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Review

Economic Analysis under the Water Framework Directive: The State of the Art and Way forward

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Abstract: Linking the improvement of water ecosystems to the use of economic concepts and instruments is one of the main innovations introduced by the EU Water Framework Directive (WFD). This should be achieved by Member States through an approach clearly linking measures and interventions to improve water ecosystems to the identified pressures on water bodies (i.e., the gap analysis) and a set of economic provisions. However, modest progress in the implementation of these provisions has been recorded over time. Therefore, this paper aims to shed new light on the current limits in the implementation of the economic analysis of the WFD, in particular in relation to the gap analysis, through a comprehensive review of grey and scientific literature on the topics of gap analysis, economic valuation of ecosystem goods and services, water pricing, and disproportionate costs of measures. General conclusions and recommendations of this analysis are that enhancing data quality, promoting consistency and interaction in economic analysis components, and embedding them pragmatically in decision-making procedures are crucial. The gap analysis plays a pivotal role in directing economic research towards relevant issues within the river basin and in guiding decision makers more effectively in the application of the economic analyses required by the WFD.

Keywords: water policy; economic evaluation; cost–benefit analysis; DPSIR approach; gap analysis

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1. Introduction

The Water Framework Directive (2000/60/EC, WFD hereafter) provides the most comprehensive legislative framework for water protection in Europe. The default objective of the WFD is to achieve good water status and good ecological status of all water bodies, initially by 2015 and then by 2027. To accomplish these goals, Member States (MS) were required to publish River Basin Management Plans (RBMPs) which contain information both on the status of their river basins and on the Programme of Measures (PoMs) they intend to implement to improve water ecosystems. Moreover, the WFD introduced a set of economic concepts and instruments. The main economic provisions refer to: (i) understanding the economic issues and tradeoffs at stake in a river basin, (ii) assessing the economic impacts of proposed measures aimed at improving water status, (iii) incentivizing an efficient use of water through water pricing policies, and (iv) assessing regions or water bodies where less stringent environmental targets need to be applied to account for economic and social impacts [1].

Since the beginning, however, the implementation of these provisions has proved challenging for most of the EU countries and modest progress has been recorded over time.

In 2015, in fact, the European Commission (EC) flagged an overall inadequate pricing of water services, especially in the agricultural sector [2]. In 2019, an assessment on the state of implementation concluded that, except for a limited number of MS, progress on the adoption of economic principles and instruments was limited [3]. Recently, in 2021, a study by the EC on the availability of economic data on WFD and Flood Directive highlighted that: (i) there is still incomplete knowledge regarding the costs needed for achieving the WFD's goals (especially economic costs); (ii) even if the WFD does not explicitly refer to any specific economic appraisal methods, fully fledged Cost–benefit Analyses (CBA) are rarely carried out by countries and Cost-Effectiveness Analyses (CEA) are often limited only to some typologies of measures; (iii) the application of the cost-recovery principle is still a challenge for MS and, except for financial costs of measures, information on other typologies of costs (e.g., operational and maintenance) are rarely available; and (iv) overall, there is limited evidence of how economic analyses have supported decision-making processes [4].

Several explanations have been provided to justify such unsatisfactory fulfillment of the economic requirements of the WFD and Berbel and Expósito [5] provide a comprehensive analysis of most of them. However, studies dealing with the economic analysis usually overlook how other types of analyses required by the WFD may affect the implementation of the economic aspects. One, above all, is the Gap Analysis (GA), which serves to quantify how far the current state of water bodies is from the good status objective of the WFD. Only when the gap is assessed is it then possible to make decisions on what measures, or combination of measures, to take and to perform an assessment of related costs and benefits [6]. However, only a few studies explicitly link the GA with the economic analysis required by the WFD [7]. Similarly to the economic analysis, the use of the GA has, in fact, been very limited since the beginning of the WFD. Through an analysis of the main implementation drawbacks after the first implementation cycle, Giakoumis et al. [8] found that for 23 out of 27 MS, performing a GA to inform the selection of measures was a problem. It is likely, in turn, that such difficulties have significantly affected the capacity to perform a sound economic analysis.

Therefore, this paper aims to shed new light on the current limits in the implementation of the economic analysis required by the WFD, in particular in relation to the GA. This objective was achieved through a comprehensive review of grey and scientific literature dealing with the implementation of the different requirements of the WFD to obtain a full picture of the state of the art. In particular, these requirements refer to: (i) the GA, (ii) the economic evaluation of ecosystem goods and services (EGSs) in order to establish how they change in response to the implementation of the PoMs, (iii) the application of the principle of full cost recovery through water pricing, and (iv) the economic appraisal of disproportionate costs of measures. These topics were selected in collaboration with the main contributor responsible for the directive's implementation in Italy, which is the River Basin District Authority (RBDA); thus, they reflect real concerns for implementing authorities.

The remainder of the paper is organized as follows: Section 2 describes the methodology adopted in this study. In Section 3 the results are reported, while Section 4 presents the discussion and conclusions.

2. Materials and Methods

Research on the topic of economic analysis under the WFD is extensive and some authors have tried to take stock of existing knowledge through literature reviews. These usually focus on specific economic requirements of the Directive, such as the use of cost-effectiveness analyses [9] or the application of the disproportionality principle [10], while there is a lack of an overall understanding of the research on economic analyses under the WFD. In our study, the focus of the review was co-decided together with the Po RBDA. In Italy, in fact, the RBDA authorities are responsible for the development of the RBMPs and, thus, for the economic analysis. Identifying the focus of a review based on the priorities and concerns of the potential users of the research is a recommended practice in social sciences [11]. As highlighted in the Po RBMP (2021–27), the issue of economic analysis still represents one of the most difficult aspects of the WFD's implementation, with a particular emphasis on "the monetary evaluation of costs and benefits linked to the use of water resources, water pricing and cost recovery, gap analysis, and the economic evaluation of the disproportionality of the costs of measures" [12]. Drawing on this consideration, and on several discussions with the Po RBD authority, four topics were identified as the focus of the review:

1. The application of the GA;
2. The economic valuation of EGSs;
3. The application of the principle of full cost recovery through water pricing;
4. The economic appraisal of disproportionate costs of measures.

Even if these topics are interconnected they cover different strands of research, with a diverse availability of data and information. Moreover, while the issue of the GA is more relevant from the point of view of the operational procedure it establishes, which required grey literature to be included in the selection of articles, on the evaluation of EGSs and pricing there is extensive debate in the literature which, in some cases, precedes and even goes beyond the implementation of the WFD. For this reason, four searches—namely one for each topic—were conducted in parallel using the main research engines (Scopus and Web of Science), but also additional sources of information depending on the topic were integrated (Table 1).

Table 1. Topics and criteria used to select relevant literature.

Topic	Additional Sources	Keywords	Total Number of Documents	Selection Criteria	N. of Selected Documents
Gap analysis	Grey literature	"gap analysis" AND "water framework directive"	8 (Scopus) 2 (Grey literature)	Scoping review	10
Economic valuations of EGSs	Environmental Valuation Reference Inventory database	"Environmental goods and services" AND/OR "water framework directive" AND "willingness to pay"; "willingness to pay" AND "freshwater ecosystem services"	135	Papers and grey literature providing descriptions of gap analysis in WFD procedures. All the significant results deriving from the quest were taken into consideration.	46
Water pricing	Grey literature	"water pricing" AND "cost recovery" AND "agriculture"; "elasticity of demand" AND "water" AND "agriculture	34	Papers providing an empirical assessment of cost recovery and/or elasticity of demand with respect to price for irrigation	20
Economic appraisal of disproportionate costs	n.a.	"water framework directive" AND disproportion * OR "cost-benefit" OR CBA OR "cost-effectiveness" OR CEA OR exemption. Limited to the subject areas of "Economics, Econometrics and Finance" AND "Business, Management and Accounting" AND "Social Sciences"	100	Papers in English providing an empirical assessment of disproportionate costs of measures. Two papers providing a general understanding of the application of disproportionality principle in EU were also included.	13

The connection between the different parts of the analysis is described in Figure 1, providing a conceptual framework to navigate the information available in the literature. From Figure 1 it is evident that the assessment of the gap should precede the definition of PoMs and the assessment of its economic viability. As explained by De Nocker et al. [13], the role of the GA in the economic appraisal required by the WFD should correspond to a scenario analysis, which is an essential step for implementing any CBA. The selection of the package of measures to be evaluated through the CBA, in fact, is based on the distance between the baseline scenario and the objectives. Figure 1 also emphasizes the connection between the GA and the well-established Determinant Pressures State Impact Resources

(DPSIR) approach. In fact, only if a clear link is established between the identified gap, its determinants, and the ensuing pressures is it possible to assess to what extent measures can improve the state of water bodies and, in turn, reduce the gap. A key challenge of this assessment concerns the economic valuation of the change in the provision of EGSs derived from the implementation of the measures that can both result in an improvement (benefit) or in a worsening (cost) of the state. Having accounted for costs and benefits, a comparison through CBA should lead to an assessment of the economic viability of planned measures or, in the opposite case, to the application of exemptions to the achievement of the objectives based on the disproportionate costs of measures (article 4.5). After the exclusion of measures assessed as "disproportionately costly", the CEA should prioritize the implementation of the remaining set of measures. More details on each step represented in Figure 1, together with the main findings of the literature review, are presented in the next sections.

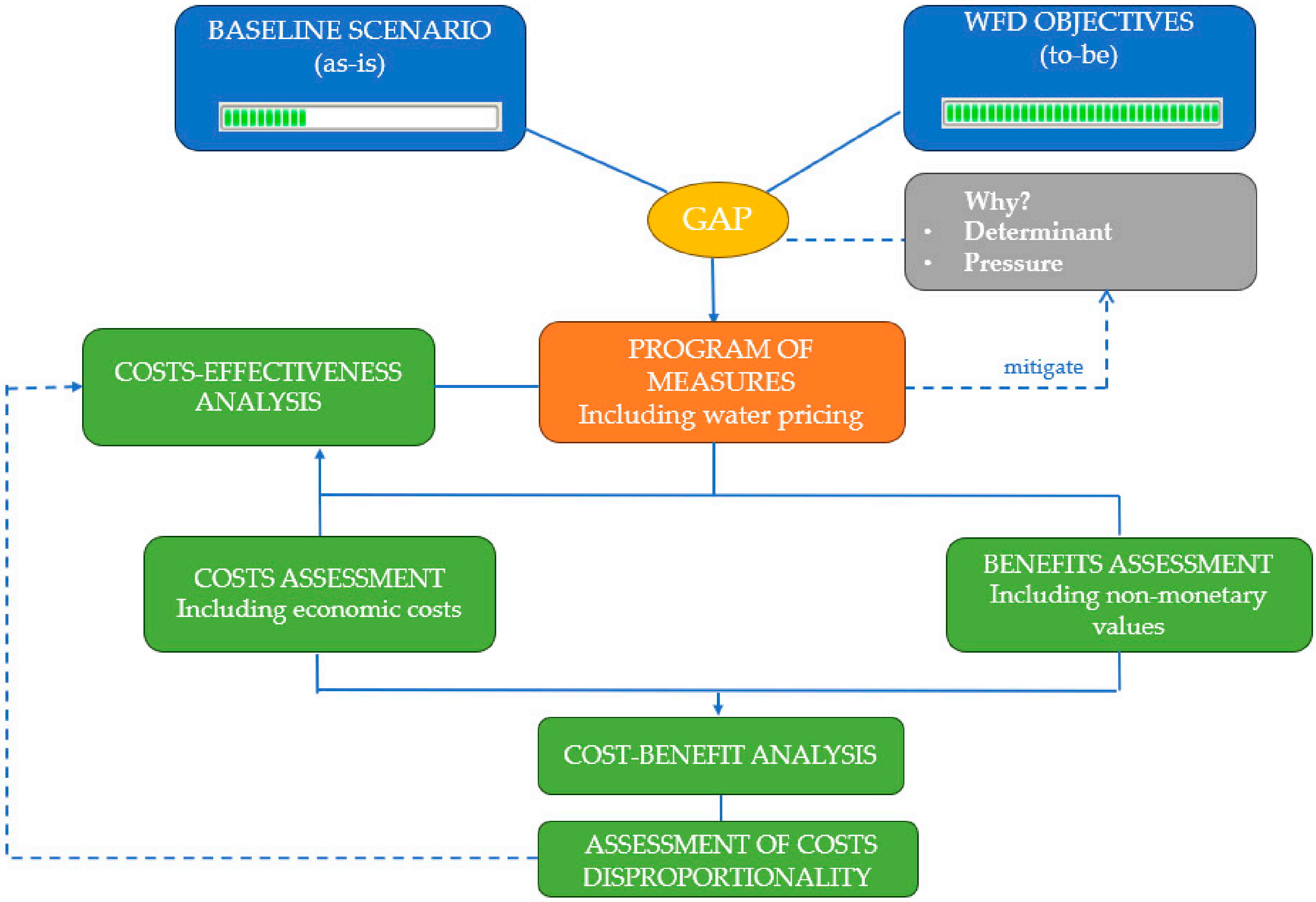

Figure 1. The gap analysis and the economic provision of the WFD.

3. Results

3.1. The Gap Analysis

Table 2 below shows a synthesis of the results obtained from the review, together with the documents derived from European legislation and grey literature. The analysis of these documents is the starting point to understand the role and the procedure of the GA as meant by the EC; this is followed by the results of the scientific review, underlining whether the GA applied by the authors is in line with the one required by the WFD or if such a term indicates another procedure.

Table 2. A brief summary of the documents analyzed in the present section.

Category	Reference	Key Points
LEGAL and GREY LIT.	WATECO [1]	First mention of the GA. Assessment of the gap between the baseline scenario and the WFD's objectives.
	COM/2015/120	The GA is said to be necessary to determine the measures for RBMPs, both in terms of costs and timing. Also, properly justifying exemptions due to technical unfeasibility or disproportionate costs is possible only based on this analysis.
SCIENTIFIC LIT.	Nikolaidis et al. [14]	GA performed on European rivers and lakes using a common set of nutrient concentrations.
	Bennetsen et al. [15]	Further examines the ecological gap, dividing it into its biotic components.
	Tsavdaridou et al. [16]	Analysis of protected areas surrounding lakes under Natura2000 network.
	Kahlert et al. [17]	Assessment of monitoring gaps in the Baltic Sea.
	Latinopoulos et al. [18]	GA of management and monitoring practices, water and quality pressures.
	Lehmann et al. [19]	GA of existing regional observation systems.
	Paz & Rinkevich [20] Weigand et al. [21]	GA of DNA barcoding.

It is worth saying that no explicit mention of a "gap analysis" is found in the text of the Directive, even if the European legislator states that "this information is necessary in order to provide a sound basis for Member States to develop programmes of measures aimed at achieving the objectives established under this Directive" (WFD, whereas 36). By this brief sentence, the main purpose of the GA is defined.

Moving to the analysis of the grey literature, the main reference for the European context is represented by WATECO [1]. The approach of this guidance to the analysis is given the name of a "three steps approach", where step 1 ("Characterizing river basin") focuses on the analysis of the current status of each river basin, step 2 ("Identifying significant water management issues") performs a GA by comparing the so-called "baseline scenario" to the objective of the WFD, and step 3 ("Identifying measure and economic impact") requires a proper economic analysis, working out the potential impacts and potential implications of the PoM, whose measures are evaluated under a CEA perspective. Based on the results obtained in the second step, a gap in water status might be identified or not, presenting two different options:

i. In a "no-gap scenario", the existing measures are considered efficient; this directly leads to an assessment of their financial implications.
ii. In a "gap scenario", MS should start identifying measures based on the drivers of pressures found in step 1 in order to develop a PoM.

Hence, according to the document WATECO, the GA is a procedure that "paves the way to the preparation of the programme of measures".

Nevertheless, in the fourth implementation report, the EC remarked that "many Member States have planned their measures based on 'what is in place and/or in the pipeline already' and 'what is feasible', without considering the current status of water bodies and the pressures identified in the RBMPs" [6]. In the same document, the EC strengthens the role of the GA by clarifying that it is an essential part in the application of the WFD because it does correspond to the economic analysis itself. In fact, the GA should serve to identify the possible measures for reaching the goals of the Directive, to assess their costs, and to identify payers.

Moving to the analysis of the scientific literature, only eight papers were derived from the adopted combination of keywords, highlighting the fact that the procedure is not particularly addressed in the scientific literature; also, it appears that the term "gap analysis" does not refer to a univocal procedure, but it can be applied to more than one practice, as shown in the following paragraphs.

In Nikolaidis et al. [14], a GA is performed to define a clear set of targets for both phosphorus and nitrogen concentrations in rivers and lakes, shared between all EU member states. Having classified each river segment and lake within the respective hydrological category, the gap between the actual average concentration of total nitrogen and phosphorus and the nutrient target for each water body segment is calculated. The results of this GA are then used to identify the extent of freshwater bodies that exceed the targets and discover the drivers of such a mismatch between the objectives of the WFD and the actual status of lakes and rivers.

Bennetsen et al. [15] propose to split the ecological gap into its main components, i.e., the biological taxa that impact on the state of the considered river body and the abiotic factors, so to obtain useful information for the managers of the river basins to use. The authors, in fact, argued that the current ecological gap provides little or no information about the drivers which bring about the gap itself since it often derives from many indexes grouped into one. This paper, together with Nikolaidis et al. [14], clearly focuses on the procedure required by the European Legislator to perform the economic analysis, exploring technical aspects that need to be further studied to better complete the PoM.

A different meaning is given to the GA in Tsavdaridou et al. [16]: the authors want to investigate the role of the Natura 2000 network in protecting European lakes by confronting the protection coverage of these water bodies and of the terrestrial area within the catchment of each lake. Instead of relating the actual water status to the good ecological status, this paper performs a GA in terms of the spatial extent of the environmental measures so that it might be considered as an ex-post analysis of the efficiency of the PoM more than an ex-ante scenario aimed at defining such measures.

An assessment of monitoring gaps in the Baltic Sea under the requirements set forth by the WFD, BSAP (Baltic Sea Action Plan), and MSFD (Marine Strategy Framework Directive) is the main topic of Kahlert et al. [17]. An identification of eight different types of gaps is followed by an analysis of their frequency of citation in scientific articles, project reports, and in stakeholder surveys. This procedure is referred to as a “holistic gap analysis” in the paper but a few connections can be found with the GA of the WFD; it shows that even in the same research field, the term GA can be applied to very different schemes.

An assessment of monitoring gaps is also found in Latinopoulos et al. [18]. In this paper, five case studies in the Mediterranean area are selected and a GA is performed on policy factors so as to improve monitoring, management, and networking practices. In particular, this analysis is considered an appropriate tool to verify the conservation status of the selected area and to both identify and evaluate their management priorities. As highlighted for Kahlert et al. [17], this type of research is more focused on the evaluation of the management practices and protocols than on the assessment of the gap in the status of water bodies.

Another GA of monitoring practices is found in the paper by Lehmann et al. [19], in which the authors apply the procedure to test the existing dataset and Earth observation systems available in the Black Sea region so as to finally model the hydrology of the entire catchment; unlike the previous papers, this study is not directly correlated to the WFD but to another European directive which is not focused on water management (INSPIRE, Directive 2007/2/EC).

Finally, the papers by Paz & Rinkevich [20] and Weigand et al. [21] perform a GA of DNA barcoding for aquatic biota, some of which are included in the list of quality elements required by the protocols of the WFD; still, the contact points between this type of procedure and the one required by the European Legislator are limited.

3.2. Economic Valuations of Ecosystem Goods and Services

This section offers an overview of research related to the economic assessment of EGSs in freshwater ecosystems. These studies can serve as the basis for estimating the costs and benefits associated with strategies and policies to manage water resources at a river basin level.

The economic assessment framework, aligned with the WFD, should revolve around the DPSIR approach. The DPSIR involves identifying causal links between specific water uses and their impacts on water resources, while also allocating estimated costs in accordance with the "polluter/user pays" principle. However, the literature review highlights that economic valuations of the EGSs of freshwater ecosystems published in scientific papers do not generally follow the DPSIR framework proposed in the WFD as the main tool to highlight the nexus between water uses and impacts. Instead, most papers focus on the EGSs (or group of EGSs) at hand without specifying the drivers that—potentially or actually—cause deterioration in the ability of freshwater ecosystems to provide such services. Moreover, due to technical aspects of the environmental valuation methods used (e.g., Contingent Valuation, CV) it is often possible to obtain a monetary estimate not for individual EGSs but rather for a cluster of very context-dependent EGSs, thus hindering the generalization of the monetary values to other contexts. For this reason, the papers were classified based on the type of services for which they provide monetary estimates and were allocated to the categories included in the Common International Classification of Ecosystem Services classification. This is also the reason some of them are repeated among different categories of ecosystem services.

It is important to emphasize that the analysis presented in this section is not intended to be comprehensive or conclusive. The analysis is based on the following assumptions: it exclusively focuses on the services provided by freshwater ecosystems, selectively choosing articles that have conducted a monetary valuation of ecosystem services using well-established econometric methodologies, and specifically selecting study contexts resembling the ecological and socioeconomic conditions of the Po River Basin.

We identified and selected a total of 46 studies (refer to Appendix A for the comprehensive list of studies) based on their relevance and according to the three primary EGSs categories: provisioning, regulation, and cultural. More specifically, we found 10 articles that assess the ecosystem services associated with drinking water supply (coded F8 in the standard international classification), 25 focusing on regulation services (R4, R5, R6, R9) and 11 studies dedicated to evaluating cultural/recreational services (C2).

It is noteworthy that the selected articles present a substantial range of diversity in terms of methodologies, timeframes, and units employed to express Willingness to Pay (WTP). Among the various methods used, stated preferences techniques are the most prevalent across nearly all ecosystem services considered, as depicted in Figure 2. To elaborate, out of the 46 studies analyzed, 23 utilized CV, 15 applied Choice Experiments (CE), 3 employed a combination of both, 3 utilized Benefit Transfer, and 2 utilized Replacement Costs.

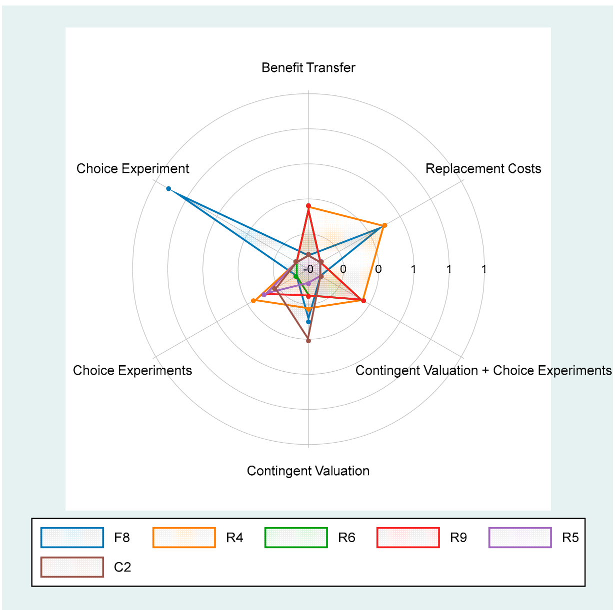

Figure 2. Evaluation methods and ESGs.

The heterogeneity is also attributed to the varying nature of the studies. Some studies focus on specific projects, aligning the estimated WTP with the project's duration. Conversely, others present hypothetical scenarios without clearly defined WTP timeframes. Consequently, the selected articles present a range of estimates due to the unique context of each source, and direct comparisons can be challenging due to the differing units of measure used for the WTP (e.g., per hectare, bill reduction, etc.). Table 3 provides descriptive statistics for a subset of 39 studies, ensuring comparability of estimated WTPs. On the whole, the average willingness to pay ranges from approximately 25 EUR per household per year for the erosion regulation service and protection from geological disruptions (R9) to 73.4 EUR for the water purification service (R4). However, it is important to note that these estimates exhibit considerable variability, as indicated by the markedly high values of the standard deviations across almost all ecosystem services.

Table 3. Average WTPs for different EGSs (selected studies).

	N	Mean	St. Dev.	Min	Max
F8—Drinking water supply	8	59.46 EUR	71.11	8.94 EUR	228.00 EUR
R4—Water purification service	8	73.39 EUR	61.86	8.94 EUR	193.65 EUR
R6—Protection from hydrological disruptions such as floods and inundation	3	72.31 EUR	47.37	38.25 EUR	126.40 EUR
R9—Biodiversity habitat service	6	37.64 EUR	31.53	15.91 EUR	98.27 EUR
R5—Erosion regulation service and protection from geological disruptions	3	24.40 EUR	5.93	18.19 EUR	30.00 EUR
C1 and C2—Cultural Services (aesthetic and recreational)	11	50.63 EUR	43.54	15.91 EUR	145.00 EUR

3.2.1. Provisioning Services

The primary EGSs within the provisioning category that were evaluated focused on drinking water supply (F8). The predominant evaluation techniques used were CV and CEs, although other methods such as replacement cost and Benefit Transfer for specific subsets of EGSs were also used to a lesser extent. Moreover, there was a broader assessment of estimates for the supply of agricultural products and biomass (e.g., Chaikaew et al. [22]). The evaluation of potable/high-quality water supply varied in approach depending on the context of the reference study. Some studies concentrated on areas facing water scarcity or increasing pollution, assessing residents' WTP for improved and reliable home drinking water supply or regenerated water for irrigation. Others, as a result of a healthy ecosystem: for example, Hein [23], evaluated the supply of drinking water from springs near a forest in the Netherlands using the replacement cost method. Regarding groundwater, Damigos et al. [24] calculated the value attributed to ecosystem services associated with non-surface water through the WTP for managed aquifer recharge (MAR) projects. In certain studies there was an assessment of the impact on human health related to the availability of water in good quality and quantity, especially in areas significantly impacted by human influence on water resources [25].

3.2.2. Regulation Services

A total of 26 economic monetary valuation articles were identified, focusing on the regulatory ecosystem services associated with freshwater ecosystems. The primary evaluation methods utilized were, once again, CE and CV.

The principal sub-categories of evaluated EGSs within this class encompass the following:

- Water purification service (R4),
- Protection from hydrological disruptions such as floods and inundation (R6),
- Biodiversity habitat service (R9),
- Erosion regulation service and protection from geological disruptions (R5),
- Carbon sequestration and local climate regulation services (R1 and R2),
- Water cycle regulation and aquifer recharge service (R3).

Concerning water purification EGSs (R4), the articles examined various aspects. In some cases, the WTP of respondents was studied for the ability of ecosystems to maintain their health and provide the service directly. For instance, Ahtiainen et al. [26] investigated the WTP of homeowners to reduce episodes of slime accumulation and excessive algae, using a CE as a methodological framework. Another study by He et al. [27] employed both the CV method and CEs to evaluate the ability of wetlands to provide multiple services, including sediment and pollutant filtration for water quality. Chen et al. [28] studied the ability of an urban river ecosystem to fulfil the purification function.

In other cases, the WTP was explored for projects (infrastructure or interventions) aiming to compensate for the diminished capacity of ecosystems to provide the water purification service or to provide it at rates necessary to purify substances entering water bodies. This category included studies such as those by Alcon et al. [29], investigating the WTP for the use of reclaimed wastewater for agricultural irrigation, and Genius et al. [30], assessing residents' WTP for the installation of a wastewater treatment plant in a rural area.

Other studies focused on evaluating the water purification service in terms of its impact on human health and the availability of sufficient or good-quality water. Examples falling into this category include research by Koundouri et al. [25], Morris & Camino [31], and La Notte et al. [32]. These studies assessed the general WTP for improvements in attributes related to water purification without specific reference to real projects.

Regarding the hydrological disturbance protection service (R6), several studies were identified, with each offering insights into various aspects of this specific ecosystem service. Birol et al. [33] assessed the flood protection service in an area that had previously experienced such phenomena. The evaluation compared the quality of the hydrological disturbance protection service when riverbanks are in their natural condition (green infrastructure) versus the current condition with artificial banks (grey infrastructure). A similar exploration of the difference in service quality between green and grey infrastructure for flood and salinization protection in the UK was conducted by Brouwer and Bateman [34]. Evaluations of EGSs R6 were also documented in Markantonis et al. [35], Morris & Camino [31], and He et al. [27]. Specifically, He et al. [27] examined the flood protection service concerning wetlands' ability to decelerate water flow. Assessments of EGSs R6 were also found in Markantonis et al. [35], Morris & Camino [31], and He et al. [27], with the latter examining the flood protection service in relation to wetlands' ability to slow water flow.

The ecosystem service of habitats for biodiversity (R9) is addressed in several studies. For instance, Bateman et al. [36] use a CV to study the ecosystem services associated with an urban river and Birol et al. [33] show, through CE, that this type of ecosystem service is less preferred than flood protection. In the study by Hanley et al. [37], the CE method was applied with a particular focus on its application within the context of the WFD. The researchers aimed to investigate the economic value linked to enhancements in three key components of ecological status, specifically focusing on the attributes that study participants evaluated. These attributes encompassed an increase in the diversity of plants, insects, and waterfowl. The study sought to quantify the economic worth associated with improvements in biodiversity and ecological health, providing valuable insights into the importance of these attributes within the context of the WFD and broader environmental considerations. A number of habitat service valuation articles for biodiversity related to the specific area of riparian areas were found. In particular, Colby and Smith-Incer [38] estimate the WTP for this service by applying the CV method. In Johnston et al. [39], the authors propose an assessment of various EGSs of riparian areas in Maryland (USA), in which the R9 service is also included. Other R9 service evaluation studies identified in this literature review come from He et al. [27], Koundouri et al. [25], Chen et al. [28], Morris & Camino [31], and Buckley et al. [40].

A few studies have been found that assess the ecosystem service R5, which relates to the regulation of erosional phenomena and protection from geological disruptions. For example, Stithou et al. [41] make an assessment of the ecosystem service of mitigation from erosion due to the presence of vegetation along riverbanks. In Hanley et al. [37] the R5 service is also assessed, referring to the presence of grey infrastructures along the river course that, in conjunction with other factors, have caused flow alterations and alterations in gravel movement and the river bottom. Chen et al. [28] also evaluated the R5 service in relation to the presence of grey infrastructures along the river course.

3.2.3. Cultural/Recreational Services

A total of 14 articles were identified that assessed cultural ecosystem services associated with freshwater aquatic ecosystems. The majority of these assessments were conducted using CV and CE. Additionally, one study by Börger et al. [42] utilized a combination of the CV method and the Travel Cost method. The studies, listed in Appendix A, covered a range of aspects related to recreational opportunities in both urban and less urbanized river ecosystems. For instance, some focused on urban river ecosystems (e.g., Bateman et al. [34], Chen et al. [28], Polyzou et al. [43], Ruperez-Moreno et al. [44], and Kourtis & Tsihrintzis [45]), while others explored less urbanized settings (e.g., Birol et al. [33,42], Börger et al. [42], Halkos & Matsiori [46]). Some studies, like Buckley et al. [40], approached the presence of rivers in the territory in a more general manner.

Two articles estimated the EGS C1, which is the aesthetic value of freshwater ecosystems, along with other ecosystem services. In Genius et al. [30], the study assessed the population's WTP to reduce the likelihood of a future wastewater treatment plant causing bad odors in the surrounding area. The surveyed population expressed a WTP equivalent to a 45% increase in the fee paid for wastewater treatment for this attribute. In Hanley et al. [37], the aesthetic value was evaluated in terms of the absence of visible discharges and wastes in the water body.

3.3. Water Pricing

Article 9 of the WFD explicitly requires MS to implement pricing policies to incentivize efficient water use and to fulfil cost recovery for water services (including environmental and resource costs) according to the "polluter pays" principle. The idea behind water pricing revolves around implementing a tax to mitigate negative externalities, where the tax amount corresponds to the "marginal social cost" caused by water usage [5]. The ultimate goal is to promote efficient use of the water resource by enforcing a price that actually reflects the value of water. However, the effectiveness of water pricing mechanisms is often undermined by several factors, including the difficulty of jointly achieving the four objectives stated in the WFD: cost recovery, polluter pays principle, incentive to save water, and economic sustainability for all.

Available assessments of water pricing in the EU indicate that only a few countries are prepared to change the pricing system to promote more efficient water use and that typically resource costs are not included in water prices [47]. This is especially true for sectors with the highest water use: Rey et al. [48] report that still no southern European MS has implemented water pricing reform for agriculture. At the European level, the fees charged for public water use do not achieve full cost recovery: a rather large share of costs is recovered in tariffs related to domestic water consumption (drinking water), while the irrigation sector reach an average cost recovery level of 50 percent in the Mediterranean area [49,50]. In general, the challenge of achieving full cost recovery appears rather daunting, but in light of this, basin authorities must increase efforts to gradually increase the share of cost recovery in conjunction with policies to comply with the quality objectives [51]. The following paragraphs set the theoretical foundation of water pricing as described in literature (Section 3.3.1) and review the selected literature focusing on one of the main elements influencing the effectiveness of this policy instrument, which is the price (in)elasticity of water demand (Section 3.3.2).

3.3.1. Principles of Water Pricing

The perception that water pricing (or user fees) is justifiable and can serve redistributive and environmental goals is almost prevalent in both the academic literature and legislation. Prices for water use should therefore be based on a few key principles, and in particular:

- Ensure the efficient use of water resources for their conservation;
- Ensure adequate revenue to fully cover operation and maintenance expenses and to cover, as far as possible, the costs of the resource;
- Ensure the economic and financial sustainability of all uses, particularly those with the highest consumption (e.g., the irrigation and energy sectors);
- Construct the pricing system in a simple way for its clear understanding by users and easy implementation by administrations;
- Construct the pricing system consistently with the country's socio-economic development policy and in a participatory manner to ensure acceptability to all stakeholders.

In this regard, Cornish et al. [52] specify what assumptions would enable the implementation of a water pricing system for irrigation as the most demanding use, responsible for 70 percent of water withdrawals at a global level [53]. The main elements that emerge from Cornish et al. [52] are (i) the price of water should be neither too low, otherwise it would have no incentive effect, nor too high as this would risk incurring so-called "disproportionate costs" (Art.4 WFD), (ii) a gradual shift toward volumetric pricing is needed; however, if the physical infrastructures to implement volumetric pricing (e.g., metering devices and conveyance channels) are prohibitively expensive, the positive social impact could be offset by the costs associated with their implementation, (iii) when low-value crops are grown for reasons beyond purely economic considerations, the price stimulus may not have the expected effect, and (vi) the allocation of water permits must be efficient and transparent to minimize transaction costs. Some of these elements raise an aspect extensively debated in the academic literature, which is how the effectiveness of water pricing is influenced by the price elasticity of demand, which is discussed in detail below.

3.3.2. The Price Elasticity of Demand

The occurrence of inelastic water demand response to water prices has been identified as one of the main reasons why tariff policies are often underutilized as a tool to reduce water use. Thus, accurately measuring this elasticity is crucial to prevent failures in water pricing policies [54]. The case of demand elasticity in the agricultural sector is particularly problematic as it is highly connected to agronomic (crop types), socioeconomic, spatial, and technological contexts (see Berbel et al. [55] for a more in-depth analysis of irrigation water pricing systems in different European contexts).

In this context, the meta-analysis by Scheierling et al. [56] revealed that farmers' demand is generally unresponsive to water prices, but the findings vary widely depending on the methodologies adopted in different empirical applications and the peculiarities of specific cases. Moreover, the response to a water price increase depends on the characteristics of the demand curve, which can be influenced by several factors.

First, different crops have different irrigation needs and, therefore, different degrees of vulnerability to changes in water availability: Water-intensive crops (such as corn or rice) tend to have lower elasticity of demand since farmers do not have alternative options to reduce water use without impacting production. Conversely, crops that are more resilient to water scarcity might have more elastic demand since it is possible to act on irrigation by decreasing water use if the tariff is high enough to motivate this change in behavior. On the other hand, there is a difference between annual crops (e.g., rice and corn) and perennials (e.g., fruit trees). In the latter case, the ability to respond to water scarcity is limited because these crops do not possess the same management flexibility as annual crops [57]. In addition, the type of crop can affect the elasticity of demand depending on the value of the crop itself. The crops that could bear higher prices are mainly high-value vegetables and fruits, which make demand inelastic since the value of the product can absorb increases in water prices. In Bartolini et al. [58], five different types of crops located in different Italian regions are considered: cereals, rice, fruit, vegetable, and citrus. The study analyses the impact that doubling the price of water (from 0.15 EUR/m^3 to 0.30 EUR/m^3) would have on some outcomes of farm activities, such as profits, number of employees, water used, etc., within five different scenarios, reflecting aspects related to agricultural policies, technologies used, and the market. The results show that in the case of crops such as rice and citrus fruits, no negligible reduction in water use is estimated (when price is doubled). This change, however, is explained by land abandonment in the case of rice and by crop conversion in the case of citrus. In contrast, the price variation does not generate any reduction in water use for fruit cultivation, being a sector linked to rather efficient technologies (drip irrigation) and more rigid crop diversification.

Second, the presence of adaptation measures or elasticity of substitution: If there are alternative sources of water, for example through rainwater harvesting or the use of recycled water, demand may be more elastic. This was the case of the city of Valencia where the structural condition of water scarcity has led to the long-standing consolidation of an irrigation system based on the reuse of wastewater [53].

Third, the availability and effectiveness of irrigation technology and infrastructure can make demand more elastic: From a theoretical point of view, efficient irrigation systems, in fact, allow less and more precise use of the water resource, and in the case of increasing prices, it is easier to adjust consumption with respect to rather more inefficient and dated irrigation systems. For this reason, a variety of incentive schemes have been created to boost the adoption of these technologies. In this respect, Koncagül et al. [53] cite cases of subsidy policies in China, the U.S., Spain, Mexico, Chile, India, Morocco, and Australia. However, the increased efficiency of these systems can lead to rebound effects. This means that the water savings derived from increased efficiency do not extend to society as a collective benefit but are internalized by farmers, who tend to expand and intensify agricultural acreage, so to increase absolute water consumption. Another possible effect of improved irrigation technology is to incentivize the change in the crop portfolio towards highly profitable crops, thereby with unexpected results in terms of water use and rural development strategies [57].

Fourth, in the short-term, demand is usually inelastic because farmers have no strategies to adapt to the new price, in contrast to the long-term when structural changes can be adopted. This means that it is advisable to implement water pricing policies according to a proactive approach, and thus under normal hydrological conditions, and not as a reactive measure during a drought period.

Massarutto [59] argues that water-use tariffs produce an effect on irrigation based on the absolute value of the price rather than by the marginal one. Therefore, the author argues that the impact of pricing can be expected to be rather modest and focused on promoting efficiency (higher value added generated) rather than environmental sustainability (reduced water use). According to Tsur et al. [60], water pricing can achieve efficient allocation of irrigation water without changing the welfare of farmers if there are guarantees that some or all of the revenue generated by the application of the tariff will remain in the area of the drawdown and be reinvested to improve efficient water use. However, the same authors conclude that water pricing has little effect on income distribution within the agricultural sector, which supports the view that this tool should be used to improve water use efficiency rather than any income distribution goals and considerations.

More recent studies have shown nonlinear elasticity behaviors with threshold effects at different price levels [61–64]. These studies confirm that water pricing policies can limit agricultural production and squeeze, if not undermine, farmers' incomes. On the other hand, excessively low water prices reduce farmers' opportunity cost of changing their consumption behavior and/or of investing in technologies that reduce demand. For example, Manos et al. [65] show some inelasticity of water demand for Greek farmers at prices below 0.03 EUR/m^3, with large reductions in water use for price scenarios of 0.11 EUR/m^3. The authors point out that against this effective reduction in water use there is also a strong economic impact due to a shift to less profitable crops. A similar result is described in Sapino et al. [62], where a shift to a water price above 0.012 EUR/m^3 would result in the replacement of rice with rain-fed crops and corn in northwest Italy.

It is worth noting that in some contexts (e.g., in many Italian regions), the implementation of a fee-setting system compliant with the principles of the WFD is further complicated by the fact that the fee for deriving public water for irrigation purposes is measured on the flow rate granted (expressed in l/s) or on the hectares of arable land. It is reasonable to assume that these measurement methods do not provide sufficient information to the final user to be a proper incentive for efficient water-use behavior. In fact, a tariff based on a volumetric (m^3) calculation would already be a useful solution to stimulate users toward more virtuous behavior [50]. This result confirms the need for transitioning to a volumetric calculation of water pricing.

3.4. Economic Appraisal of Disproportionate Costs

The issue of assessing the disproportionality of costs is still partially unexplored in the literature. As emphasized by Boeuf and Fritsch [66] in their meta-analysis of approximately 90 publications related to the WFD, central themes in economic analyses such as CEA, CBA, and the application of exemptions for disproportionate costs remain under-studied in the literature and would benefit from further investigation. Therefore, this section is dedicated to studies where the assessment of disproportionality is the core issue under investigation. Following the classification adopted by Macháč et al. [67], existing approaches to the appraisal of disproportionate costs can be divided into three categories:

- Approaches based on the affordability criterion assessing society's actual capacity to contribute to the financing of measures.
- Approaches based on the monetary CBA that present a monetary evaluation of benefits.
- Approaches based on a criterial CBA that evaluate benefits not in monetary terms but according to a set of criteria and qualitative/quantitative indicators.

The EC has clarified that affordability cannot be adopted as the only criterion guiding the decision on the disproportionality [3]. Hence, the focus of this review was on analyses dealing with both types of CBA, which, however, often include the same measure of affordability.

Overall, monetary CBA studies are lightly more numerous than the ones using criterial CBA (Table 4). In summary, the main differences between monetary CBA and criterial CBA relate to: (i) the methods used to measure the benefits associated with the implementation of measures and (ii) the chosen comparison criterion (or threshold) beyond which costs are considered disproportionate. Regarding the latter point, both approaches agree that the final choice of the threshold is a purely political decision, with economic analysis providing support.

Table 4. List of reviewed studies.

Authors	Year	Method
Bolinches A., De Stefano L., Paredes-Arquiola J. [68]	2020	Criterial CBA
Macháč J., Brabec J., Vojáček O. [69]	2020	Criterial CBA
Macháč J., Brabec J. [70]	2018	Monetary and Criterial CBA
Boeuf B., Fritsch O., Martin-Ortega J. [71]	2018	Not empirical, introduction to the topic
Klauer B., Schiller J., Sigel K. [72]	2017	Criterial CBA
Klauer B., Sigel K., Schiller J. [7]	2016	Criterial CBA
Feuillette S., Levrel H., Boeuf B., Blanquart S., Gorin O., Monaco G., Penisson B., Robichon S. [73]	2016	Monetary CBA
Feuillette S., Levrel H., Blanquart S., Gorin O., Monaco G., Penisson B., Robichon S. [74]	2015	Monetary CBA

Table 4. *Cont.*

Authors	Year	Method
Martin-Ortega J., Skuras D., Perni A., Holen S., Psaltopoulos D. [10]	2014	Not empirical, introduction to the topic
Galioto F., Marconi V., Raggi M., Viaggi D. [75]	2013	Monetary CBA
Jensen C.L., Jacobsen B.H., Olsen S.B., Dubgaard A., Hasler B. [76]	2013	Monetary CBA
Vinten A.J.A., Martin-Ortega J., Glenk K., Booth P., Balana B.B., MacLeod M., Lago M., Moran D., Jones M. [77]	2012	Monetary CBA
Molinos-Senante M., Hernández-Sancho F., Sala-Garrido R. [78]	2011	Monetary CBA

3.4.1. Monetary CBA

This type of analysis is based on the comparison of the costs of implementing an intervention (in this case the PoM or part of it) with the benefits of achieving good ecological status or potential. The most complex issue in this type of analysis concerns how to quantify the benefits in monetary terms, even though other aspects are also relevant to this analysis, which are the type of costs included, the decision on the discount rate, and the threshold beyond which the costs are considered disproportionate. Regarding the first aspect, the WATECO [1] clarified that the costs to be considered should include both financial and economic costs. In terms of discount rate, given the social and intergenerational implications of the implementation of the WFD, the same guidelines suggest testing the results with different discount rates [1]. Regarding the threshold, in projects that have social impacts, the economic performance indicator considered most appropriate is the Economic Net Present Value (ENPV), obtained as the difference between benefits and total discounted social costs [79]. Alternatively, there are also other indicators, although less in use, such as the Internal Rate of Return or the ratio of Benefits to Discounted Costs. When using ENPV, a project is considered optimal from the point of view of social welfare when it has a positive ENPV, while in the case of the benefit–cost ratio, the project is economically efficient if the ratio is greater than one.

The review of the studies considered in this paper is organized around these four aspects (benefits, costs, discount rate, and threshold) and the main outputs are summarized in Table 5.

Table 5. Studies dealing with monetary CBA.

Author	Cost Estimation	Benefit Estimation	Discount Rate	Threshold
Vinten et al. [77]	CEA curves of marginal abatement costs	Choice experiment	3.5%	-
Feuillette et al. [73]	Financial costs	Benefit transfer	4% (it is unclear whether they use the same rate for all water bodies)	Disproportionate costs if benefits < 80% of costs
Jensen et al. [76]	Financial costs converted to consumer prices	Benefit transfer	Costs discounted at 6%	Range of possible benefits and costs to avoid over/underestimates
Galioto et al. [75]	Direct investment and missed income opportunities	Benefit transfer	5%	The authors suggest that decisions are taken considering a combination of indicators derived from both CEA and CBA
Molinos-Senante et al. [78]	Financial costs	Distance function approach	3%, 2%	-
Birol et al. [33]	Financial costs	Choice experiment	3.5%, 6%, decreasing rate	-

In most of the studies, cost estimates are based on the financial costs of measures, while environmental and resource costs are not included. Regarding environmental costs, this choice is explained by the fact that the measures considered in the studies aim at achieving a good water status and therefore are supposed not to cause environmental damages. For instance, Vinten et al. [77] study measures to mitigate phosphorus (P) pollution in Scottish lakes, while Galioto et al. [75] evaluated measures to tackle qualitative (pollutants) and quantitative (withdrawals) pressures in a region of northeast Italy. Jensen et al. [76] provide calculations of the welfare economic cost of measures since they account for the distortion effects associated with the tax financing of the implementation. It is noteworthy also that except for the work of Feuillette et al. [73] and Jensen et al. [76], which analyze the whole PoMs at a country level (for France and Denmark, respectively), all the other studies focus their analyses on individual or a cluster of measures; thus, a full assessment of the costs of PoMs is rather absent in the literature.

The methods applied for the estimation of the benefits derived from the implementation of measures are the same described in Section 3.2. However, in the studies dealing with the assessment of disproportionate costs, the prevailing method is benefit transfer [73,75,76]. The work of Molinos-Senante et al. [78] is an exception because the benefits, derived from measures to improve the efficiency of water treatment plants in a catchment area in Spain, are estimated through the distance function approach. The latter is based on the concept of the shadow price generated by the treatment plants. While the plants produce pollutants through water purification, they avoid the environmental impacts that would occur if the waste was released directly into the environment. The shadow price of pollutants is therefore equivalent to the environmental damage avoided by the plants, and the distance function approach makes it possible to estimate the increase in benefits from the improvement in plant efficiency. The approaches based on avoided cost by a measure has the advantage of being relatively easy to apply, but at the same time may incur the risk of underestimating the costs related to environmental damage [1]. This is why the authors emphasize the importance of considering this method as an alternative to, and not a substitute for, stated preference methods [26]. Within the category of studies applying stated preference methods to support decisions on disproportionate costs, we analyzed the papers of Vinten et al. [77] and Birol et al. [33]. The study by Birol et al. [33], in particular, is interesting for what concerns the third aspect of the monetary CBA, which is the adopted discount rate. The authors evaluate the economic viability of a project to recharge a depleted aquifer whose duration for the full realization of benefits is estimated at 200 years. In addition to applying two different constant discount rates (3.5% and 6%), the authors also use a decreasing discount rate, which decreases over the years. The authors conclude that the net benefits exceed the costs regardless of the discount rate adopted, but that with the decreasing rate the benefits outweigh those obtained at constant rates, suggesting an important policy recommendation when evaluating long-term projects [33]. Most of the papers, however, consider different discount rates in the sensitivity analysis to account for uncertainty.

Finally, concerning the comparison criteria (i.e., the threshold), given the uncertainties that characterize estimates of costs and benefits, some studies opt for less stringent criteria. For example, in the French approach to CBA described by Feuillette et al. [73], costs are deemed disproportionate to benefits when the latter are less than 80% of the costs. However, as the authors themselves point out, establishing this threshold implies a certain level of arbitrariness [73]. To reduce subjectivity, Galioto et al. [75] suggest basing decisions on a combination of indicators derived from the CEA and the CBA. According to the authors, intervention priorities should be identified for those areas that present the best estimates in both analytical tools. Jensen et al. [76], on the other hand, do not set a threshold but adopt an approach they call 'cautious' as it aims to reduce the risk of overestimating benefits and underestimating costs. In their ENPV calculations, they use the lowest value of benefits and the highest value of costs in a range of possible values of the estimates. The results of this "pessimistic" approach are then compared with different values of the benefits through sensitivity analysis.

3.4.2. Criterial CBA

Studies belonging to this second type of CBA developed in Germany as a response to the need for implementing administrations to adopt less costly and complex approaches to assessing the disproportionality of costs [7]. In criterial CBA, in fact, benefits are not assessed in monetary terms but according to a set of previously defined (qualitative/quantitative) criteria. Another feature of these studies is that the assessment of disproportionality is made by comparing the costs of implementing the measures with an indicator that approximates the ability of society to cover the necessary costs [68].

The so-called "New Leipzig approach" or benchmark approach is a procedure developed by Klauer et al. [7] and tested on a group of seven water bodies in Germany, then on of the entire water bodies of a German federal state [72], in a river basin in the Czech Republic [70], and in a Spanish river basin [68]. It is worth saying that the benchmark approach has also been taken as a reference methodology in the Italian national guidelines for the assessment of disproportionated costs of WFD measures [80].

There are some differences between these different applications but the core idea of this approach is to determine for each water body (or group of water bodies) a specific disproportionality threshold that is compared to the costs to achieve the WFD's objectives [7]. Klauer et al. [7,72] clarify that some preliminary steps are required to conduct this analysis, which are the calculation of the national average past expenditure on water protection normalized for the river basin/catchment area and an appraisal of the costs of measures. After that, the method foresees the creation of a parameter called the Effort Factor (EF), which identifies the additional effort that is required for the achievement of the good state objective for each water body or group of water bodies. The EF, in turn, is defined as a function of two parameters: the "objective function", that is the gap between the current state and the WFD targets, which is determined through a set of quality indicators with the assumption that the higher the gap the greater the benefits, and the "additional benefits", which are positive side-effects coming from increased water state determined through expert-based consultation conducted with a standardized scoring method. To obtain the water body (or per group of water bodies) cost threshold for disproportionality, the EF is multiplied for the average past public expenditure on water protection, normalized for the catchment area. According to the authors, this water-body cost threshold should not exceed the average past expenditure on water protection by 50%, otherwise the disproportionality criterion applies. The authors clarify that the decision of setting the threshold at 50% is intrinsically political and should be agreed with the implementing authorities.

Among the main limitations of this approach, Macháč and Brabec [70] note that the use of past expenditure as an indicator can be misleading as it does not necessarily reflect the resources used to achieve the environmental objectives of the WFD. On the other hand, by comparing this approach with a monetary CBA in the same study area, the authors point out that this approach is less complex and quicker to perform and recommend its application in water bodies that do not have critical conditions or as a preliminary disproportionality analysis.

Other weaknesses concern the calculation of the objective distance, which in the study of Klauer et al. [7] is based only on a limited set of parameters, and the subjectivity of the evaluation of the additional benefits. To overcome these limitations, Bolinches et al. [68] propose (i) to refer to Eurostat data, corrected for the GDP of the study area, for the calculation of past expenditure, (ii) to eliminate subjective elements such as the additional benefits as well as the definition of a threshold that, according to the authors, should be at the EU level, and (iii) that the objective distance is based on the whole chemical, biological, hydromorphological, and physico-chemical elements included in the WFD for the assessment of the water body status.

Regardless of these differences, the authors agree that the benchmark approach can be a valid support for decisions on disproportionate costs only for those cases that do not present particular criticalities or as a preliminary screening to a more in-depth evaluation.

4. Discussion and Conclusions

This paper provides a comprehensive review of the grey and scientific literature dealing with the implementation of the economic requirements and the application of the GA under the EU WFD. The four topics addressed in this review still represent significant challenges for the implementing authorities and the analysis confirms that they are also open issues in the academic debate. For each topic, a summary of the main research gaps is presented below, together with a brief discussion on possible future research directions.

The Gap Analysis. This review has found a low number of studies dealing with the GA under the WFD and with different approaches and applications; such a result is partly due to the limited adoption of this method by the implementing authorities, as reported by the EC. A closer connection between the RBMPs and the GA itself should enhance its application in both the scientific and grey literature. Future research might focus on investigating and comparing the GA methodologies effectively implemented in each MS and assessing their performances.

The economic valuation of EGS. The literature corroborates the findings of numerous similar analyses on the economic valuation of ecosystem services. It underlines significant heterogeneity in the estimated WTP, attributable to the various methodologies employed and the specific contexts in which each study is applied. Moreover, in the context of the WFD and the application of the DPSIR, it is evident that most of the studies are not fully compliant with the approach as they are dedicated to the estimation of one or more ecosystem services instead of the pressure exerted on water resources. These results clearly make the application of the Benefit Transfer method more complex and expose the practitioners to the risk of errors, in some cases significant, in transferring the monetary valuation. This risk undermines the potential use of Benefit Transfer in CBAs and policy evaluation. As a result, additional effort from scholars and practitioners is necessary. On one hand, there is a need to encourage the implementation of primary studies that adhere to the DPSIR approach. On the other hand, maintaining a consistent and systematic update of literature reviews pertaining to the economic valuation of ecosystem services is essential to improve future applications of meta-analyses and the Benefit Transfer method.

Water pricing. Given the specificity of the results in terms of variation in demand to the price stimulus (with potentially negative effects in some areas), pricing needs to be included in a broader set of tools to be used according to local specificities from environmental, hydrological, land-use, and socio-economic perspectives. Quotas, subsidies, water-use planning and stakeholder involvement in the shared use of the resource cannot be placed on a secondary level in the preparation of large-scale management policies. In this regard, hydro-economic models are emerging to provide useful analytical tools for assessing the effectiveness of water resource management at the basin level. The advantage of these models is that they are able to consider different dimensions of the system, such as agronomic, hydrologic, environmental, and economic conditions in order to provide a comprehensive and basin-specific assessment of water policy options in a climate change context [57]. Certainly, in a future likely characterized by a rise in extreme events, the effectiveness of water pricing policies may diminish if not integrated within a broader management framework that engages users through participatory systems [61].

Disproportionate costs of measures. CBAs have been carried out in a very heterogeneous way across EU countries, partly because of the lack of a clear direction provided by the EC. This resulted in the adoption of different parameters, both across and within MS, for the decisions regarding the use of exemptions for disproportionate costs (article 4.5). The findings of this review corroborate the argument of Berbel and Expósito [61] of the need to advance towards harmonized CBA approaches among MS, at least with regard to the inclusion of aspects related to equity and the affordability of users/polluters, the consideration of a wide range of benefits associated with the measures that are not strictly related to water, the involvement of stakeholders, and the assessment of uncertainty about the effectiveness of the measures under changing future conditions. This need for harmonized approaches is made even more relevant by the fact that some countries, for instance Germany and Italy, are opting for practitioner-oriented approaches. On the one hand, these approaches have the advantage of moderate data requirements and less complex analytical skills; on the other hand, they are based on some arbitrary choices (on the evaluation of benefits) and heterogeneous parameters (the use of national past expenditures on water protection). Given these limits, we argue that a monetary evaluation of non-market benefits should remain the prevailing method for the decision on disproportionate costs of measures, at least for those cases presenting more critical aspects. However, some aspects of the criterial CBA could be maintained and integrated: for instance, the criterial CBA explicitly foresees the evaluation of the gap (through the "objective function") as one of the parameters to determine how much public expenditure is worth spending to restore water bodies. Moreover, compared to monetary CBA studies, criterial CBA is applied to the entire PoMs and not only to some individual measure.

Besides the four topics above, a few general conclusions can be drawn. Overall, one issue remains the feasibility of the ambition of economic analysis in the WFD, after more than 20 years of experience. After the Fitness check in 2019 [81], the European Commission confirmed that the WFD is broadly fit for purpose and that "the objectives of the directives are as relevant now as they were at the time of the adoption, if not more". In 2020, the EC announced that the WFD would not be revised (as it was initially expected), while it adopted a proposal to revise the lists of pollutants in surface water and groundwater [82].

At this point, as highlighted in the Fitness Check, the main focus for the following years is on the implementation of measures, together with the assessment of their own effectiveness. Based on this consideration and on the results presented in this review, a few priorities aimed at better exploiting the potential of economic analysis and the GA as support tools are summarized below:

- First of all, in order to perform a sound GA, more consistent and complete datasets should be developed by competent authorities, both in terms of monitored parameters and measures effectively taken. Particularly, environmental quality data require a punctual and precise monitoring program. This would allow the GA to better identify potential measures to address the mismatch in the status of water bodies so as to focus economic studies on the relevant problems of river basins. Furthermore, the GA would perform better if implemented in such a way that gaps are expressed in technical parameters (e.g., unit of fertilizer needing reduction) rather than in abstract distance calculations from the good status objective.
- Such a consideration is closely connected to the adoption of research approaches that do integrate the analytical requirements of the WFD, highlighting the nexus between the assessment of the *ex-ante* status of water bodies and the economic evaluation of the PoM put in place. This might turn out to be of a particular interest to identify the structural struggles in the implementation of the environmental goals of the WFD throughout different MS.
- There is the need to promote better consistency and interaction among the different components of the economic analysis, which can best be achieved through a more pragmatic embedding of the economic analysis in the decision-making procedures.
- From the procedural point of view, this highlights that timing (early start) and integration of economic analysis into the process, as well as the recognition that the economic analysis is not a substitute for the political process, are key for meaningful results.

Lastly, the new climate context contributes to changing the conditions for economic evaluation. Increasing risks related to climate change pose at least two main challenges for the implementation of both economic analysis and the GA. On one hand, climate change brings more uncertainty in the evaluation of the costs and benefits related to planned measures. On the other, the increasing frequency of extreme events (e.g., floods and droughts) urges a constant monitoring of environmental parameters and, in some cases, may require re-evaluating and re-considering the effectiveness of planned measures. In this regards, scenario and sensitivity analyses are increasingly needed to support decision making by providing information on the optimal level of activation of policy measures with respect to expected environmental damages generated by climate change.

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Abbreviations

BSAP	Baltic Sea Action Plan
CBA	Cost–benefit Analysis
CE	Choice Experiments
CEA	Cost-Effectiveness Analyses
CV	Contingent Valuation
DPSIR	Determinant Pressures State Impact Resources
EC	European Commission
EGS	Ecosystem Goods and Services
ENPV	Economic Net Present Value
GA	Gap Analysis
MAR	Managed Aquifer Recharge
MS	Member State
MSFD	Marine Strategy Framework Directive
PoM	Programme of Measures
RBDA	River Basin District Authority
RBMP	River Basin Management Plans
WFD	Water Framework Directive
WTP	Willingness to Pay

Appendix A

Table A1. List of studies selected.

Authors	Method	ESG	Estimated WTP (EUR 2020)			Unit of Measure and Country
			min	max	mean	
Damigos et al. (2017)	CV	F8	43.7	70.66		EUR/year/household (Italy)
Genius et al. (2008)	CV	F8	8.69	14.42		EUR/year/household (Greece)
Genius et al. (2012)	CE	F8	23%	39%		% bill increase/ trimester/household (Greece)

Table A1. *Cont.*

Authors	Method	ESG	Estimated WTP (EUR 2020)			Unit of Measure and Country
			min	max	mean	
Hein (2011)	Replacement Costs	F8			0.47	EUR/m^3 (Netherlands)
Martin-Ortega & Berbel (2010)	CE	F8	11.23	86.471		EUR/year/household (Spain)
Polyzou et al. (2011)	CV	F8			66.86	EUR/year/household (Greece)
Beaumais et al. (2010)	CV	F8	106.20	414.546		EUR/year/household (Italy)
Ruperez-Moreno et al. (2015)	CV	F8			19.58	EUR/year/household (Spain)
Koundouri et al. (2014)	CE	F8	46.38	48.35		EUR/year/household (Greece)
del Saz-Salazar et al. (2016)	CV	F8	8.53	10.00		EUR/year/household (Spain)
Ahtiainen et al. (2015)	CE	R4	64.27	126.07 [1]		EUR/year/household (Finland)
Alcon et al. (2010)	CV	R4	60.37	77.89		EUR/year/household (Spain)
Genius et al. (2008)	CV	R4	8.69	14.42		EUR/year/household (Greece)
Genius et al. (2012)	CE	R4			0.26	% bill increase/household/ trimester (Greece)
Morris & Camino (2011)	Benefit Transfer	R4			417.41	£/ha/year (UK)
He et al. (2017)	CV + CE	R4			65.60	$CAN/year/household (Canada)
La Notte et al. (2015)	Replacement Costs	R4			45.24	EUR/km (Italy)
Koundouri et al. (2014) [2]	CE	R4	39.64	49.18 [3]		EUR/year/household (Greece)
Chen et al. (2017)	CE	R4			67.58	EUR/year/household (Belgium)
del Saz-Salazar et al. (2016)	CV	R4	10.55	12.38		EUR/year/household (Spain)
Börger et al. (2021)	CV	R4			134.42	EUR/year/household (UE)
Brouwer & Bateman (2005) [4]	CV	R6	90.07	512.32		£/year/household (UK)
Markantonis et al. (2013)	Contingent Valuation	R6			59.42	EUR/year/household (Greece)
He et al. (2017)	Contingent Valuation + Choice Experiments	R6	12.95			$CAN/year/household (Canada)
Morris & Camino (2011)	Benefit Transfer	R6			90.63	£/ha/year (UK)
Bateman et al. (2006)	Contingent Valuation	R9	20.59	49.09 [5]	32.68	£/year/household (UK)
Morris & Camino (2011)	Benefit Transfer	R9			483.36	£/ha/year (UK)
Hanley et al. (2006)	Choice Experiments	R9			36.76	£/year/household (UK)
He et al. (2017)	Contingent Valuation + Choice Experiments	R9			33.29	$CAN/year/ household (Canada)
Koundouri et al. (2014) [2]	Choice Experiments	R9	39.64	49.18		EUR/year/household (Greece)
Chen et al. (2017)	Choice Experiments	R9			31.02	EUR/year/household (Belgium)
Buckley et al. (2016)	Contingent Valuation	R9			19.47	EUR/year/person (Ireland)
Stithou et al. (2012)	Choice Experiments	R5			30	EUR/year/household (Ireland)

Table A1. *Cont.*

Authors	Method	ESG	Estimated WTP (EUR 2020)			Unit of Measure and Country
			min	max	mean	
Hanley et al. (2006)	Choice Experiments	R5			34.17	£/year/household (UK)
Chen et al. (2017)	Choice Experiments	R5			27.7	EUR/year/household (Belgium)
Kourtis & Tsihrintzis (2017)	Contingent Valuation	C2	11.54	49.89		EUR/year/person (Greece)
Brouwer & Bateman (2005) [4]	Contingent Valuation	C2	89.43	508.69		£/year/household (UK)
Bateman et al. (2006) [5]	Contingent Valuation	C2	20.59	49.09	32.68	£/year/household (UK)

Note(s): [1] Median WTP for the reduction in blue-green algae and slime. [2] The attribute evaluated in the study deals with ecological status and is defined as a set of different aspects related to services R4 and R9; the extremes of the range indicate the WTP for the maximum achievable improvements for the reference attribute in the two municipalities considered in the study. [3] The range indicates the WTP for the maximum achievable improvements for the reference attribute in the two municipalities considered in the study. [4] The range of WTP estimates is quite wide because the first estimate was obtained with different econometric models; the authors themselves point out in the body of the article the great difference between the estimates obtained with different model types. In addition, it should be noticed that, according to the methodology used, the WTP presented is for two classes of EGS jointly evaluated (R6 and C2). [5] Values of average WTP depending on whether small, medium, and large changes in river ecosystem quality are considered.

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