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Justifying exemptions through policy appraisal: ecological ambitions and water policy in France and the United Kingdom

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Abstract

The Water Framework Directive aims to achieve 'good status' for all water bodies in the European Union. However, exemption clauses enable member states to delay protective measures and to lower water quality objectives. The ambiguity of exemption clauses has led to a plurality of approaches across the continent. They differ as to their political objectives, i.e., the overall ambition displayed in implementing the Directive, and to their methodological choices, i.e., the analytical tools used to justify exemptions. This article argues that those political and methodological dimensions influence each other. Relying on a framework of analysis that integrates key recommendations from the literature, we explore the usage and justification of exemptions in two countries, the United Kingdom and France. Our analysis suggests that analytical methods were often decided so as to reflect the ecological ambitions of a country, and some methodological choices seem to have had unintended consequences for water quality objectives. We conclude that economic methods should be adapted so that they take into account, rather than ignore, the political ambitions of a country in the field of water.

Keywords: Affordability; Cost–benefit analysis; Disproportionate costs; Exemptions; Water Framework Directive

Introduction

The Water Framework Directive (WFD, 2000/60/EC) represents a major shift in EU water policy from isolated attempts to reduce pollution from various *specific* sources and clearly *defined* types of water usage towards a more *holistic* approach. The Directive recommends or makes compulsory water management principles such as river basin management, public participation and economic analysis, with a view to preventing any further deterioration and achieving 'good status' for all surface and groundwater bodies.

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Specifically, the WFD obliges EU member states to draft River Basin Management Plans (RBMP), which specify water quality objectives for individual water bodies and justify exemptions. Programmes of Measures, published at the same time, identify the actions required to achieve these objectives. Water authorities operate within 6-year management cycles; this includes the initial drafting, update and implementation of RBMPs and Programmes of Measures. The first cycle started in 2009 when the first RBMPs were published. The second cycle began in 2015 with the update of the plans. The third cycle will last from 2021 to 2027.

However, exemption clauses enable EU member states to delay protective measures for up to 12 years (Art. 4.4 WFD) or to lower water quality objectives for individual water bodies, i.e., to reach 'less stringent objectives' (Art. 4.5 WFD). Member states may resort to these exemptions under three circumstances: if natural conditions are unfavourable, if the achievement of good status is technically infeasible, or if the associated costs are disproportionately high. They may also deteriorate water body quality to pursue projects of major general interest (Art. 4.7 WFD).

This article focuses on exemptions related to deadline extensions and less stringent objectives based on disproportionate costs only. Exemption clauses were widely used across Europe: for instance, up until 2009, deadline extensions were granted for 40% of all surface water bodies and for 11% of all groundwater bodies (European Commission, 2012b). Obviously, the use of exemptions has a major impact on the degree to which the overall aim of the WFD will be achieved. At the time of writing, more than 15 years after the WFD came into force, many EU countries are still a far cry from achieving good water status. Back in 2012, the European Commission (2012b) had estimated that only 53% of all water bodies would reach a good status by 2015. More up-to-date data are not yet available, but we have little reason to assume that these estimates were wrong. There are many reasons for this including technical (e.g., lack of knowledge), political (e.g., lack of incentive pricing) and economic difficulties (e.g., financial restraints) (European Commission, 2012a; Stanley *et al.*, 2012; Levraut, 2013). Yet exemptions certainly play a role here.

The term 'disproportionate costs' is somewhat ambiguous and the process of justifying exemptions not very well defined (Görlach & Pielen, 2007). This can be traced back to political disagreements during the negotiation phase of the Directive, almost 20 years ago. Even today, the legal status of the overall aim of 'good status', the extent to which exemptions should be relied on, and the economic tools used to justify disproportionality are still in dispute (Boeuf *et al.*, 2016). This has resulted in a plurality of approaches: on the one hand, member states differ greatly as to the overall ambition displayed in WFD implementation, i.e., the degree to which they would make use of exemption clauses (Bourblanc *et al.*, 2013). In other words, we observe diversity as to the *political* aspects of WFD implementation. On the other hand, EU member states rely on very different analytical tools to justify the presence of 'disproportionate costs', one of the conditions for an exemption clause (van der Veeren, 2010; Gómez-Limón & Martin-Ortega, 2013; Dehnhardt, 2014; Martin-Ortega *et al.*, 2014; Feuillette *et al.*, 2016). This suggests a high degree of diversity with regards to the *methodological* aspects of WFD implementation.

This article argues that *political* and *methodological* aspects are interrelated and cannot be separated from each other. Political ambitions may influence which analytical tools are used – and how; and tools, far from being purely technical and neutral, may have intended and unintended consequences for the political ambitions on the ground (Lascoumes & Le Gales, 2007). We will show that the ambitions of EU member states related to WFD implementation have shaped the analytical tools used and that these choices have influenced the protection standards of individual water bodies. Based on original data and extensive fieldwork in two EU member states, the United Kingdom (UK) and France, this article explores a widely understudied phenomenon: the politics of exemptions in WFD implementation and the role of 'disproportionate costs' therein.

In this way we introduce a novel argument to the literature on WFD implementation. So far, in-depth studies on the actual use and justification of exemptions and their relationship to the political ambitions displayed by a country remain scarce (Boeuf & Fritsch, 2016). Existing research tends to provide broad overviews across Europe (e.g., Görlach & Pielen, 2007; Klauer *et al*., 2007; Martin-Ortega *et al*., 2014; Maia, 2017). Some of them are already outdated. WFD management activities are organised in 6-year cycles, and works such as Gómez-Limón & Martin-Ortega (2013) explored the first management cycle from 2009 to 2015 only (and even here mainly the first 2 or 3 years). We know little about the second cycle and how water managers took into account feedback from the first management cycle. In fact, we are not aware of any study that has already looked into the second WFD cycle (i.e., 2015 to 2021). Other works offer recommendations based on academic experiments (e.g., Del Saz-Salazar *et al*., 2009; Vinten *et al*., 2012; Galioto *et al*., 2013; Perni & Martínez-Paz, 2013; Martin-Ortega *et al*., 2015; Klauer *et al*., 2016, 2017; Machac & Slavikova, 2016). Obviously, these works may provide great benefits to practitioners and researchers, but they say little about what is happening on the ground.

Analytical framework

The WFD does not properly define what 'disproportionate costs' are and how disproportionality should be established. Two methods – and thus two interpretations of this term – emerged from discussions at EU level. The costs of protective measures could be compared to the benefits provided to society through the improvement in water quality: disproportionality as a result of a cost–benefit analysis (CBA). Alternatively, costs could be compared to the ability of stakeholders to pay for protective measures: disproportionality as the inability of various sectors or polluters to afford the measures (Boeuf *et al*., 2016).

There is a rich academic literature making recommendations on how to undertake disproportionality analyses (Brouwer, 2008; Del Saz-Salazar *et al*., 2009; Martin-Ortega, 2012; Galioto *et al*., 2013; Gómez-Limón & Martin-Ortega, 2013; Martin-Ortega *et al*., 2014; Feuillette *et al*., 2016; Klauer *et al*., 2016, 2017). While these studies differ in important ways, they have one thing in common: they acknowledge that CBA and affordability tests are multi-dimensional. Essentially, the comparison of benefits and costs lies at the heart of every CBA, and so does the juxtaposition of costs and available resources in affordability tests. In order to carry out those tests, however, environmental economists are required to consider a range of decisions which govern how precisely the method will be put into practice (Pearce *et al*., 2006; Davidson, 2014). Our argument is that these decisions not only define the operationalisation of the method, but may also influence analytical outcomes. The contents of these decisions form what we call here the 'dimensions' of CBA and affordability tests.

We select five dimensions from the literature: scale, screening, costs and benefits data, uncertainty, and additional parameters. They were selected for three reasons: First, they are comprehensive, i.e., taken together, they cover all the technical aspects related to CBA, to affordability tests, or to both. Second, they may be applied globally and enable cross-country comparisons. Third, they all depend on the degree of ambition displayed by an EU member state for implementing the WFD, and their precise operationalisation may influence the process of setting objectives.

The overall function of these dimensions in this research therefore is to *unpack* two complex analytical tools – CBA and affordability tests – and to provide the signposts needed to understand the application of these tools in diverse empirical settings. The above dimensions have no normative meaning here, i.e., we use them to anatomise, dissect and examine rather than to assess and evaluate. In doing so, these dimensions provide a structure for our case study analysis and lay the foundation for the argument that we wish to make: First, we compare the choices that water managers in England and France have made with regards to each dimension. Second, we explore the relationship between these choices and the political ambition displayed by each country. We describe these five dimensions below.

Scale

Both CBA and affordability tests are performed on a specific geographical perimeter. In the case of WFD implementation, at least four hydrographical units could be considered: the water body, the catchment or sub-catchment, the river basin, or the national scale.

Screening

CBA and affordability tests could be performed systematically and consistently for each hydrographical unit. Alternatively, one may attempt to limit the number of units analysed or to reduce the depth of the analyses. Preliminary screenings support a decision here and, in doing so, save resources. For example, water managers may want to identify hydrographical units where implementation costs are likely to be disproportionately high.

Costs and benefits data

Data are a necessary input to both CBA and affordability tests. Here, we focus on costs and benefits data. They may be assessed qualitatively, quantitatively (but not monetised) or monetarily. Costs include investment, operating, administrative and environmental costs as well as income reductions. Benefits involve market and non-market benefits and typically inform CBA only. Finally, we examine whether benefit transfers were used. Benefit transfers apply benefit values estimated from a particular location to another location with similar characteristics. This method is often used when local data are unavailable, but it comes with obvious methodological weaknesses (Klauer *et al.*, 2016).

Uncertainty

Uncertainty is a common feature of environmental policy-making processes. In WFD water management, this may refer to the status of water bodies (and therefore to the nature and costs of measures that should be implemented), the effectiveness and efficiency of measures, input data, the monetisation of benefits and costs, and methodological limitations related to the use of benefit transfers. Here, we consider whether and how these uncertainties have been taken into account when assessing disproportionality.

Additional parameters

We consider here various methodological decisions to operationalise CBA and affordability tests. For CBA, this includes the cost–benefit ratio, i.e., the threshold distinguishing proportionally and disproportionally high costs. Economic theory suggests that the cost–benefit ratio should be 1. We also consider the rate used to discount future benefits and costs. Discount rates respond, among others, to the insight that many people prefer short-term over long-term gains and long-term over short-term costs. A high discount rate gives more weight to current expenses while a low discount rate favours long-term benefits. Therefore, the discount rate has an ethical dimension because it determines the extent to which the interests of future generations are taken into account (Martin-Ortega *et al*., 2014, 2015). We also study which categories of users, criteria and thresholds were used in affordability tests.

Data and methods

This article compares the UK (specifically England) and France, two countries that have relied extensively on disproportionate costs to justify exemptions (Levraut, 2013; Environment Agency, 2015).

In England, economic analyses were performed consistently across the country, up until 2015 at national and after 2015 at catchment level. We therefore explore the national level, one representative river basin and one equally representative catchment: the Humber basin and the Aire and Calder catchment, respectively.

Economic analyses in France, on the other hand, differed significantly from one river basin to another. Consequently, this research focuses on the national and the river basin level whereby all river basins in mainland France and Corsica were investigated, namely Adour-Garonne, Corsica, Loire-Brittany, Meuse, Rhine, Rhone and Coastal Mediterranean, Sambre, Scheldt, and Seine-Normandy. We do not take into account the French overseas territories.

This research examines the first and the second WFD management cycle, i.e., economic analyses carried out to support the 2009 and 2015 RBMPs. To this end, we analysed 77 policy documents drafted between 2003 and 2016 by policy-makers at the local, regional and national level in the UK and France as well as at EU level. Furthermore, we conducted, transcribed and analysed 32 semi-structured interviews with state and non-state actors directly involved in the implementation of the WFD in these two countries. Tables A and B in the Supplementary materials (available with the online version of this paper) provide a complete list of interviewees and policy documents.

Political ambitions and objective setting in England and France

This section discusses the general ambition displayed by England and France during the implementation of the WFD. RBMPs and Programmes of Measures are ‘ambitious’ when they set objectives that are significantly higher than the initial situation – and ‘cautious’ when this is not the case. We use the terms ‘ambitious’ and ‘cautious’ neutrally, with no positive or negative connotations.

England

In each constituent part of the UK – England, Northern Ireland, Scotland and Wales – a designated non-departmental public body manages the water environment and, therefore, produces RBMPs and performs economic analyses. In England this is the Environment Agency (EA), which carried out this task from six regional offices until 2014 and, since then, from 14 area offices. The Department for Environment, Food and Rural Affairs (Defra) is legally responsible for the timely and correct implementation of the WFD. Defra's Secretary of State approves the final RBMPs, including the WFD water quality objectives (INT-EN01). This suggests that the preparation of RBMPs in England is very much centralised.

Water managers in England take a cautious and pragmatic approach to WFD implementation. In the first management cycle, 26% of all surface water bodies were monitored to have a good or high ecological status or potential. The aim was to reach good ecological status in 30% of all water bodies by 2015. In the second cycle, however, the EA aimed to increase the proportion of surface water bodies with a good ecological status from 17% monitored in 2015 to 21% in 2021, and to reach a less stringent objective for ecological status in 25% of all cases (Environment Agency, 2015). This could suggest that water quality deteriorated between 2009 and 2015. However, the changed figures are mainly due to a re-designation of water bodies, resulting in a decrease in the overall number of water bodies, and to more comprehensive monitoring data from further investigations. Moreover, if water managers were uncertain whether necessary measures could really be implemented, they resorted more systematically to exemptions in the second cycle, specifically deadline extensions (INT-EN01).

This suggests that water managers in England interpret the WFD as an obligation to *aim* to achieve good status (except for exemptions), i.e., a 'best effort approach' (Bourblanc *et al.*, 2013: p. 1457). In other words, the English approach to the WFD aims to avoid over-implementing the Directive – also known as gold-plating (Jans *et al.*, 2009). This stands in contrast to the politically motivated ambition to implement the WFD beyond minimum requirements in France, as we will explain later.

According to Bourblanc *et al.* (2013: p. 1465), 'the more politicians and policy makers feel they are held accountable by EU institutions, the more the level of ambition will be adjusted to the perceived adequate implementation process in front of the EU'. Water managers in England see the implementation of the Programmes of Measures, rather than the achievement of good water status, as a legally binding requirement. They therefore prefer to adopt Programmes of Measures that are likely to be implemented even if – or better, exactly because – they display a certain lack of ambition (Dieperink *et al.*, 2012; INT-EN04).

The degree of caution expressed here is highly compatible with the reluctant position that the UK has generally taken towards European integration and the level of scepticism shown as to the benefits the EU can provide to member states. The UK government has always sought to avoid 'gold-plating' during the transposition and implementation of EU law and, to this end, encouraged ministries, departments and independent regulatory agencies to apply EU standards to the minimum so as to minimise costs and efforts where they are not justified in terms of benefits (Knill, 2001; Wurzel, 2002; Fritsch, 2011; UK Government, 2015).

France

Water management in France is decentralised, which is why the river basin level deserves particular attention. In each basin, a River Basin Committee brings together elected policy-makers at the local level (40% of all seats), water users (industry and commerce, agriculture, recreation, environmental movements, water consumers, 40%) and non-elected officials from local authorities (20%). Supported by one of the six water agencies – public bodies operating at regional level under the responsibility of the Ministry of Environment – each Committee defines the water management priorities in their basin, establishes the overall aim (i.e., the percentage of water bodies that should reach good status by the next deadline) and recommends the budget available to implement the Programme of Measures (INT-FR07, INT-FR10, INT-FR18, INT-FR23, INT-FR25, INT-FR27). The water agencies determine the water quality objectives for individual water bodies. The River Basin Coordinating Prefect, a state representative at the regional level, then approves the RBMP (Levraut, 2013). The Ministry of Environment coordinates this work, being legally responsible for the implementation of the WFD (Levraut, 2013).

In contrast to water managers in England, authorities in France generally set ambitious water quality goals which have been more difficult to achieve (Levraut, 2013). The *Grenelle de l'environnement*, a political convention that included members of civil society and took place in 2007, decided that two-thirds (in practice 64%) of all surface water bodies should be in good ecological status by 2015. This effectively translates into a legally binding commitment to restrict the use of exemptions to one-third of all surface water bodies or less – an ambitious, symbolic target that had a major influence on the planning process at river basin level (INT-FR12). In 2009, 41% of all surface water bodies were already in good ecological status (Levraut, 2013). France aimed to increase this figure by another 23%. In 2015, only 44% of all surface water bodies were in good ecological status, and the new aim was to improve this figure to 66% by 2021 (INT-FR17). However, figures of water bodies in good status are not quite comparable between the first and second cycles. This is because the guidelines to assess the status of water bodies have changed in between. Both in the first and the second cycle, water managers preferred deadline extensions over less stringent objectives to justify exemptions (INT-FR17).

Bourblanc *et al.* (2013: p. 1449) offer several reasons for the different approaches taken in England and France. The 'visibility of the policy process', not the least thanks to the highly political, public role played by the *Grenelle de l'Environnement,* is of particular importance when it comes to understanding the high ambitions pursued in France. Another factor is 'the division of responsibilities'. Although the River Basin Committees, supported by the water agencies, set the objectives, the Committees are not responsible for their achievement and funding. Usually, local authorities are in charge of implementing the measures. River Basin Committees therefore do not necessarily feel accountable for the objectives they set. The authors also argue that accountability towards the European Commission matters. In contrast to the UK, pro-European sentiments are a defining element of France's international identity, and the country is genuinely committed to achieving policy goals set at EU level. It should be noted, though, that its performance has always been somewhat less impressive in the environmental field. The European Commission repeatedly initiated infringement procedures against France, and it is plausible to assume that the high ambitions pursued by France in the water sector were and are an attempt to improve its reputation (Bourblanc *et al*., 2013).

In short, the UK and French approaches to the WFD stem from two different policy and administrative stances. We will now show how the economic analyses performed to justify exemptions reflect these differences.

Operationalising disproportionality analyses

In our two case studies, the logic behind exemptions and their justification differed substantially. We also observe evolution over time, i.e., between the two management cycles.

In England, water managers primarily referred to the uncertain status of water bodies to justify exemptions in the first cycle (Environment Agency, 2009). Uncertainty comes with the risk that costs would outweigh benefits and that public investments be misspent for unnecessary or ineffective measures. Water managers thus favoured deadline extensions to collect more data on the status of water bodies and spread the costs of measures over time (Defra, 2009). In the first cycle, economic analyses therefore played a minor role only in exemption-related decisions. The European Commission and environmental movements criticised this extensive reliance on uncertainty as a basis for exemptions (INT-EN10; INT-EN18). Defra responded by publishing a statement of position which, among others, committed to enhance their water quality data so as to avoid legal action from the WWF and the Angling Trust (INT-EN10; INT-EN15; INT-EN18). In the second management cycle, economic analyses played a more prominent role. The EA trained their area staff to perform CBA on each catchment and used these analyses to define the level of ambition (good status or less stringent). When funding was not readily available for necessary measures, Defra would apply for a deadline extension (INT-EN01).

In France, River Basin Committees were constrained in so far as they were obliged to pursue the national target set by the *Grenelle de l'Environnement*, according to which two-thirds of all surface water bodies were to be in good ecological status by 2015 (INT-FR12). Economic analyses therefore were not only designed to identify and justify cases of exemptions, but also to limit their number. However, we observe a considerable degree of variation across river basins as to the methods used to justify the use of exemptions. Analysts performed over 700 CBA in total (Feuillette *et al.*, 2016). Water managers largely preferred deadline extensions over less stringent objectives in order to stick to higher ambitions. At the end of the first cycle, the European Commission criticised France for the lack of available justification for exemptions (Levraut, 2013). In the second management cycle, the Ministry attempted to harmonise methods across river basins and asked to make economic analyses publicly available (INT-FR17). However, not all water agencies complied.

We now apply our framework of analysis to each country. We offer a summary of our findings in Table 1 and provide additional information in Table C in the Supplementary materials (available with the online version of this paper).

Scale

Water managers in England and France operated at different scales to perform economic analyses and set water quality objectives. In the first management cycle, analysts in England mainly performed economic appraisals at the national or river basin scale as part of an impact assessment of the RBMPs (INT-EN01). In the second cycle, EA staff performed CBA at sub-catchment scale (the number of water bodies within these sub-catchments varied), close to each other or with similar activities impacting them (INT-EN05). In France, the water agencies conducted CBA and affordability assessments at the water body, catchment (groups of around ten water bodies) or river basin scale (INT-FR02; INT-FR09; INT-FR14; INT-FR22; INT-FR23; INT-FR27). While the EA tried to optimise the scale used for the analysis in the second cycle in order to balance the level of detail with the number of analyses, authorities in France were less concerned about this aspect.

However, scale matters. On the one hand, authorities operating at larger scales reduce the number of analyses and therefore save time and resources. Moreover, analyses at larger scales reduce the risks of double counting costs and benefits that apply to several water bodies (INT-EN05). To illustrate, let us consider a factory that is located at a particular water body and that pollutes another water body as well.

Table 1. Synthesis of findings.

Dimension	England	France
Approach	Cautious (−)	Ambitious (+)
Scale	1st cycle: national and river basin (potentially -). 2nd cycle: sub-catchments (0)	Both 1st and 2nd cycle: water body, catchment and river basin level (+/-)
Screening	1st cycle: decision trees, no in-depth analysis. 2nd cycle: 'triage' approach consisting of a qualitative description of measures that impact on ecosystem services, stage 1: CBA with NWEBS benefit values, stage 1 + : CBA with wider benefits, stage 2: site-specific valuation. (overall: potentially -)	Both 1st and 2nd cycle: various criteria used including the ability to pay, cost thresholds, past expenditures and non-priority measure (+)
Data	1st cycle: range of costs not monetised (+), NWEBS benefit values (+). 2nd cycle: more costs assessed (0), NWEBS and qualitative assessment of ecosystem services (+)	Both 1st and 2nd cycle: incomplete database of benefits (−), use of benefit transfers (−), benefit values applied to population densities (−)
Uncertainty	Both 1st and 2nd cycle: uncertainty in favour of deadline extensions (−)	Both 1st and 2nd cycle: uncertainty in favour of good status (+)
Additional parameters	CBA used to justify less stringent objectives	CBA used to justify deadline extensions and in a few cases less stringent objectives
	Both 1st and 2nd cycle: discount rate 3.5% over 30 years, then 3%; if 0.5 < cost–benefit ratio < 1.5 in stage 1, perform stage 1+ (2nd cycle) (0)	Both 1st and 2nd cycle: cost–benefit ratio = 0.8 (+). Discount rate: 1st cycle: 4% (−), 2nd cycle: 2.5% (+)
	Affordability: disproportionate burdens: 2nd cycle: deadline extensions set when no secure funding was available (−)	Affordability: Both 1st and 2nd cycle: criteria and thresholds used (+ when used in addition to CBA to set deadline extensions, in this case, both analyses had to show negative results, 0 when affordability was a sufficient criteria to set a deadline extension)

Reducing the pollution load, for example by building a treatment plant, will incur costs for the factory. These costs would be considered for the water body where the factory is located. However, the benefits accrue to both water bodies. The overall analysis would be faulty if the analyst took into account these costs in CBA for both water bodies: this would be double counting. On the other hand, analyses at smaller scales may consider more robust local data. The catchment scale thus seems to be ideal if one wants to increase the robustness of the analysis and avoid an overestimation of costs or benefits. At the same time, this practical problem raises legal questions: Art. 4.4 and Art. 4.5 WFD require decision-making and reporting at the water body scale. However, there is disagreement as to whether the underlying analysis must be performed at the water body scale as well. So far, this ambiguity has not yet been resolved legally.

Screening procedure

In order to assess whether measures to improve the quality of each water body would incur disproportionate costs, economists have the choice between detailed disproportionality analyses on each hydrological unit or screening procedures. The latter enable analysts to sort and group cases, but also to select the water bodies on which a detailed assessment should be undertaken. Due to time and resource constraints, both countries used screening procedures; however, their screening processes differed substantially.

In England, in the first cycle, water managers used decision trees to sort cases and decide upon exemptions and their justification: unfavourable natural conditions, technical infeasibility, or disproportionate costs (see Figure 1). Analyses related to disproportionate costs were usually performed at national level, i.e., showed little context sensitivity, and were generally not very detailed (Defra & Environment Agency, 2009).

In the second management cycle, area EA offices applied a step-wise procedure or 'triage approach' to perform in-depth analyses only if they were absolutely necessary and the expected impacts high (Environment Agency, 2014: p. 8). In a first step, analysts would identify and describe the potential impacts of different bundles of measures; no monetisation was envisaged at this stage. They estimated the expected (dis)benefits using a scale from 'significant' to 'noticeable' and 'no net change', and compared them to the 'do-nothing option'. The second step, a 'stage 1' valuation, took into account a range of monetised benefits and explored which bundles of measures were particularly cost-beneficial or not.

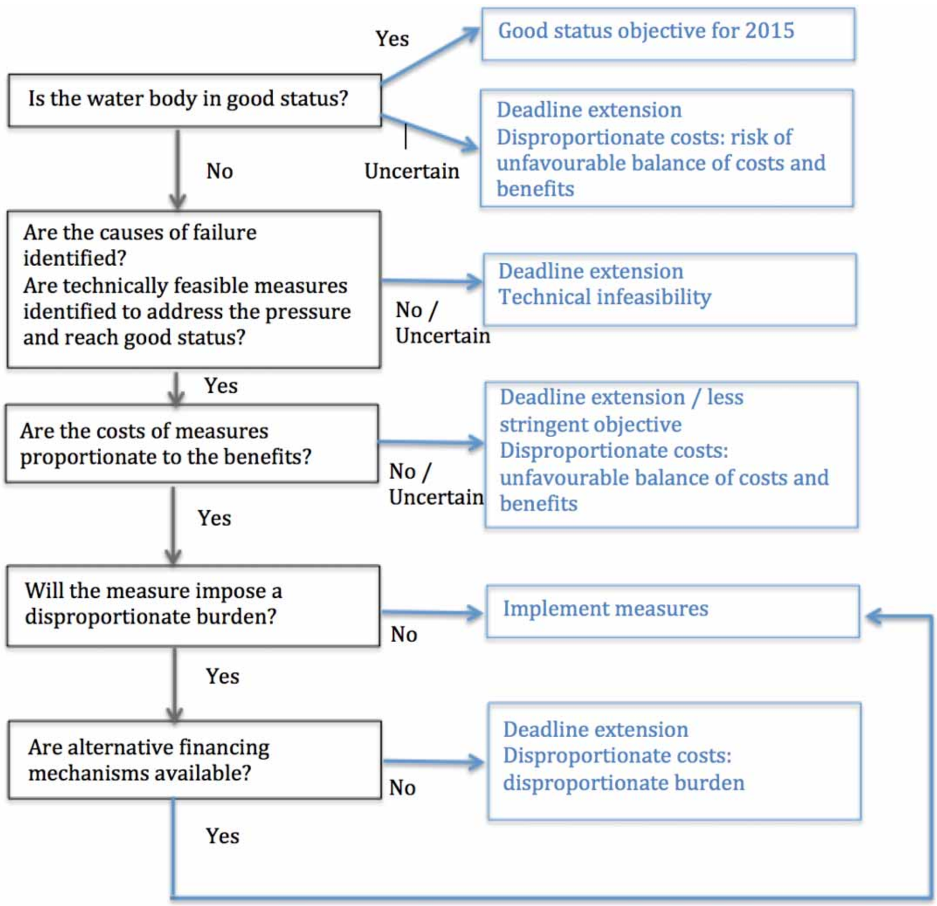

Fig. 1. Summary of the main steps used by the EA in its decision trees to decide on exemptions in the first management cycle. *Source:* authors.

If necessary, a 'stage 1 + ' valuation was performed. This analysis included a more comprehensive range of monetised benefits identified during the qualitative description. Finally, analysts could perform a 'stage 2' site-specific valuation if the previous results were inconclusive (Environment Agency, 2014) (see Figure 2). This advanced appraisal method was rarely used in practice, since stage 1+ analyses were usually satisfactory (INT-EN01).

The water agencies in France used different screening criteria. This included stakeholder ability to pay, the costs of measures compared to past expenditures, particularly high costs incurred by a specific type of measure, and cost thresholds (INT-FR09; INT-FR23, INT-FR27). In the second management cycle, national guidance recommended CBA when measures were not a priority and where affordability tests produced negative results (Commissariat Général au Développement Durable, 2014).

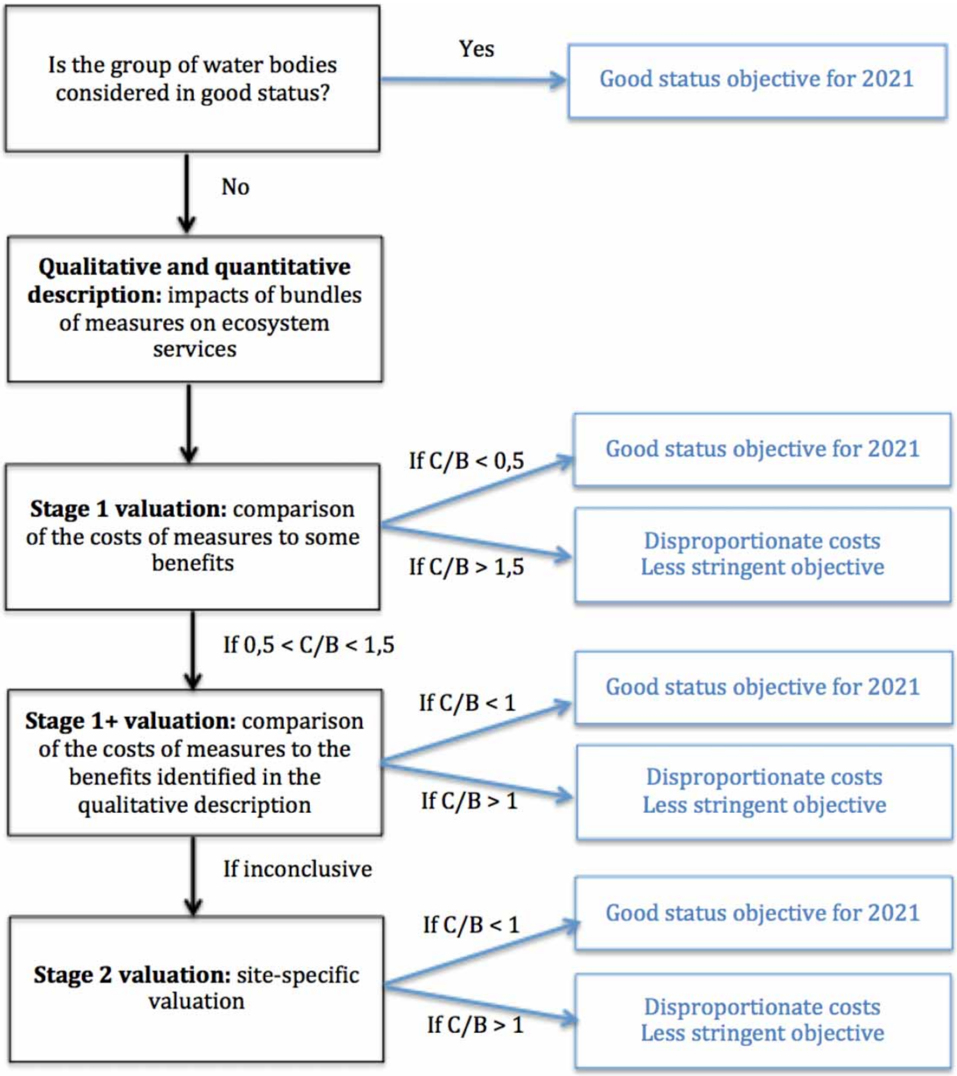

Fig. 2. Main steps used by the EA to decide on exemptions in the second management cycle. *Source:* authors.

Screening procedures may have a profound impact on management decisions. In England, the EA used a staged approach to determine the depth of the analysis. Analysts thus undertook a more or less comprehensive CBA for most water bodies. Because 'stage 1' valuations did not take into account the full range of benefits, this process could lead to the exemption of water bodies where protective measures would actually come with a positive cost–benefit ratio. In other words, the EA's staged procedure, relying on a subset of potentially available data, resulted in a more cautious approach when it came to objectives and exemptions. That said, a preliminary study published by the EA (2013) concluded that the results of 'stage 1' valuations did not significantly differ from more in-depth assessments. Consequently, the relevance of this factor should not be overestimated.

In France, however, analysts used screening criteria to select water bodies on which to perform a CBA. This approach had the advantage of reducing the number of analyses to be performed. However, it also limited the potential number of exemptions. As such, it favoured a more ambitious interpretation of the WFD. For example, applying a cost threshold means that measures with low costs, but also potentially low benefits, were not eligible for an exemption. The diversity of screening criteria used in France also shows that they are more relevant if tailored to local characteristics. In the Rhone and Coastal Mediterranean basin a cost threshold was used due to the geography of the river basin. While protective measures were inexpensive in mountainous areas with low human pressures on water bodies, actions were costly in densely populated and industrialised cities (INT-FR27). Another example is Loire-Brittany, where water pollution through agriculture is a major problem, which was therefore explicitly flagged up for an economic appraisal (INT-FR23).

Costs and benefits data

Costs and benefits data constitute a crucial input to economic analysis. They may differ as to their nature (qualitative, quantitative or monetary), their source, their quality and their scale. All these characteristics may influence water management decisions.

In the first management cycle, the EA extracted cost-related data from water company business plans (INT-EN03; INT-EN06), earlier impact assessments and in-house sources; for instance, data collected through permits. However, analysts did not consider all costs (Defra, 2009). In the second cycle, the Agency tried to broaden the data available to the analyses (INT-EN03) based on in-house sources and used a database on agricultural activities and pollutants that would subsequently inform CBA (INT-EN08). Although EA staff were encouraged to use local costs (INT-EN05), analysts often relied on national databases that did not always accurately reflect local realities (INT-EN21). With regards to benefits, the EA relied on the National Water Environment Benefits Survey (NWEBS), which elicited preference values from 1,487 people in 50 locations and valued aesthetic, biodiversity and recreational benefits of water status improvement. In the first management cycle, EA economists used these values to prepare national and regional impact assessments (Metcalfe *et al.*, 2012; Environment Agency, 2013). In the second cycle, EA staff integrated an updated version of these benefit values into the stage 1 valuation process (Environment Agency, 2014). Furthermore, a qualitative assessment was made to better take into account non-monetised and non-market benefits. As a cogent example, the concept of ecosystem services, which informed valuations, was used to frame this plurality of benefits in assessments (INT-EN05; INT-EN09).

In France, economists calculated investment and maintenance costs based on databases developed by the water agencies, expert assessments, in-house and external studies and local data (INT-FR06;

INT-FR09; INT-FR23). In the second management cycle, water agencies enhanced the quality and quantity of their data on costs, in particular through additional studies, e.g. on hydromorphological measures (INT-FR27; INT-FR25). When it comes to benefits, the Ministry prepared a systematic review of valuation studies so as to build a national database of non-market benefits (angling, kayaking, bathing, windsurfing, hiking, observing, boating) and non-use values (property values). Market benefits mainly refer to the costs saved on drinking water treatment and generally weighted for more than 50% of the total benefits (Feuillette *et al*., 2016). Unfortunately, the Ministry only found about 40 studies and was unable to assess many categories of benefits. It then saved those benefit values that could be extracted from the academic literature, as incomplete as they were, in a Microsoft Excel tool designed to perform the CBA (Feuillette *et al*., 2016; INT-FR13). Consequently, some benefit categories, in particular non-market benefits, were not systematically considered during the CBA although they constitute, in an ideal world, an important element of disproportionality analyses. In order to establish the benefits of protective measures in a specific water body, the analyst would then select the most relevant non-market benefit values and multiply the Ministry's default value by the number of water users. The Ministry suggested the use of local data sources to establish the number of water users, such as surveys on site visits. In practice, however, analysts relied on generic figures of the population near a water body (Feuillette *et al*., 2016). Some water agencies also prepared local studies to improve the data (INT-FR09; INT-FR23; INT-FR27). In the second management cycle, the Ministry updated its systematic review through the inclusion of new publications, although they were not numerous (Commissariat Général au Développement Durable, 2014).

The approach followed in England seems to have favoured more ambitious water quality objectives than the one pursued in France. This is because EA staff did not take into account all the costs related to the achievement of good water status while the parallel usage of NWEBS and additional qualitative analyses provided a comprehensive overview of all the benefits. Unsurprisingly, this approach increased the cost–benefit ratio. In France, in contrast, the database on benefits was patchy, and non-market benefits were rarely taken into account, favouring a less ambitious implementation of the WFD. This factor may partly explain why only 25% of all CBA had a negative cost–benefit ratio in England (Environment Agency, 2015), as compared to 75% in France. Obviously, this conflicted with the high ambitions associated with WFD implementation in France. Water economists therefore criticised the method used for the valuation of benefits and promoted a more qualitative approach (Feuillette *et al*., 2016).

Using benefit transfers seems to be unavoidable if one faces a large number of water bodies. However, analysts in England appear to apply this method in a more accurate way than in France. This may explain why economists in the French water agencies criticised the use of benefit transfers. The basis on which authorities in France applied benefit values was particularly problematic. Analysts would use the number of residents near a water body, so that areas with a smaller population density were heavily penalised (Feuillette *et al*., 2016). This approach favoured a less ambitious implementation of the WFD. We do not make similar observations in England where analyses were carried out at the catchment rather than the water body scale. This is because average population densities are generally more homogenous at larger hydrographic scales. Moreover, analysts at the local level included upper bound benefits values and looked at wider benefits for scarcely populated areas with a high non-use value. Finally, EA staff did not only consider upstream–downstream issues in their economic analyses, but also during the planning process (monitoring and determination of the water status and subsequent measures) (INT-EN05). The use of benefit transfers was thus less problematic in England than in France.

Uncertainty

Both countries considered uncertainties during the whole planning process. This includes uncertainties related to the status of water bodies, to activities impacting the aquatic environment and to the efficiency of measures. However, England and France responded very differently to their presence, and these responses reflect the different ambitions of these countries associated with WFD implementation.

In the first cycle, the inability to accurately assess the current status of water bodies, the reasons for a degraded status and the necessary measures were a key reason for water managers in England to request exemptions based on disproportionate costs. Obviously, uncertainties related to the water status may result in uncertainties as to the nature, effectiveness and efficiency of measures taken to improve water bodies (Environment Agency, 2009). Accordingly, analysts were trying to avoid the possibility that the costs outweigh the benefits if inappropriate and inefficient measures were to be taken. In order to win time for additional research, regulators preferred deadline extensions to less stringent objectives (Defra, 2009). Although in the second management cycle uncertainty was less central to disproportionality analysis, EA staff continued to take into account uncertainties when they prepared the 2015 RBMPs. For example, they discounted benefit values based on their level of confidence in the data describing the water status (INT-EN08). Consequently, EA analysts took uncertainties into account to avoid misspending (Defra & Environment Agency, 2009), resulting in a cautious approach to setting water quality objectives.

In line with the French commitment to implement the WFD to a high standard, the overall approach was to avoid exemptions for less stringent objectives unless the impossibility of reaching good status by 2027 had been proven (Ministère de L'Ecologie de l'Energie du Développement Durable et de la Mer, 2009). Consequently, most exemptions requested were deadline extensions. As in England, the idea was to gain time to increase the scientific knowledge base. Water agencies even pursued the objective of good status for several water bodies characterised by high degrees of uncertainty (Levraut, 2013; INT-FR10). Moreover, analysts used a cost–benefit ratio of 0.8 to account for the possibility that benefit values were underestimated, resulting in rather ambitious objectives in case of uncertainty (Ministère de L'Ecologie de l'Energie du Développement Durable et de la Mer, 2009).

Additional parameters

Several additional parameters were used in both countries to operationalise the CBA and the affordability tests. These include the discount rate and the cost–benefit ratio in CBA and various indicators and thresholds in affordability tests.

In England, analysts used a discount rate of 3.5% for the first 30 years and 3% for any subsequent years, in accordance with guidance from the Treasury (HM Treasury, 2003). The cost–benefit ratio was primarily used in stage 1 valuations in screening procedures: if the cost–benefit ratio was between 0.5 and 1.5, economists would perform a stage 1+ valuation (Environment Agency, 2014). In France, analysts used a cost–benefit ratio of 0.8 and a discount rate of 4% over 30 years in the first management cycle and of 2.5% in the second (Commissariat Général au Développement Durable, 2014).

The discount rate used in France in the first management cycle was thus higher than in England. This resulted in a higher number of exemptions in France, because it valued future benefits less. However, France changed the discount rate in the second cycle; in fact, it is lower than in England now. This change favoured more ambitious water quality objectives and is well in line with the ambitious take on WFD implementation in France. In England, the discount rate was medium, remained stable over time and therefore had only a moderate impact on the result of the analyses. In doing so, England followed the conventional approach, taken from welfare economics, of determining economic efficiency when the benefit–cost ratio is greater than 1, i.e., when discounted benefits outweigh discounted costs. In contrast, water managers in France chose a cost–benefit ratio below 1, which favoured benefits over costs, i.e., more ambitious targets.

Regulators in England interpreted the inability to pay as a 'disproportionate burden' (Defra & Environment Agency, 2009: p. 8). In the first management cycle, EA analysts used this argument to justify exemptions in two cases only. The first one concerns water bodies polluted by abandoned mines. Analysts decided to spread costs over time so that expenditures would match available public funding. The second case relates to water bodies awaiting the installation of fish passes. Deadline extensions then served to gain time with a view to identifying additional sources of funding in the public and private sector (Defra & Environment Agency, 2009).

Water managers relied much more extensively on disproportionate burdens in the second cycle. They set the 2021 objectives on the basis of Programmes of Measures that could be delivered with budgets and policies that were already in place. Measures with no reliable and credible funding were not presumed to be deliverable. The authorities did not consider other, insecure funding sources at this stage. This practice is at variance with previous agreements at EU level. So far, the European Commission has not commented on its lawfulness. For example, the financial amount that the water industry may spend on environmental protection measures is agreed together with Ofwat, the regulating body of the privatised water and sewerage industry, in so-called periodic reviews. These processes take place every 5 years and are disconnected from the WFD management cycle (INT-EN07; INT-EN16; INT-EN18). Consequently, it is difficult to anticipate how much the water industry will be able to spend on WFD measures in the future. Likewise, achieving good water status may require additional legislative activities, budgetary reallocations, funding applications to the Treasury, and decisions taken by other ministries and government departments, all having uncertain outcomes. While exemptions based on less stringent objectives relied on economic analyses, exemptions requesting an extension of deadlines were based on affordability tests (INT-EN01). The question of who would pay for those measures was, first and foremost, explored in impact assessments (INT-EN09). Our analysis suggests that a majority of the costs would be borne by the water industry and national government. Consequently, the English approach to affordability was extremely cautious, in line with the British take on WFD implementation. The 2021 objectives set were based on secure funds and existing policies.

In France, water agencies assessed the ability to pay thanks to a set of indicators for each sector and defined thresholds in order to determine when costs would be seen as disproportionate. To illustrate, costs were considered unaffordable for households if the water bill exceeded 3% of their income (Ministère de L'Ecologie de l'Energie du Développement Durable et de la Mer, 2009). The Water Agency Rhine-Meuse used a particularly elaborate method to assess affordability: for each sector, economists would assess the costs of protective measures. Several indicators would then be calculated and thresholds applied. Those had been agreed prior to the assessment with the River Basin Committee and affected stakeholders (INT-FR14). A more detailed assessment of the indicators and thresholds used in Rhine-Meuse is available in the Supplementary materials (available with the online version of this paper).

Authorities in France assessed affordability in very different ways. Affordability tests did not refer to the availability of funding, but to indicators developed for each sector or stakeholder. This approach was much more ambitious than the British one, particularly in river basins where affordability tests were used in combination with CBA results. In such cases, action would be taken even if the costs were higher than the benefits, as long as there was evidence that stakeholders could afford protective measures. Some water agencies, however, were not fully convinced by the indicators and thresholds used (see the Supplementary materials for an example of the gross operating surplus of farmers). Those thresholds were often considered to be non-discriminating, i.e., almost all measures would then be above or below the threshold (INT-FR22; INT-FR23; INT-FR27). The case of Rhine-Meuse is thus particularly interesting because the Agency chose indicators and thresholds that were specifically tailored to local circumstances and the stakeholders' concerns. Thanks to this analysis, economists in France took into account distributional effects and the impacts of the costs of measures on each sector.

Summary of our findings

Table 1 summarises our findings for England and France and indicates whether methodological choices resulted in more ambitious (+), more cautious (−) or neutral (0) water quality objectives.

To sum up, our analysis shows that the above five dimensions do affect the results of disproportionality analyses and may serve to set more or less exemptions:

- Scale influences the number of analyses performed, the risk of double-counting benefits and costs and the robustness of data used in the analysis. In our view, the catchment scale is preferable here.
- Screening procedures determine the depth of the analysis performed and, in doing so, the degree of precision of costs or benefits data. Furthermore, screening procedures, if strictly used, reduce the number of analyses and therefore of potential exemptions.
- The quality and quantity of data related to benefits and costs has, according to our analysis, the greatest impact on the result of CBA. The lack of benefits data and the sensitivity of the analyses to the population living near a water body largely explain the numerous negative CBA results in France. This is independent from the discount rate and the cost–benefit ratio.
- Uncertainties are used in two contradictory ways: as an argument to justify exemptions, with a view to avoiding disproportionally costly measures, or to set ambitious aims for individual water bodies because an exemption cannot be justified on the basis of the data available.
- Finally, inability to pay can either be used alone to support deadline extensions, thus making the justification easier, or, on the contrary, in addition to CBA to diminish the number of possible exemptions.

As argued above, data related to costs and benefits appeared to have the greatest impact on the results of economic analyses. Surprisingly, it is the only dimension where England generally displayed greater ambition than France. Nevertheless, in England, benefits are more likely to be higher than the costs. Because the outcomes of those analyses were not in line with the general approach towards WFD implementation dominant in France, French regulators, favouring ambitious water quality targets, complemented CBA with additional criteria to tilt the scale against the use of exemptions. This includes requirements to identify additional arguments for exemptions; for instance, unfavourable natural conditions or technical infeasibility, the use of thresholds to limit the overall number of water bodies associated with disproportionate costs, and combinations of CBA and affordability tests. Overall, the high number of CBA displaying higher costs than benefits in France has certainly been a cause for distrust towards the use of CBA in WFD implementation in France.

Obviously, decisions taken with regards to the above five dimensions were also subject to more general constraints, i.e., factors unrelated to the WFD. Three factors play a role here and deserve more attention in future studies: first, resource constraints, explaining the poor method used on benefits valuation in France; second, the presence of statutory guidelines on economic analyses in general; and finally, attitudes about the usefulness of economic appraisal methods in public policy on a broader level.

Conclusion

Our article has explored the use of economic analysis to justify exemptions during the implementation of the WFD in England and France. Relying on an analytical framework consisting of five dimensions – scale, screening, benefit and costs data, uncertainty, and additional parameters – we show that the two countries rely on economic analysis, that their operationalisation differs, that these differences reflect, to some extent at least, political ambitions in the field of water policy and, finally, that the usage of economic analysis influences the process of setting water quality objectives. All this suggests that policy appraisal tools have a political dimension and are not, and cannot be, neutral when it comes to aiding decision-makers.

This argument departs from the mainstream narrative put forward in environmental economics according to which analytical tools such as CBA are politically neutral, if applied correctly by the textbook (Owens *et al*., 2004). Economic analyses lose this neutrality only as a result of inaccurate and flawed usages by practitioners. Instead, this article builds on an emerging research agenda in public policy and political science exploring the political dimension of policy appraisal in legislation and programme implementation (McGarity, 1991; Turnpenny *et al*., 2008; Cashmore *et al*., 2010; Coletti & Radaelli, 2013; Fritsch *et al*., 2017). The specific usage of policy appraisal tools can, intentionally or unintentionally, shape the outputs of political decision-making processes (Dunlop *et al*., 2012) and, in fact, support almost contradictory political aims. However, this argument has rarely been spelt out in detail in an interdisciplinary water policy context.

We contribute to extant scholarship by suggesting three pathways – related to *input*, *process* and *output* – through which economic analyses may influence water policy decisions, thereby bringing in another degree of sophistication to previous work on policy appraisal. First, screening processes are useful examples to highlight the importance of data inclusion rules in economic analysis – they basically alter the range of materials defining the *input* of the analysis, thereby answering the question of *what* is actually analysed. Second, we provide evidence for variance in the interpretation of uncertainties, the choice of the cost–benefit ratio, the discount rate, thresholds in affordability tests, and other *process*-related features of economic analysis. The way data are processed, decisions are taken and key concepts interpreted may tip the scale one way or another – referring to the *how* question of economic analysis. Finally, tools come with different degrees of *precision* and *soundness* of analysis. Consequently, methodological choices influence the *output* of water policy decisions. This includes various aspects, but most importantly, the degree of ambition and the affected parties – the *to what end* and *who*. Examples include the challenges related to benefit transfers and the scale at which analyses are performed. Future research could address these questions in more detail and reflect in more depth upon factors explaining specific methodological choices in economic analyses, both in the water sector and beyond.

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