive instruments for long-term water sustainability.

Taxonomía de los instrumentos económicos de política hídrica y sus efectos dinámicos para una gestión sostenible del agua

RESUMEN: Los instrumentos económicos se utilizan ampliamente en la política ambiental, pero su aplicación en la gobernanza del agua a menudo carece de claridad contextual. Este artículo desarrolla una taxonomía integrada de instrumentos económicos para la gestión sostenible del agua y propone un marco para evaluar el desempeño temporal. Las intervenciones se caracterizan por el tipo de implementación (única, discontinua o continua) y el tipo de respuesta (estática o dinámica, con efectos reforzadores o erosivos). Esta combinación clarifica cómo los ajustes conductuales e institucionales condicionan los resultados, apoyando el diseño y la evaluación de instrumentos robustos y adaptables.

KEYWORDS / PALABRAS CLAVE: Dynamic effect, economic instruments, taxonomy, water governance / Efecto dinámico, instrumentos económicos, taxonomía, gobernanza del agua.

JEL Classification / Clasificación JEL: C35, D12, I12, M33.

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1. Introduction

The increasing global demand for water resources, driven by population growth and expanding economic activities such as agriculture, energy production, and industrial processes, is placing unprecedented pressure on the world's limited water supplies (Strong *et al.*, 2020). At the same time, climate change and environmental constraints are projected to reduce water availability, intensifying competition for water and exacerbating imbalances between supply and demand (Intergovernmental Panel on Climate Change, 2023). In light of these developments, the effective management of water scarcity has become a crucial priority, not only to preserve the ecological health of water bodies but also to achieve broader environmental and socio-economic objectives, as set out in strategies such as those of the European Union (European Commission, 2024). Addressing these challenges requires instruments capable of operating across sectors and scales, while coping with uncertainty, user heterogeneity, and evolving constraints. Consequently, water governance increasingly relies on policy mixes that combine technical, legal, regulatory, and economic instruments (EIs) (Hernández-Pacheco Algaba *et al.*, 2024).

EIs have been central to water policy since the adoption of the Water Framework Directive (Dir. 2000/60/EC) and have attracted growing attention as a promising approach for addressing water scarcity by aligning individual decisions with collectively agreed objectives (Lago *et al.*, 2015; Rey *et al.*, 2018). More specifically, they aim to steer behaviour towards public goals such as ensuring reliable water quantity and quality and mitigating water-related risks. This is achieved by creating incentives or disincentives for particular actions (such as consumption or pollution), while allowing users to choose how to respond to these mechanisms. Their implementation rests on three core ideas: incentives, motivation, and voluntary choice (Lago *et al.*, 2015). This flexibility is often presented as a key advantage in water governance, where uniform rules may be ill-suited to heterogeneous users and where the ability to accommodate local conditions is important for both the effectiveness and political feasibility of such goals.

EI performance, however, is strongly contingent on its design and implementation, including equity impacts and ecological constraints (Organisation for Economic Co-operation and Development, 2015). A central implication is that 'what works' cannot be inferred from the instrument label alone: the same instrument category may perform very differently depending on the context, administrative capacity, and users' behavioural responses (Organisation for Economic Co-operation and Development, 2015; Rey *et al.*, 2018). In practice, this means that EIs can generate trade-offs: instruments designed to improve efficiency may raise equity concerns; measures that create stronger water conservation incentives at farm or household level may not translate into basin-level water savings if broader allocation rules and hydrological feedbacks are not accounted for; and instruments that appear technically sound may face limited acceptance or weak compliance if perceived as unfair or poorly aligned with local constraints (Organisation for Economic Co-operation and

Development, 2015). These implementation realities underscore the importance of analytical frameworks that can support design choices, clarify assumptions and improve comparability across contexts.

A further, often under-specified challenge is that EI impacts are frequently time-dependent. Responses can vary over the short and long term as users gradually adjust practices and technologies, learn about the instrument, and adapt their behaviour. Empirical evidence shows that these adjustments can produce dynamic effects that either enhance or weaken performance over time –for example, through rebound dynamics, substitution across sources, or diminishing marginal responses as users exhaust low-cost adaptation options (Scheierling *et al.*, 2006; Huffaker, 2008; Adamson & Loch, 2014; Perry & Steduto, 2017; Liu *et al.*, 2019). More broadly, socio-political responses and compliance dynamics can shift as stakeholders adjust their expectations, governance capacity evolves, or policy mixes change, thereby affecting both the instrument durability and outcome (Muradian *et al.*, 2013). In this context, an EI's temporal profile is not a secondary feature but a core design feature: whether an instrument is conceived as a single intervention, a periodic adjustment, or a continuous mechanism can affect the nature of adaptation processes and the long-term policy trajectory.

Against this background, the objective of this paper is twofold. First, it develops and presents an integrated taxonomy that provides a consistent classification of economic instruments, facilitating comparability across contexts and clarifying what instrument labels capture. The taxonomy serves as a practical reference for organising the diversity of economic instruments used in water governance. Second, it proposes a conceptual framework to examine how the impacts of economic instruments may evolve over time.

The novelty lies in incorporating a temporal dimension into the analysis by distinguishing interventions according to their temporal design and the nature of responses over time. The temporal dimension is operationalised through two classification axes. First, interventions are distinguished by their temporal implementation type: single measures applied at a specific point in time, discontinuous measures applied intermittently, and continuous measures maintained over time. Second, interventions are distinguished by the nature of the response: a static response denotes effects that do not generate material spillover effects or subsequent responses beyond the initial change. This allows us to assess whether their effectiveness is likely to persist or decline as behavioural, institutional, or socio-political adaptations alter the initial effects, either by reinforcing (positive dynamics) or eroding them (negative dynamics). Together, these dimensions provide a parsimonious framework for describing policy trajectories and long-term performance, rather than focusing solely on short-term outcomes.

The proposed taxonomy, together with the temporal framework, is intended as a practical analytical tool. It is designed to help policymakers, river basin and irrigation

authorities, and water-sector analysts compare instruments beyond their labels, anticipate whether effectiveness is likely to be robust or to erode through adaptation, and identify when complementary measures or iterative adjustments may be required. For researchers, it provides a structured basis for synthesising evidence across cases and for framing empirical evaluations in a way that distinguishes short-run effects from longer-term trajectories.

The following section reviews the role of taxonomies in the study of economic instruments and presents a detailed analysis and classification of economic instruments. Section 3 introduces the concept of dynamic effects as a key factor influencing the effectiveness of economic instruments. Finally, section 4 discusses the findings, while section 5 concludes.

2. An integrated taxonomy of economic instruments in water policy

Economic instruments have become central to water policy as mechanisms that guide behaviour through pricing, incentives, or financial transfers, rather than through prescriptive regulations (Delacámara *et al.*, 2014; Hadjigeorgalis, 2009; Lago *et al.*, 2015; Montilla-López *et al.*, 2016). The role of these instruments has expanded as traditional allocation systems, rooted in basic water pricing and cost-recovery models, have struggled to address increasing water scarcity and evolving environmental pressures (Leflaive, 2024; Berbel & Expósito, 2020). A substantial body of literature has examined water pricing, markets, regulatory frameworks, and demand-management tools across diverse contexts, providing extensive evidence of their potential to improve efficiency, support conservation, and promote sustainable resource use.

Among these instruments, taxonomies provide structured frameworks for organising complex domains, thus supporting systematic comparison and cumulative learning (Nickerson *et al.*, 2013). In policy settings, taxonomies can improve transparency in instrument choice by clarifying how different tools operate, the assumptions underlying them, and how they are interrelated. In economics and public policy, taxonomic approaches are widely used to reduce conceptual ambiguity, improve communication among stakeholders, and enable more coherent synthesis of evidence across a diversity of cases.

In environmental policy, numerous classification efforts, such as OECD taxonomies (Organisation for Economic Co-operation and Development, 2009) and studies comparing regulatory, price-based, and market-based mechanisms, highlight the importance of structured frameworks for understanding how policy instruments operate and supporting evidence-based decision-making (Crespi *et al.*, 2015; Duval, 2008; Goulder & Parry, 2008; Pacheco-Vega, 2020; Intergovernmental Panel on Climate Change, 2015; Pannell, 2008; Whitten *et al.*, 2003; Revesz & Stavins, 2004). While the literature is extensive, the classification of EIs specifically applied to water

remains more fragmented, spanning pricing reforms, cost-recovery mechanisms, tradable permits, agricultural subsidies, payments for ecosystem services, and instruments to reduce market frictions (Expósito & Berbel, 2024). Divergent definitions of what constitutes an EI add further complexity, particularly regarding hybrid or novel approaches.

This paper provides a unified taxonomy of water-related EIs designed to clarify their mechanisms, support comparisons across instruments, and guide the selection of appropriate tools under different policy circumstances. The adoption of a broad definition of EIs (Lago *et al.*, 2015; Gómez Gómez *et al.*, 2018; Rey *et al.*, 2018) improves communication and engagement among policymakers, stakeholders, and the general public by providing a structured, accessible framework for discussing EIs. This allows us to capture both well-established mechanisms and innovative or hybrid tools increasingly used in practice.

The taxonomy was developed through a structured, two-step procedure. First, a panel of specialists from the GOVAQUA project (*Governance innovations for a transition to sustainable and equitable water use in Europe*)¹ jointly identified a broad set of economic instruments considered relevant to current policy and management contexts. The panel comprises approximately 45 senior researchers and practitioners with expertise in water governance, environmental and resource economics, and hydrology, drawn from a diverse range of European institutions, including universities, public research agencies, environmental organisations, and water management bodies. This collaborative process ensured that the initial list reflected both practical relevance and the diversity of approaches used across different regions internationally.

In a second stage, an extensive review of academic and technical literature was undertaken to document, refine, and substantiate the instruments identified by the panel. The review covered publications up to 2024 and drew primarily on peer-reviewed journals indexed in Web of Science and Scopus, complemented by grey literature including technical reports and policy documents from international organisations (e.g., OECD, European Commission, FAO). Searches were conducted instrument by instrument focused on empirical and conceptual studies addressing the design, implementation, and performance of economic instruments in water policy across diverse geographical and institutional contexts. Inclusion criteria prioritised studies providing evidence on instrument mechanisms, behavioural responses, or policy outcomes; purely descriptive or tangential references were excluded. No formal systematic review protocol was applied, as the objective was to achieve broad coverage and contextual depth rather than exhaustive quantitative synthesis. This two-step methodology ensured that the final compilation is firmly grounded in existing evidence while remaining attentive to emerging developments in the field.

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The resulting taxonomy organises EIs into coherent categories and clarifies their functional characteristics. Table 1 summarises the taxonomy and provides references to documented case studies and empirical applications that illustrate practical successes, limitations, and implementation challenges across contexts.

TABLE 1

Economic instruments in water policy

Economic instruments	Example	References
Price-based instruments		
Abstraction charges	England and Wales (based on seasons)	Rey <i>et al.</i> (2018); Finney (2013).
Flat rate	Edwards Aquifer licensing (Texas, USA)	Molle & Closas (2020b); Finney (2013).
Volumetric rate	Emilia Romagna (Italy)	Pronti & Berbel (2023).
Tax	Danish pesticide tax	Berbel <i>et al.</i> (2019a); Böcker & Finger (2016).
Emissions charges	German effluent tax	Möller-Gulland <i>et al.</i> (2015).
Non-compliance fees	Compliance fees	Loch <i>et al.</i> (2020).
Subsidy/subsidy removal		
Subsidy for capital	Subsidies for water efficiency (Australia, Spain, and USA)	Berbel <i>et al.</i> (2019b).
Subsidy for exploitation/removal	EU-CAP ecoschemes, Environmental Quality Incentives Program (EQIP)	European Commission (2023). Natural Resources Conservation Service (2025).
Payment for ecosystem services	Vittel (France), New York Catskills (USA)	Zanella <i>et al.</i> (2014); Perrot-Maître (2006).
Trading schemes		
Tradable rights (water trade)	Bulletin board market (Colorado), clearinghouse, and sealed bid double auctions	Hadjigeorgalis (2009).
	Water markets: Australia, USA, and Spain Option contract: Spain, USA	Pujol <i>et al.</i> (2006); Qureshi <i>et al.</i> (2011); Rey <i>et al.</i> (2016); Palomo-Hierro <i>et al.</i> (2016).
Water banks	California Drought Emergency Water Bank in 1991, Idaho, Montana, Colorado, Texas, Oregon, Washington Water Trust (USA)	Montilla-López <i>et al.</i> (2016); Dellapenna (2000); Clifford <i>et al.</i> (2004); Megdal <i>et al.</i> (2014).
Buyback	Murray-Darling Basin (Australia) Segura, Júcar, and Guadiana (Spain)	Pérez-Blanco & Gutiérrez-Martín (2017); Montilla-López <i>et al.</i> (2016); Palomo-Hierro <i>et al.</i> (2016).
Credit schemes	Ohio Water Quality Trading (USA)	Tabaichount <i>et al.</i> (2019); Rinaudo <i>et al.</i> (2016); Kieser & McCarthy (2015).

TABLE 1 (CONT.)

Economic instruments in water policy

Economic instruments	Example	References
Information policies (value chain)		
Product certification schemes	WaterSense label (USA) Blue Angel (Germany)	U.S. Environmental Protection Agency (2025). Blue Angel (2025).
Green nudges	Neste River basin (France)	Chabé-Ferret <i>et al.</i> (2019); Lu <i>et al.</i> (2019).
Water stewardship certifications	AWS International Water Stewardship Standard. Sustainable Program for Irrigation and Groundwater Use (SPRING)	Alliance for water stewardship (2026). GLOBALG.A.P (2025).
Voluntary actions		
Sector agreement	Data centre water usage	Judge (2022).
Common property resources		
OUGC, CUAS, Nebraska Nature	OUGC (France), CUAS (Spain), NRD (Nebraska)	Adelman (2003); Rouillard <i>et al.</i> (2021).

Note: OUGC: Organisme unique de gestion collective (single collective management body); CUAS: Comunidad de usuarios de aguas subterráneas (groundwater users’ associations); NRD: Natural Resources Districts.

Source: Own elaboration.

The following instrument descriptions are presented in a concise, taxonomy-oriented manner focusing on defining features relevant for classification rather than providing exhaustive reviews of design variants or empirical performance, which is beyond the scope of this study.

2.1. Price-based instruments

Price-based economic instruments are financial mechanisms implemented by authorities to manage and regulate water use and quality. These instruments encourage sustainable behaviour by implementing tariffs, taxes, or charges to promote water-use efficiency, internalise environmental and social costs, and ensure full cost recovery for water services (Lago *et al.*, 2015). The three major instruments are abstraction charges, emissions charges, and non-compliance fees.

Abstraction charges are fees imposed by authorities or private suppliers for the use of conventional or non-conventional water. They may be levied as a flat fee (e.g., per hectare), on a volumetric basis, or as a two-part tariff combining fixed and volumetric components. Depending on their design, these charges can reflect financial cost

recovery and/or include elements that account for resource rents or external costs (Finney, 2013)

Emissions charges, wastewater effluent taxes, or discharge fees are levied to discourage pollution by charging based on pollutant volume and concentration. Revenues are often earmarked to support investments in pollution abatement and water-quality improvement (Möller-Gulland *et al.*, 2015).

Non-compliance fees are penalties imposed on users who breach abstraction or discharge requirements. They are intended to deter non-compliance and to compensate, at least in part, for the negative impacts associated with illegal or harmful activities (Loch *et al.*, 2020).

2.2. Subsidy/subsidy removal

A subsidy can be understood as the implicit (or explicit) transfer created when users pay less than the marginal cost of supply, or less than the full economic cost where environmental costs are taken into account (Tiwari & Dinar, 2002). In water policy, subsidies for water conservation include subsidies to capital (investment) or to exploitation.

Subsidy removal targets situations where subsidised irrigation water prices weaken incentives for efficient use. By increasing the effective price signal, removing subsidies can reduce overuse and, where relevant, associated pressures on water quality; fiscal savings may also be redirected towards investments in water-efficient technologies (Tiwari & Dinar, 2002; European Court of Auditors, 2021).

Payments for ecosystem services (PES) are incentive-based arrangements intended to promote ecosystem protection by compensating land or resource managers for practices that sustain or enhance the provision of ecosystem services (Muradian *et al.*, 2013; Perrot-Maitre, 2006). The core rationale is that beneficiaries of ecosystem services (e.g., downstream water users) provide compensation to those whose management decisions influence the continued delivery of those services.

2.3. Trading schemes

Trading schemes are organised arrangements that involve the transfer of water rights or allocations between users to improve allocative efficiency and flexibility under changing conditions of scarcity (Pujol *et al.*, 2006). Tradable rights (water trading) range from relatively simple bilateral exchanges to more formal cap-and-trade-style systems, where a cap on use is set at basin level and allocations (or entitlements) can be transferred subject to technical and regulatory constraints (Delacámara *et al.*, 2014; Lago *et al.*, 2015).

Water banks are intermediary-based arrangements in which an administrative agency or designated operator facilitates trading by matching buyers and sellers and standardising transactions. Such arrangements may be operated by public or private agencies (Montilla-López *et al.*, 2016). Water banking is used to manage supply variability by storing water in wet periods (in aquifers or surface reservoirs) for later recovery during periods of deficit, and is particularly common in groundwater settings (Megdal *et al.*, 2014).

The buyback or reacquisition of water rights involves the purchase of permanent water rights, typically through tender processes, to reduce over-allocation and reassign water to environmental or other priority uses. This type of trading scheme is usually voluntary and implemented with pre-defined budget or price constraints to manage costs (Grafton & Wheeler, 2018; Montilla-López *et al.*, 2016).

Water credits are a relatively novel approach in which users generate tradable credits by conserving water or reducing use, which can then be sold to other users who need additional water resources. Credits are intended to create financial incentives for conservation actions, although implementation remains limited (Abu Madi *et al.*, 2008).

Finally, tradable emission permits apply a similar cap-and-trade logic to water quality by setting a limit on total pollutant loads, allocating discharge permits, and allowing trading among users. Under the right circumstances, these tradable permits have the potential to deliver environmental improvements more rapidly and at a lower cost than alternative policy instruments, generating potential efficiency gains and strengthening incentives for innovation (Shortle, 2013). However, some studies have critically evaluated the effectiveness of water-quality trading, highlighting challenges and limitations in practice (Wang *et al.*, 2024; Vangipuram *et al.*, 2025).

2.4. Information policies (value chain)

In water governance, ‘information policies’ in value-chain contexts refer to non-price approaches that aim to influence behaviour. These include improving transparency, signalling performance, and shaping norms or procurement requirements.

Product certification schemes define auditable criteria intended to promote responsible water management (including diffuse pollution control), typically focusing on efficiency, reuse, sustainable sourcing, and risk management. However, water-footprint indicators can be misleading without robust contextualisation (Fereres *et al.*, 2017; Perry, 2014).

Green nudges are behavioural, non-price interventions implemented by an authority, utility, or programme operator. Nudges seek to reduce water use by changing

salience and social norms, such as social comparison, environmental education, and other approaches (Binet *et al.*, 2024).

Water stewardship certifications are voluntary assurance frameworks that signal water management practices regarded as socially equitable, environmentally sustainable, and economically beneficial. They typically combine site-level requirements with catchment-oriented actions and stakeholder engagement processes (Alliance for Water Stewardship, 2023).

2.5. Voluntary actions

Voluntary actions (including voluntary agreements) are commitment-based mechanisms in which organisations or users voluntarily adopt specified practices or targets, rather than responding to prices or behavioural feedback. Participation is commonly motivated by reputational considerations, buyer expectations, collective norms, or broader self-regulatory incentives, as discussed in the literature on voluntary environmental programmes (Heinz, 2002; Pedersen *et al.*, 2015).

2.6. Common property resources

The conversion of privately-owned resources, such as water entitlements held by stakeholders, into ‘common properties’ is used in some aquifers as a critical strategy to ensure the sustainability of endangered and overexploited resources. This approach, often referred to as ‘co-management’, involves shared responsibilities between users and the state, which is typically represented by government agencies (Molle & Closas, 2020a).

3. The dynamic effect of economic instruments

3.1. Static vs. dynamic responses

The concept of ‘static response’ refers to the capacity of an instrument to produce changes without causing subsequent or spillover effects. In this context, a static intervention, such as a buyback, can modify the allocation of water resources without triggering additional consequences or ripple effects. This type of intervention results in immediate, direct outcomes that do not lead to further unintended or cascading impacts within the system. In contrast, when EIs cause behavioural changes, they can be classified as either short-term static or long-term static depending on the duration of the response. Consistent with their temporary nature, short-term policies exert effects over a brief period. In contrast, long-term policies produce effects that persist beyond their implementation and can therefore be described as ‘robust’.

Conversely, dynamic interventions are characterised by unwanted effects that can reduce the policy’s overall effectiveness and potentially cause the system to revert

to its initial state. These unwanted effects are related to the adaptive response of the water users. The term ‘adaptive response’ refers to the way in which these policies influence users’ behaviour over time and can take two distinct forms:

- **Positive dynamic response.** This refers to a situation in which users’ behaviour aligns with the intended changes set out in the policy. In such cases, behavioural adjustments support the intended outcomes of the intervention, contributing positively to its overall effectiveness. These responses not only enhance the desired outcomes but also support the policy’s long-term sustainability.
- **Negative dynamic response.** This occurs when users adjust their behaviour in ways that undermine or diminish the effectiveness of a policy. This can occur when users alter their practices in response to an intervention in ways that counter the intended objectives, such as by increasing water consumption or adopting less sustainable practices. A notable example is the ‘rebound effect’; a concept originally developed in the energy sector and later applied to water policy, where instruments such as water conservation subsidies can produce unintended consequences, thereby reducing the overall impact of the policy (Berbel *et al.*, 2018).

As regards dynamic adaptation to policy instruments, both positive and negative responses can influence the intended policy outcomes. For instance, when the objective is to reduce water abstraction, a continuous intervention such as water pricing typically shows lower short-term than long-term demand elasticity (Scheierling *et al.*, 2006). Long-term adaptations (e.g., changes in irrigation systems or the selection of crops with higher apparent water productivity) constitute a ‘positive dynamic response’ as they reinforce the desired policy effect. In contrast, interventions such as water conservation subsidies aimed at improving water-use efficiency can trigger a negative dynamic response where farmers exploit the resulting ‘water abstraction savings’ to expand irrigated areas or intensify existing ones as a spontaneous rational behaviour. Such unintended outcomes can be prevented through strict and decisive policy governance.

3.2. Single, continuous, and discontinuous interventions

Another dimension for classifying interventions is to consider the duration of implementation. There are three types of interventions: single, discontinuous, and continuous. Single (or one-off) interventions are applied at a specific point in time and typically address immediate or short-term issues. These interventions are limited in scope and duration, focusing on achieving a particular objective within a defined timeframe (i.e., a buyback). Discontinuous interventions are those that occur intermittently or periodically, with intervals or gaps between applications. These interventions are often designed to address recurring challenges but do not

require constant engagement or intervention, allowing for periodic adjustments or assessments (e.g., best-practice subsidies). Finally, continuous interventions are those that, once applied, are sustained over a prolonged period. These interventions are typically aimed at addressing long-term challenges (i.e., water pricing).

3.3. Classification of EIs based on temporal intervention and response type

Figure 1 illustrates how different policy trajectories unfold over time depending on the type of intervention (single in Figure 1a and continuous or discontinuous in Figure 1b) and the type of response (static or dynamic). The figure displays generic cases in which the implementation of an EI influences the indicator (initially set at 100) and how this effect evolves over time.

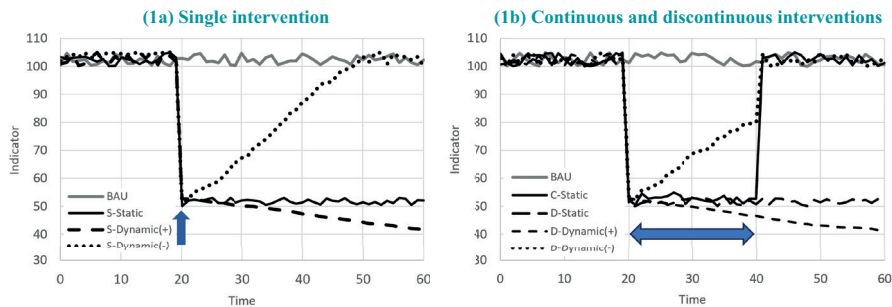
Figure 1a depicts the scenario in which no intervention is implemented (i.e., business as usual [BAU]) alongside three types of responses. The first corresponds to a single static intervention, resulting in a robust response that persists over time. Additionally, the figure includes two dynamic interventions: a positive one and a negative one. A positive dynamic intervention reinforces the initial effect over time, ultimately exhibiting robustness in terms of its long-term persistence. In contrast, in a negative dynamic intervention, unintended effects or secondary consequences cause the indicator to revert to its original level over time.

Figure 1b depicts continuous and discontinuous interventions. Static continuous interventions remain effective only while the policy is active and revert to the initial state once discontinued. In contrast, static discontinuous interventions demonstrate long-term robustness due to their ability to modify behaviour through intermittent applications (e.g., green nudges). Additionally, dynamic interventions are categorised as either negative (i.e., having a limited short-term effect that eventually dissipates) or positive (i.e., user responses reinforce the intervention, rendering it robust over time).

Table 2 presents examples of EI responses and intervention types. Notably, the taxonomy excludes single, static, short-term interventions, as their nature precludes their existence. A purely static, single intervention loses its effect the very moment it is implemented, rendering it conceptually meaningless. Consequently, within the category of single interventions, only two types of responses are relevant: robust long-term responses and negative dynamic responses, in which the effect gradually diminishes over time. Likewise, by definition, a continuous static intervention cannot exist.

FIGURE 1.

Types of trajectories for policy interventions



* BAU: Business as usual; S-Static: Single Static (robust); S-Dynamic (+): Single positive dynamic (robust); S-Dynamic (-): Single negative dynamic; C-Static: Continuous static short term; D-Static: Discontinuous static permanent response (robust); D-Dynamic (+): Discontinuous positive dynamic long term (robust); Discontinuous negative dynamic.

Source: Own elaboration.

TABLE 2

Examples of EIs responses and intervention types

Intervention type	Static response (short-term effect)	Dynamic adaptation response		Static permanent (robust)
		Negative adaptation (reduced impact)	Positive adaptation (amplified impact)	
Single	N/A		Green nudges	Buyback
Discontinuous	Water bank (leasing) Subsidies for exploitation	Subsidies for water conservation	Non-compliance fees	Green nudges
Continuous	N/A	Water rights trade	Volumetric rate** Tax Emissions charges PES Tradable emission permit Water stewardship certifications	Abstraction charges Flat rate Common property resources (OUGC, CUAS) Product certification schemes Sector agreement

Note: **Short-term is lower than long-term elasticity of demand. PES: payments for ecosystem services; OUGC: Organisme unique de gestion collective (single collective management body); CUAS: Comunidad de usuarios de aguas subterráneas (groundwater users' associations)

Source: Own elaboration.

3.4. Examples of the dynamic effect of economic instruments

In what follows, several examples illustrate how interventions, differentiated by their temporal scope, can elicit distinct adaptive behaviours and long-term impacts. By analysing several examples, the study explores how users respond to interventions over extended periods. Such long-term effects are frequently neglected in analyses that focus solely on short-term outcomes.

Volumetric water pricing as a continuous intervention: a case of continuous positive dynamic adaptation

Volumetric water pricing provides an illustrative example of a continuous intervention, as volumetric charging maintains a marginal cost signal for each unit of water consumed over time. An empirically grounded illustration is provided by the Central Emilia Irrigation Water District (CEWD) in Emilia-Romagna, Italy, where the introduction of volumetric charging enables an ex post assessment of farmers' behavioural responses using observed delivery and administrative data (Pronti & Berbel, 2023).

The wider literature indicates that irrigation water demand is often relatively inelastic in the short run, reflecting limited scope for immediate adjustments in crop choice or irrigation infrastructure (Scheierling *et al.*, 2006; Molle & Berkoff, 2007). Nonetheless, short-run optimisation responses can occur under certain conditions, including reductions in irrigation doses as marginal water prices increase (Sapino *et al.*, 2022). Over longer horizons, sustained exposure to marginal pricing can lead to more substantial adjustments in management practices and, where feasible, technological changes, thereby reinforcing the intended conservation effect (Scheierling *et al.*, 2006).

From a cost-structure perspective, volumetric charging shifts water from a largely fixed-cost component to a variable input cost. This strengthens the link between water use and production decisions and incentivises economisation where marginal returns are lower (Zilberman & de Gorter, 1984; Scheierling *et al.*, 2006). In the case of the CEWD of Italy, Pronti & Berbel (2023) have reported sizeable reductions in water use among farmers transitioning from flat-rate to volumetric charging consistent with a positive dynamic response in which ongoing price exposure supports progressive adjustment over time (Berbel *et al.*, 2019b).

Capital subsidies as a discontinuous intervention: balancing immediate savings and negative adaptive behaviours

Capital subsidies for irrigation modernisation (e.g., support for drip conversion or conveyance upgrades) provide a typical example of a discontinuous intervention, as they are implemented through discrete programme rounds and induce one-off

investment decisions. Such subsidies can reduce the water applied per hectare and are therefore often presented as ‘water saving’ in the short run (Berbel *et al.*, 2018). However, a substantial literature cautions that these apparent savings may be offset over time by negative dynamic responses –including rebound and scale effects– when efficiency improvements enable expansion, intensification, or shifts in production that increase basin-level consumptive use (Huffaker, 2008; Adamson & Loch, 2014; Liu *et al.*, 2019).

A well-documented illustration is provided by Ward & Pulido-Velazquez (2008), who examine drip-irrigation subsidy scenarios in the Upper Rio Grande Basin using an integrated basin-scale hydroeconomic optimisation model. Their results show that subsidies can reduce applied water while increasing total water depletion (evapotranspiration), largely because improved application efficiency reduces return flows and aquifer recharge that would otherwise remain available within the system. This finding reinforces a key design implication highlighted in the broader evidence base: capital subsidies are more likely to deliver durable water savings when accompanied by governance arrangements that define conservation in terms of consumptive use/depletions, constrain re-appropriation of ‘saved’ water, and are supported by effective monitoring and enforcement (Huffaker, 2008; Ward & Pulido-Velazquez, 2008; Berbel *et al.*, 2015).

Buyback as a single intervention: a case of robust response and long-term water allocation stability

Buybacks, as the permanent reacquisition of water rights, can be classified as a single intervention at the entitlement level, since each transaction permanently transfers rights from consumptive users to environmental holders. Although buyback programmes are often implemented through multiple tender rounds, their defining feature is the irreversible reallocation of rights once acquired (Grafton & Wheeler, 2018; Montilla-López *et al.*, 2016). A prominent illustration is the Restoring the Balance programme in the Murray–Darling Basin, where the Australian Government purchased water entitlements (and related products) from willing sellers to increase environmental water availability and support long-term ecological objectives (Wheeler & Cheesman, 2013; Dixon *et al.*, 2012).

The principal strength of buyback lies in its capacity to secure enduring changes in allocation with relatively limited ongoing administrative intervention once rights are transferred and held for environmental purposes (Crase *et al.*, 2012). Because the entitlement is removed from consumptive control, environmental gains are less exposed to behavioural ‘offsetting’ responses than instruments that operate through management changes while leaving rights unchanged. Buyback programmes can also contribute to price discovery in water markets, thereby informing policymakers about the opportunity costs of reallocation (Wittwer, 2011; Wheeler & Cheesman, 2013). At the same time, concerns regarding adverse rural impacts (e.g., employment

losses or farm exit) have been raised, although empirical evidence suggests that such effects are not necessarily systematic or as large as sometimes claimed (Wheeler & Cheesman, 2013).

Overall, buybacks exemplify a single intervention with robust long-term implications for allocation, provided that the programme design (e.g., tender rules and targeting) supports environmental effectiveness and avoids unnecessary market distortions (Montilla-López *et al.*, 2016; Grafton & Wheeler, 2018).

Sector agreements as a continuous intervention: strengthening collective commitment for long-term water sustainability

Sector agreements are voluntary but structured arrangements designed to foster collective action for sustainable water management. They typically involve industry associations, public authorities, and other stakeholders committing to agreed objectives and measures, often as an alternative or complement to prescriptive regulation (Heinz, 2002). Within the taxonomy adopted in this paper, sector agreements are considered a continuous intervention, as they depend on sustained commitments over extended periods and ongoing governance arrangements, such as monitoring, reporting, and periodic review, to maintain performance.

In the short term, sector agreements can support relatively rapid improvements in practice by coordinating sector-wide guidance and encouraging the adoption of measures that reduce pressures on water resources (e.g., improved nutrient management, reduced pollutant loads, and best practices). Illustrative evidence from the Netherlands suggests that cooperative arrangements between farmers and water actors have been associated with measurable progress on nitrate-related objectives (Heinz, 2002). Similarly, in France, agreements facilitated by water agencies have been used to encourage agricultural sectors to limit pesticide runoff into river basins, thereby reducing treatment burdens and downstream impacts (Aubry *et al.*, 2014). In addition to immediate changes in practices, such agreements can also enhance awareness and knowledge sharing within sectors, which may increase the likelihood of sustained adoption.

Over longer horizons, sector agreements may contribute to more durable outcomes when participants internalise agreed norms and embed them in routine operational decisions. Heinz (2002) reported cases in Germany in which improvements in water quality management persisted beyond the formal agreement period, indicating the potential for longer-term stabilisation of practices under supportive institutional conditions. At the same time, reliance on voluntary compliance can create vulnerability to performance erosion, particularly where monitoring is weak or incentives are insufficient. Consequently, the effectiveness of sector agreements depends critically on transparent accountability mechanisms and periodic renegotiation, and may be further strengthened by linking participation to credible

economic incentives (Organisation for Economic Co-operation and Development, 2001; Marcus *et al.*, 2024).

Green nudges (social comparison) as a single or discontinuous intervention: a case of positive dynamic adaptation

Social comparison nudges are non-price behavioural interventions that provide users structured feedback on their own water use relative to a reference group (or efficiency benchmark), thereby leveraging salience and social norms to encourage conservation (Croson & Treich, 2014; Binet *et al.*, 2024). In the taxonomy used in this study, nudges can be considered either single interventions or discontinuous interventions delivered repeatedly. This distinction is important not only for the implementation profile but also for the likely trajectory of responses.

When implemented as a single intervention, the evidence indicates that effects are often modest or short-lived and may, in certain contexts, induce offsetting behavioural adjustments consistent with a negative dynamic response. By contrast, when nudges are applied consistently and reinforced over time, repeated feedback helps maintain salience and supports learning processes, thereby increasing the likelihood of sustained conservation behaviour and a more durable behavioural trajectory (Binet *et al.*, 2024; Schubert, 2017).

An empirically grounded agricultural illustration is provided by Chabé-Ferret *et al.* (2019), who conducted a randomised controlled trial among irrigators in south-west France using metered data. Treated farmers received weekly SMS messages over four months reporting their own water consumption and the average consumption in their watershed—an archetypal discontinuous social-comparison nudge. While the intervention did not produce large average reductions in total water use, it did generate measurable behavioural adjustments at the upper tail: the proportion of farmers consuming more than 80 % of their quota fell from approximately 10 % to 3 %, indicating a shift away from extreme water use. In the same line, other studies indicate that when consumers receive information comparing their water use to that of more efficient peers, they tend to reduce their consumption (Ferraro *et al.*, 2011; Fielding *et al.*, 2013; Schultz *et al.*, 2016; Seyranian *et al.*, 2015; Strong & Goemans, 2014; Brent & Wichman, 2022).

Water banks as a discontinuous intervention: a mechanism for static reallocation without long-term adaptive effects

Water banks are classified as discontinuous interventions because they operate through intermittent activation and discrete transaction rounds (e.g., seasonal calls, drought programmes, or time-bounded trading windows) rather than applying a continuous incentive signal. In the taxonomy considered here, water banks are associated primarily with a static short-term response, since their effects arise from

the immediate reallocation achieved through transfers (typically temporary leases or allocations) and do not, in themselves, rely on behavioural adaptation processes unfolding over time (Montilla-López *et al.*, 2016; Carney *et al.*, 2021).

A well-documented example is the California Drought Emergency Water Bank of 1991, which was established as a centrally coordinated mechanism to procure and reallocate water during severe drought. By organising purchases and subsequent allocations through an administratively managed platform, the water bank enabled rapid transfers across user groups (including agricultural and urban users, as well as environmental needs), thereby providing short-term allocative flexibility without altering underlying entitlements on a permanent basis (Lund & Israel, 1992). In this sense, effectiveness derives from institutional intermediation by standardising transactions, reducing search and negotiation frictions, and accelerating reallocation under acute scarcity, rather than from inducing progressive changes in water-use behaviour (McCann *et al.*, 2005; Montilla-López *et al.*, 2016).

Nevertheless, the performance of water banks depends critically on enabling governance conditions, including clearly defined rights and accounting rules, transparent operating procedures, and sufficient administrative capacity to verify and execute transfers (Carney *et al.*, 2021). When these conditions are met, bank-type platforms can also reduce information frictions and improve matching and price discovery, thus supporting more efficient reallocation during periods when additional flexibility is most needed (Chen *et al.*, 2024).

4. Discussion

Many countries have reformed their water allocation systems in response to droughts and scarcity and society's growing environmental concerns. These reforms aim to increase water management flexibility and efficiency as existing allocation regimes often lack adaptability and effectiveness (Rey *et al.*, 2018). A central challenge for both policy design and empirical assessment is that EI performance is often time-dependent and context-contingent, which makes comparisons across cases difficult and contributes to inconsistent findings in the literature. The taxonomy proposed in this paper responds to this challenge by providing an explicitly temporal classification that distinguishes instruments by a) intervention type (single, discontinuous, continuous) and b) response type (static short-term, dynamic adaptation, static permanent/robust). The analytical value of this framework lies in its capacity to organise empirical observations in terms of trajectories, rather than treating policy effects as inherently stable once an instrument is introduced.

The examples summarised in Table 2 illustrate how the framework aids the interpretation of heterogeneous evidence. Instruments that appear similar in mechanism can exhibit markedly different trajectories depending on whether they elicit adaptive responses. For instance, irrigation efficiency investments subsidised

through discontinuous programmes may reduce water applied at the field level but fail to generate basin-scale savings if users expand production or return flows are reduced—effects often described as rebound or system feedbacks (Perry & Steduto, 2017; Ward & Pulido-Velazquez, 2008; Huffaker, 2008; Adamson & Loch, 2014). Within this framework, such outcomes are understood as dynamic adaptive responses to discontinuous interventions, rather than anomalies. Similarly, the literature on water trading indicates that market-mediated reallocations can improve flexibility and efficiency (Pujol *et al.*, 2006), yet distributional outcomes and long-term sustainability can vary across contexts (Qureshi *et al.*, 2011). Framing these instruments through the lens of response trajectories helps distinguish cases where trading operates primarily as a static reallocative mechanism from those where behavioural and institutional interactions drive dynamic outcomes.

Beyond its ex-post interpretation, the double-axis classification also offers ex-ante design insights. By distinguishing between intervention and response types, it helps anticipate where negative adaptation is most likely and, consequently, where complementary measures may be needed from the outset. In this way, the framework can inform the design of coherent policy mixes by indicating where instruments are likely to require reinforcement to prevent erosion of effects.

The framework also helps distinguish instruments that produce enduring changes in allocation from those that require repeated implementation to sustain their effects. Buybacks, for example, can deliver robust outcomes at the entitlement level by permanently reallocating rights away from consumptive use, thereby limiting the scope for subsequent behavioural offsetting (Grafton & Wheeler, 2018). By contrast, instruments such as green nudges may deliver stronger and more persistent effects when implemented discontinuously, while one-off nudges may be weaker and more heterogeneous in their trajectories (Binet *et al.*, 2024; Chabé-Ferret *et al.*, 2019; Perry & Steduto, 2017).

A relevant example of a dynamic unintended consequence is the policy of subsidising water-saving investments. If strict governance measures are not implemented, there is a serious risk that water savings from efficiency improvements will increase overall consumption due to the so-called ‘rebound effect’: the ‘saved’ water is often reallocated to expand irrigated areas or to shift to more water-intensive crops. Empirical evidence confirms this pattern (Perry & Steduto, 2017). In Europe, the European Court of Auditors (2021) found that CAP-funded irrigation efficiency measures frequently led to higher, rather than lower, water use. However, the rebound effect can be mitigated through robust governance. This includes measures such as defining water rights in terms of consumptive use to prevent appropriation of return flows, mandating that water savings be retained for environmental or public purposes—such as the required abstraction reductions in Spanish basins, which have shown to be effective (Berbel *et al.*, 2019b)—and fostering collaborative agreements among stakeholders, as illustrated by contrasting experiences in Kansas (Zwickle *et*

al., 2021). Without such ex-ante regulatory design and ex-post monitoring, efficiency policies risk undermining their objective of reducing abstractions.

Several limitations warrant emphasis. First, classification is inherently an abstraction. Indeed, many instruments allow multiple design variants and may fall into different categories depending on institutional conditions and how conservation or compliance is defined and measured. Second, empirical identification of response type can be sensitive to the evaluation time horizon and the availability of data that capture longer-run adjustment processes. Third, instrument trajectories are shaped by both their institutional setting and interactions with other measures within the broader policy mix. The framework is intentionally parsimonious and is not intended to represent causal interactions among instruments. Rather, it provides a structured lens for identifying where complementary governance conditions are likely to be decisive in determining whether the observed responses reinforce or erode the intended effect. Finally, political and social feasibility can influence trajectories. Stakeholder opposition and legitimacy concerns may constrain implementation even when instruments are technically well specified, as illustrated by debates over large-scale transfers and their associated ecological and socio-economic concerns (Medellín-Azuara *et al.*, 2013). These considerations underscore the fact that the temporal framework is best understood as an analytical tool for structuring observation and comparison, rather than as a deterministic predictor of outcomes.

5. Conclusions

This paper proposes a streamlined taxonomy of economic instruments in water environmental policy to enhance conceptual clarity and enable more consistent comparisons across a diverse set of instruments and contexts. Its central contribution is the explicit incorporation of a temporal framework that distinguishes instruments by their intervention profile and the nature of responses over time. By shifting attention from instrument labels to trajectories, the framework provides a structured approach to interpret why instruments that share similar mechanisms may nevertheless exhibit markedly different long-run performance.

The temporal framework is designed to be both policy-relevant and analytical. By making temporal profiles and response trajectories explicit, it supports more anticipatory reasoning about instrument choice and design, including whether effects are likely to persist, require reinforcement, or be vulnerable to erosion through adaptive behaviour. In this way, it offers a coherent basis for discussing the conditions under which EIs contribute to sustainable water management objectives and for structuring evaluation beyond short-run outcomes.

Overall, the taxonomy and temporal framework are intended as a practical tool to aid communication among researchers, policymakers, and stakeholders, as well as to strengthen the evidentiary foundation for designing and assessing EIs in water governance.

Future research could extend the framework in three complementary directions. First, further work is needed to more systematically examine the interplay between EIs and institutional frameworks, including how rights structures, accounting conventions, and enforcement capacity condition response trajectories over time. Second, additional empirical studies that evaluate real-world applications across diverse socio-economic and hydrological settings would strengthen the external validity and support more consistent coding of instruments into trajectory categories. Third, integrating interdisciplinary approaches, particularly behavioural economics, system dynamics, and participatory governance, could improve understanding of how behavioural responses, feedbacks, and legitimacy dynamics shape long-run policy performance and how policy mixes can be designed to foster adaptive and resilient water management strategies.

Authorship contributions

Esther Díaz-Cano: Methodology, investigation, visualization, writing – original draft and writing – review & editing. **Julio Berbel:** Conceptualization, writing – original draft and funding acquisition. **Nazaret M. Montilla-López:** Methodology, investigation and writing - original draft. **Carlos Gutiérrez-Martín:** Investigation, supervision and funding acquisition.

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Declaration of Conflict of Interest

The authors declare no conflict of interest

Data Availability

No datasets were generated or analyzed during the current study.

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