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The use of cost–benefit analysis in environmental policies: Some issues raised by the Water Framework Directive implementation in France

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ABSTRACT

This paper aims to practically contribute to the literature on the use of cost–benefit analysis (CBA) and economic evaluation in environmental decision-making through a practical case study: the implementation of the Water Framework Directive (WFD) in France, for the first cycle (2010–2015). The WFD requires that Member States achieve "good status" for all water bodies in 2015. However, exemptions can apply, if justified, on natural, technical or economic reasons. For the latter, EU guidance documents recommend to use CBA. In France, the water agencies carried out 710 CBAs on proposed restoration projects for water bodies. This article reports on this experience. Issues concerning these analyses are discussed, especially the assessment of non-market benefits. Finally, this article questions the use of economic analysis in the implementation of environmental policy.

1. Introduction

1.1. Cost–benefit analysis and valuation in environmental policy-making

1.1.1. Context

Both economics and law offer normative benchmarks on the way public policies should be implemented (Kirat, 2012). The legal benchmark relies on laws adopted by citizens' representatives, the economic one on social welfare reflecting individual preferences. The latter can be used against the former when it helps policy-makers to balance the costs of public policies, e.g. related to health, transport, environment, with the social benefits generated by these policies. In particular, cost–benefit analysis (CBA) is applied to environmental policies in order to compare the social costs and benefits of legal environmental norms (Hansjuergens, 2004; Hansson, 2007). Its use has grown in Europe since the mid-1980s (Börger et al., 2014; Pearce et al., 2006).

Two categories of economic values are usually differentiated to perform a CBA (Heal, 2000): the first is use values (divided into direct use, e.g. angling, and indirect use, e.g. flood control) and the second is non-use values (divided into bequest for future generations, altruistic and existence values of the biodiversity components, e.g. the existence value of wild species).

Assessing use values through monetary indicators is relatively easy when they are connected with market prices (e.g. production of drinking water), but it is more complex to capture the non-market benefits (e.g. recreational fishing or bird watching). To estimate non-market benefits, three types of valuation methods can be used (Barbier et al., 2009): cost-based methods (cost of avoided damages, replacement costs, substitute costs, restoration costs, impact on productivity); revealed preference methods

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(hedonic pricing, travel cost method); stated preference methods which employ hypothetical markets (willingness to pay (WTP) based on contingent valuation or choice experiments).

Stated preference methods are the only way to capture non-use values in monetary units (Heinzerling, 2011; Levrel et al., 2012). Thanks to these methods, it is possible to assess how much people would be willing to pay (or accept as compensation) for conserving (or degrading) a component of the environment from which they do not benefit, but which has value by virtue of people expressing an attachment to its existence.

To facilitate the implementation of these conventional valuation methods, the benefits transfer method is increasingly used in order to carry out a large number of CBA at low costs (Johnston and Rosenberger, 2010). This method involves applying to a given site a monetary value obtained from another site – ideally similar regarding its biophysical and socio-economic characteristics – with adjustments if necessary. The method is now widely used as it can lead to substantial savings.

Once the values for all benefits estimated (provided that duplications are avoided), they are aggregated to give the "total economic value" of the targeted environmental components (CGDD, 2011; Chevassus-au-Louis et al., 2009).

Although the literature often mentions an increasing use of CBA and economic valuation to support environmental decision-making, some authors argue that their actual utilisation remains limited in practice (Laurans et al., 2013; Nyborg, 2014; Posner and Adler, 2000; Salles, 2011). Few concrete examples have been published, and knowledge on whether and how they are used, and on their practical limitations, is still lacking (Laurans et al., 2013; McKenzie et al., 2014).

1.1.2. Objective of this paper

First of all, this paper intends to give a concrete example of economic valuations applied to an environmental policy, i.e. the Water Framework Directive (WFD) implementation. Second, this article aims to illustrate the methodological difficulties raised by CBA and the uncertainties affecting the assessment process. Lastly, this paper will discuss the strengths and limitations of using an economic benchmark in environmental policy implementation.

1.2. Background: the Water Framework Directive and cost–benefit analysis

1.2.1. Economic analysis in the WFD

One notable exception of an environmental public policy where monetary valuations are broadly used is the European WFD (Laurans, 2006). This Directive, published in 2000, requires Member States to achieve "good status" for their "water bodies" (lakes, stretches of water courses, groundwater and coastal water) by 2015. Exemptions from deadlines or objectives may be justified on the basis of three types of arguments: "technical" (no technical means exist to achieve the objective), "natural" (the natural environment response time is such that the deadline cannot be met) and "economic" (the cost of a project, or group of measures, that should be implemented to achieve good status is considered as disproportionately costly). In the following sections, we focus on the latter. According to Article 4 of the WFD, disproportionate costs can justify extending deadlines for good status achievement to 2021 or 2027 and setting less stringent environmental objectives than good status.

1.2.2. Exemptions and disproportionate costs in the WFD

The WFD therefore requires the use of economic analysis to support public decision-making, and to assess the risks of disproportionate costs supported by society, arising from its implementation. However, the WFD does not specify which criteria and methods should be used to assess and decide whether costs are disproportionate. Yet, water quality targets can be altered through exemptions based on this analysis. Biases in competition could arise between Member States through differences in interpretations. At the river basin or even at a more local level, it is clear that studying whether a project is disproportionately costly will expose tensions between environmental services delivered by the project and the changes necessary to achieve them. The most appropriate method to use to assess whether costs are disproportionate has been debated by Member States' Water Directors. In this debate, some countries have supported the assessment of stakeholders' financial ability to bear the costs (e.g. Denmark), whereas others have preferred CBA (e.g. England) (Martin-Ortega et al., 2014).

Balancing the legal benchmark for good status against an analysis of the costs and benefits of achieving this objective for society may appear entirely justified. If the social cost of an environmental measure is deemed to be "disproportionate", then it should be possible to reduce its scope. This explains why this precaution was explicitly mentioned in the Directive. However, applying this criterion entails a risk of substantially reducing the environmental scope of the legal framework. As a consequence, it is essential that Member States calculate these costs and benefits in a robust way.

2. Method

2.1. The case study

Materials used in this paper are based on the French implementation of 710 CBA (corresponding to water bodies or groups of water bodies) aiming to assess whether the costs of reaching good status for water is "disproportionate" or not. The water agencies economists (authors of this paper) carried out these CBA to justify exemptions during the first WFD implementation cycle. They also had to present these results for discussion and take stakeholders' feedback into account. Consequently, data and results rely on the authors' own experience.

2.1.1. Water governance and the French water agencies system

The water agencies are responsible for the WFD implementation. Created in 1964, they coordinate water policy at the river basin level in order to maintain or restore water quality. They levy taxes on water uses and grant financial support to project leaders seeking to improve water management. The level of subsidy provided and the amount of tax levied are defined and approved by a Basin Committee, i.e. a "Water Parliament", which includes representatives from local governments, water users and the central government. In this framework, the Basin Committee defines the objectives of the river basin management plans. They are submitted for approval to the coordinating State representative for the basin.

2.1.2. The approach for disproportionate costs in France

In France, guidance on exemptions was published in October 2009 detailing the method to be used[1] for disproportionate costs assessment. Monetary valuation and CBA should be applied to water bodies for which measures were "likely to incur disproportionate costs" (on financial capacity criterion), if exemptions could not be granted due to a lack of technical feasibility or to natural conditions. When benefits were less than 80% of costs, it was considered that costs were disproportionate. This rule, decided at

[1] Ministère de l'Ecologie, de l'Energie, du Développement Durable et de la Mer, 2009. Guide Méthodologique de justification des exemptions prévues par la directive cadre sur l'eau, pp. 1–54.

the national level, was based on the fact that benefits should be at least equal to costs to be considered as "proportionate". However, due to uncertainties in the analysis, a margin of 20% was adopted. This rule is discussed below.

2.2. Valuation methods used by water agencies economists

Water agencies economists have used several methods to assess the benefits of restoring water good status. The avoidance cost method was applied in order to value the reduction in water treatment costs. Contingent valuation was used to assess the increase of users utility coming from water quality improvement. Limited use was made of the travel cost method. To carry out a large number of CBA, benefits transfer was mainly employed.

To facilitate the CBA process and reduce the cost of the WFD implementation, the French Ministry of Environment provided water agencies with a spreadsheet. This enabled valuation using benefits transfer and made the calculation of Benefits/Costs (B/C) ratios easier. The spreadsheet included simplified mean values obtained from about forty French studies that linked a change in environmental quality with a change in value. The aim was to avoid the delays and costs that could arise when conducting on-site stated preference studies to monetise the non-market benefits of restoration projects (MEDD, 2005).

Using the spreadsheet to value the benefits of a specific project requires entering information on the type of environment concerned (watercourse, lake, coastal or groundwater), then selecting the type of expected benefits (e.g. improvements in populations of wild fish species). Depending on the criteria entered, the tool selects the benefits estimates that are most relevant to the information entered on the watercourse (e.g. its size), often with a minimum and a maximum value that the economist can accept or reject. These unit values are applied to the population assumed to benefit from the ecosystem services that the watercourse offers. By multiplying the unit value by the population concerned, the benefits associated with certain ecosystem services are obtained. For example, kayakers' willingness to pay (WTP) for an environmental improvement, as estimated for an original study, is applied to the number of estimated kayakers using the section of watercourse. The method is the same for benefits to anglers, swimmers and walkers. When the data are uncertain, either because they are available at a less detailed level or because the estimate at the given level includes uncertainty, then estimates are broken down further into estimation brackets. The economist also enters data on project costs (including investments lifetime, operational costs), again using "brackets" to reflect uncertainty if necessary. Finally, a comparison is made of the costs and benefits of the project, as described for the example in the box below (Large, 2008).

Box: Example of CBA for the "Béthune and Arques" river basin (2008):

The "Béthune and Arques" river basin comprises three water bodies including a coastal river in Normandie, which flows out onto two beaches and incorporates a leisure amenity further upstream. Its initial status was poor, mainly due to low fish diversity and an excess of phosphorus from agriculture that had degraded water quality. For this case study, the costs of measures to restore good water status during the first cycle of the WFD were estimated at €235 M (present value), 85% of which were recurring costs (for river maintenance) and 15% were for one-off investments (improving wastewater treatment at a treatment plant).

To use the WTP estimates from benefits transfer to value the improvements in ecosystem services that arise from the project, information is needed on the population of beneficiaries. In our example, the economist selected the population of the river basin, using estimates of the number of users (kayakers, swimmers, etc.) at the regional or departmental levels, or at the site level when there was no exact data on the specific interest of this watercourse in relation to a defined geographic area.

As can be seen, the costs and benefits compared are estimated in very different ways. The finding of the analysis was that estimates of the total benefits for the period 2010–2040 (based on facilities' lifetime) were far lower than the total costs: the present value[2] of benefits was estimated at €18.2 M, whereas costs were €235 M.

As shown in Table 1, the method and results given here are representative of the majority of CBA carried out for the first WFD cycle in France.

3. Results

Across France, there are 11,523 surface[3] water bodies. The pre-selection process based on financial capacity, as recommended by the national guidelines, reduced the final number of CBA to 710, at the scale of water bodies or groups of water bodies (small river basins). The analyses were carried out at the level of the seven main river basins that make up the national territory. Table 1 summarises the procedures they adopted. The majority of analyses performed were "desk CBA", based on available data describing the watershed and its local water uses. They were often carried out with the Ministry of Environment spreadsheet. Consequently, benefits transfer was the main method used to value non-market benefits. Because the willingness to pay was rather stable in these different studies (CGDD, 2013), the main stake was the size of the population from which it was possible to extend the results. A small proportion (0.4%) of CBA did not use benefits transfer, but was based on on-site WTP surveys (e.g. "Loire-Bretagne" and "Adour-Garonne")). This costly and time-consuming option was used when the site was of particular interest and when the project was to be discussed beyond the purely technical realm.

Table 1 shows that although the national method was followed overall, the river basins applied it differently. Differences included the undertaking of monetary valuations, the use (or non-use) of the spreadsheet and of valuations in discussions and into the decision-making process. Another and unavoidable difference included the choice of the population for values extrapolation, as discussed below. Some economists used CBA as a mean of retrospective justification and conducted the analysis on a large number of water bodies without deep field analysis; others conducted only a small number of in-depth analyses that were often used to inform decision-making; some others opted for a mixed system; and finally, a minority of water agencies preferred to focus as little as possible on the economic criterion as a motive for exemption, mainly due to uncertainties on benefits values.

4. Discussion

CBA has obvious potential to improve economic efficiency in resource allocation and enhance transparency and rationality in decision-making, not the least because CBA forces decision-makers to make public their implicit assumptions and uncertainties

[2] With a discount rate of 4% over the 30 years of the calculation.
[3] As opposed to groundwater bodies.

Table 1
Comparison of cost–benefit analysis (CBA) conducted in the different river basins by the six water agencies.

	Seine-Normandie	Loire Bretagne	Adour-Garonne	Rhin Meuse	Rhône-Méditerranée et Corse (two basins but one agency)	Artois-Picardie
Strategy/Economic exemption	Pre-selection through financial capacity criterion, then "desk CBA"[a] on groups of water bodies in risk of not achieving good status. When $B > 0.8C$, refined financial capacity analysis.	Pre-selection through financial capacity criterion, then "desk CBA" on groups of water bodies in risk of not achieving good status. When $B > 0.8C$, refined financial capacity analysis. 3 in-depth field CBA conducted on "sensitive" water bodies.	Priority given to technical and natural causes. 4 in-depth field CBA conducted on 7 water bodies (sometimes including downstream water bodies).	Pre-selection through financial capacity criterion, then "desk CBA" on groups of water bodies in risk of not achieving good status. When $B > 0.8C$, refined financial capacity analysis, and objectives modified if necessary.	Global cost criterion (threshold €10 M), then "desk CBA" on groups of water bodies in risk of not achieving good status. Refined when B/C ratio between 0.65 and 0.95.	Priority given to technical and natural causes. Economic criterion only used to support other criteria.
Number of CBA conducted on surface water bodies (rivers) (total: 710 water bodies or groups of water bodies)	55 CBA on groups of water bodies (covering 92 water bodies)	150 CBA (on groups of water bodies, water bodies and by issue) and 3 more detailed benefit assessments (coastal sites, wetlands and estuaries) with field surveys.	4 CBA, impacting 7 water bodies	277 CBA (on water bodies)	192 CBA (on groups of water bodies)	29 CBA
B/C average	0.48	0.5	1.4	0.17	4.87	0.44
B/C average when $B < 0.8C$	0.27	0.4	0.13	0.14	0.32	0.34
% cases where $B > 0.8C$	10%	49%	66%	7%	44%	0%
Used in discussion	Not used in prior discussions	In a dozen of cases, arguments used at local meetings	Used in discussion	Used in discussion.	Not used in prior discussions.	Not used in prior discussions.
Used in decision-making	Not used in decisions. Retrospective justification for a decision on an exemption taken locally according to feasibility.	In most cases, retrospective justification.	Not used in decision-making.	Used for decision-making.	Not used in decision-making. Economic reason never used on its own. Retrospective justification for a decision to request exemption.	Not used in decision-making. Retrospective justification after the event for a decision to request exemption. Economic reason never used on its own and $C > B$ in all cases
Objective changed on the basis of CBA results.	No change on the basis of CBA results.	In a minority of cases, change of objective after analysis of contributory capacity.	Objective unchanged (exemption maintained even when $B > 0.8C$).	Objective was changed for about 20 water bodies	No change on the basis of CBA result	No change on the basis of CBA result
% of surface water bodies with extended deadline	48%	39%	38%	64%	38%	85%
Of which % of extended deadlines justified on the basis of disproportionate costs[b]	72% (with other kinds of justifications)	60%	60%	60%	22% (with other kinds of justifications)	48% (with other kinds of justifications))

B: benefits; C: costs.
[a] No specific field study was carried out for "desk CBA". They were conducted based on the available data and using the Ministry's spreadsheet.
[b] The water agencies are in charge of supporting decision-making regarding WFD objectives at the local level. As economists working in the water agencies, some authors of this paper witnessed this process.

(Buehler, 2012; Dehnhardt, 2014; Turner, 2006). Comparing the costs paid by society with the benefits derived from a project seems a sensible measure and a necessary safeguard against projects where social costs are greater than benefits. However, in practice, methodological and practical issues arise.

4.1. The real role of valuations on exemption processes

In France, due to legal interpretations, compromise and pragmatism, CBA were considered as the key argument to justify exemptions on economic grounds. Constraints of time and costs have led, as shown in Table 1, to a majority of "desk CBA".

Moreover, deadline extensions appear as very easy to justify through CBA, since benefits usually do not weigh very heavily in the balance. **Three quarters of the 710 CBA showed considerably lower benefits than costs**. If disproportionate cost assessments had been used to set targets level in river basins, the level would have been much lower than the objective considered as "acceptable" for local authorities, given that it was voted by the basin committees.

In Artois-Picardie, where relatively detailed CBA were carried out, the water agency, in agreement with the State, chose to base exemptions not solely on "disproportionate costs". Although economic analyses would have been sufficient to justify exemptions, the preferred approach was to combine disproportionate costs with technical feasibility and natural conditions. This decision was made because the water agency considered the economic analysis as too weak to justify such a decision, in particular due to the uncertainty linked with the valuation of non-market benefits.

4.2. Issues raised when comparing costs and benefits of environmental projects

The literature highlights **methodological biases** of valuation methods, especially when estimating non-market values. Contingent valuations and benefits transfer are heavily criticised for lacking robustness and producing estimates with high levels of uncertainty (Horowitz and McConnell, 2002; Levrel et al., 2012; TEEB, 2010). For instance, regarding contingent valuation, respondents often lack knowledge on ecological issues and may adopt strategic behaviours (Hanley, 2001; Hanley and Shogren, 2005; Nunes and van der Bergh, 2001). Benefit transfers, although less time and resource consuming, are also disputed due to their inaccuracy in most situations (Brouwer, 2000; Iovanna and Griffiths, 2006).

Moreover, while costs are usually overestimated, non-quantifiable benefits tend to be disregarded (Ackerman and Heinzerling, 2002). The complexity of both estimating environmental effects and valuing non-market benefits in monetary terms, leads to a risk of underestimating the overall benefits (Salles, 2011; Shapiro and Schroeder, 2008). In particular, some ecosystem services are difficult to quantify; there is little to compare them with and they may be subject to many uncertainties (Heal, 2000; TEEB, 2010). Thus, a good water status achievement cannot be easily expressed in terms of total economic value.

The limits and the uncertainty associated to the monetary valuation have been taken into account in our case by setting the benefit/cost ratio at 80%. This choice is based on the fact that costs are often over-estimated and benefits coming from Nature often under-estimated, as mentioned above. If the principle makes sense, it is questionable to know why it has been decided to choose 80% as the proper ratio. It is obvious that this benchmark could have been set at 60% or at 40%. Nevertheless, Table 1 shows that the average B/C ratio appears as very low in most river basins, especially for the cases where $B < 0.8C$. Consequently, the 80% decision rule had a very limited impact on the number of exemptions. We explain this result by the fact that WTP does not sufficiently take into account non-market benefits. Furthermore, all cases where $B > 0.8C$ were linked to a high population density and a relatively low cost of the project, even for in-depth field CBA.

The variation between studies in unit values for WTP and the different approaches used to conduct CBA clearly shows that these assessments cannot necessarily be replicated in the same way as calculations in biophysics or accounting used to assess costs. In addition, in the case of environmental projects, CBA consists in comparing values of very different nature: on the one hand, costs that are financial and that may lead to real expenditure if the project is finally implemented, and on the other hand, benefits that are partly assessed, thanks to powerful (but uncertain) methods but that are not financial values.

4.3. The key role of the reference population selected

Another difficulty that arises in conducting CBA is the choice of the relevant population to which the WTP is applied. Who benefits from the environmental improvements of a restoration project is not obvious. The issue is that aggregate values of WTP can vary significantly depending on the population selected (Hanley et al., 2003; Pearce et al., 2006). Some works have shown that WTP tended to decay with distance from site, but this relationship is questioned in the case of non-use values (Hanley et al., 2003).

The water agencies economists were confronted with this issue. They could not know how to select the population of beneficiaries to a project and how to adjust WTP values. There is indeed no general rule to define with rigour the "perimeter of interest" for a restoration project at the water body level, in particular to know whether more distant populations care as much about the restoration project and its non-market benefits. Taking into account all these aspects for each study site would have required an unrealistic number of costly studies. Moreover, other factors related to the population (e.g. income, age, education) or to the study site (e.g. ecological substitutes, costs and extent of the restoration project) theoretically influence WTP (Bateman et al., 2011; Loomis and Rosenberger, 2006). Defining the population of beneficiaries is consequently very difficult even with in-depth field studies (CGDD, 2011). As a consequence, unit benefit values were simply multiplied by the number of inhabitants living close to the water body, in a quite arbitrary way. This simple choice heavily penalised rural watersheds, where benefits estimations were much lower due to smaller population densities.

Let us take the example of two basins in the south of France, the Vidourle, with 42,000 inhabitants, and the Lez-Mosson, with 414,000 inhabitants (this basin includes the city of Montpellier). Both the costs of restoring good water status and the qualitative ecological benefits at the river basin scale were similar for these two basins. For the Vidourle, the main aim was to restore open spaces around watercourses and the coast, and actively tackle agricultural pollution. For the Lez-Mosson, the goal was to tackle urban pollution. CBA concluded that for the Vidourle, restoring good water quality would cost €27.6 M, whereas benefit values were only €13.4 M, despite important tourism and highly developed canoeing and kayaking activities. The project was therefore considered as disproportionately costly. For the Lez-Mosson, the costs were estimated at €27.4 M and the benefits at €125 M simply by virtue of Montpellier's population. The costs were therefore not considered as disproportionate. To conclude, the disproportionate costs argument was put forward only for the Vidourle, while other arguments (technical, natural) were used in the case of the Lez-Mosson to justify an exemption.

In this example, there are great variations in the values obtained, due to differences in the beneficiary populations and high differences in population density.

4.4. Unexpected strategic uses of monetary valuations

In addition to the limitations mentioned above, we must consider the potential undesirable effects of valuations when used in the implementation of public environmental policies. The existing literature underlines strategic uses of CBA (Hockley, 2014), especially as an instrument to support a political position or some private interests (Damart and Roy, 2009).

The authors of this paper have also noticed strategic uses of these monetary valuations by private firms. In one French basin, once the river basin management plan was officially adopted by the local Basin Committee (which included objectives to be achieved by 2015), industrial firms located around water bodies that were considered able to achieve good status objectives by 2015, but had not undergone economic analyses, asked for a CBA to be carried out. The aim was to prove that the benefits of the restoration projects affecting them did not sufficiently outweigh the costs. The dilemma is easy to imagine for the water agencies, given the context of uncertainty, the valuation weaknesses, the low credibility of these analyses, and knowing that CBA had been used to justify exemptions and not to reinforce quality objectives. The situation was problematic since the water agencies operate within a governance framework that gives almost as much power to water users (including industries) as to the State. Consequently, stakeholders who did not want to see restoration projects adopted could use disproportionate costs to justify exemptions. A tool used as a retrospective pseudo-justification was turned into a tool used against sustainable water management by some commercial stakeholders unwilling to make further investments. A situation such as this could get bogged down in legal proceedings if a private stakeholder decided to go to court, as was the case in USA, regarding the monetary valuation methods used to calculate compensation after an oil spill, in the 1990s (Thompson, 2002). This risk is amplified by the greater power of economic compared to environmental lobbies, in France and at the European level, as shown by observatories like the Corporate Europe Observatory.[4]

Finally, as shown in Table 1, CBA was seldom used to change the objectives, and in many cases, the economic argument was not used alone. This is due to the fact that the results of CBA did not appear as strong enough, even when in-depth field studies were conducted, because of the various limitations discussed above.

5. Conclusion

This article has shown some of the tensions that exist between legal and economic benchmarks in the implementation of environmental public policies. Although both can provide the basis for public debate on the relevance of actions to restore good water status, it is important to scrutinise their quality.

Like any evaluation technique, whether multi-criteria or CBA, the lack of information on interactions in the ecological system leads to limited and biased results, due to the high complexity of ecosystems (Hanley, 2001; Nunes and van der Bergh, 2001) and the irreversibility of ecosystem damages once a critical threshold is reached (Pindyck, 2000; UNEP, 2011). But beyond this limitation, CBA requires specific methods to express environmental services in monetised benefits that add even more uncertainty. This is particularly true when it comes to environmental long term or hidden benefits that can hardly be perceived by the population (e.g. biodiversity or groundwater quality). This is probably the main weakness of the technique. According to the French water agency economists' feedback, it seems that the robustness of the economic benchmark can be called into question. Although the costs of restoring good water status can be evaluated using simple accounting measurements of expenditure, the same does not apply when valuing the benefits to be weighed against these costs in order to assess whether or not they are disproportionate. Evaluating the full benefits derived from restoring water quality depends on two methods and one extrapolation parameter, which can lead to great variations in the estimates:

- the contingent valuation method, which is the most debated method in the economics community, but also the most costly to implement if value transfer is not used;
- the benefit/value transfer method, which is clearly the cheapest method to provide an economic valuation of ecological restoration benefits, but also with the highest error level in the value estimated;
- the size of the population to which the extrapolation of individual values can be applied depends on the knowledge available on the links between the ecosystem services delivered by restoration actions and their real use by local populations. Unfortunately, the exact number of ecosystem services users that will benefit from the water restoration project is difficult to know and its estimation is certainly very fragile.

These limitations create areas of uncertainty, which leave room for a wide variety of possible interpretations and may ultimately be strategically used by stakeholders with a vested interest in tipping the valuation in one direction or another. This type of situation has already been the source of considerable controversy in USA (Thompson, 2002). Of course the same types of strategies can be used with the legal norms. And it seems interesting to balance these legal norms with economic values. However, our analysis leads to question whether such monetary valuation is appropriate for assessing the disproportionality of implementing an environmental water policy and if a multi-criteria analysis of the benefits provided by this policy would not provide a better framework to discuss these disproportionate costs.

5.1. Future prospects

Ultimately, the non-market benefits a project can bring to a community should preferably be based on non-monetary assessments, crossing quantitative and qualitative indicators. Every effort should then be made to conduct a real debate covering all values, and not only those that can be converted into euros. Initial feedback from other Member States, e.g. through the European research project ESAWADI,[5] showed a similar reluctance to use CBA, in particular in Portugal and Germany. Other methods were preferred, such as multi-criteria analysis or the "Leipzig approach", which does not entail monetising benefits, but instead makes a "semi-qualitative" analysis based on expert opinion (Blancher et al., 2013). Such methods would enhance the transparency of the decision, which is ultimately political.

Specifically, it seems important:

- to define non-market benefits in non-monetary units such as biophysical quantities, number of users, etc.;
- to clearly identify the nature of these values (use but non-market value, heritage value, existence value, etc.);

[4] http://corporateeurope.org/fr.

[5] The ESAWADI project analysed the added value of valuing ecosystem services for decision-making in the context of the WFD.

– to use monetary valuation when it makes sense for policy-makers (concrete and observable benefits or costs) to build up arguments in support of environmental law (Levrel et al., 2014);
– to consider, as highlighted by other authors (Laurans et al., 2001), economic valuations as "social processes to construct values", in contrast to the standard methods which assume that valuation can reveal pre-existing values.

To achieve this, politicians should take the lead in guaranteeing that all decisions are the result of a democratic negotiation process based on both monetary and non-monetary indicators (especially for non-market benefits).

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at http://dx.doi.org/10.1016/j.envsci.2015.12.002.

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