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The Theory and Practice of Water Pricing and Cost Recovery in the Water Framework Directive

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ABSTRACT: Article 9 of the Water Framework Directive (WFD) requires member states to take account not only of the principle of cost recovery of water services, including environmental and resource costs (ERCs), but also of the use of water pricing as an environmental policy instrument; nevertheless, no common methodology exists for the estimation of financial costs, nor is there a practical definition of ERC. The review of public evidence and scientific research regarding the effect of pricing on demand shows the limitations of water pricing and the need to integrate pricing and non-pricing instruments. Cost recovery remains a convenient policy for the financing of existing and future water infrastructures. This study offers a brief discussion on the theory and practice of pricing in Article 9 of the WFD and proposes the adoption of a more realistic approach to the implementation of cost recovery, one which abandons the unrealistic objective of monetisation of ERCs and proposes alternatives to the current emphasis on water pricing as a component of water resources management.

KEYWORDS: Water Framework Directive, cost recovery, water pricing, affordability, environmental and resource costs

INTRODUCTION

Between 2000 and 2017, in the European Union (EU), water abstraction and economic growth have decoupled; during this period, according to Eurostat sources, the value added of the economy as a whole has grown by 59% while water abstraction has decreased by 17% (EEA, 2019). This fact may be explained by the confluence of many factors; these include a growing compliance cost arising from the settlement of new operational standards in water services such as drinking water, sewage and conveyance efficiency; from technical progress such as improvement in equipment and know-how by farmers, industries and utilities; from increasing social awareness regarding the quantity and quality of water; and from a significant increase in water prices during this period (EEA, 2019). Through the WFD, the EU introduced a wide range of economic concepts and instruments designed to improve the management of water resources and to achieve the good water and good ecological status of all water bodies. The initial goals of the WFD (Article 4 and the Preamble) were to be accomplished by the end of the first cycle in 2015; the second cycle will conclude in 2021 and the third implementation phase will end in 2027. The WFD is a breakthrough legal instrument that is aimed at coupling ecological vision with an economic ideal (Bouleau, 2008). This paper will focus specifically on the implications of Article 9 of the WFD regarding the cost recovery principle (CRP) and its implementation through the use of economic instruments such as water pricing.

Article 9 of the WFD requires member states to take into account the principle of cost recovery of water services, including not only capital and investment (C&I) and the costs of operation and maintenance (O&M) of water infrastructures and services (both C&I and O&M costs are commonly denoted as financial costs), but also environmental and resource costs (ERCs). Environmental costs usually refer to the costs of environmental damage imposed by water users (and beneficiaries), while resource costs represent opportunity costs due to resource depletion. Article 9 is probably one of the most quoted normative sections of the directive and the political and economic foundations of the cost recovery principle deserve attention.

The idea behind the use of prices to manage demand is supported by the Pigouvian tradition, which advocates for the use of taxes to control negative externalities derived from any economic activity. According to this tradition, tax must be equal to the 'marginal social damage' that the activity generates; in practice, however, the computation of the value of marginal social damage remains unfeasible due to our inability to quantify the concept. This paradigm was central to the adoption of the 'polluter pays principle' (PPP) in the Treaty on the Functioning of the European Union (TFEU) (Article 191[2]); it functioned as a basic guide for environmental policy and was also recognised in the 1992 Rio Declaration. In fact, the inclusion of this principle in the WFD goes beyond pollution as a negative externality and, in cases of environmental damage, also includes compensatory measures on the basis of 'resource user pays' (Unnerstall, 2007). Article 9 of the WFD relates the PPP to the principle of cost recovery in the sense that it requires an 'adequate contribution' by alternative water users towards the full recovery of the costs, including ERCs. The term 'polluter pays' should thus be broadly interpreted to include the negative externalities of any water use on the environment and the sustainability of both the quantity and quality of the water resource; this is problematic if no adequate and standardised methodology is available. In contrast, positive externalities such as ecosystem services provided by irrigation should also be considered and promoted through, for example, a system of subsidies, thereby adopting a more systematic approach.

In relation to cost recovery, the European Court of Justice has ruled that member states may decide which type and design of economic instruments are to be implemented, as long as WFD objectives are met; these may include new taxation instruments and public tariffs (Court of Justice of the European Union, 2014). The European Commission (EC) and many other stakeholders nevertheless advocate for the strict application of Article 9, which involves the implementation of pricing instruments for full cost recovery in all its dimensions, financial – including C&I and O&M costs – as well as ERC.

This paper is organised as follows. The next section draws on the origin of Article 9 of the WFD and its implications for EU water policy. The subsequent section reviews the basis for water pricing as the main economic instrument for water policy; this is followed by a section that illustrates how these principles are implemented in practice by EU member states. The final section reviews and discusses the evidence regarding the implementation of the CRP and proposes a way forward for the improvement of the use of economic instruments in the context of Article 9, whose aim is to achieve sustainable management of water resources.

THE COST RECOVERY PRINCIPLE AND THE ORIGIN OF ARTICLE 9

Article 9 of the WFD states the necessity of taking the CRP into account in the economic analysis presented in Article 5 and Annex III considering the polluter pays principle. It also states that water pricing policies provide adequate incentives for users to increase efficiency in water use and to contribute towards achieving WFD environmental objectives. The PPP. and water pricing have thus become the main issues regarding to the implementation of the CRP as it is defined by the WFD.

The definition of the CRP and its implementation in water services has proved to be a conflicted issue in the relationship between the EC and the European Parliament (Kaika and Page, 2003). Lindhout and van den Broek (2014) analyse the evolution of the CRP, beginning with the European Commission's proposal which states that, "[member states] ensure full cost recovery for all costs for services provided for water uses overall and by economic sectors (…)" (EC, 1997).

The wording of the proposal was finalised three years later, following negotiations between the European Parliament, the Council, and the Commission; it stated that, "[member states] shall take account of the cost recovery principle of water services, including environmental and resource costs".

The changes introduced through the negotiation process are relevant since they imply a change in the scope of the above statement. First, the Council differentiates between the original term 'water uses' and the final term 'water service'. The concept of water services is restricted to abstraction, emission and wastewater issues (Article 2.38). Water uses (Article 2.39), on the other hand, cover a wider range of activities since "it includes any other activity (…) having a significant impact on the status of water"; it therefore includes other economic activities that influence water resources, such as hydropower and navigation. It should be noted that, according to the Water Blueprint (EC, 2012), the main pressure on EU waters is from hydromorphological modifications. However, at the start of adoption of the WFD, few countries had experience in water economics. For that purpose, a specific guidance on water economics (WATECO) was developed in 2001-04 under the CIS process, contributing to strengthening and homogenisation of the economic knowledge in the field of water throughout Europe. WATECO group of experts proposed the enlargement of Article 2.38, such that 'water services' also included hydromorphological modifications of water bodies (such as reservoirs) that manage water supply and flood protection, and also included navigation activities and facilities for energy production purposes or agricultural drainage (WATECO, 2002). Nevertheless, the specific focus of the CRP in irrigation, urban and other consumptive water uses was already established in the WFD text, and it was difficult to amend the directive through the use of a guidance document.

Second, the original wording of "ensure full cost recovery" (EC, 1997) became "take account of the cost recover principle" (WFD) as a result of EU parliamentary debates and the pressure exerted by interest groups and some member states to gain a certain degree of freedom in the implementation of Article 9. Additionally, the concept of "including environmental and resource costs" is introduced in the directive without a proper definition of the concept. WATECO documents attempt to clarify the concept but, as discussed below, the practical implementation (and assessment) of the ERC remains subject to debate.

Finally, a 'northern' bias may be evident in the fact that 'water uses' became 'water services'; a result of this change in wording was that critical pressures such as navigation and hydropower – like agriculture, industry and households – were removed from cost-recovery sectors. This northern bias puts the focus on consumptive uses such as irrigation, which is more widespread in Southern European member states. This may also be a way to avoid a serious analysis of the main pressure on water bodies, which is hydromorphological (EC, 2012).

In this context, the question should be posed and further analysed as to whether these wording changes are purely the result of random choices of language by EU institutions or whether they are the result of pressure by member states and stakeholders; the answer to this question, however, falls outside the scope of this paper.

Finally, it should be mentioned that paragraph 4 of Article 9 the WFD limits the coercive nature of Article 9; it states that "Member States shall not be in breach of this Directive if they decide in accordance with established practices not to apply the provisions of paragraph 1". It is worth noting that, with the dispute in 2014 between the Commission and Germany, this lack of a harmonised definition of the cost recovery concept has even reached the Court of Justice of the European Union (Court of Justice of the European Union, 2014).

WATER PRICING AS AN ECONOMIC INSTRUMENT FOR COST RECOVERY

According to the EEA glossary, water pricing is defined as, "applying a monetary rate or value at which water can be bought or sold" (EEA, 2013), although other broader definitions, such as that proposed by ARCADIS (2012), describe pricing in the water sector as a process of "monetising the abstraction, use, or pollution of water".

The use of prices and environmental taxes is well established in the Pigouvian economic theory; it is considered to be a silver bullet that is capable of changing agents' behaviour with the aim of reaching an efficiency optimum. Obviously, assessing the "marginal social damage" to be handled through taxation presents a Herculean task. As a pragmatic approach to the evaluation problem, Baumol and Oates (1971) proposed the definition of standards of environmental quality (such as wastewater discharge) and the imposition of a set of charges (for example, wastewater tariffs). These authors also identified measures to achieve a predefined environmental objective at minimum cost; in other words, instead of the 'optimiser' approach, they defend a 'satisfier' approach. Simon (1978) observed that decision makers could satisfy either by finding optimal solutions for a simplified world or by finding satisfactory solutions for a more realistic world.

Economic theory also states that the effectiveness of water pricing as a policy instrument is determined by the sensitivity of the response in the economic agent's demand. This is usually observed by the estimation of price elasticity of the demand function which is the demand response to price changes (through, for example, taxation) in the short term.

Price elasticity of demand is likely to be less in the short term, when decisions are constrained, than it is in the longer term when more adjustments are possible (Johnston, 1968). The existing literature shows that there are differences in estimated short- and long-term price elasticities that are largely explained by the higher rigidity of the demand function to short-term price changes. Estimations of long-term price elasticities are, however, technically more difficult to obtain; the existing literature is scant and is focused mainly on the urban sector (Reynaud and Romano, 2018). There are some references to this in the literature on the agricultural sector; they include that by Wheeler et al. (2015) who, based on an analysis of the Australian water market, estimated an average short-term elasticity of – 0.52 and a long-term elasticity of – 0.89. Short-term elasticities are usually found to be smaller than their long-term counterparts, which suggests that consumers might need time to adjust water-using capital stocks and to learn about the effects of water use on their bills.

Urban water

Hundreds of water-demand elasticities have been reported in the literature related to the urban (or household) sector, the majority of which refer to the short term. One example involves the study by Espey et al. (1997), who estimated a mean demand elasticity of – 0.51 (that is, when price increases by 1%, water demand decreases by 0.51%); this was based on a sample of 124 elicited price elasticities. Alternatively, Dalhuisen et al. (2003), after adding a further 172 observations to the study of Espey et al. (1997), obtained a mean elasticity of – 0.41. Sebri (2014) obtained a mean elasticity of – 0.37 from a sample of 638 elicited elasticities. In the case of the EU member states, Reynaud (2015) offered a review of household-demand elasticities which yielded heterogeneous findings depending on the member state analysed and on the data sample. Martínez-Espiñeira (2007) for Spain, found that for a panel of representative municipalities the price elasticity of demand was around −0.10 in the short run and −0.50 in the long run. In the case of France, the short- and long-term price elasticities reported by Nauges and Thomas (2003) were, respectively, −0.26 and −0.40. Musolesi and Nosvelli (2007) found that short- and long-term elasticities for the Italian household water demand were equal to −0.27 and −0.47, respectively.

A recent and ambitious report by the European Environmental Agency (EEA) states that, "In some of the case studies, price does not appear to be a significant determinant of water demand. (…). Water pricing still remains a key instrument in achieving cost recovery for water services to ensure the maintenance and financing of existing and future water infrastructure" (EEA, 2017).

In addition to prices, certain other economic instruments can induce changes in consumer behaviour. Ferraro and Price (2013), in a massive natural field experiment, found important complementarities between pecuniary and non-pecuniary strategies, especially social comparison messages; this confirms the conclusion of the EEA that "water demand management strategies need to find the right mix of pricing and non-pricing instruments" (EEA, 2017).

In summarising the findings regarding household users, water pricing does not seem to be efficient as a stand-alone measure for the management of urban water demand. According to the mentioned study (ibid), urban price elasticities are generally low and consequently high price increases are required to induce water savings in the short term; this opens up the issue of affordability (impact in poorer social groups), which will be discussed in the next section. Cost recovery is nevertheless required to both maintain infrastructure and to finance existing and future water infrastructure; this is particularly the case when considering that scarcity and the variability of water may increase in the future as predicted by many climate change scenarios. This issue is discussed further in the following section.

Industry

In order to quote recent estimations in the European context, the case of France should be mentioned. Reynaud (2003) found an average industrial-demand elasticity of – 0.29; this varied from – 0.095 for the alcohol and beverage industry to – 0.734 for the extractive industry. Dupont and Renzetti (2001) found an average elasticity of – 0.77 for a wide range of industrial sectors in Canada, and in the case of Spain's hotels, Arbués et al. (2010) found that the water-demand elasticity was – 0.38. As can be observed, the range of elasticities is wider than those reported for household domestic use and depends on the specificities of the industrial sector that is being considered; nevertheless, the increasing pressure imposed by the implementation of social compliance programmes and principles by industrial firms is moving this sector towards optimisation in the use of water resources. Many industrial sectors, in fact, are claiming to be 'water neutral', especially those sectors with a high price elasticity; they are thus demonstrating a rapid response to price increases and the adoption of technology. Finally, it is also worth noting that some of the technological changes are driven by considerations of social responsibility and public image which go beyond strict cost accounting calculations.

Agriculture

The irrigation sector is the main water user in the EU. Agriculture, forestry and fishing (mainly irrigation) sector was responsible for 58% of water abstraction by economic sectors in 2017b at the European level (EEA,2020). There is abundant literature from around the world estimating the water-demand response to price changes and farmers' willingness to pay; Scheierling et al. (2006), for example, conducted a meta-analysis of irrigation-water price elasticity and found a mean price elasticity of -0.48, which depended on the presence of high-value crops such as citrus trees and vegetables, and on the level of pre-existing prices. This study also found that elasticity estimates depend on the methodology applied, for example, mathematical programming, field research and/or econometric methods. The number of EU studies is extensive; it includes that by Bartolini et al. (2007), Manos et al. (2006), and de Frutos Cachorro et al. (2017).

This large body of literature shows evidence of a wide range of elasticities, from highly elastic responses in the case of agricultural commodities in water-abundant regions with low water prices (generally a flat rate) to inelastic responses in the case of high-value crops such as fruits and vegetables. In water-scarce regions, farmers generally apply deficit irrigation techniques, that is, they apply amounts of water that are below irrigation needs; the consequence is the generation of a water-demand curve with a threshold price below which the demand curve is vertical and therefore unresponsive to price increases (de Fraiture and Perry, 2007). Certain reported cases estimate threshold values of as high as €1.0/m^3 before farmers start reducing water use (Expósito and Berbel, 2017).

In summary, analysis of the published evidence, shows that water pricing may constitute an efficient economic tool in those regions where water prices are low and where water is abundant and there is generalised over-irrigation. In these regions, small price increases may have a large effect in terms of water savings achieved in both the short and long term. In contrast, in water-stressed regions where irrigation modernisation programmes have produced water savings and where the implementation of deficit irrigation is generalised, increases in the price of water significantly affect farmers' incomes before they lead to further water savings.

THE PRACTICE OF COST RECOVERY INCLUDING ENVIRONMENTAL AND RESOURCE COSTS

The practice of cost recovery is mainly focused on the analysis of financial cost. Financial cost – as compared to environmental or resource costs which have no common methodology for their recovery – lends itself more readily to calculation; its recovery is also necessary to the financing of present and future needs such as water infrastructure. This section discusses the practical challenges of recovery of these various costs.

Financial costs

Prices of any commodity are captured by markets, which follow the demand/supply principles described in economic theory. In the case of the water sector, price determination based on financial costs – that is, capital and investment, and operation and maintenance – present unique difficulties. This task is made difficult by market imperfections related to the singular characteristics of water resources and service providers, together with the lack of standardised accounting practices such as depreciation and computation of capital costs (Massarutto, 2007). Spain is the only EU country that has implemented the trade of water entitlements (Montilla-López et al., 2016), which allows the determination of real market prices for water as an interaction between demand and supply forces (Schwabe et al., 2020). In general, 'water price', in the EU context, merely refers to water tariffs (or publicly set prices) that have been established by an administrative procedure that strives to estimate the cost of services and then distribute this cost among the various users. In the case of publicly owned infrastructure and a public service-provider, tariffs (or prices) seldom reflect all costs to users; the economic agents therefore fail to internalise all the financial costs – C&I and O&M – that are required to deliver the service. According to Rey et al. (2019), there is evidence from France, Italy and Spain that higher cost recovery levels are typically observed where collective water management is implemented; this is true in the case of water user associations (WUAs) in the irrigation sector which are also responsible for infrastructure management.

The EEA (2013) report offers a summary on the status of cost recovery implementation which shows large disparities between member states; nevertheless, this report proposes no common methodology, since the criteria for the selection of economic instruments is also unclear and the information sources are only secondary. Financial cost can only be estimated; it also requires subjective decisions such as the depreciation rate, the type of financial costs involved, and the distribution of general expenses among users. The analysis of certain member state reports shows differences of from 25 to 50 years in the depreciation rates of their infrastructure; this produces variations in the estimated cost. A critical point regarding the implementation of the WFD is the level of cost recovery. We have mentioned the lack of harmonisation and the information gaps at the EU 28 level; certain data is nevertheless available from national reports, and Table 1 offers several recent estimations. As shown, EU member states register a wide variability in cost recovery rates; in the case of the agricultural sector, C&I costs register a lower cost recovery rate than do O&M costs, mainly due to public subsidies for irrigation infrastructure. In the case of the urban sector, cost recovery rates are generally higher in the supply and sanitation services.

Table 1. Estimations of the percent of cost recovery in selected countries.

Country/sector	Agriculture (surface water)			Urban sector	
	C&I (%)	O&M (%)	Total (%)	Water supply (%)	Waste water (%)
Spain	56	≈ 100	≈ 85	74	74
Portugal	23	≈ 100	65	80	46
France	15-60	≈ 100	≈ 85	75	75
Austria				84	84
Bulgaria				48	55
Italy				44	44

Source: Figures for the agricultural sector are from Berbel et al. (2019); figures for the urban sector are from Reynaud (2016).

No specific details regarding cost recovery and pricing were given in the last report by the European Commission on the implementation of River Basin Management Plans (RBMP) in EU member states (EC, 2019a). Commenting on the analysis of second-cycle RBMPs (2015-2021), it concludes that limited changes in water pricing policies have taken place in the implementation of the provisions of Article 9. In the report, information on the incentive function of water pricing and the adequate contribution of the different water uses is often rather limited and generic; methodologies to calculate costs are insufficiently documented; and essential information is missing. The implementation of Article 9 therefore remains incomplete and cost recovery is not always applied. Finally, the report concludes that "[the] EC could support the development of more consistent methodologies" (EC, 2019a).

Despite these different estimations and the fact that the guidelines for second-cycle RBMPs were set out in 2015, no common standard accountancy methodology has yet been proposed by the EC. If all these technical problems can be overcome, then a reliable estimation of financial cost recovery can be obtained. Although the EEA report offers certain calculations, it contains inconsistencies and fails to rely on a sound common methodology (EEA, 2013). An alternative proposal for a common methodology is offered by Borrego-Marín et al. (2016). It is based on the UN System of Environmental Economic Accounting (SEEA) framework and may be used as a standard system; its level of analysis, however, is at the national and river basin scales, which is excessively large for any practical decision-making to be carried out with regard to prices in the water sector.

Certain progress has been made towards volumetric charging; this is generally operated through two-tier tariff systems, one reflects the fixed costs of water conveyance and another is based on actual withdrawals and use (EC, 2019b). What is clear is that water pricing does not affect behaviour when the tariff is designed as a flat rate, a situation that is frequently found in the irrigation sector across the EU; it is also clear that only the implementation of volumetric charges may induce agents to internalise cost and to adjust by reducing abstraction. This has been effective in the household and industrial sectors; there is, however, still room for improvement in the irrigation sector, where volumetric charges may reduce water use despite the short-term inelasticity of water-demand functions in southern EU countries due to water scarcity.

The principle of cost recovery is complementary to water pricing as an incentive for sustainable resource use; it is also necessary for financing present and future water infrastructures and service maintenance costs. Several services, such as storage, distribution and sanitation, can be related to certain economic users who profit from the service; meanwhile, other services such as flood prevention may be supported with general taxation since they can be understood as a common benefit for multiple users, including the environment.

Article 9 of the WFD has been in full effect since 2010; now, finally – ten years later – the European Commission recognises the lack of a common methodology for defining and estimating the cost of water services. A 2019 report states that, with regard to "defining water services, calculating financial costs [and] metering (…), significant gaps remain in translating these improved elements of economic analysis into concrete measures and achieving more harmonised approaches" (EC, 2019b). Many member states have nevertheless upgraded their water pricing policies in anticipation of the Common Provisions Regulation for the European Structural and Investment Funds; among other measures, this requires metering and cost recovery to finance irrigation modernisation. The economic crisis that followed the financial crash of 2008, and the increasing supervision of member states by the EC, have further accelerated the reform of water tariffs in many member states, including Portugal, Italy and Cyprus (for a detailed analysis, see Berbel et al., 2019); this has involved the introduction of legal reforms to facilitate compulsory water abstraction metering, fulfilment of the CRP, and the establishment of an 'ecotax' to increase water tariffs in areas sensitive to persistent water scarcity.

The issue of joint cost allocation still appears, however, even if a good estimation of the financial cost of water services is achieved. This is a generalised problem when sharing the cost of water services, such as when a multipurpose reservoir serves various policy goals, consumptive uses such as irrigation and municipal supply, economic non-consumptive uses such as industrial cooling, hydroelectric power and navigation, and public goods such as flood control; in such cases, cost estimation is a complex task which is generally based on judgment. In the case of Spain and Portugal, the cost-distribution formula for multipurpose reservoirs is based on a 1 to 3 ratio between irrigation and non-irrigation uses; in France, this ratio is close to 1 to 4.5. Justification of this ratio is grounded in the higher level of guarantee for priority users (non-irrigation users) in case of drought events (Berbel et al., 2019).

The political decision to distribute multipurpose and joint services illustrates the problem of cross-subsidisation; this is a situation in which certain users (or general taxpayers) pay a greater share of the costs while others pay less than their service costs. As explained earlier, this cross-subsidisation issue is unavoidable in the complex issue of estimating multifunctional services. There is also the problem of the locational distribution of cost; this refers to the setting of regional or local water supply and sanitation tariffs where remote (rural) users are charged less than the full cost of their water services and are subsidised by urban water users who benefit from the economy of scale that characterises water supply and treatment plants. In this respect, cross-subsidies may improve the implementation of measures and benefit the good status of water bodies. This fact illustrates the need to define a level for 'cost recovery' estimation at local, regional, basin, national, sector or other scales. In all cases, recovery of financial costs should be implemented for water services in order to ensure the maintenance and financing of existing and future water infrastructures; this is the case despite the fact that the effect of this implementation on water-demand management may be limited due to low short-term price elasticities. A clear and common methodology is also needed for these financial cost estimations. Part of the explanation for the lack of a comprehensive methodology and the persistent information gaps with regard to financial costs is found in the high level of attention and discussion devoted to the controversial issue of ERC monetarisation; this is analysed in the next subsection.

Environmental and resource costs

EU institutional literature states that ERC assessment is a feasible task. It includes comments such as, "It would be particularly useful to have a system, standardised to a certain degree across EU member states, that indicates which areas of environmental and resource costs are covered, and the level of coverage" (EEA, 2013); also, "significant gaps remain in translating these improved elements of economic analysis into concrete measures and achieving more harmonised approaches to estimate and integrate environmental and resource costs" (EC, 2019b).

Twenty years have passed since WFD approval, yet there is still no practical proposal for the estimation of ERC and no member state has been able to develop a methodology to evaluate this concept. The EC supports the need for 'calculation' to ensure that ERCs are internalised, but it has yet to offer a clear methodological proposal. This contrasts with the success achieved in other complex domains such as the monitoring of environmental flows (EC, 2015). In this respect, Gawel (2014) provides several arguments against the EC's emphasis on ERC; the main argument is that, because of the lack of accurate data on environmental resource costs, it is not feasible to translate ERC into monetary terms; they further argue that it is difficult to use the polluter pays principle to allocate costs among individual users.

Certain EU countries, including Spain, France and Germany, have adopted the pragmatic approach of including ERC in the cost of measures to achieve good environmental status. this approach, we feel, follows the theoretical and pragmatic approach of Baumol and Oates (1971). Portugal, Italy, France and Cyprus have applied an ecotax in order to partially internalise ERC; the average tax for water abstraction in these member states is approximately €0.02/m^3, with higher figures in the case of areas sensitive to scarcity. Paradoxically, the Netherlands (in 2008) and the German state of Baden-Württemberg (in 2011) eliminated such taxes (Berbel et al., 2019).

The 1976 German Wastewater Charge, on the other hand, was a more pragmatic approach which significantly improved the state of water treatment (Möller-Gulland et al., 2015); it demonstrated that ERCs can be implemented without performing the exhausting task of their calculation. As stated in a 2019 EC report, "in many cases these costs are considered as calculated through their internalisation, without a primary effort to actually estimate them" (EC, 2019b).

The principal pressures from agriculture involve diffuse pollution and irrigation abstraction; the first of these affects a larger part of the EU territory and is caused mainly by rainfed and livestock farming. These issues cannot be managed solely by water pricing as taxes on fertiliser and quantitative controls for nutrient balance are not directly related to water abstractions.

In summary, we believe that the unachievable goal of 'monetising' ERCs has wasted energy and that it is headed towards an impasse in the practical implementation of Article 9 of the WFD. The section below aims to propose a way forward with respect to this issue.

Hydropower and cost recovery

The hydropower sector and navigation are not explicitly mentioned for cost recovery estimation under Article 9. Hydropower plants seldom pay any kind of charge to manage the 'thermal pollution' from cooling water and other general costs of basin management (EEA, 2013). The review of hydropower royalties in the EU, in fact, shows large disparities (Glachant et al., 2014), while Pineau et al. (2017) conclude that, in general, taxation of hydropower is determined arbitrarily and water use is usually undervalued. The case of hydropower is currently under scrutiny by several member states. In January 2019, Sweden approved legislation that aims to reduce hydropower licences to 40 years in order to improve the environmental status of river catchments. A key measure involves the creation of the Hydroelectric Environmental Fund (Vattenkraftensmiljofond), which aims to gather SEK 10 billion (around €1 billion), with an additional €0.3 billion supplied by hydropower plant owners. The fund is now in operation and constitutes an example of a pragmatic approach to ERC internalisation without requiring the heroic effort of monetisation.

DISCUSSION: 20 YEARS OF WFD, ARTICLE 9 AND BEYOND

It is now 20 years since the initial implementation of the WFD and 10 years since Article 9 came into force. According to the published second-cycle RBMPs (2015-2021), the level of financial cost recovery reports an average estimation of 75 to 80% for the majority of services, sectors and countries; clearly, these figures are cautious since they are based on data submitted by each member state and follow various definitions and methodologies. The report by the EEA (2013) provides estimates of cost recovery; it lacks a common methodology, however, and consequently cannot be used for comparison purposes. According to the reported levels of cost recovery there is scope for improvement, as financial costs are still not fully recovered according to the available evidence.

According to a 2019 EU report, volumetric charges are in place for 58% of all reported water services across the EU (EC, 2019b). The agricultural sector – mainly irrigated farming – appears to be the economic sector with the lowest levels of cost recovery. The good news is that the use of volumetric pricing in the irrigation sector, which was negligible in the 1990s, is becoming the dominant system in several EU countries, including Spain, Italy and Cyprus. Technological changes in irrigation such as, for example, drip irrigation, combined with conversion from a flat tariff to a two-tier system (fixed cost plus volumetric charges) has led to an average reduction of water abstraction by 25 to 50% in Spain (Espinosa-Tasón et al., 2020) and Italy (Ponti and Berbel, 2020). This reduced water abstraction has a positive impact on water bodies, both surface and groundwater; it presents a positive development in itself since it guarantees the avoidance of any rebound effect in water consumption (understood – as per Perry et al., 2017 – as increased consumption after the efficiency measures are implemented).

The bad news is that the level of cost recovery is still low in certain agricultural sectors; this is especially true for self-services (mainly groundwater abstraction) that seldom pay any tariff and are generally unmonitored and therefore not controlled by public administrations such as river basin authorities. Water pricing in the case of overallocated aquifers is probably less effective than governance measures along the lines of the *Organisme unique de gestion collective* (French Collective Management Association). Measures to monitor, control and protect the aquifer should nevertheless be paid for by the user through a levy or tariff, although this is not the case in most of the overexploited aquifers of the EU.

The reason behind the low level of cost recovery in agriculture is explained by policy makers' reluctance to increase tariffs and water taxation in the agricultural sector and by the question of affordability in this sector; in the majority of EU regions, agriculture is subject to increasing competition and decreasing income. During the winter of 2019-2020, protests by farmers in France, Ireland, Germany and Spain expressed the discontent of those working in this sector regarding policy changes; in their opinion, policy changes such as reform of the Common Agricultural Policy and trade preference agreements could threaten their livelihoods. The ability of those in the agricultural sector to influence politics is well known.

Household water charges usually represent a small percentage of average household income (or GDP per capita); water costs range from 0.2% of income in Oslo to 3.5% in Bucharest (EEA, 2019). A review of household pricing by the Joint Research Centre (JRC) concluded that the implementation of the full financial cost recovery (FCR) principle does not lead to substantial water affordability issues in most of the EU member states; however, the case of Bulgaria and, to a lesser extent Estonia, France and the UK, constitutes an exception since poor households (i.e. households belonging to the first income decile) must devote more than 3% of their income to paying their water and wastewater bills under an FCR regime (Reynaud, 2016). In the case of the UK, 15% of households declare that they are struggling to pay their water bills; this is especially the case for those located in South West England and Wales, where bills are particularly high and where 60% of households lack a water meter (OFWAT, 2011). The design of municipal water tariffs requires the balancing of multiple objectives, such as sufficiency and equity principles that do not overly burden poorer social groups, and societal economic efficiency. Evidence suggests that quantity signals, rather than pricing schemes based on block tariffs, provide a directive towards a specific behavioural response (Nauges and Whittington, 2017). Affordability issues, unfortunately, are most often used as the criteria to delay decision-making and implementation in both the urban and agricultural sectors; affordability criteria are also used to determine exemptions and derogations, such as the definition of disproportionate costs (Boeuf et al., 2016).

We conclude that the implementation of the WFD and of cost recovery principles may be eased with the adoption of a pragmatic approach, and that such an approach should be adopted in order to avoid the unfeasible task of monetising ERCs.

THE WAY FORWARD

The idea of water pricing and cost recovery was originated in the 1990s; it was understood as a silver bullet for solving problems involving the efficient allocation of water resources and reducing demand to more sustainable levels. In our opinion, it should be a necessary, but not sufficient, condition for the improvement of resource use and environmental services. Garrick et al. (2020) propose that three phases can be observed in economic thinking in water resources management. The first phase, which arose in the second half of the 20th century, focused on supply augmentation; the second phase, at the turn of the 21st century, focused on water pricing and water markets as the 'textbook solution' to handling water scarcity and achieving higher resource efficiency; the third and current phase recognises system complexity. Water pricing and cost recovery thus belong to the "second phase of economic thinking", according to these authors; they feel that we have already entered the third phase, whereby water as an economic commodity with multiple competing and non-competing uses, variability, uncertainty, and territorial and social impacts demonstrates that water management is a highly complex issue. They hold that the experience of water scarcity challenges faced by developed economies such as California, Australia and the EU has demonstrated the need for better planning and governance in order to address equity and other policy objectives (Berbel and Esteban, 2019). A similar conclusion is reached by Albiac et al. (2020); they argue that water pricing is limited as an instrument for water reallocation, and consider the use of policy instruments such as water markets and institutional cooperation to be more operational for water reallocation.

As for the aforementioned suggestion of the need to enter a 'third phase' in water policy thinking, water markets and water banks can provide a highly effective solution; we should, however, also learn from mistakes such as the Chile Water Code (approved in 1981) which failed to consider the environmental flow when allocating resources, and the errors committed in Australia, which continues to have problems guaranteeing resources for the environment. Water markets essentially consist of the decision to 'cap and trade', that is, to define a limit on the total usable amount of water resources, to allocate resources to users, and to grant flexibility in the trading of water rights. Critical for the sustainability of the model are the constraints imposed by the practical decisions on the level of 'cap' and the need to maintain an environmental flow (Hanemann and Young, 2020).

California, after centuries of mismanagement whereby certain aquifers and rivers have been overallocated by 1000%, is now tackling a legal revolution. The State has recently implemented the Sustainable Groundwater Management Act (SGMA) which involves legal reform that forces groundwater self-regulation, the definition of abstraction limits, and the regulation of water reuse, storage and recharging. It is an approach that is based on the self-regulation of local stakeholders under state monitoring and includes a long-term dynamic adaptation approach where quantifiable sustainability metrics constitute a key component of sustainable groundwater management (Dumas, 2019). The management of the Beauce aquifer in France presents a related approach in the EU (de Frutos Cachorro et al., 2017); it essentially requires the dynamic limitation of total abstractions, the sharing of available resources with a definition of environmental flow and priorities, and flexible trade between stakeholders. The increasing efficiency of remote sensing and sensors, and improvement in costs, make the monitoring and enforcement of abstraction limits more accessible (Loch et al., 2020).

Finally, it is worth mentioning that some non-consumptive uses such as energy uses (for example, refrigeration for power plants and hydropower) have not been subject to a detailed economic analysis despite having serious negative hydromorphological impacts. In this regard, the aforementioned

Hydropower Environmental Fund (Sweden) aims to internalise the negative impacts of the hydropower sector without the need to carry out a heroic estimation of these externalities.

CONCLUSION

The implementation of cost recovery should be seen as a general principle that follows the PPP included in the Treaty on the Functioning of the European Union. It is, furthermore, essential to guarantee funds for future and current investments, including water quality, prevention of extreme events, drought management, and reliability of systems; cost recovery may also exert influence on the efficient allocation of resources as an incentive towards behavioural change. This review has highlighted two distinct difficulties in the implementation of the CRP: the fuzzy and impractical nature of ERCs and the low price elasticity of the majority of water uses, especially in water scarce areas that are – paradoxically – the locations where water savings are most needed.

Water pricing is not the 'best and only instrument' for water reallocation; furthermore, the goal of good environmental status, which is the guiding principle of the WFD, can be achieved through the support of the economic instruments mentioned above, such as cost-effectiveness and cost-benefit analysis and the analysis of disproportionality decisions. We have, moreover, entered a new phase in water economics and policy. In this phase, the complexity of water systems should be better acknowledged through the use of a variety of instruments; the local involvement of the main stakeholders in the achievement of environmental and social goals should be promoted by the state; and clear, transparent and measurable objectives should be defined by reliable and practical methodologies. Once goals and resource limits are imposed, flexibility towards water reallocation can be introduced by using water markets and public water banks with clear functioning and equitable rules.

To conclude, the approval of environmental legislation such as that of the WFD constitutes collective action that involves costly negotiation and lobbying between multiple social groups. During the first 20 years of WFD implementation, there have been both lights and shadows. Article 9 illustrates both the difficulty of shifting from theory to practice and the need to enter a new phase that is characterised by a combination of various tools and instruments such as voluntary agreements, technology adoption, trade in water rights, multi-actor governance, planning, and control measures; this new phase is one where water pricing can play a role that is complementary to other economic instruments and takes a humbler and more pragmatic approach.

Water pricing should most certainly not constitute the only way to oblige the water user (and polluter) to pay for the cost recovery of water services; indeed, when it comes to ERCs, regulatory instruments and governance measures may prove to be more effective. Effective enforcement of Article 9 of the WFD may include a range of instruments such as groundwater abstraction limits, agricultural-environmental compliance initiatives (voluntary and mandatory), bans and discharge restrictions on certain pollutants, and obligations to restore or compensate for environmental damage.

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