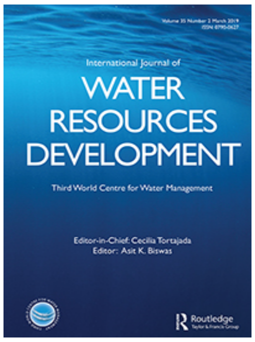

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Role of economic instruments in water allocation reform: lessons from Europe

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ABSTRACT
A growing number of countries are reforming their water allocation regimes through the use of economic instruments. This article analyzes the performance of economic instruments in water allocation reforms compared against their original design objectives in five European countries: England, France, Italy, Spain and the Netherlands. We identify the strengths of, barriers to and unintended consequences of economic instruments in the varying socio-economic, legal, institutional and biophysical context in each case study area, and use this evidence to draw out underlying common guidelines and recommendations. These lessons will help improve the effectiveness of future reforms while supporting more efficient water resources allocation.

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Introduction

In many countries in Europe, population growth coupled with changing income distributions, and the longer-term threat due to human-induced climate change, is increasing competition for water resources, resulting in supply–demand imbalances (Intergovernmental Panel on Climate Change, 2014). While new or enlarged water resource infrastructure (e.g. reservoirs, boreholes) to expand the supply base can assist in the short term, expansion will eventually exhaust the limited capacity of water resources to support new diversions at affordable prices, a process known as river basin closure (Falkenmark & Molden, 2008). Where the costs of expanding supply exceed the economic benefits of marginal uses, policy makers are expected to shift priorities towards making water withdrawals compatible with available resources (Randall, 1981). In this context, a growing number of countries are reforming their water allocation regimes – the set of laws, rules or common practices that determine who is able to use water resources, including how much they can abstract, for what purpose, and when and where withdrawals are permitted (Organisation for Economic Co-operation and

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Development [OECD], 2015a). *Economic instruments* or incentives are expected to play a major role in these reforms (Lago, Mysiak, Gómez, Delacámara, & Maziotis, 2015).

According to the Dublin Principles, managing water as an economic good is instrumental in achieving efficient and equitable use and encouraging conservation and protection of water resources. Yet, water allocation regimes often distribute resources on the basis of historical rights or queueing (Chong & Sunding, 2006), and are plagued with rigidities (e.g. non-transferability), inconsistencies (e.g. due to lack of regulatory coordination) and information gaps (e.g. environmental performance) (OECD, 2015a), which all together hinder the prioritization of uses in accordance with their economic value. This is particularly relevant in agriculture, the largest global water user and the sector where the marginal uses of water resources are most concentrated (Food and Agriculture Organization, 2013). As river basins close and economic and environmental impacts grow, existing water allocation regimes, which typically lack flexibility are no longer fit for purpose. This has led to attempts to improve their performance through the use of economic instruments (Gleick, 2002; Wutich et al., 2014). Here we define economic instruments for water management as those incentives designed to align individual behaviour with the public objectives of achieving reliable quantity and quality of water and mitigating water-related risks (Delacámara et al., 2014; Gómez, Pérez-Blanco, Adamson, & Loch, 2017; Lago et al., 2015). Incentive compatibility is thus key to their success (Delacámara, Gómez, & Maestu, 2015a). However, where private incentives conflict with water policy objectives, economic instruments can fail or even backfire. This is reported to be happening in Australia, where environmental flows have been reduced after the implementation of market incentives to enhance efficiency among commercial users (Connor & Kaczan, 2013), and in Spain, where subsidies for irrigation modernization have led to increased water consumption (Berbel, Gómez-Limon, & Gutiérrez-Martín, 2017; Rodríguez Díaz, Urrestarazu, Poyato, & Montesinos, 2012) and withdrawals (Gutiérrez-Martín & Gómez, 2011) rather than reducing demand.

This article reviews and assesses the design, implementation and achievements of economic instruments in the context of water allocation reforms in Europe, to draw out useful insights and lessons to enhance their future uptake and performance. In a literature review, four key steps were undertaken. First, we reviewed major water allocation reforms to identify their specific objectives, focusing on five case studies in Europe, including Spain, Italy, the Netherlands, France and England. Each case study was critiqued qualitatively regarding: (1) existing pressures on freshwater resources; (2) the water regulatory framework, existing water allocation systems and recent reforms; (3) the use of economic instruments in the mix of water management arrangements; and (4) objectives and performance of their economic instruments. Second, we critically analyzed the role of economic instruments in achieving water allocation reform objectives in each case study. The analysis extended beyond the water charging and market mechanisms widely addressed in the literature to include voluntary agreements, buyback, subsidies and insurance. We then identified critical variables that enabled or prevented the adoption and success of economic instruments. Finally, while acknowledging that the design of water allocation is typically context-specific, we compared the strengths of, barriers to and unintended consequences of economic instruments across the varying socio-economic, legal, institutional and biophysical contexts to draw out underlying common guidelines and general recommendations. Lessons learned from this critical review are relevant not only to other European countries involved in a similar implementation process in the context of the European Water Framework Directive (WFD; European

Commission [EC], 2000), but also for other countries with interests in developing a deeper understanding of the strengths, weaknesses and implementation barriers of economic instruments.

European water policy and recent regulatory reforms

Water regulatory framework: the EU context

The use of economic instruments to promote economic efficiency in water resources management was one of the boldest recommendations of the WFD, with major implications for water management in Europe (Gómez-limón & Martín-Ortega, 2013). The use of economics thus transcends its traditional supportive role (funding) to become part of a transformational policy that overcomes incremental (e.g. grey infrastructure) responses in water management. Failure to integrate economic instruments in water reforms in Europe has been acknowledged as one of the reasons for the under-achievement of the WFD objectives (EC, 2012a).

The WFD set out a new paradigm for EU water policy. The overarching goal of achieving 'good ecological status' of water bodies was to be accomplished through a series of objectives that included preventing further deterioration of aquatic and water-dependent ecosystems; promoting sustainable water use based on a long-term protection of available resources; enhancing protection and improvement of the aquatic environment; ensuring the reduction of groundwater pollution; and contributing to mitigating the effects of floods and droughts. Management of European waters was to be implemented at a river basin scale (hydrological unit) through river basin management plans that detailed how the WFD objectives were to be attained for that specific basin within the required timescale.

Economic analysis and principles were set to play a key role in the transposition of the WFD to the specific characteristics of each member state, region or basin in the context of the Common Implementation Strategy (EC, 2003). In Article 5 (on economic analysis) and Article 9 (on pricing) the directive calls for sound economic analysis to support the development of economic instruments to fund programmes of measures but also to affect the behaviour of individuals. For example, Article 9 states that 'water-pricing policies provide adequate incentives for users to use water resource efficiently, and thereby contribute to the environmental objectives of this Directive' (EC, 2000, p. 13). Levies, and by extension other economic instruments, are thus desirable not only for their contribution to cost recovery and the enforcement of the polluter-pays principle but also for their ability to align individual decisions with the objective of achieving the good ecological status of European waters. Article 5 calls for an estimation of the costs of water services and an informed (economic) assessment to find the most cost-effective combination of measures to achieve the objectives of the directive, which, according to Preamble 38 and Annex VI, shall use 'economic instruments'.

The Blueprint to Safeguard Europe's Water Resources (EC, 2012a) identified the insufficient use of economic instruments as a key reason for inadequate implementation and integration of water policy objectives, and highlighted that 'not putting a price on a scarce resource like water can be regarded as an environmentally-harmful subsidy'. The blueprint also considered water markets as a tool that could help improve water-use efficiency and overcome water stress, provided that a cap on use was implemented and enforced. Periodic reports from the EU Action on Water Scarcity and Droughts identified some advances in the introduction of water tariffs in a number of member states and the extension of metering devices in most of them, but noted that 'economic instruments have not been widely used by Member States thus far' (EC, 2007, p. 5; EC, 2011). Furthermore, important design flaws were present in those existing economic instruments, pointing towards a mismatch between the purpose and design of the instrument and the objectives of the directive (EC, 2012b). Although the EU regulatory background acknowledges the relevant role of economic instruments in achieving WFD objectives, performance assessments and EU institutional reports to date display a narrow focus on tariffs and disregard other economic instruments already in place in several countries (e.g. European Environment Agency [EEA], 2017), which equates economic instruments to pricing). This narrow focus hinders the ability of European institutions to inform an efficient and effective water policy, which requires a sensible combination of (economic) instruments for water management adapted to the needs and characteristics of each basin (EC, 2012a).

EU institutions also have other means (beyond water policy) to underpin compliance with WFD objectives, notably agricultural policy. The EU allocates nearly 40% of its budget to agriculture, a sector that represents more than 50% of total water abstractions in southern Europe, rising to more than 80% in some regions (EC, 2017; EEA, 2009). The Common Agricultural Policy (CAP) 2014–2020 updated and strengthened its 'second pillar' (EC, 2005), acknowledging the provision of valuable public environmental goods and services by rural areas (European Union [EU], 2013a). This so-called greening of the CAP defined some funding available under the second pillar to be conditional on meeting environmental targets (including those of the WFD); for example by ensuring that adequate charging mechanisms were in place where irrigation modernization investments were subsidized (EU, 2013a). The new CAP also introduced a strong ecological component in its 'first pillar', allocating part of the direct payments to those farmers who complied with predetermined environmental objectives, and continued the decoupling of environmentally and socially harmful subsidies (EU, 2013b). Other CAP initiatives such as the income stabilization tool foresee the subsidy of mutual funds to enhance the uptake of comprehensive insurance policies, which under some conditions can substitute for unsustainable withdrawals from natural capital (groundwater) (EU, 2013a).

Pressures on water resources

Water availability is uneven across Europe. Whereas water scarcity and droughts have traditionally been an issue for some river basins in southern Mediterranean countries, flooding is regarded as the most important risk for the quantitative management of water resources in most central, eastern and northern countries. Nevertheless, the pressure on water resources is increasing due to changing patterns of consumption, ambitious environmental objectives and the initial impacts of a changing climate in many areas across Europe (Dai, 2013; EC, 2011; EEA, 2012; Gudmundsson & Seneviratne, 2016; Prudhomme et al., 2014), with the Water Exploitation Index ('mean annual total demand for freshwater divided by the long-term average freshwater resources', EEA, 2016) reaching more than 20% in some basins (Figure 1).

Water availability pressures are also gaining importance and government attention in northern and central Europe, as evidenced by recent water-allocation reforms in various member states (e.g. the UK: Department for Environment, Food & Rural Affairs [Defra], 2011).

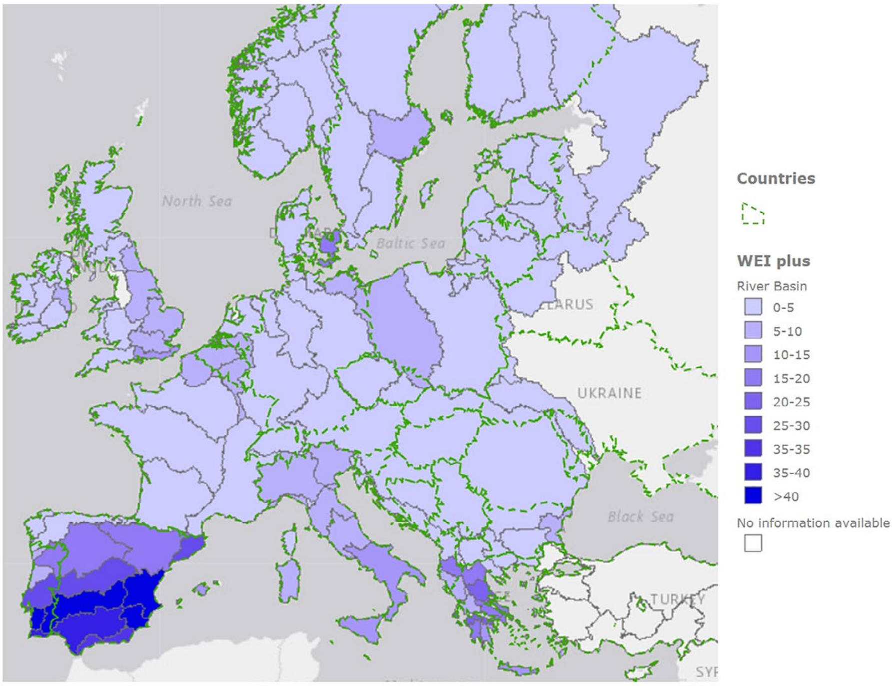

Figure 1. Map of Water Exploitation Index (%) for European countries, summer 2014 (EEA, 2016).

The countries included in this review reflect different water resources challenges, which determine the ongoing evolution of the water abstraction licensing system and their relative maturity. In England, significant pressures on water resources affect both the aquatic environment and available water supplies. Many catchments have little or no water available for additional abstraction licensing, and many abstractors face seasonal restrictions during dry periods (Environment Agency, 2008; Rey, Holman, & Knox, 2017). Resource availability is also becoming more variable (and less predictable) as the country is affected by more frequent drought episodes, with severe impacts in all sectors (Marsh, 2007; Rey et al., 2016). Drought and shortages in freshwater supply are also emerging in the Netherlands. Droughts have occurred in recent years (e.g. 1976, 2003, 2005, and 2011), with significant associated economic impacts, which are expected to occur more frequently in future (Klijn, Van Velsen, Ter Maat, & Hunink, 2012; RIZA, 2005). At the same time, competition across water users is intensifying due to socio-economic and demographic developments, e.g., rising demand for electricity and irrigation water use (Klijn et al., 2012; RIZA, 2005).

In France, the water supply can meet the demand in most of the provinces in an average hydrological year. However, water stress is increasing, especially in the south, and this trend is expected to continue due to the impact of climate change and socio-economic development (Giuntoli, Maugis, & Renard, 2012). In these areas, the increasing frequency of summer droughts means that irrigation water demand exceeds the supply almost every summer. Consequently, a growing number of provinces are now limiting water withdrawals through emergency measures – although the recurring nature of 'emergencies' may justify reforming the water allocation regime in over-abstracted areas (Ministère de l'écologie, de l'énergie, du développement [MEEDDAT], 2008).

Water resources in Spain are limited and scarce and vary significantly from year to year (Garrote, Iglesias, & Flores, 2009). The high variability and uneven distribution of water and its scarcity throughout the country have led to intensive management of water resources, especially for agriculture, through water infrastructure investment (Estrela & Vargas, 2012). Demand is also rising due to demographic shifts, economic development, and lifestyle changes. Rising water demand is also a concern in Italy, where water use has risen steadily in recent decades. In some areas, the volume of authorized abstraction licenses already exceeds average water availability, although some is not used (sleeping licences) (Santato, Mysiak, & Pérez-Blanco, 2016). Water allocation mismanagement thus becomes more apparent during periods of drought, which have become increasingly frequent since the turn of the century (Castellari et al., 2014).

Allocation systems, recent reforms and their objectives across case studies

The influence of European regulations and guidelines, coupled with a greater pressure on water resources, has motivated governments to rethink the way water has traditionally been allocated among competing sectors. This article focuses on five countries in Europe, each with different water availability problems, regulatory systems and allocation mechanisms (Table 1). Over the last two decades, these countries have reformed their water allocation systems, often after a major drought that highlighted the need for greater flexibility and efficiency (Figure 2). In each country, the case for implementing economic instruments for water allocation is increasing, playing a key role in current and planned allocation reforms.

Application of economic instruments to reform water allocation in Europe

The first fundamental theorem of welfare economics, which states that laissez-faire markets tend to a Pareto-optimal resources allocation, is often seen as an argument for non-intervention. Self-interest and free markets, it is argued, should be enough to drive the allocation of resources towards efficiency, according to Adam Smith's (1776) 'invisible hand' hypothesis. This postulate renders policy intervention in resources (re)allocation, including water, unnecessary and even counterproductive (Mendelsohn, 2016). However, this construct also relies on several assumptions that do not often hold true in real-life markets. For example, imperfect information and transaction-cost problems mean that markets are far from Pareto-optimal (Stiglitz & Greenwald, 1986). Thus, laissez-faire markets typically lead to externalities, costs or benefits accruing to third parties. Moreover, even if all assumptions hold, Pareto-optimality is not the same as desirability. For example, Pareto-efficiency may lead to inequitable and biased allocations of resources to specific groups, raising distributive and ethical issues.

The second fundamental theorem of welfare economics states that advances in fairness and policy acceptability are not at odds with economic efficiency, and can be attained through lump-sum transfers that correct undesirable outcomes, such as biased property rights, while leaving all agents in the market better off (Kaldor-Hicks improvement). However, in other cases, barriers of varying nature prevent this. This is the case for deeply entrenched water rights as a result of existing laws, customs and actions by relevant groups of interest. Achieving the greatest collective good requires active management by an institutional agent capable of solving complex water-allocation problems, so that externalities are addressed,

Table 1. Case study description.

Case study	Water regulatory framework summary	Existing water allocation systems	Recent/ongoing water abstraction licensing reform	Objectives of the reform
England	• Defra deals with water policy issues • Water Resources Act 1991 and the Water Act 2003 are the main legislation controlling water abstraction. New amendment of the Water Act in 2014. • Public water supply was privatized in 1989.	• Abstraction licences granted by the Environmental Agency to abstract more than 20 m^3 of water a day (time-limited; first come first served) (2008, 2013) • The licence has conditions to protect other water users and the environment. • Current licensing system set up in the 1960s (not flexible enough to respond to current pressures).	• Reform under design (Defra, 2011, 2016a, 2016b). It will be implemented by the 2020s. • 'Brexit' will pose new challenges and opportunities for future water policy in the absence of EU directive-related obligations.	1. Flexibility to address short-term water availability issues; 2. Long-term sustainable management supporting growth and investment. Main changes: quicker and easier trading, implementation of water-share accounting frameworks in certain catchments, and review of the abstraction charging system (Defra, 2016b).
Netherlands	• The state manages the 'main water management system'. The national government coordinates WFD implementation and is responsible for national policy. • Regional water authorities manage the regional system (operational management, including planning, licensing, enforcement and evaluation) and groundwater. • Ten drinking water companies, semi-public bodies, operate under private law.	• Water allocation system set out in response to the 1976 drought and updated after the 2003 drought (RWS, 2011), giving the environment a more prominent position. • During shortages, a 'sequence of priorities' system determines the allocation of freshwater to specific user categories. • Groundwater abstraction: permits are needed for industrial/agricultural abstractions of >150,000 m^3/y, and a permit is needed for water supply of geothermal energy storage. • Use of surface water is free of charge and permits.	Water abstraction reforms are recommended and their feasibility/impact is being investigated in selected case studies as part of the Delta programme (Deltacommissaris, 2012).	The central aim of these reforms is to increase the flexibility of water allocation at the least cost to society and in an inclusive way.

(*Continued*)

Table 1. (*Continued*).

Case study	Water regulatory framework summary	Existing water allocation systems	Recent/ongoing water abstraction licensing reform	Objectives of the reform
Spain	• River Basin Management Districts are the main water allocation authority. They distribute available resources among water uses through a licensing agreement. • Drought Commissions are created during drought to allocate available resources among users, trying to minimize impacts. • The 2001 Water Act (based mostly on the 1985 Water Act, with slight modifications) determines allocation rules.	• All rights attached to surface waters are 30-year concessions. For those rights under 7000 m^3/y, and in the case of irrigation, water is bundled with land (bundled rights). • 'Use it or lose it' is applicable, although this has rarely been enforced. • Rights differ in the priority of their access to water depending on the type of use. • Environmental flows are not considered a user but a constraint that has to be met.	The most significant amendments included water trading in 1999 and adaptation to the WFD in 2003.	In 1999 the Spanish Water Law was reformed to allow water use right holders to exchange their water concessions for temporary or long-term contracts. The 2003 adaptation to the law only added that river basin management plans should be adapted to the WFD. Since then all such plans have included the main points of the WFD (especially preventing deterioration, seeking sustainability of resource management, and recovering costs).
Italy	• A command-and-control approach is typically adopted to manage droughts, in which the Ministry of Environment sets specific thresholds for each use, with sanctions for non-compliance. • The 2003 drought event in the Po River Basin District opened the way for establishment of a coordinated approach, in which water restrictions are defined through consensual participatory processes in the context of a Drought Steering Committee.	• Incomplete or missing information on the characteristics of water abstraction licenses (for 40%+ of the licences in the Po RBD). • Over-allocation, but sleeping licences prevent structural problems. • Temporary limitations can be enacted during prolonged droughts (states of emergency). • Water tariffs paid are earmarked for improving and maintaining the good ecological status of water bodies, but this is not actually implemented.	• Devolution of water abstraction licensing and management and environmental protection. • Transposition of the EU WFD through the 2006 Environmental Code (Legislative Decree 152/2006). • Following success in the Po RBD, Drought Steering Committees to be deployed in other RBDs in Italy. • Charges for the recovery of environmental and resource costs proposed but put on hold (AAEGSI, 2014).	De jure objectives of the ongoing reform follow EU guidelines: good ecological status and cost recovery, particularly for environmental and resource costs. De facto gaps persist in implementation.
France	• Local Water Committees define maximum extractable volume (maxEV) in areas with structural deficit. • In areas where irrigation is significant farmers shall form a Single Water Users' Association to limit withdrawals to less than their share of maxEV (MEEDDT, 2008).	• From command and control to self-control. • Collective management: shift from centralized management of individual withdrawals to decentralized management of collective withdrawals (Montginoul et al., 2016).	• Implementation of the 2006 French water law. • Implementation of the WFD. • Establishment of collective management of water withdrawals in areas of structural deficit.	EU WFD Good status transposed for quantitative management as: ensure balance between withdrawals and available resources at the local level to ensure that supply of water uses and environmental objectives are achieved in four years out of five for withdrawals and maintain extractions below maxEV.

economic outcomes are maximized, and overall water allotments are aligned with water policy objectives (McCann, 2013; Rausser, Swinnen, Zusman, & Approaches, 2011). Economic instruments are thus a means towards this goal. The following section analyzes how economic instruments can contribute to achieving the objectives of water allocation reform in each case study (Table 2).

Water charges

In contrast to prices that are set in a market environment, water charges in the EU follow an administrative procedure that in many cases does not internalize the costs of the water infrastructure required to deliver the service – let alone environmental and resource costs. This is particularly evident in the irrigation sector, where implicit subsidies in the form of insufficient cost-recovery levels are widespread (EEA, 2013). As a result, water levies are insufficient to pass on the costs of externalities to users in Europe and elsewhere (EEA, 2013; OECD, 2009). Where cost-recovery levels are higher, this usually reflects the users' ability to pay and typically has limited impact on withdrawals. For example, the Guadalquivir River basin has the highest (water infrastructure) cost-recovery ratio in Spain (98%) but displays an inelastic response to charges (Gutiérrez-Martín, Blanco, Gómez, & Berbel, 2014) and has one of the highest Water Exploitation Indices in Europe (Figure 1). This situation can be aggravated where revenues raised through water levies are not directed towards water policies that mitigate the negative impacts of over-allocation, as in Italy (Santato et al., 2016). Levies can be further reduced to incentivize the adoption of desirable water management arrangements, as in the development of the French Single Water Users' Association (Organisme unique de gestion collective) (MEEDDAT, 2008). Such subsidies can ease institutional transformation and compliance with maximum abstractions, but they prevent the identification of the actual cost of water; in the longer term they may also delay adaptation to the risks of river basin closure (Colby & Bush, 1987).

Despite the issues identified above, growing adoption of metering devices underpinned by regulatory and planning instruments, and the empowerment of water users' associations and collective management (e.g., Po River Basin Authority, Po River Basin Agency, 2003), has underpinned transition towards volumetric charging. In all five case studies, a growing number of water levies are defined in two tranches: one that reflects the fixed costs of water conveyance; and the other based on actual withdrawals and use (EEA, 2013). Metering also enables temporary levies on water use that charge abstractors in accordance with season (incremental charges), thus introducing incentives for water conservation during dry periods. This is the case in England and Wales, where abstraction charges are based on season – although this model is now transitioning towards charges based on reliability (Defra, 2016a). Evidence from Italy and Spain shows that higher cost-recovery levels are typically observed where collective water management is in place (Maestu, Del Villar, & Ministerio de Medio Ambiente, 2007; URBER, 2015), suggesting that the devolution of water management to users' associations in countries such as France may have aided cost recovery. Conversely, higher charges constitute an additional cost, which may be rejected by farmers (Rinaudo & Hérivaux, 2014).

In 2012, the European Commission commenced infringement procedures against nine member states 'for their narrow interpretation of the concept of water services' (EC, 2012a, p. 10), which was allegedly 'hindering progress in implementing cost recovery policies

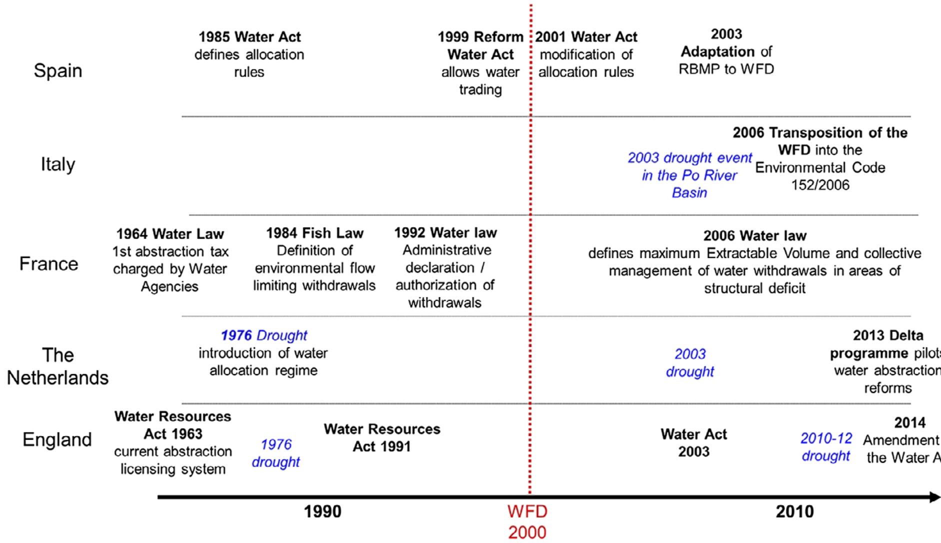

Figure 2. Timeline of water allocation reforms and introduction of economic instruments across case studies.

beyond drinking water and sanitation'. However, the European Court of Justice ruled that member states may decide which economic instruments and design are to be implemented, as long as they meet WFD objectives (Jääskinen, 2014). This could be interpreted as recognition of the role of instruments such as payment for ecosystem services (PES) or subsidies; but the polluter-pays principle pervades environmental legislation. This has not stopped widespread de facto reliance on a benefit-based approach to address environmental challenges, especially through subsidies. Mature water economies with inelastic supply often witness autonomous adaptation and water reallocation to more productive uses, as growing demand limits water availability. At this point, though, charges may be a less cost-effective tool in addressing (urgent) scarcity problems due to more inelastic responses and larger local, regional and economy-wide economic impacts (Pérez-Blanco, Standardi, Mysiak, Parrado, & Gutiérrez-Martín, 2016). Eventually, higher levies may lead to capital losses (e.g. perennial plants) and, in some cases, to farm exit (Wheeler, Loch, Zuo, & Bjornlund, 2014). In compliance with Article 9 of the WFD, charges should anticipate environmental and economic tipping points through a transformational and proactive approach. Yet, the transaction costs associated with transforming institutions are not negligible and can block transition – calling for complementary (economic) instruments.

Payment for ecosystem services

Initially set up to communicate the role that ecosystems play in human well-being (Armsworth et al., 2007), PES schemes have developed over the last three decades into a powerful tool for economic decision making on ecosystem services (Engel, Pagiola, & Wunder, 2008; Gómez-Baggethun, de Groot, Lomas, & Montes, 2010; Landell-Mills & Porras, 2005; Pagiola, 2008; Pagiola & Platais, 2007; Wunder, 2015). PES schemes are defined as voluntary and conditional transactions over well-defined ecosystem services between at least one supplier and one user (Wunder, 2005), following the rationale that the

Table 2. Economic instruments in the context of the water allocation reform.

Instrument	Definition	What can the economic instrument deliver for water allocation reform objectives?	Case studies where instrument is implemented				
			FR	IT	NL	SP	EN
Charges	Levies on water use related to conveyance and storage services and the opportunity cost of the resource. They can be earmarked (tariff) or not (tax).	Compliance with the law of demand implies that water charges are by definition effective. Where enforced, higher levies will eventually reduce withdrawals and enhance environmental/ecological flows. If demand is inelastic, charges become effective revenue-raising tools that can be earmarked for the restoration of water bodies.	X	X	X	X	X
Payment for environmental services	Conditional payments offered to water users in exchange for the voluntary provision of some sort of ecological service.	The beneficiary of an existing or potential watershed ecological service (be it a public or private agent) can exchange financial value for the protection, enhancement or rehabilitation of those services from economic users.	X	X	X	X	X
Subsidies	Financial aid or support to enhance the supply of positive externalities (e.g. through ecological flows). Can be explicit (price supports, subsidized loans, direct payments) or implicit (reduced regulation, tax/charges relief).	Subsidized loans for the adoption of more efficient water infrastructure can contribute to water allocation reform objectives through qualitative improvements, restoration of valuable services, and reclaiming non-recoverable runoff/percolation. Price supports have been also used to promote adoption of a collective water management approach that adjusts caps to available resources. Greening of the CAP to decouple direct payments from crop yields also relieved pressure on degraded agricultural inputs such as water. Implicit subsidies (e.g. through low cost-recovery levels) are also widespread, although they have a negative impact on the ecological status of water bodies.	X	X	X	X	X
Water trading	There is no unambiguous definition of what should be considered a water market (Brown, 2006). In the context of water allocation reform, water trading can be defined as an institutional setting that allows temporary or permanent transfers of (marketable) water rights across agents, places and time, in exchange for pecuniary compensation.	Through active management of the institutional and regulatory framework of markets, trading can reallocate water to more productive uses that make possible both (a) compensating those economic users who end up worse-off and (b) conserving part of the water traded for the environment (in-kind tax). Payment for environmental services can be also operated through auctions in a market environment, as in water purchase tenders (buyback). This active institutional management is not feasible in informal markets, which often introduce incentives for water overuse.				X	X

(*Continued*)

Table 2. (*Continued*).

Instrument	Definition	What can the economic instrument deliver for water allocation reform objectives?	Case studies where instrument is implemented				
			FR	IT	NL	SP	EN
Insurance	Insurance is the most commonly used instrument for financial protection against risk-contingent losses, in which 'the insured party or policyholder transfers the cost of potential loss to the insurer in exchange for monetary compensation known as a premium' (DRMKC, 2017). The insurer thus absorbs, pools and diversifies the individual risks acquired from policyholders, making them assessable and manageable.	Insurance may replace the depletion of natural capital aquifers through the constitution of a financial fund that protects policyholders when a drought with predefined characteristics hits.	X	X		X	
Voluntary agreements	Non-pecuniary and voluntary incentives, based on opportunities for individual profit or loss mitigation, to enhance negotiated arrangements among agents to achieve public policy objectives. They exclude rewards, penalties and other regulated obligations.	Voluntary and participatory reallocation processes can be a more cost-effective way of complying with temporary or permanent water reallocation, and are increasingly used at a local (e.g. users' associations) and basin scale (e.g. drought steering committees). Transformational solutions such as green infrastructure can create relevant opportunities for individual profit and can be adopted voluntarily and without pecuniary exchanges where there is a Pareto improvement, i.e., where all sides end up better-off as compared to the counterfactual scenario.	X	X	X	X	X

beneficiaries of service provision compensate the providers (Gómez-Baggethun et al., 2010). The ecosystem services that relate to freshwater resources encompass 'the benefits to people that are produced by terrestrial-ecosystem effects on freshwater' (Brauman, Daily, Duarte, & Mooney, 2007, p. 66). Such benefits can be categorized into four groups: provisioning, e.g. in-stream water supply; regulating, e.g. water purification; supporting, e.g. maintenance of aquatic habitats that produce services; and cultural, e.g. recreation (Reynaud & Lanzanova, 2017). PES schemes can be government-financed (buyer as third party), private-user-financed (buyer as end-user) or utility-financed (buyer confronted with user fees or tariffs from a public or regulated private utility) (Porras, Alyward, & Dengel, 2013).

PES schemes have been widely implemented outside Europe (e.g. Asquith, Vargas, & Wunder, 2008; Brouwer, Tesfaye, Pauw, & Pepping, 2011; Kosoy, Martinez-Tuna, Muradian, & Martínez-Alier, 2007; Martín-Ortega, Ojea, & Roux, 2013; Pagiola, 2008; Postel & Thompson, 2005; Wunder & Albán, 2008) and to a growing extent within Europe; they exist in all case studies reported here. PES schemes have been promoted in the EU Biodiversity Strategy to 2020, and their potential is further highlighted in the Roadmap to a Resource Efficient Europe (EC, 2015). But the success of such schemes is highly dependent on the political, socio-cultural and institutional context in which they are implemented (Muradian et al., 2013). Key factors for their success include the following (OECD, 2010). The first is a well-defined framework. Most importantly, the goals to be achieved by implementing the PES scheme should be clearly defined and agreed to by the different participants. It is also important also to have initial property rights clearly defined, along with who is able to sell, and who would like to buy. Clear insight into the sellers and buyers, their objectives and their means of finance is a third factor that defines the success of a PES. Finally, a robust framework for monitoring and evaluation not only facilitates the endurance of a PES system but also enables PES systems to act as exemplar studies and may boost implementation in other areas.

A number of factors/threats must also be accounted for or dealt with to maximize PES financial efficiency. First, potential perverse incentives should be dealt with. Moreover, the potential of social inefficiency and lack of additionality as obstacles that PES programmes might experience, while emphasizing the importance of targeting applicants to maximize PES' financial efficiency (Engel et al., 2008; Ferraro, Pattanayak, Demmer, Starkey, & Telfer, 2006).

In England, recent studies (Defra, 2013; Reed et al., 2013) have summarized successful PES schemes relating to the supply of freshwater resources. For example, SCaMP I (north-west England), Upstream Thinking (south-west England) and Wessex Water (south-west England) were all set up to improve water quality via land management measures targeting farmers in each catchment area (Defra, 2013). In France, Vittel (Nestlé Waters) serves as a best-practice example. To improve water quality, Vittel signed contracts with other farmers to use more sustainable dairy farming techniques and to improve farm facilities (Perrot-Maître, 2006). Pure PES schemes (buyer as end-user) do not exist in the water sector in Italy (Pettenella, Vidale, Gatto, & Secco, 2012), but PES-like schemes driven by public authorities are well established, related to hydropower generation, tap-water provision and mineral water supply. PES in Italy were set up to compensate the water opportunity costs for local populations and to enhance water quantity and/or quality via changing land (here: forest) management practices (Pettenella et al., 2012). Multiple examples also exist of recently implemented PES(-like) schemes in the Netherlands targeting water quality and/or quantity (Franken, Van Der Meulen, Kwakernaak, Bos, & Lenselink, 2016; Linderhof, De Blaeij, & Polman, 2009; van der Meulen, Neubauer, Brils, & Borowski, 2012; Verburg & Seines, 2014). Visitor payback schemes are applied, for example, in Nationaal Park de Hoge Veluwe, while financial payment schemes exist between the Dutch government and various water supply utilities (WaterNet, PWN, Vitens) to support their land and water management practices in nature areas (Verburg & Seines, 2014). In Spain, significant investments are being made to put buyback schemes in place (€ 829.9 million for 2007–2027) in, for example, the Upper Júcar River basin, the Segura River basin (Garrido, Rey, & Calatrava, 2012), and the Upper Guadiana River basin (Pérez-Blanco & Gutiérrez-Martín, 2017). These aim to restore environmental flows while overcoming resistance from farmers through financial compensation and compensating for other possible negative feedbacks (Garrido et al., 2012). These examples highlight the added value of PES in identifying integrated solutions, providing new funding streams for measures that are otherwise difficult to finance, and providing broader support for management and policy by the general public. PES schemes are also perceived as more flexible, more easily applied and more cost-effective than other instruments, like command-and-control measures (Ferraro & Simpson, 2002), although ex-post evaluations are often lacking.

Non-monetary voluntary agreements

Voluntary agreements, negotiated among agents, rely on truly voluntary (i.e. excluding rewards, penalties and other regulated obligations) and non-monetary incentives (as opposed to PES) to take action or adopt practices that benefit agents individually while contributing to solving water-allocation issues (Lago et al., 2015). Voluntary agreements are a common practice in solving water allocation problems in Europe, where markets are rare and pecuniary compensation is unfeasible in most settings, following the polluter-pays principle (Lindhout & Van den Broek, 2014). In Spain, users and institutions in the drought commission of a river basin authority can negotiate and voluntarily agree on restrictions or alternative measures to apply during drought events (Carmona et al., 2017). If the board fails to achieve an ecological status in compliance with predefined minimum thresholds, then a conventional command-and-control approach is typically adopted (Bielza, Conte, Dittmann, Gallego, & Stroblmair, 2008), which usually results in less efficient allocation of the resource and higher losses. After a 12-year experiment in the Po River Basin District, Italy has recently validated the use of voluntary agreements and is now transitioning from a conventional command-and-control to a mixed approach across all its river basins. In England, during recent drought events (e.g., 2010–12), the Environment Agency and farmers agreed on voluntary restrictions on abstraction for irrigation to avoid further mandatory restrictions in the worst-hit areas of the country (Rey et al., 2017).

Voluntary agreements also work as a tool for water management at the local level, most notably among irrigators, who typically reallocate water during dry periods through collective negotiations via farmer associations. This provides additional flexibility and the option to mitigate losses as opposed to a proportional reduction in water allotments. An example is the French Single Collective Water Management Association, which defines the rules for reallocating water among its users within the maximum threshold set by the local water committees or public institutions on a common agreement. The requirement to create a

Single Collective Water Management Association to control water withdrawals for irrigation is imposed by the regulation in areas of structural deficit, but it opens the way to voluntary agreements negotiated among the farmers, as they have the freedom to decide on the allocation and management rules of the association. These associations also benefit from incentives from the local authorities in the form of reduced water charges.

Voluntary agreements can also be used for the adoption of innovative transformational solutions that create win-win opportunities, such as green infrastructure and nature-based solutions. Evidence shows that when technological, institutional and/or legal barriers fade, users have incentives to engage in agreements for their adoption, in several cases due to the relevant non-monetary benefits expected, such as better qualitative and quantitative status of water bodies (e.g. Baró et al., 2016; Demuzere et al., 2014; Mazza et al., 2011). For example, research conducted in the Lower Ebro River in Spain showed that the release of artificial pulse floods by the hydropower operator to partially restore the river regime could also mitigate the clogging of intakes for hydropower generation at a lower cost than conventional mechanical removal methods (Gómez, Pérez-Blanco, & Batalla, 2014). In England, such voluntary agreements have been observed among farmers and water utilities to limit the use of fertilizers and improve water quality, as in the agreement between Wessex Water and farmers in Dorset (Viavattene, McCarthy, Green, & Pardoe, 2015).

Subsidies

Subsidies are used to enhance the provision of positive externalities, as they help reduce production costs, shift supply downwards, reduce prices, and increase the number of goods traded and related externalities. During water shortages, over-allocation can result in a negative externality. Following economic theory, this problem is best addressed through charges that make the polluter pay. However, other factors, such as actions from relevant groups of interest or affordability issues, can lead policy makers to adopt alternative solutions (Rausser et al., 2011). This political economy of water helps explain why, while water charges are rarely used to recover environmental and resource costs, subsidies to water-reliant activities are the most widespread economic instrument for water management in Europe (European Environment Agency, 2013; OECD, 2013), despite a dubious contribution to achieving water policy objectives (EC, 2012a).

Subsidies can be explicit, through price support, subsidized loans and direct payments; or implicit, through reduced regulation and/or tax or charges relief. Implicit subsidies in Europe are typically exerted through insufficient cost recovery at different levels (water works, environmental and resource costs), with loose monitoring and enforcement of allocations. Cost recovery of resource and opportunity costs is reported to be inadequate and represents an 'environmentally-harmful subsidy' (EC, 2012a, p. 10) in all EU member states. Cost-recovery levels for water works, which in the UK and the Netherlands are close to 100%, range between 60% and 80% for households and around 50% for irrigation in Spain and Italy (EEA, 2013). Poor monitoring and enforcement also make illegal withdrawals inexpensive (these are forms of subsidy) and can aggravate water availability problems. For example, in one of the most over-exploited basins in Europe, the Segura River basin (Spain), the area of irrigated land grew by 6,500 ha/y between 1990 and 2000, a surprising figure considering that the granting of new concessions had been forbidden by law since 1986 due to persistent over-exploitation problems (WWF, 2006). Some estimates suggest that up to 40% of the irrigated area in the most water-scarce areas in the basin was informal (Pérez-Blanco & Gómez, 2013).

Explicit subsidies are also widespread in Europe, particularly in the agricultural sector (EU, 2003, 2013b). Together with tariffs to protect farmers from cheaper imports, the CAP historically relied on price supports through purchases of surplus food and minimum price thresholds (EU, 2003). The artificially high prices encouraged food surpluses, increased demand for agricultural inputs such as water, and depleted EU financial and natural resources. The cost and inefficiency of the policy eventually led to the substitution of price supports by direct payments, a process known as decoupling.

Another explicit subsidy that has played a major role in Europe and elsewhere is subsidized loans, typically used to engage farmers in irrigation modernization programmes. The largest irrigation modernization programme in Europe took place in Spain, with a total investment of € 7368 million between 2002 and 2008, of which 60.1% was via public funds and 39.9% privately funded. The objective was to save 3,662 hm^3 water annually through reduced water withdrawals (López-Gunn, Mayor, & Dumont, 2012). However, evidence clearly shows that this investment, together with other irrigation programmes, has not reduced pressures on water bodies (Berbel & Mateos, 2014; Rodríguez Díaz et al., 2012). A recent study reviewing international evidence concluded that 'reductions in water consumption by irrigated agriculture will not come from the technology itself', and that other instruments 'like limiting water allocation will be needed to ensure a sustainable level of water use' (Food and Agriculture Organization, 2017, p. v).

The Tinbergen principle states that to achieve targets for a certain number of objectives, an equal number of instruments is necessary. Of equal importance, the Tinbergen principle also stated that the design of a successful policy requires a clear differentiation between the objective, namely the variables policy makers want to affect, and the variables that the policy maker can control directly or instrument (Tinbergen, 1952). The assignment principle states that policy makers should assign each instrument to the pursuit of a specific policy target, and avoid using that instrument to pursue a second target (Mundell, 1960, 1962). The underperformance, if not failure, of subsidies in Europe seems to validate these principles. Implicit subsidies and subsidized loans for irrigation modernization pursue the double objective of reducing water use while guaranteeing a stable income to farmers (one instrument, two targets), but evidence shows they succeed only in the latter – at a higher cost than available alternatives. The transition from price supports to direct payments tackles the problem of income stability in a more effective way (income instability is not only explained by prices, but also production volatility), while avoiding negative feedbacks on the environment. Ultimately, restoring the balance in over-exploited basins demands instruments specifically designed to reduce use; these can be complemented with others that then address distributive issues and compensate those who lose out (Young, 2014).

Water trading

A water market can be defined as 'an institutional framework which allows water right holders, under certain established rules, to transfer their rights to other economic agents or water users, receiving an economic compensation in exchange' (Sumpsi, Garrido, Blanco, Varela, & Iglesias, 1998, p. 73). In water-scarce areas such as Australia, California, Spain and Chile, water markets have been used to buy back water for the environment, to increase reliability of urban water supply, and to reallocate water from low-value to high-value crops (Garrido & Gomez-Ramos, 2009; Grafton, Landry, Libecap, McGlennon, & Brien, 2010; Hanak & Jezdimirovic, 2016; Palomo-Hierro, Gómez-Limón, & Riesgo, 2015; Wheeler et al., 2014). With the expected effects of climate change on water supplies and demands looming, some authors have referred to water markets as a cost-efficient adaptation mechanism in Europe (Escriva-Bou, Pulido-Velazquez, & Pulido-Velazquez, 2017). However, water markets can also generate significant damage to local economies that has to be accounted for and mitigated if necessary (Doherty & Smith, 2012). Therefore, clarity in water rights and a sound regulatory framework are essential for a successful water market (Wheeler, Loch, Crase, Young, & Grafton, 2017).

The five case studies presented here illustrate a variety of contrasting situations. In Spain, water markets have been fully developed, although they are spatially and temporarily limited and conditioned to a formal enactment by royal decree. In England, trading is possible but not widespread, although there is increasing interest, given growing pressures on water bodies. In France and Italy, trading is not allowed, and legal and political reasoning have prevented its implementation. In the Netherlands, trading is not currently permitted, and its implementation has not been recently discussed.

Water trading was the centrepiece of the 1999 reform of the Water Act in Spain (Boletín Oficial del Estado, 1999). The reform included two ways to exchange public water use rights: right-holders that voluntarily agree on specific terms of trade and jointly file a request to lease out for a number of years the water to which right-holders are entitled; and water banks (or water exchange centres, as they are called in the 1999 Water Act), a clearinghouse for buyers and sellers (Garrido et al., 2012). Prior to this reform only private rights could be formally traded; water flows pumped from private wells could be leased, auctioned or sold (Rey, 2014). Following the 1999 reform, several water rights exchanges have taken place in the Spanish territory, involving different water users, water sources and basins. However, the overall performance of water markets in Spain has been below expectations (Garrido et al., 2012). Water markets have been operative only during drought periods, and even then, trading activity accounted for less than 5% of total water use (Palomo-Hierro et al., 2015). Various barriers to trade have been identified in Spain, including lack of information; high transaction costs; temporal limitations of trading allowances; lack of certainty and property right protection in the definition of water rights; and a lack of clarity in the conditions under which exchanges involving more than one region could be made (Calatrava & Martínez-Granados, 2017; Palomo-Hierro et al., 2015; Rey, 2014; Wheeler et al., 2017). Informal water markets also exist in Spain, particularly in areas of intense water scarcity and high-economic-value water uses, like the Mediterranean south-eastern coast. They have been developed mostly at the local scale through a variety of institutional arrangements, which do not always clearly align with current legislation (Hernández-Mora & Del Moral, 2015).

In England, the transfer of rights between licence holders is possible but not straightforward. Short-term exchanges are de facto unfeasible under standard procedures due to the time required for approval. Several barriers to trade have been identified (Environment Agency & Ofwat, 2008), including lack of a visible market; an inability to see the value added in trading; lack of understanding of the process; hoarding for future uncertainty; the existence of alternatives to trading; the feasibility of making suitable trades; the restrictions placed on trading and the reduction of rights at the point of trade; the difficulty and complexity of the trading process; and broader barriers which prevent the market from developing, for example obstacles to upstream competition, such as the current access price. Since then, other studies have been undertaken to inform abstraction licensing reforms, and to improve basin interconnections, in order to enhance water trading (Cave, 2009; Defra, 2011; House of Commons, 2012).

In Italy and France, water trading faces significant opposition, substantiated through legal and political barriers. In Italy, a referendum in 2011 rejected a legislative decree (3 April 2006, n. 152, Gazzetta Ufficiale della Repubblica Italia, 2006), which would have introduced the option to 'adequately' remunerate capital investments by water utilities and managers. Its revocation complicates the introduction of marketable permits in the Italian context. In France, the Social and Economic Council concluded in 1991 that water markets are not desirable in France as they conflict with the understanding of water as more than a purely economic good. Since then, the establishment of tradable quotas has been investigated and evaluated in different contexts and through different formats to highlight its potential benefits in France (Rinaudo, Montginoul, Varanda, & Bento, 2012; Strosser & Montginoul, 2001). However, it has not been implemented until now, mainly due to farmers' rejection and political reluctance (Figureau, Montginoul, & Rinaudo, 2015; Montginoul & Rinaudo, 2009). Finally, cultural, social, legal, institutional and physical infrastructure barriers have collectively limited the implementation of water trading mechanisms. Designing efficient market institutions to replace traditional water allocation rules is understandably a challenging exercise (Garrido, 2007).

Insurance

Water stocks underground are a de facto insurance against droughts in the absence of financial arrangements. This is the case in many agricultural basins in the Mediterranean area, where water deficit during drought periods is (partially) covered through (illegal) over-abstraction, mostly from dependable aquifers that are difficult to monitor (Perez-Blanco & Gomez, 2014). Although crop insurance might not be an economic instrument for water management per se, it could play a key role in addressing the incentives towards aquifer overdraft during dry periods. The EU has adopted mostly classic agricultural insurance schemes, with commercial insurance usually supplied by private insurers, except in Greece and Cyprus, where insurance is supplied by the public sector (Bielza et al., 2008). Drought insurance coverage in Europe has traditionally been limited. Until recently, only insurance schemes in Spain, Italy, France and Austria have compensated drought-related losses in rainfed agriculture. Emerging mutual funds that hedge policyholders against income (instead of yield) losses, and thus provide drought coverage also in irrigated agriculture, have been subsidized by the EU CAP European Agricultural Fund for Rural Development since 2014. The fund offers compensation for up to 65% of the indemnities paid, provided that (1) the indemnities compensate for less than 70% of the foregone income, and (2) the income drop is above 30% of a three-year average based on the preceding three years or the five-year period excluding the highest and lowest entry annual income (EU, 2013c). Mutual funds are also eligible for EU financial contributions related to the administrative costs of setting up the mutual fund, and to financial contributions to premiums for crop insurance against several risks (including adverse climatic events or outbreak of a disease or pest infestation or an environmental incident which destroys more than 30% of the average annual production of a farmer in a three-year period).

Spain and Italy have taken relevant steps to insure irrigated agriculture against drought losses. In Spain, an extensive and heavily subsidized insurance programme hedges farmers against the main natural risks, e.g., hail, frost and flood (Antón & Kimura, 2011).The programme is subsidized by the Ministry of Agriculture and (to a lesser extent) by regional governments. Costs are subsidized by up to 45%, depending on the type of commodity and the terms of the contract (Meuwissen, Huirne, & Skees, 2003). The implementation of drought insurance policies in irrigated agriculture through traditional yield insurance (Perez-Blanco & Gomez, 2014), index-based insurance (Maestro, Bielza, & Garrido, 2016; Ruiz, Bielza, Garrido, & Iglesias, 2015), and income insurance (Pérez-Blanco, Delacámara, & Gómez, 2015) has been explored by a series of studies commissioned by the pool of agricultural insurance firms and the Ministry of Agriculture over the last decade, but it still represents a challenge for companies and institutions because of the complexity of its design and implementation (Maestro et al., 2016; Ruiz et al., 2015).

Conversely, Italy has pioneered the development and support of mutual funds in Europe: Decree 102/2004 (Gazzetta Ufficiale della Repubblica Italia, 2004) and Law 388/2000 (Gazzetta Ufficiale della Repubblica Italia, 2000) established that mutual funds constituted by the so-called Comisioni di Difesa (Defense Commission) were eligible for public funding. As a result, mutual funds emerged in Italy even without public support in different agricultural sub-sectors. Although they did not receive final approval from the European Commission, the new CAP 2014-2020 is likely to relaunch mutual funds and increase their (still marginal) relevance.

Policy recommendations: enablers and barriers for implementation, and transferability

As highlighted above, it is critical to carefully study the multiple aspects and dimensions relating to the implementation of economic instruments as a tool for water allocation reform. First, it is necessary to recognize the strengths of each economic instrument and its potential to contribute to the reform objectives. Second, it is critical to identify the barriers that might inhibit the success of the instrument. Third, it is essential to consider all the unintended consequences that economic instruments can create in the water-allocation regime. The comparative analysis of the strengths, barriers and unintended consequences of existing economic instruments within different socio-economic, legal, institutional and biophysical contexts in the selected case-study areas helps elicit underlying common guidelines that can determine success. Table 3 highlights the main strengths, barriers and unintended consequences based on a comprehensive review of European experiences, together with specific recommendations.

Drawing from our analysis and Table 3, we formulate below a series of recommendations to inform the implementation of economic instruments in water allocation reform in Europe, and elsewhere.

Regulatory framework

Water rights are the heart of any allocation system, and essential for successful reallocation (Meinzen-Dick & Bakker, 2000; Pigram, 1999). As long as the resource is plentiful, there is little pressure to define or enforce water rights. When water becomes scarcer and competition for it increases, as is happening in most countries around Europe (see section on water resource pressures), property rights can clarify expectations, limit uncertainty and reduce conflicts (Bruns & Meinzen-Dick, 2005). As summarized in Table 1, each case study has a different allocation system designed for its specific context. A major problem is that, currently, around the world water is often allocated based on institutions established when water was not considered a scarce resource (Frederick, 2001). There is a need for well-defined property rights such that they are completely specified, monitored, enforceable, transferable and legally secured (Saliba & Bush, 1987). In order to do so, we need to ensure the following.

Rights should account for hydrological variability and dynamics, and be defined as shares of harvestable resources to ensure flexibility and legal security. Rights should be 'consistent with the way water is stored, and how it flows through landscapes', accounting for return (and escape) flows, connectivity between water bodies, and environmental uses (Young, 2014, p. 34). Rights need to be defined as shares of harvestable resources, rather than in absolute terms, as is the case in Europe now (OECD, 2015a). In this context, risk assignment and management is also critical: if users are assigned 100% of water supply risk (no compensation is provided in the event of a drought), planning is likely to lead to dynamically efficient outcomes. On the other hand, full risk assignment may be undesirable, if not unfeasible, in the EU context, which has solidarity among its essential values (EU, 2012).

Water entitlements should be separated from land. The Tinbergen and assignment principles imply that an accurate and successful policy design necessitates one instrument for the pursuit of each objective. A prerequisite for correct policy design is thus de jure differentiation between land and water rights so that agricultural and water policy objectives are achieved through ad hoc instruments designed in an efficient and effective way. A clear example of this problem is the reacquisition of water rights in the Guadiana River basin in Spain. Since the right to use water in Spain is bundled with the land, policy makers addressed this shortcoming by targeting the land market and acquiring land rights instead of water rights (Garrido et al., 2012). Of the water rights bought, 85% were 'paper rights' not exercised by users, which consequently did not contribute to solving the over-allocation problem. Despite some other issues (see below), the Australian buyback scheme successfully unbundled water from land and managed to develop a water market to address overcompensation in agricultural water buyback more effectively (Iftekhar, Tisdell, & Connor, 2013).

Thresholds for water licenses should refer to consumption, not only withdrawals. Following efficiency improvements, any potential 'real water saving' that enhances environmental flows (Perry, 2011) can be offset by the incentive to increase income through higher water consumption (i.e. the water fraction that evapotranspires), even if withdrawals decrease (Gómez & Pérez-Blanco, 2014; Huffaker, 2008). Evidence on irrigation modernization programmes from countries like the United States (Pfeiffer & Lin, 2014; Scheierling, Young, & Cardon, 2006), Australia (Adamson & Loch, 2014; Grafton, 2017), China (Kendy, Molden, Steenhuis, & Liu, 2003), Tanzania (Lankford, 2004), Pakistan (Ahmad, Turral, Masih, Giordano, & Masood, 2007), and Spain and other Mediterranean countries (Food and Agriculture Organization, 2017), among others, shows that this apparently paradoxical outcome is the norm unless restrictions on consumptive use are established. Adequate monitoring and return-flow accounting, complemented with a sensible regulatory framework that limits consumptive use, is necessary to prevent depletion of water bodies. This holds for almost every situation with the exception of escape-flow regimes (i.e. water flows to a sink; Huffaker, Whittlesey, & Huffaker, 2003).

Table 3. Strengths, barriers, unintended consequences and recommendations for economic instruments for water management, based on the review of case studies.

	Strengths	Barriers	Unintended consequences	Suggested steps
Water charges	• Effectiveness • Enforcement of the polluter-pays principle	• Resistance from groups of interest and related transaction costs • Willingness and ability to pay	• Income losses in agriculture • Economy-wide impacts • Redistributive impacts	• Strengthen institutions • Enforce regulation and metering • Complement with decoupled subsidies to compensate users that end up worse-off
Payment for environmental services	• Limited transaction costs • Flexible and cost-effective	• Infringement of the polluter-pays principle • Willingness to pay among potential buyers (fairness) • Ability to pay, including budgetary constraints	• Crowding-out of intrinsic motivations to protect ecosystems • Equity issues	• Robust monitoring and reporting • Performance-based payments • Ensure adequate enforcement and permanence
Subsidies	• Redistribution towards equity • Limited transaction costs	• Infringement of the polluter-pays principle • Budgetary constraints	• Dichotomic design may increase withdrawals • Low effectiveness and cost-effectiveness	• Decouple from water use • Use explicit subsidies • Complement with effective instruments in constraining use (e.g. charges)
Water trading	• Allows reallocation of water resources • Enables buyback • Improves economic efficiency	• High transaction costs • Institutional/legal complexity • Acceptability	• Asymmetric impacts on local economies (e.g. food industry) • Impacts on downstream users • Can increase water use and consumption (e.g. due to technical efficiency gaps or sleeping licences) • Illegal, uncontrolled markets	• Inform an objective public debate • Enhance legal security, also for the environment • Streamline legal permitting of trading • Create online information on marketing opportunities • Permit only to sell consumptive fraction
Insurance	• Limits withdrawals that are difficult to monitor • Privately funded	• Willingness and ability to pay • Often subsidized: budgetary constraints, infringement of polluter-pays principle	• Over- and under-subsidization • Equity issues	• Assess willingness to pay and risk • Target subsidies • Progressively scale up successful pilot studies
Voluntary agreements	• Flexible • Acceptable • Inexpensive	• Low performance if incentives not properly defined • Limited to win-win situations • Technological, institutional and/or legal barriers	• Can delay action during droughts • Exclusion of some users	• Expand evidence base • Build institutional and legal security • Public monitoring • Clear, predefined rules

Legal uncertainty on the conflict between the polluter-pays principle and the beneficiary-pays principle deserves careful consideration. Although the polluter-pays principle pervades environmental legislation, differing interpretations of the legal *acquis* have resulted in a benefit-based approach. Examples of this interpretation plague water realpolitik in Europe and elsewhere. For example, multi-million-dollar investments have been recently committed to water buyback in areas such as Australia's Murray-Darling Basin (AUD 3.1 billion for 2009–24), south-east Spain (€ 829.9 million for 2007–27) and in the US, notably California (USD 547 million during 1987–2011, 55% of which after 2003) (Department of Sustainability, Environment, Water, Population and Communities, 2016). The fairness of the beneficiary-pays approach is questionable, despite its pragmatism, and EU institutions seem committed to advancing towards a full-fledged polluter-pays principle in Europe (Lindhout & Van den Broek, 2014). A recent example involves a ruling by the National Court of Spain (Tribunal Supremo) against the Jucar River Basin Management Plan because it asked municipalities to pay for the cost of changing their water supply source from ground to surface water, which was necessitated by farmers polluting the groundwater source with nitrates and pesticides (Boletín Oficial del Estado, 2017).

Development of a public register to enhance transparency. Its absence is often cited as one of the causes of water resource over-allocation (Bruns & Meinzen-Dick, 2005). In the Po River Basin District in Italy, the amount of resources allocated through water rights is often loosely defined (e.g. average withdrawals) or not specified at all, and metering devices are often not in place. The absence of accessible registers complicates the implementation of environmental impact assessments for new licences and prevents prioritization of uses according to their full economic value. This has resulted in a de jure over-allocation, which does not translate into de facto scarcity due to the large number of sleeping licences (Santato et al., 2016), although most recently increasingly intense and recurrent droughts have made this problem apparent (Castellari et al., 2014). In Australia, those basins where water trading activity has not developed significantly are the ones where public registers, trading platforms and market information are also less developed (Defra, 2012), suggesting that revealing the full economic value of water may underpin the development of registers.

Integration of economic instruments into broader water policies

Economic instruments are a means to an end. Any private gain through the deployment of economic instruments should be accompanied by societal gains through the achievement of water policy objectives (incentive compatibility). This is illustrated for instance in the implementation of PES schemes in the Netherlands, UK, France, and Italy (see review of PES in previous section) and outside Europe (e.g., Martín-Ortega et al., 2013), where farmers have been compensated for reducing or avoiding fertilizer application and guaranteeing water quality in specific areas based on the avoided cost for water utilities and expected societal environmental benefits.

Economic instruments cannot be designed in isolation but only in combination with other tools and policies. They must be enhanced and promoted through complementary measures that guarantee their success and avoid externalities, relying on the Tinbergen and assignment principles. The Australian reform is often cited as an example of international best practices in this regard (OECD, 2015a), and recent guidelines for water policy reform in the UK and western US states took stock from them (Young, 2012, 2015).

Policy sequencing should be carefully designed to enhance cost-effectiveness. Several studies of buyback programmes in the US, Spain and Australia suggest that the cost of the policy is likely to increase where water markets among private users are in place and the full economic value of water has been revealed, thus hampering cost-effectiveness (Hanak & Stryjewski, 2012; Iftekhar et al., 2013; Pérez-Blanco & Gutiérrez-Martín, 2017). An example of cost-effective combination and sequencing of water policies is given by voluntary agreements (see review of this instrument in previous section) established among stakeholders of drought commissions in Spain, which allow for cost-effective agreements prior to the implementation of a command-and-control approach; or in the case of the French water users' association, where water abstractions reforms set up the conditions for the implementation of economic instruments among water users, but leave them the autonomy to decide on the type of instruments they want to implement within their association to comply with targets while minimising losses.

Avoidance of undesirable outcomes

Economic instruments might create undesired externalities. To avoid these undesirable outcomes, all the potential socio-economic and environmental effects have to be properly evaluated and, if possible, avoided. In cases where third-party effects are identified but unavoidable, mitigation strategies are necessary to reallocate policy benefits and/or to avoid inequitable outcomes. Table 3 summarizes the main negative externalities of economic instruments in the case studies. These unintended consequences are specific to each context and instrument, although some general recommendations can be drawn.

Economic instruments, and water policy overall, should be carefully designed to avoid unintended socio-economic (notably redistributive) and environmental consequences. In places like California and Spain, water reallocation and more specifically trading are subject to the primary restriction of the 'no-injury rule': a water transfer must not result in an injury to other legal uses of water. Typically this means that water right holders may transfer only the amount of water that results from a reduction in the consumptive use of their water right (Escriva-bou, Mccann, Hanak, Lund, & Jezdimirovic, 2016). Socio-economic consequences might arise from water transfers if there is an economic activity linked to the water traded. These effects need to be properly assessed. The substantial search and information, bargaining and policy enforcement costs involved in the process have led the Spanish administration to apply the rule of 'positive silence' (Williamson, 1998), meaning that if no response is provided in a given period of time, the reallocation can take place – a major loophole which traders have often resorted to, as in the Tagus-Segura inter-basin trading scheme (Delacámara, Pérez-Blanco, Ibáñez, & Gómez, 2015b).

Where third-party effects are properly identified but unavoidable, mitigation strategies are necessary to reallocate policy benefits and/or to avoid inequitable outcomes. Where the policy benefits are greater than the unavoidable damages, or inequitable outcomes result from the reallocation of water, monetary (or other type of) compensation should be established. As an example, the Metropolitan Water District and the Palo Verde Irrigation District, both in Southern California, established a USD 6 million local development fund to mitigate the impact of the water transfer on the Palo Verde Valley, after signing a 35-year agreement to fallow annually between 2,400 and 107,000 hectares of Palo Verde Irrigation District land, depending on Metropolitan Water District needs (Doherty & Smith, 2012). In the Upper

Guadiana in Spain, water reallocation from agriculture to the environment through the buy-back programme (€829.9 million) was complemented with €2.1 billion for diverse flanking measures to subsidize economic diversification and new transportation, communication and energy infrastructure that mitigated negative spillover on the local economy (Confederación Hidrográfica del Guadiana, 2008).

Institutional capability

The implementation of economic instruments demands knowledge of the costs and benefits of economic instruments for users (including the environment) and other affected agents locally and in the wider economy; but also of the laws, the customs and the political power balance that define transaction costs and the range of actions that institutions can realistically implement. To this end, the total cost of implementing economic instruments should be assessed by aggregating the net costs to users and third-party effects, and institutional transaction costs (Marshall, 2013). Yet, although some recent studies have measured the transaction costs of water policy reform in the US, Australia and South Africa, there is still no empirical base on these costs in Europe (Garrick, McCann, & Pannell, 2013).

Stakeholder engagement and transparency

The declaration of the Dublin Principles in 1992 at the International Conference on Water and the Environment clearly stated the important role of participatory approaches in water management. According to Integrated Water Resources Management principles, water development and management should be based on a participatory approach, involving users, planners and policy makers at all levels (Solanes, 1998). Early engagement and transparent and accountable decision making are key to acceptability (Delacámara et al., 2014; Organisation for Economic Co-operation and Development, 2015b). For instance, in England, abstraction reform is being designed through consultation with stakeholders (Defra, 2016a).

Stakeholder engagement in the design and use of economic instruments to achieve the policy goal is essential. Economic instruments are often misunderstood and hindered by misconceptions. Lack of understanding of the aims, features and outcomes of economic instruments could hinder their success. This has been the case for water trading in many parts of the world, where water users are hesitant to participate in the market due to a lack of understanding of the system and the fear that their water licence must be reviewed by the water authority if they are selling the water to someone else (Garrido et al., 2012).

Transparency, including the provision of useful and timely data and information, can help eliminate barriers. Apart from reducing over-abstraction and over-allocation, a central register of water use and the encouragement of metering will help reduce stakeholders' reluctance towards economic instruments.

Conclusions

The water scarcity- and drought-related challenges we face today should be interpreted as a problem of governance, which is being and will be further aggravated by climate change. Sensible and conscientious concerted action is necessary to achieve the greatest collective good through economic instruments for water management. Economic instruments are gaining momentum in European countries with different water-scarcity levels, allocation systems and regulations, revealing common and site-specific challenges that need to be addressed to ensure a meaningful contribution to water policy objectives and a successful implementation.

This study first reviewed the performance of economic instruments in water allocation reform against original design objectives in five European countries: England, France, Italy, Spain and the Netherlands. It then identified the strengths of, barriers to and unintended consequences of economic instruments within the varying socio-economic, legal, institutional and biophysical contexts in the case-study areas, and drew out underlying common guidelines and general recommendations.

These guidelines and recommendations respond to our assessment of European case studies, but also consider experiences in other parts of the world (especially California and Australia). These guidelines and recommendations offer valuable insights to inform the mainstreaming of economic instruments in water allocation reform processes elsewhere in Europe, and also in other countries or international river basins with similar characteristics. Our study reveals that there is no 'fit-for-all' solution to water-allocation challenges, and every economic instrument for water management has strengths and limitations that have to be accounted for when redesigning allocation systems. Furthermore, economic instruments are not panaceas to address the challenges at hand; quite the contrary, they should be designed considering the particularities of the study area and in concert with regulatory and/or engineering solutions where these can enhance their impact or outperform them. In the process of designing economic instruments, it is also important to acknowledge the bounded rationality of individuals: particularly in the case of water, policy decisions are made in a context of (deep) uncertainty and contingent on the institutional infrastructure at hand. This makes the involvement of relevant stakeholders of paramount importance, to ensure that policy goals and means are the result of consensual agreement in the society and coordinated with other policy spheres.

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References

Adamson, D., & Loch, A. (2014). Possible negative feedbacks from 'gold-plating' irrigation infrastructure. *Agricultural Water Management, 145*, 134–144. doi:10.1016/j.agwat.2013.09.022

Ahmad, M-D., Turral, H., Masih, I., Giordano, M., & Masood, Z. (2007). *Water saving technologies: Myths and realities revealed in Pakistan rice-wheat systems* (Reports No. H039765). Colombo, Sri Lanka: IWMI Research.

Antón, J., & Kimura, S. (2011). *Risk management in agriculture in Spain* (OECD Food Agriculture and Fisheries Papers No. 43). Paris, France: OECD Publishing.

Armsworth, P. R., Chan, K. M. A., Daily, G. C., Ehrlich, P. R., Kremen, C., Ricketts, T. H., & Sanjayan, M. A. (2007). Ecosystem-service science and the way forward for conservation. *Conservation Biology, 21*, 1383–1384. doi:10.1111/j.1523-1739.2007.00821.x

Asquith, N. M., Vargas, M. T., & Wunder, S. (2008). Selling two environmental services: In-kind payments for bird habitat and watershed protection in Los Negros, Bolivia. *Ecological Economics, 65*, 675–684. doi:10.1016/j.ecolecon.2007.12.014

Baró, F., Bugter, R., Gómez-Baggethun, E., Hauck, J., Kopperoinen, L., Liquete, C., & Potschin, M. B. (2016). Green infrastructure. In M. Potschin & K. Jax (Eds.), *OpenNESS ecosystem services reference book* (pp. 1-6). EC FP7 Grant Agreement no. 308428. Retrieved from www.openness-project.eu/library/reference-book

Berbel, J., Gómez-Limon, C., & Gutiérrez-Martín, J. A. (2017). Moderninzacion de regadios y ahorro de agua [Irrigation modernization and water savings]. In J. Berbel & C. Gutiérrez-Martín (Eds.), *Efectos de la modernizacion de regadios en Espana* (pp. 185–220). Almería, Spain: Cajamar.

Berbel, J., & Mateos, L. (2014). Does investment in irrigation technology necessarily generate rebound effects? A simulation analysis based on an agro-economic model. *Agricultural Systems, 128*, 25–34. doi:10.1016/j.agsy.2014.04.002

Bielza, M., Conte, C. G. C., Dittmann, C., Gallego, J., & Stroblmair, J. (2008). *Agricultural Insurance Schemes*. Ispra, Italy: European Commission.

BOE. (2017). Sentencia de 23 de marzo de 2017, de la Sala Tercera del Tribunal Supremo, que estima en parte el recurso contencioso-administrativo interpuesto contra el Plan Hidrológico de la Demarcación Hidrográfica del Júcar aprobado por el Real Decreto 595/2014, de 2017 [Sentence of the 23rd of March 2017, Room 3 of the Supreme Court, on the Jucar Hydrological Plan, approved by the Royal Decree 595/2014]. Tribunal Supremo.

Boletín Oficial del Estado. (1999). *Ley 46/1999, de 13 de diciembre, de modificación de la Ley 29/1985, de 2 de agosto, de Aguas* [Law 46/1999 of December 13th, for the modification of the Water Law 29/1985 of August 2nd]. Madrid, Spain: Jefatura del Estado.

Brauman, K. A., Daily, G. C., Duarte, T. K., & Mooney, H. A. (2007). The nature and value of ecosystem services: An overview highlighting hydrologic services. *Annual Review of Environment and Resources, 32*, 67–98. doi:10.1146/annurev.energy.32.031306.102758

Brouwer, R., Tesfaye, A., Pauw, P., & Pepping, G. C. (2011). Meta-analysis of institutional-economic factors explaining the environmental performance of payments for watershed services. *Environmental Conservation, 38*, 380–392. doi:10.1017/S0376892911000543

Brown, T. C. (2006). Trends in water market activity and price in the western United States. *Water Resources Research*, 42, W09402.

Bruns, B. R., & Meinzen-Dick, R. (2005). Frameworks for water rights: An overview of institutional options. In B. R. Bruns, C. Ringler, & R. Meinzen-Dick (Eds.), *Water rights reform: Lessons for institutional design* (pp. 3–25). Washington D.C., MD: International Food Policy Research Institute.

Calatrava, J., & Martínez-Granados, D. (2017). The limited success of formal water markets in the Segura river basin, Spain. *International Journal of Water Resources Development, 46*, 1–21. doi:10.1080/07900627.2017.1378628

Carmona, M., Manez, M., Andreu, J., Pulido-Velazquez, M., Haro-Monteagudo, D., Lopez-Nicolas, A., & Cremades, R. (2017). Assessing the effectiveness of multi-sector partnerships to manage droughts: The case of the Jucar river basin. *Earth's Future, 5*(7), 750–770. doi:10.1002/eft2.2017.5.issue-7

Castellari, S., Venturini, S., Giordano, F., Ballarin Denti, A., Bigano, A., Bindi, M., … Guzzetti, M. (2014). *Elementi per una Strategia Nazionale di Adattamento ai Cambiamenti Climatici* [Elements for a national climate change adaptation strategy]. Rome, Italy: Ministero dell'Ambiente e della Tutela del Territorio e del Mare.

Cave, M. (2009). *Independent review of competition and innovation in water markets: Final report*. London: Defra.

Chong, H., & Sunding, D. (2006). Water markets and trading. *Annual Review of Environment and Resources, 31*, 239–264. doi:10.1146/annurev.energy.31.020105.100323

Colby, B., & Bush, D. B. (1987). *Water markets in theory and practice: Market transfers, water values, and public policy*. Boulder, CO: Westview Press.
Confederación Hidrográfica del Guadiana. (2008). *Plan especial del Alto Gaudiana* [Alto Guadiana special plan]. Badajoz, Spain: Confederación Hidrográfica del Guadiana.
Connor, J. D., & Kaczan, D. (2013). Principles for economically efficient and environmentally sustainable water markets: The Australian experience. In K. Schwabe, J. Albiac, J. D. Connor, R. M. Hassan, & L. Meza González (Eds.), *Drought in arid and semi-arid regions: A multi-disciplinary and cross-country perspective* (pp. 357–374). Dordrecht, The Netherlands: Springer. doi:10.1007/978-94-007-6636-5
Dai, A. (2013). Increasing drought under global warming in observations and models. *Nature Climate Change, 3*(1), 52–58. doi:10.1038/nclimate1633
Delacámara, G., Dworak, T., Gómez, C. M., Lago, M., Maziotis, A., Rouillard, J., & Strosser, P. (2014). *Guidance on the design and development of economic policy instruments in European water policy* (Deliverable 5.3). Madrid, Spain: EPI-Water - Evaluating Economic Policy Instruments for Sustainable Water Management in Europe.
Delacámara, G., Gómez, C. M., & Maestu, J. (2015a). Water trading opportunities and challenges in Europe. In K. Burnett, R. Howitt, J. Roumasset, & C. Wada (Eds.), *Routledge handbook of water economics and institutions* (pp. 281–294). London: Routledge.
Delacámara, G., Pérez-Blanco, C. D., Ibáñez, E., & Gómez, C. M. (2015b). Water trading in the Tagus river basin (Spain). In M. Lago, J. Mysiak, C. M. Gómez, G. Delacámara, & A. Maziotis (Eds.), *Use of economic instruments in water policy: Insights from international experience* (pp. 249–263). Heidelberg, Germany: Springer. doi:10.1007/978-3-319-18287-2
Deltacommissaris. (2012). *Deltaprogramma 2013: Werk aan de delta. - De weg naar deltabeslissingen* [2013 Delta Programme: Work on the Delta - The road to Delta decisions]. The Netherlands: Deltacommissaris.
Demuzere, M., Orru, K., Heidrich, O., Olazabal, E., Geneletti, D., Orru, H., … Faehnle, M. (2014). Mitigating and adapting to climate change: Multi-functional and multi-scale assessment of green urban infrastructure. *Journal of Environmental Management, 146*, 107–115. doi:10.1016/j.jenvman.2014.07.025
Department for Environment, Food and Rural Affairs. (2011). *Water for life*. London: Defra.
Department for Environment, Food and Rural Affairs. (2012). *Review of international abstraction regulation*. London: Defra.
Department for Environment, Food and Rural Affairs. (2013). *Payments for ecosystem services: A best practice guide* (p. 85). London: Defra.
Department for Environment, Food and Rural Affairs. (2016a). *UK government reponse to consultation on reforming the water abstraction management system*. London: Defra.
Department for Environment, Food and Rural Affairs [Defra]. (2016b). *Water abstraction management reform in England What would reform mean for abstractors?* London, UK: Defra.
Department of Sustainability, Environment, Water, Population and Communities. (2016). *Progress of water recovery under the restoring the balance in the Murray-Darling basin program*. Canberra, Australia: DSEWPAC.
Doherty, T., & Smith, R. (2012). *Water transfers in the West: Projects, trends, and leading practices in voluntary water trading*. Denver, CO: The Western Governors' Association and Western States Water Council.
Engel, S., Pagiola, S., & Wunder, S. (2008). Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological Economics, 65*, 663–674. doi:10.1016/j.ecolecon.2008.03.011
Environment Agency. (2008). *Water resources in England and Wales - Current state and future pressures*. London: Environment Agency.
Environment Agency. (2013). *Managing water abstraction*. London, UK: Environment Agency.
Environment Agency & Ofwat. (2008). *Exploring views on the potential for more active water rights trading*. London: Environment Agency & Ofwat.
Escriva-bou, A., Mccann, H., Hanak, E., Lund, J., & Jezdimirovic, J. (2016). *Accounting for California's water*. San Francisco, CA: Public Policy Institute California.

Escriva-Bou, A., Pulido-Velazquez, M., & Pulido-Velazquez, D. (2017). Economic value of climate change adaptation strategies for water management in Spain's Jucar basin. *Journal of Water Resources Planning and Management, 143*(5), 04017005. doi:10.1061/(ASCE)WR.1943-5452.0000735
Estrela, T., & Vargas, E. (2012). Drought management plans in the European Union. The case of Spain. *Water Resources Management, 26*, 1537–1553. doi:10.1007/s11269-011-9971-2
European Commission. (2000). *Water framework directive 2000/60/EC, council directive*. Brussels, Belgium: European Commission.
European Commission. (2003). *Economics and the environment. The implementation challenge of the water framework directive (guideline document no. 1), common implementation strategy for the water framework directive (2000/60/EC)*. Brussels, Belgium: European Commission.
European Commission. (2005). *Council regulation (EC) No 1698/2005 of 20 September 2005 on support for rural development by the European Agricultural Fund for Rural Development (EAFRD), regulation*. Brussels, Belgium: European Commission.
European Commission. (2007). *Communication from the commission to the European parliament and the council - Addressing the challenge of water scarcity and droughts in the European Union* (Communication No. COM(2007) 0414 final). Brussels, Belgium: European Commission.
European Commission. (2011). *Report from the commission to the European parliament and the council. Third follow up report to the communication on water scarcity and droughts in the European Union* (Communication No. COM(2011) 0133 final). Brussels, Belgium: European Commission.
European Commission. (2012a). *EU blueprint to Safeguard Europe's water Resources* (Communication No. COM(2012) 673 final). Brussels, Belgium: European Commission.
European Commission. (2012b). *Report from the commission to the European parliament and the council on the implementation of the water framework directive (2000/60/EC) river basin management plans* (Communication No. COM(2012) 0670 final). Brussels, Belgium: European Commission.
European Commission. (2015). *Ecosystem services and the environment* (In-depth Report 11 produced for the European Commission, DG Environment). Brussels, Belgium: European Commission.
European Commission. (2017). *Multiannual financial framework 2014–2020 and the financing of the CAP*. Brussels, Belgium: European Commission.
European Environment Agency. (2009). *Water resources across Europe – Confronting water scarcity and drought* (EEA Report No 2/2009). Copenhagen, Denmark: European Environment Agency.
European Environment Agency. (2012). *Climate change, impacts and vulnerability in Europe 2012: An indicator-based report*. Copenhagen, Denmark: European Environment Agency.
European Environment Agency. (2013). *Assessment of cost recovery through water pricing* (Report No. 16/2013). Copenhagen, Denmark: European Environment Agency.
European Environment Agency. (2016). *Water exploitation index*. Copenhagen, Denmark: European Environment Agency.
European Environment Agency. (2017). *Water management in Europe : Price and non-price approaches to water conservation*. Copenhagen, Denmark: European Environment Agency.
European Union. (2003). Council regulation (EC) No 1782/2003. Official Journal of the European Union, L 270.
European Union. (2012). Consolidated version of the Treat on European Union.
European Union. (2013a). Regulation (EU) No 1305/2013 of the European Parliament and of the Council of 17 December 2013 on support for rural development by the European Agricultural Fund for Rural Development (EAFRD) and repealing Council Regulation (EC) No 1698/2005 . Official Journal of the European Union - Regulation.
European Union. (2013b). Regulation (EU) No 1307/2013 of the European Parliament and of the Council of 17 December 2013 establishing rules for direct payments to farmers under support schemes within the framework of the common agricultural policy and repealing Council Regulation. Official Journal of the European Union - Regulation.
European Union. (2013c). Regulation (EU) No 1305/2013 of the European Parliament and of the Council of 17 December 2013 on support for rural development by the European Agricultural Fund for Rural Development (EAFRD) and repealing Council Regulation (EC), Pub. L. No. No 1698/2005 (2005).
Falkenmark, M., & Molden, D. (2008). Wake up to realities of river basin closure. *International Journal of Water Resources Development, 24*, 201–215. doi:10.1080/07900620701723570

Ferraro, P. J., Pattanayak, S. K., Demmer, J., Starkey, M., & Telfer, P. (2006). Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLoS Biology, 4*, e105. doi:10.1371/journal.pbio.0040105

Ferraro, P. J., & Simpson, R. D. (2002). The cost-effectiveness of conservation payments. *Land Economics, 78*, 339–353. doi:10.2307/3146894

Figureau, A. G., Montginoul, M., & Rinaudo, J. D. (2015). Policy instruments for decentralized management of agricultural groundwater abstraction: A participatory evaluation. *Ecological Economics, 119*, 147–157. doi:10.1016/j.ecolecon.2015.08.011

Food and Agriculture Organization. (2013). *The state of food and agriculture 2012*. Rome, Italy: FAO.

Food and Agriculture Organization. (2017). *Does improved irrigation technology save water? Does improved irrigation technology save water?*. Rome, Italy: FAO.

Franken, R., Van Der Meulen, S., Kwakernaak, C., Bos, M., & Lenselink, G. (2016). *Added value of ecosystem services for the Delta Programme*. The Hague, The Netherlands: PBL Netherlands Environmental Assessment Agency.

Frederick, K. D. (2001). Water marketing: Obstacles and opportunities. *Forum for Applied Research and Public Policy, Spring, 200*, 55–62.

Garrick, D., McCann, L., & Pannell, D. J. (2013). Transaction costs and environmental policy: Taking stock, looking forward. *Ecological Economics, 88*, 182–184. doi:10.1016/j.ecolecon.2012.12.022

Garrido, A. (2007). Water markets design and evidence from experimental economics. *Environmental and Resource Economics, 38*, 311–330. doi:10.1007/s10640-006-9077-0

Garrido, A., & Gomez-Ramos, A. (2009). Risk management instruments supporting drought planning and policy. In A. Iglesias, L. Garrote, A. Cancelliere, F. Cubillo, & D. Wilhite (Eds.), *Coping with drought risk in agriculture and water supply systems* (pp. 133–151). Dordrecht, The Netherlands: Springer. doi:10.1007/978-1-4020-9045-5

Garrido, A., Rey, D., & Calatrava, J. (2012). Water trading in Spain. In L. De Stefano & M. R. Llamas (Eds.), *Water, agriculture and the environment in Spain* (pp. 205–216). Leiden, The Netherlands: CRC Press/Balkema.10.1201/b13078

Garrote, L., Iglesias, A., & Flores, F. (2009). Development of drought management plans in Spain. In A. Iglesias, A. Cancelliere, D. A. Wilhite, L. Garrote, & F. Cubillo (Eds.), *Coping with drought risk in agriculture and water supply systems* (pp. 175–186). Dordrecht, The Netherlands: Springer. doi:10.1007/978-1-4020-9045-5

Gazzetta Ufficiale della Repubblica Italia. (2000). Legge 23 dicembre 2000, n. 388. Disposizioni per la formazione del bilancio annuale e pluriennale dello Stato (legge finanziaria 2001).

Gazzetta Ufficiale della Repubblica Italia. (2004). Decreto Legislativo 29 marzo 2004, n. 102. Interventi finanziari a sostegno delle imprese agricole, a norma dell'articolo 1, comma 2, lettera i), della legge 7 marzo 2003, n. 38.

Gazzetta Ufficiale della Repubblica Italia. (2006). Decreto Legislativo n. 152: Codice dell'Ambiente [Legislative Decree n 152: Environmental Regulation]. *Gazzetta Ufficiale,* Serie Generale n 88.

Giuntoli, I., Maugis, P., & Renard, B. (2012). Evolutions observées dans les débits des rivières en France [Observed evolution of river flows in France]. *Sélection d'un réseau de référence et analyse de l'évolution temporelle des régimes des 40 dernières années* [Selected of a Reference Network and Analysis of the Temporal Evolution of Regimes Over the Last 40 Years], *40*, 1-8.

Gleick, P. H. (2002). Soft water paths. *Nature, 418*, 373–373. doi:10.1038/418373a

Gómez, C. M., & Pérez-Blanco, C. D. (2014). Simple myths and basic maths about greening irrigation. *Water Resources Management, 28*, 4035–4044. doi:10.1007/s11269-014-0725-9

Gómez, C. M., Pérez-Blanco, C. D., & Batalla, R. J. (2014). Tradeoffs in river restoration: Flushing flows vs. hydropower generation in the Lower Ebro river. *Journal of Hydrology, 518*, 130–139. doi:10.1016/j.jhydrol.2013.08.029

Gómez, C. M., Pérez-Blanco, D., Adamson, D., & Loch, A. (2017). Managing water scarcity at a river basin scale with economic instruments. *Water Economics and Policy*, 1750004. doi:10.1142/S2382624X17500047

Gómez-Baggethun, E., de Groot, R., Lomas, P. L., & Montes, C. (2010). The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. *Ecological Economics, 69*, 1209–1218. doi:10.1016/j.ecolecon.2009.11.007

Gómez-limón, C., & Martín-Ortega, J. (2013). The economic analysis in the implementation of the water-framework directive in Spain. *International Journal of River Basin Management, 11*, 301–310. doi:10.1080/15715124.2013.823977
Grafton, R. Q. (2017). Editorial water reform and planning in the Murray-Darling basin, Australia. *Water Economics and Policy, 3*, 1–18.
Grafton, R. Q., Landry, C., Libecap, G. D., McGlennon, S., & Brien, R. O. (2010). An integrated assessment of water markets. *Review of Environmental Economics and Policy, 5*, 219–239.
Gudmundsson, L., & Seneviratne, S. I. (2016). Anthropogenic climate change affects meteorological drought risk in Europe. *Environmental Research Letters, 11*, 044005. doi:10.1088/1748-9326/11/4/044005
Gutiérrez-Martín, C., Blanco, C., Gómez, C., & Berbel, J. (2014). Price volatility and water demand in agriculture: A case study of the Guadalquivir river basin (Spain). In T. Bournaris, J. Berbel, B. Manos, & D. Viaggi (Eds.), *Economics of water management in agriculture* (pp. 319–348). Boca Raton, FL: CRC Press.
Gutiérrez-Martín, C., & Gómez, C. M. (2011). Assessing irrigation efficiency improvements by using a preference revelation model. *Spanish Journal of Agricultural Research, 9*, 1009–1020. doi:10.5424/sjar/20110904-514-10
Hanak, E., & Jezdimirovic, J. (2016). *California's water market*. San Francisco, CA: Public Policy Institute California.
Hanak, E., & Stryjewski, E. (2012). *California's water market, by the numbers, update 2012* (Techncal Appendix (Report - Appendix No. Update November 2012)). San Francisco, CA: Public Policy Institute California.
Hernández-Mora, N., & Del Moral, L. (2015). Developing markets for water reallocation: Revisiting the experience of Spanish water mercantilización. *Geoforum, 62*, 143–155. doi:10.1016/j.geoforum.2015.04.011
House of Commons. (2012). *The water white paper. Environment food and rural affairs committee*. London: House of Commons.
Huffaker, R. (2008). Conservation potential of agricultural water conservation subsidies. *Water Resources Research, 44*, 1–8. doi:10.1029/2007WR006183
Huffaker, R., & Whittlesey, N. (2003). International journal of water a theoretical analysis of economic incentive policies encouraging agricultural water conservation. *International Journal of Water Resources Development, 19*, 37–53. doi:10.1080/713672724
Iftekhar, M. S., Tisdell, J. G., & Connor, J. D. (2013). Effects of competition on environmental water buyback auctions. *Agricultural Water Management, 127*, 59–73. doi:10.1016/j.agwat.2013.05.015
Intergovernmental Panel on Climate Change. (2014). *Climate change 2014: Synthesis report*. Geneva, Switzerland: IPCC.
Jääskinen, N. (2014). *Case C-525/12 European Commission v Federal Republic of Germany*. Brussels, Belgium: European Commission.
Kendy, E., Molden, D. J., Steenhuis, T. S., & Liu, C. (2003). *Policies drain the North China Plain: Agricultural policy and groundwater depletion in Luancheng county, 1949-2000* (Research Reports No. 71). Colombo, Sri Lanka: IWMI.
Klijn, F., Van Velsen, E., Ter Maat, J., & Hunink, J. C. (2012). *Zoetwatervoorziening in Nederland* [Freshwater supply in The Netherlands]. Overijssel, The Netherlands: Deltares.
Kosoy, N., Martinez-Tuna, M., Muradian, R., & Martínez-Alier, J. (2007). Payments for environmental services in watersheds: Insights from a comparative study of three cases in Central America. *Ecological Economics, 61*, 446–455. doi:10.1016/j.ecolecon.2006.03.016
Lago, M., Mysiak, J., Gómez, C. M., Delacámara, G., & Maziotis, A. (2015). *Use of economic instruments in water policy: Insights from international experiences*. Heidelberg, Germany: Springer.
Landell-Mills, N., & Porras, I. T. (2005, May). Silver bullet or fools' gold? A global review of markets for forest environmental services and their impact on the poor (Instruments for sustainable private sector forestry series). In G-C. Roman, W. Griswold, & B. Nuseibeh (Eds.), *Proceedings of the 27th international conference on software engineering* (pp. 35-35). St. Louis, MO: ICSE.
Lankford, B. (2004). Irrigation improvement projects in Tanzania; scale impacts and policy implications. *Water Policy, 6*, 89–102.

Linderhof, V., De Blaeij, A., & Polman, N. (2009). *Betalen voor ecosysteemdiensten: Een interessante aanvulling op het waterprijsbeleid?* [Payment for ecosystem services: An interesting addition to water pricing policies?]. Wageningen, The Netherlands: LEI Wageningen UR.

Lindhout, P. E., & Van den Broek, B. (2014). The polluter pays principle: Guidelines for cost recovery and burden sharing in the case law of the European court of justice. *Utrecht Law Review, 10*, 46–59. doi:10.18352/ulr.268

López-Gunn, E., Mayor, B., & Dumont, A. (2012). Implications of the modernization of irrigation. In L. De Stefano & M. R. Llamas (Eds.) *Water, agriculture and the environment in Spain: Can we square the circle?*(pp. 241–255). Leiden, The Netherlands: CRC Press/Balkema.

Maestro, T., Bielza, M., & Garrido, A. (2016). Hydrological drought index insurance for irrigation districts in Spain. *Spanish Journal of Agricultural Research, 14*(3), 1–14. doi: 10.5424/sjar/2016143-8981

Maestu, J., & Del Villar, A., & Ministerio de Medio Ambiente. (2007). *Precios y Costes de los Servicios del Agua en España Informe Integrado de Recuperación de Costes de los Servicios de Agua en España* [Prices and costs of water servicies in Spain. Report of the water services cost recovery in Spain]. Madrid, Spain: Ministerio de Medio Ambiente.

Marsh, T. (2007). The 2004–2006 drought in southern Britain. *Weather, 62*, 191–196. doi:10.1002/(ISSN)1477-8696

Marshall, G. R. (2013). Transaction costs, collective action and adaptation in managing complex social–ecological systems. *Ecological Economics, 88*, 185–194. doi:10.1016/j.ecolecon.2012.12.030

Martín-Ortega, J., Ojea, E., & Roux, C. (2013). Payments for water ecosystem services in Latin America: A literature review and conceptual model. *Ecosystem Services, 6*, 122–132. doi:10.1016/j.ecoser.2013.09.008

Mazza, L., Bennett, G., De Nocker, L., Gantioler, S., Losarcos, L., Margerison, C., … van Diggelen, R. (2011). *Green infrastructure implementation and efficiency* (Final Report for the European Commission, DG Environment on Contract ENV.B.2/SER/2010/0059). London: IEEP.

McCann, L. (2013). Transaction costs and environmental policy design. *Ecological Economics, 88*, 253–262. doi:10.1016/j.ecolecon.2012.12.012

Meinzen-Dick, R., & Bakker, M. (2000). Water rights and multiple water uses: Framework and application to Kirindi Oya irrigation system Sri Lanka. *Irrigation and Drainage Systems, 15*, 129–148.

Mendelsohn, R. (2016). Adaptation, climate change, agriculture, and water. *Choices, 31*, 1–7.

van der Meulen, S., Neubauer, L., Brils, J., & Borowski, I. (2012). *Towards practical implementation of the ecosystem services (ES) concept in transboundary water management*. The Netherlands: Deltares.

Meuwissen, M. P. M., Huirne, R. B. M., & Skees, J. R. (2003). Income insurance in European agriculture. *EuroChoices, 2*, 12–17. doi:10.1111/j.1746-692X.2003.tb00037.x

Ministère de l'écologie, de l'énergie, du développement durable et du Territoire. (2008). *Circulaire du 30 juin 2008 relative à la résorption des déficits quantitatifs en matière de prélèvement d'eau et gestion collective des prélèvements d'irrigation NOR : DEVO0815432C, Bulletin officiel du Ministère de l'écologie, de l'énergie, du développement*. Paris, France: MEEDDAT.

Montginoul, M., & Rinaudo, J. D. (2009). Quels instruments pour gérer les prélèvements individuels en eau souterraine? Le cas du Roussillon [What instruments to manage individual groundwater abstraction? The case of Rousillon]. *Économie rurale, 2*, 40–56. doi:10.4000/economierurale

Montginoul, M., Rinaudo, J.-D., Brozović, N., & Donoso, G. (2016). Controlling groundwater exploitation through economic instruments: Current practices, challenges and innovative approaches. In A. J. Jakeman, O. Barreteau, R. J. Hunt, J.-D. Rinaudo & A. Ross (Eds.), *Integrated groundwater management: Concepts, approaches and challenges* (pp. 551–581). Cham: Springer International Publishing.

Mundell, R. A. (1960). The monetary dynamics of international adjustment under fixed and flexible exchange rates. *The Quarterly Journal of Economics, 74*, 227. doi:10.2307/1884252

Mundell, R. A. (1962). The appropriate use of monetary and fiscal policy for internal and external stability. *Staff Papers - International Monetary Fund, 9*, 70–79. doi:10.2307/3866082

Muradian, R., Arsel, M., Pellegrini, L., Adaman, F., Aguilar, B., Agarwal, B., … Urama, K. (2013). Payments for ecosystem services and the fatal attraction of win-win solutions. *Conservation Letters, 6*, 274–279. doi:10.1111/conl.2013.6.issue-4

Organisation for Economic Co-operation and Development (2009). *Managing water for all: An OECD perspective on pricing and financing*. Paris, France: OECD.

Organisation for Economic Co-operation and Development. (2010). *Enhancing the cost-effectiveness of payments for ecosystem services*. Paris, France: OECD.
Organisation for Economic Co-operation and Development. (2013). *Water security for better lives*. Paris, France: OECD.
Organisation for Economic Co-operation and Development. (2015a). *Water resources allocation: Sharing risks and opportunities*. Paris, France: OECD.
Organisation for Economic Co-operation and Development. (2015b). *OECD principles on water governance*. Paris, France: OECD.
Pagiola, S. (2008). Payments for environmental services in Costa Rica. *Ecological Economics, 65*, 712–724. doi:10.1016/j.ecolecon.2007.07.033
Pagiola, S., & Platais, G. (2007). *Payments for environmental services: From theory to practice*. Washington, MD: World Bank.
Palomo-Hierro, S., Gómez-Limón, J. A., & Riesgo, L. (2015). Water markets in Spain: Performance and challenges. *Water, 7*, 652–678. doi:10.3390/w7020652
Pérez-Blanco, C. D., & Gómez, C. M. (2013). Designing optimum insurance schemes to reduce water overexploitation during drought events: A case study of La Campiña, Guadalquivir River Basin, Spain. *Journal of Environmental Economics and Policy, 2*, 1–15. doi:10.1080/21606544.2012.745232
Perez-Blanco, C. D., & Gomez, C. M. (2014). *An integrated risk assessment model for the implementation of drought insurance markets in Spain*. Venice, Italy: FEEM.
Pérez-Blanco, C. D., & Gutiérrez-Martín, C. (2017). Buy me a river: Use of multi-attribute non-linear utility functions to address overcompensation in agricultural water buyback. *Agricultural Water Management, 190*, 6–20. doi:10.1016/j.agwat.2017.05.006
Pérez-Blanco, C. D., Delacámara, G., & Gómez, C. M. (2015). Revealing the willingness to pay for income insurance in agriculture. *Journal of Risk Research*, *19*, 1–21.
Pérez-Blanco, C. D., Standardi, G., Mysiak, J., Parrado, R., & Gutiérrez-Martín, C. (2016). Incremental water charging in agriculture. A case study of the Regione Emilia Romagna in Italy. *Environmental Modelling & Software, 78*, 202–215. doi:10.1016/j.envsoft.2015.12.016
Perrot-Maître, D. (2006). *The vittel payments for ecosystem services: A 'perfect' PES case?*. London: International Institute for Environment and Development.
Perry, C. (2011). Accounting for water use: Terminology and implications for saving water and increasing production. *Agricultural Water Management, 98*, 1840–1846. doi:10.1016/j.agwat.2010.10.002
Pettenella, D., Vidale, E., Gatto, P., & Secco, L. (2012). Paying for water-related forest services: A survey on Italian payment mechanisms. *iForest - Biogeosciences and Forestry*, 210–215. doi:10.3832/ifor0626-005
Pfeiffer, L., & Lin, C. Y. C. (2014). Does efficient irrigation technology lead to reduced groundwater extraction? Empirical evidence. *Journal of Environmental Economics and Management, 67*, 189–208. doi:10.1016/j.jeem.2013.12.002
Pigram, J. J. (1999). Economic instruments in the management of Australia's water resources: A critical view. *International Journal of Water Resources Development, 15*, 493–509. doi:10.1080/07900629948727
Po River Basin Agency. (2003). *Protocollo di Intesa* [Memorandum of understanding]. Parma, Italy: Po River Basin Authority.
Porras, I., Alyward, B., & Dengel, J. (2013). *Monitoring payments for watershed services schemes in developing countries*. London: IIED.
Postel, S. L., & Thompson, B. H. (2005). Watershed protection: Capturing the benefits of nature's water supply services. *Natural Resources Forum, 29*, 98–108. doi:10.1111/narf.2005.29.issue-2
Prudhomme, C., Giuntoli, I., Robinson, E. L., Clark, D. B., Arnell, N. W., Dankers, R., … Wisser, D. (2014). Hydrological droughts in the 21st century, hotspots and uncertainties from a global multimodel ensemble experiment. *Proceedings of the National Academy of Sciences, 111*, 3262–3267. doi:10.1073/pnas.1222473110
Randall, A. (1981). Property entitlements and pricing policies for a maturing water economy. *Australian Journal of Agricultural Economics, 25*, 195–220. doi:10.1111/j.1467-8489.1981.tb00398.x
Rausser, G. C., Swinnen, J. F. M., Zusman, P., & Approaches, A. (2011). *Political power and economic policy*. New York, NY: Cambridge University Press. doi:10.1017/CBO9780511978661

Reed, M. S., Hubacek, K., Bonn, A., Burt, T. P., Holden, J., Stringer, L. C., … Worrall, F. (2013). Anticipating and managing future trade-offs and complementarities between ecosystem services. *Ecology and Society, 18*(1), 5.
Rey, D. (2014). *Water option contracts for reducing water supply risks: An application to the Tagus Segura Transfer* (Doctoral Dissertation). Retrieved from: http://oa.upm.es/30924/1/DOLORES_REY_VICARIO.pdf
Rey, D., Holman, I. P., Daccache, A., Morris, J., Weatherhead, E. K., & Knox, J. W. (2016). Modelling and mapping the economic value of supplemental irrigation in a humid climate. *Agricultural Water Management, 173*, 13–22. doi:10.1016/j.agwat.2016.04.017
Rey, D., Holman, I. P., & Knox, J. W. (2017). Developing drought resilience in irrigated agriculture in the face of increasing water scarcity. *Regional Environmental Change, 17*, 1527–1540. doi:10.1007/s10113-017-1116-6
Reynaud, A., & Lanzanova, D. (2017). A global meta-analysis of the value of ecosystem services provided by lakes. *Ecological Economics, 137*, 184–194. doi:10.1016/j.ecolecon.2017.03.001
Rinaudo, J.-D., & Hérivaux, C. (2014). Quels instruments pour une gestion collective des prélèvements individuels en eau pour l'irrigation ? [What instruments for the collective management of individual water abstraction for irrigation?] *Économie Rurale, 310*, 40-56.
Rinaudo, J. D., Montginoul, M., Varanda, M., & Bento, S. (2012). Envisioning innovative groundwater regulation policies through scenario workshops in France and Portugal. *Irrigation and Drainage, 61*, 65–74. doi:10.1002/ird.v61.S1
RIZA. (2005). *Aard, ernst en omvang van watertekorten in Nederland* [Nature, severity and extent of water shortages in The Netherlands]. The Netherlands: RIZA.
Rodríguez Díaz, J. A., Urrestarazu, L. P., Poyato, E. C., & Montesinos, P. (2012). Modernizing water distribution networks. *Outlook on Agriculture, 41*, 229–236.
Ruiz, J., Bielza, M., Garrido, A., & Iglesias, A. (2015). Dealing with drought in irrigated agriculture through insurance schemes: An application to an irrigation district in Southern Spain. *Spanish Journal of Agricultural Research, 13*(4), 1–15.
RWS. (2011). *Water management in the Netherlands*. The Netherlands: Rijkswaterstaat Ministry of Infrastructure and the Environment.
Saliba, B. C., & Bush, D. (1987). *Water markets in theory and practice: Market transfers, water values and public policy*. Boulder (CO): Westview Press.
Santato, S., Mysiak, J., & Pérez-Blanco, C. D. (2016). The water abstraction license regime in Italy: A case for reform? *Water (Switzerland), 8*, 1–15.
Scheierling, S. M., Young, R. A., & Cardon, G. E. (2006). Public subsidies for water-conserving irrigation investments: Hydrologic, agronomic, and economic assessment. *Water Resources Research, 42*, 1–11.
Smith, A. (1776). *The Glasgow edition of the works and correspondence of Adam Smith, Vol. 2: An inquiry into the nature and causes of the wealth of nations*, Vol. 1. London: W. Strahan and T. Cadell. doi:10.1093/actrade/9780199269563.book.1
Solanes, M. (1998). *Integrated water management from the perspectie of the Dublin principles*. Santiago, Chile: CEPAL.
Stiglitz, J. E., & Greenwald, B. C. (1986). Externalities in economies with imperfect information and incomplete markets. *The Quarterly Journal of Economic, 101*, 229–264.
Strosser, P., & Montginoul, M. (2001). Vers des marchés de l'eau en France ? Quelques éléments de réflexion [Towars water markets in France? Some elements of reflection]. *Annales Des Mines, Juillet, 2001*, 13–31.
Sumpsi, J. M., Garrido, A., Blanco, M., Varela, C., & Iglesias, E. (1998). *Economía y Política de Gestión del Agua en la Agricultura* [Agricultural water management economics and policy]. Madrid, Spain: MAPA and Mundi-Prensa.
Tinbergen, J. (1952). *On the theory of economic policy*. Amsterdam, The Netherlands: North-Holland Publishing Company.
URBER. (2015). Bilancio economic finanziario delle bonifiche Emilia Romagna. Unione Reg. Delle Bonifiche Emilia Romagna. [Financial economic balance of the Emilia Romagna reclamation]. Retrieved from http://www.urber.it/consorzi-cartina.asp

Verburg, R., & Seines, T. (2014). *The ecosystem services approach as an instrument for action, (WOT paper 30)*. Wageningen, The Netherlands: Wageningen UR.

Viavattene, C., McCarthy, S., Green, C., & Pardoe, J. (2015). Cooperative agreements between water supply companies and farmers in Dorset (E). In M. Lago, J. Mysiak, C. M. Gómez, G. Delacámara, & A. Maziotis (Eds.), *Use of economic instruments in water policy: Insights from international experience* (pp. 325–335). Heidelberg, Germany: Springer.

Wheeler, S., Loch, A., Zuo, A., & Bjornlund, H. (2014). Reviewing the adoption and impact of water markets in the Murray-Darling basin, Australia. *Journal of Hydrology, 518*, 28–41. doi:10.1016/j.jhydrol.2013.09.019

Wheeler, S. A., Loch, A., Crase, L., Young, M., & Grafton, R. Q. (2017). Developing a water market readiness assessment framework. *Journal of Hydrology, 552*, 807–820. doi:10.1016/j.jhydrol.2017.07.010

Williamson, O. E. (1998). Transaction cost economics: How it works; where it is headed. *De Economist, 146*, 23–58. doi:10.1023/A:1003263908567

Wunder, S. (2005). *Payments for environmental services: Some nuts and bolts* (Occasional Paper, 42). Jakarta, Indonesia: CIFOR.

Wunder, S. (2015). Revisiting the concept of payments for environmental services. *Ecological Economics, 117*, 234–243. doi:10.1016/j.ecolecon.2014.08.016

Wunder, S., & Albán, M. (2008). Decentralized payments for environmental services: The cases of Pimampiro and PROFAFOR in Ecuador. *Ecological Economics, 65*, 685–698. doi:10.1016/j.ecolecon.2007.11.004

Wutich, A., White, A. C., White, D. D., Larson, K. L., Brewis, A., & Roberts, C. (2014). Hard paths, soft paths or no paths? Cross-cultural perceptions of water solutions. *Hydrology and Earth System Sciences, 18*, 109–120. doi:10.5194/hess-18-109-2014

WWF. (2006). *Illegal water use in Spain: Causes, effects and solutions*. Madrid, Spain: WWF.

Young, M. (2012). *Towards a generic framework for the abstraction and utilisation of water in England and Wales*. London: UCL Environment Institute.

Young, M. (2014). Designing water abstraction regimes for an ever-changing and ever-varying future. *Agricultural Water Management, 145*, 32–38. doi:10.1016/j.agwat.2013.12.002

Young, M. (2015). *Unbundling water rights: A blueprint for development of robust water allocation systems in the Western United States* (ReportNI R.15-01). Durham, NC: Duke University.