Scholarly Publisher
RS Global Sp. z O.O.
ISNI: 0000 0004 8495 2390

Dolna 17, Warsaw, Poland 00-773
Tel: +48 226 0 227 03
Email: editorial_office@rsglobal.pl

JOURNAL	International Journal of Innovative Technologies in Economy
p-ISSN	2412-8368
e-ISSN	2414-1305
PUBLISHER	RS Global Sp. z O.O., Poland

ARTICLE TITLE	ECONOMIC ASSESSMENT OF THE IMPLEMENTATION MEASURES OF EUROPEAN WATER FRAMEWORK DIRECTIVE
AUTHOR(S)	Yuli Radev, Desislava Simeonova, Reneta Barneva, Lisa Walters
ARTICLE INFO	Yuli Radev, Desislava Simeonova, Reneta Barneva, Lisa Walters. (2022) Economic Assessment of the Implementation Measures of European Water Framework Directive. International Journal of Innovative Technologies in Economy. 2(38). doi: 10.31435/rsglobal_ijite/30062022/7814
DOI	https://doi.org/10.31435/rsglobal_ijite/30062022/7814
RECEIVED	24 March 2022
ACCEPTED	16 May 2022
PUBLISHED	20 May 2022
LICENSE	This work is licensed under a **Creative Commons Attribution 4.0 International License.**

ECONOMIC ASSESSMENT OF THE IMPLEMENTATION MEASURES OF EUROPEAN WATER FRAMEWORK DIRECTIVE

Yuli Radev, *University of Mining and Geology, Sofia, Bulgaria*
Desislava Simeonova, *University of Mining and Geology, Sofia, Bulgaria*
Reneta Barneva, *State University of New York at Fredonia, USA*
Lisa Walters, *State University of New York at Fredonia, USA*

DOI: https://doi.org/10.31435/rsglobal_ijite/30062022/7814

ARTICLE INFO

Received 24 March 2022
Accepted 16 May 2022
Published 20 May 2022

KEYWORDS

European Water Framework Directive (WFD), cost effectiveness analysis (CEA), cost-benefit analysis (CBA), East Aegean Region (Bulgaria).

ABSTRACT

In this article we analyze the measures against pollution in river basins that follow the European Water Framework Directive (European Commission, 2000) and propose a methodology for assessing their economic effectiveness. Compared to other similar studies (Berbel et al., 2018), the presented methodology has been developed and tested in rivers where water pollution is a result of mining activities. In terms of economic theory, the methodology can be summarized as follows: The cost effectiveness analysis used to select the optimal mix of costs is integrated into the cost-benefit analysis to assess the cost-effectiveness of the proposed measures. The methodology has been tested on a case study of the East Aegean Region and recommendations for the region have been made for the next five-year period of the Directive.

Citation: Yuli Radev, Desislava Simeonova, Reneta Barneva, Lisa Walters. (2022) Economic Assessment of the Implementation Measures of European Water Framework Directive. *International Journal of Innovative Technologies in Economy*. 2(38). doi: 10.31435/rsglobal_ijite/30062022/7814

Introduction. Water is an indispensable resource for human beings. It can also be considered as a complex economic good and a key driver of sustainable growth and development (Goswami & Bisht, 2017). The European Water Framework Directive (WFD) sets out the principles guiding the policies of the European Union (EU) Member States as well as the choice of economic instruments for controlling the use of water resources on the base of methods for economic assessment and adopting the *Principle of Recovering Full Cost* and the *Polluter Pays Principle.*

According to the WFD, the European Union's water quality objectives were to be achieved by 2015. Potential extensions of the deadline are allowed either for reasons of technical feasibility or because of a disproportionate cost (European Commission, 2000; Article 4, paragraphs 4, 5, 7). These reasons justify the possibility of extending the deadline for achieving *Good Environmental Status* (GES) by 2027. If the costs are disproportionate, lower targets may be established, to achieve *Acceptable Ecological Status*.

The European Commission has developed several methodological rules to carry out an economic analysis (European Commission, 2009). These rules offer various assessment tools, depending on the different strategies and policies of the individual EU member states (Brouwer, 2008). The evaluation itself supports the process of political decision-making and provides the necessary transparency (Voulvoulis et al., 2017). However, the general nature of the rules does not define the practical procedures, which each country could use to assess the benefits and costs associated with a list of necessary measures (Jensen et al., 2013). In an attempt to resolve this problem, a number of articles appeared, including Postle et al. (2004), Berbel et al. (2011), and Jensen et al. (2013). They offer rules and criteria for economic evaluation that (at least to some extent) limit the influence of the subjective factor. Specific for the cost-benefit analysis presented here is the focus on the individual sources of pressure on the water bodies without taking into account the multiple impacts of the applied measures. Martin-Ortega et al. (2013) fairly criticized this approach suggesting the economic analysis to include such elements as the spatial and temporal scale of the evaluation, the cost-benefit sharing, and uncertainty.

Analyzing the WFD methodological guidelines and the good research practices, in this paper we propose a methodology to assess the cost-effectiveness of the measures for restoring the ecological status of water bodies. As an illustration, we consider its application in the case study of the East Aegean Region. The methodology was tested in mining and ore processing regions, but could also be applied to other regions and water areas.

Although it is not possible to construct a generic framework, most empirical studies on the cost-effectiveness of WFD implementation contain three mandatory elements:

(1) Economic assessment methods necessary to quantify the social, economic, and environmental effects;

(2) Comparative criteria and threshold values; and

(3) Spatial scale of the analysis (water body, part of a water body, pool, sub-basin, administrative district or a particular region) (Ward, 2009).

Our study addresses these three elements.

The paper is organized as follows: the next section describes the methodology for the economic assessment of measures for implementation of WFD. Then the case study of its application in the East Aegean Region is considered. Finally the limitations of the methodology are discussed.

Methodology for economic assessment of WFD implementation measures. The most commonly used methods for economic evaluation of large investment projects for environmental purposes are cost effectiveness analysis (CEA) and cost-benefit analysis (CBA). The CEA compares the monetary values of the costs and the physical benefits of the measures taken (i.e., the costs are compared with the reduced level of pollution), while the CBA compares the cost-benefit monetary costs (i.e., costs are compared to the direct and indirect benefits of the improved environmental status). CEA avoids the discrepancies related to the monetization of some intangible assets, such as the environment, and is therefore a preferred tool in the comparative analysis of alternative measures. As the CBA method is assessing not only costs, but also tangible and intangible assets, it is appropriate for an overall assessment of the economic effectiveness of the adopted measures or combinations of measures.

The preferences of European researches to the CEA method probably are due to the fact that the evaluation of benefits has encountered a number of difficulties. However, in some studies in the UK, Scotland, France (Seine and Normandy), the Netherlands, and Denmark, the CBA method was used. To reduce the effort needed to assess the benefits, economists use two strategies. The first one includes limiting the application of CBA for those water bodies for which CEA results conflicts with the expectations of local stakeholders. The second one is based on the assessing of benefits in other studies with similar objectives and similar conditions. The first strategy was implemented by Postle et al. (2004) in England and Interwies et al. (2005) in Scotland, while the second strategy, also known as the Benefit Transfer Method (TM), was used by Laurans (2006) and Jensen et al. (2013) in France and Denmark, respectively.

For efficient using of CBA and/or CEA methods reliable benchmarks and thresholds are needed. The results of the CBA are presented as a difference or a cost-benefit ratio. An investment project is economically viable when the difference is greater than zero or the ratio is greater than one. In general, the results of the CBA and CEA approaches should lead to the same conclusions. However, the CEA results are strongly dependent on the conditions under which the comparison is performed. Therefore, through additional indicators different perspectives of obtained results are examined. For example, the cost of implementing the measures could be compared with the benefits from lower level of environmental pressure in the area of the whole region, as well as the financial capabilities of the economic players who are expected to meet the costs (Berbel et al., 2018).

According the CEA method, the cost of achieving GES (Good Ecological Status) is effective if it is lower than the respective threshold. If the cost is over the threshold, it is necessary to either reformulate the time horizon (European Commission, 2000; Article 4 (4)) or recommend measures with less ambitious environmental objectives (European Commission, 2000; Article 4 (5)). Ordinarily, the economists associate the thresholds to household budget. Therefore, the thresholds usually range from 2.5% to 4% of the per capita income (Borkey, 2006, p. 12). However, as we mention above the WFD does not define either the type of indicators or the level of the appropriate thresholds, leaving the choice exclusively at the discretion of the local authorities (Voulvoulis et al., 2017).

The lack of information and/or the high level of uncertainty of some key technical and economic indicators may justify lower thresholds than those normally required in the CBA. Such adjustments increase the role of the subjective factor, thus compromising the confidence in the assessment method. One of the ways to overcome this problem is to combine the indicators of the two methods – CEA and CBA. In such cases, the policy prescriptions should be directed toward intervention in areas with best assessments from both methods.

The choice of evaluation methods and benchmarks is complemented by the choice of the most appropriate scale of economic analysis. According to the Common Implementation Strategy of the WFD (European Commission, 2003), the water body is the reference unit for achieving the target water status and represents the minimum scale at which each EU Member State has to identify the sources of pollution and measures for surface and groundwater rehabilitation. However, the "optimal scale" of the analysis is not defined either in the official recommendations or in the other empirical studies. Therefore, the scale of analysis is a key factor in the final assessment of the economic efficiency. For example, costs which are too high for a particular body of water may be acceptable at a higher scale of analysis.

The larger areas generally allow for otherwise inexpensive economies of scale, as well as for more accurate assessment of the local conditions. It is therefore important to identify such regions that are homogeneous both in terms of natural and of socio-economic conditions. In relation to this, Stemplewski et al. (2008) recommended that water bodies are aggregated into sub-basins. According to the authors, it is within this framework that the technical problems related to the scale of analysis are minimized and, at the same time, consistent evaluations are obtained.

The approach in our methodology can be summarized as follows. The effectiveness of WFD measures is assessed in terms of target water status and is limited to pre-selection of the measures by which this status can be achieved in the most effective way. There are two parallel analyses – of cost and of benefits. The cost estimate is calculated after selecting the set of measures and calculating their unit value and total value. After that, the measures are revised until the minimum level of expenditure is reached, i.e., the level at which no more savings are possible. The value of the benefits is assessed on the basis of an a priori classification of the positive effects of reaching the target status. The benefits and costs of the individual water body are assessed; then the benefits and costs of a part of a water body, or of a combination of parts of underground and surface water bodies are assessed. After that, a staged aggregation is undertaken to cover the entire water area. Efficiency analysis is performed at each level using CEA (Cost Effectiveness Analysis) and CBA (Cost Benefit Analysis) methods.

The presented methodology is appropriate to assess the proportionality of the costs and the technical inability leading to a temporary derogation. The latter is often interpreted as a compromise to efficient measures, the realization of which will take longer to reduce the pollution to the desired levels. The proportionality/disproportionality assessment depends from the desired status of water and the chosen measures against pollution. Proportionality means measures leading to the objective status in a cost-effective manner.

Application of the methodology for the east Aegean region.

As an illustration of the methodology, its application to the case study of the East Aegean Region is presented below.

Water Status in the East Aegean Region.

Regional Inspectorates of Environment and Water (RIEWs) determine to what extent individual economic sectors (agriculture, communal, and industrial) are responsible for the various forms of pressure (qualitative/quantitative, point/non-point) over the water bodies in the country. As a rule, the majority of point sources of pollution are due to the industrial sector which generates organohalogen and metal pollutants. Agriculture and livestock cause diffusion pollution (nitrogen, phosphorus, pesticides) and may have quantitative impact (over-exploitation of water resources). Morphological changes, in turn, are mainly associated with the extraction of inert materials.

The functional purpose of each body of water could be classified as drinking water, bathing water, and fish/shellfish water. Bathing water in the East Aegean basin is generally of good ecological quality, while fish/shellfish water is declining in quality as a result of the outgoing natural processes. The impact of various factors threatens the fish/shellfish water and worsens the quality of drinking water in most plain areas in the region.

Before assessing measures for different water bodies, we have to assume one of the two hypotheses about interconnections between them:

(1) The water body interacts with (almost) all water bodies in the area and its restoration is part of the aggregate restoration of the river basin or one of the three geographic areas in the region.

(2) The water body only interacts with adjacent water bodies, so its assessment focuses on an individual aggregate within the basin. In this case, the cross influence of the impact from the connected aggregates is also taken into account.

An intermediate variant of the two extreme hypotheses is an aggregate comprising surface water bodies and groundwater flowing these bodies.

In the preliminary study of contaminated by mining enterprises water in the East Aegean region, presented in the subsection *Estimates of the benefits of anti-pollution measures from mining and ore processing*, the intermediate hypothesis was accepted. Such a choice is justified, because the impact on the water bodies in this specific case is carried out by point sources of heavy metals and each measure is directed to a specific source of pollution (von Schiller et al., 2017). Unfortunately this approach ignores the possible diffusion of heavy metals (Yanger, 2001).

To simplify the methodology, we assume two degrees of ecological status – "lower" and "good or better." In most district management plans of the Basin Directorates, however, the ecological status of the water bodies, and hence the assessment of the benefits of the measures taken, is carried out on a five-level Likert scale.

The plurality of complementary/substitute measures is determined by the sources of pressure that would affect with highest probability the environmental status of water bodies in the region. The choice of the measures is coordinated with all the stakeholders, including the Regional Environment Agency managers and water protection programs. The process of selection excludes those already adopted under other related directives (e.g., the Nitrate Directive, 2020 or the Habitats Directive, 2020).

According to the Basin Directorate of the city of Plovdiv, nutrients, pesticides, heavy metals, morphological changes and over-use of water are the main threats to the water resources in the East Aegean region. The degree of impact varies: 44% is the threat of nitrogen and phosphorus concentrations, 3% of pesticides, 5% of heavy metals, 6% of water scarcity, and 15% of morphological changes.

The strongest pressure on the environmental status of water bodies in the region are the nitrogen and phosphorus from the agriculture and utilities sectors and, to a lesser extent, the industrial point-sources of pollution. The utility sector also worsens the ecological status of water mainly in urban areas.

In farming, pollutants are managed with voluntary savings (e.g., fertilizers are replaced with manure; see Rural Development, 2020) and mandatory restrictions on the use of fertilizers in nitrate vulnerable zones (Nitrate Directive, 2020) are being addressed. Additional restrictions are imposed by the measures of implementing of the program for extensive development of this sector. Actually the reduced intensity lowers the nutrient content of both surface and groundwater bodies (Macgregor and Warren, 2016). As for the pesticides, following Directive 2009/90 (EC), three types of measures for recovering chemical status of water bodies have been taken: (1) a ban; (2) dosing; and (3) substitution.

In the utilities sector, improvement was made in the efficiency of the existing treatment plants. This improvement activity facilitates the reduction of landfill waste.

The industrial sector, including the mining enterprises, is the major source of quality pressure. Depending on the distribution of different categories of chemicals, construction of new treatment plants for heavy metals and hydrocarbons, as well as re-cultivation of the industrial zones contaminated with organohalogen, is recommended. The extraction of inert materials from the nearby rivers is a source of morphological changes. Therefore, the industrial sector is also a potential threat for water scarcity (the main source of quantitative pressure, however, is the agriculture). In such cases, the measures include the construction of sewage treatment plants and distribution networks for drainage water.

Rules for Assessing the Costs and Benefits of the Proposed Measures.

In the proposed methodology, the costs and the benefits are involved not in their net present value, but in their annuity equivalents. Annuity values are interpreted as single or total costs and benefits averaged over one year. In this manner, the difficulties in comparison of measures that would produce future effects without initial costs are avoided.

The costs in the methodology are estimated in the same way as the way they are reported when activating measures for achieving GES in each particular sector. This means that all transfers (taxes or subsidies) from one economic sector or player to another one are excluded and there is no distinction between financial and economic costs. Such an approach is compatible with the *Polluter*

Pays Principle, but it is preferred because there is a lack of sufficient data on the actual impact of the multiple measures (who pays for what) and hence it is impossible to carry out more accurate analyses.

In cost estimation procedures, key factors are the consumer price index, the cost of the capital, and the time horizon representing the operational life of the investment. In most economic analyses of the WFD, cost of the capital of 2.5-4% is assumed. Because of the higher risk and risk premium, we increased this rate to 5%. This higher rate was also recommended by the European Commission for the period 2009-2015. It is also assumed that the European practice should be analyzed in a 30-year time horizon. According to the economics theory, the rate and the time horizon should correctly reflect the opportunity cost to achieve GES for future generations.

Most of the data needed for evaluation of investment projects are available from previous publications as well as from the technical information provided by the Basin Directorate of East Aegean Region. In the utility sector, a unit cost for the modernization of the treatment plants and the construction of new sewage systems was estimated. In the industrial sector, including the case of heavy metal pollution, the average annuity cost of construction of modern wastewater treatment plants with different capacity was determined.

A serious barrier to accurately assessing the measures is the inability of obtaining comparable results for the different sources of pollution. A commonly used method for solving this problem is employing corrective procedures. The nitrogen and the phosphorus from nutrients, for example, are measured in P-equivalents, with the nitrogen value divided by ten to account for its lower ecological (eutrophication) effect.

For the assessment of unit costs in the construction or reconstruction of sewage treatment plants, a similar approach applies to mining and chemical industries. Heavy metal compounds are converted into comparable units thanks to the freshwater Aquatic Eco-toxity Potential Index (fAETP). This index was introduced by Huijbregts et al. (2000), with the idea that all substances should be presented in terms of one reference substance. For heavy metals one unit of 1.4-dichlorobenzene (para-dichlorobenzene) equals one unit of fAETP. Based on this, the relative weights of 181 elements are determined. The fAETP values of the most common ones are as follows: para-dichlorobenzene=1; mercury=1700; cadmium=1500; lead=9.6; zinc=920; copper=1200; nickel=3200; chromium=28; arsenic=210 (Van Soesbergen et al., 2008).

The unit costs per cubic meter of purified water are obtained by dividing the total cost by the annual flow (0.04 €/mc^3 in microfiltration, 0.05 €/mc^3 in precipitation). The flow rate of the treatment plants is 5000 m^3/day for chemical industry companies or 1000 m3/day for metallurgic industry companies. The estimates of the cost per cubic meter depend on the total costs and on the level of functionality of the basic technology.

The ultimate goal of the Basin Directorates in Bulgaria is to develop a comprehensive catalog of measure and unit costs for their implementation, including a specialized catalog on pollution from mining and processing plants and we believe that our work is a contribution to this goal.

Benefits assessments are carried out in accordance with Annex I of Guide No 20 (European Commission, 2009). Several categories of values (use value, non-use value, side effects from other sectors, and cross-effects from other environmental projects) are listed in the Guide. Systematic analysis of all aspects of the benefits requires an extremely large and costly study. Moreover, some of the benefits listed are difficult to present not only in monetary terms but also in physical terms. Only the categories of use and non-use values are evaluated in the presented methodology, using additional checklist for some categories.

In a similar way as in the cost analysis, several important assumptions were accepted in the benefit analysis:

(1) First, when assessing the non-use value, it is assumed that the benefits are generated only when the water bodies reach a good status. In economic terms it means that the effects of the range outweigh the effects of scale. This assumption justifies the two-level status of water: unsatisfactory status and satisfactory status. However, with this assumption we miss the opportunity to analyze intermediate levels of status improvement.

(2) The second assumption is related to the economic effect of substitution. This effect suggests that the estimated value of the water body depends on the presence of substituting water bodies. In economic theory with the increase of the substitution opportunities, the value of the product decreases. In the case of water bodies, this is true when non-use values are site specific. In the present

methodology, however, the non–use value refers to the whole area and it is apportioned to the separate water body depending on its relative share in the total water reservoir of the region. Therefore, such substitution effects do not occur (they are equal to zero).

(3) The third assumption is about the effect of the distance: an increase in the distance changes the established correlations. However, our methodology does not take into account this effect, although in some cases there are deviations in the assessment of the benefits (Bateman et al., 2006).

Some assumptions have been made regarding the method of assessment of the non-use value – the so-called benefit transfer (BT) method. With this method, estimates of the non-use values in the explored area are calculated on the basis of the results of investigations in other areas. Such adaptation processes usually generate distortions. The closer the two areas from socio-economic point of view, the smaller the distortions. Ideally, the areas would be from the same country. Due to the lack of studies of the WFD in Bulgaria, for the assessment of the benefits in the East Aegean Region through the BT method, the results of the studies of the Northern Italian region of Emilia-Romagna are used. When using the BT method, it is assumed that there is no distortion of the estimates.

Technically, the assessment of the non-use value requires methods based on the interpretation of the economic perceptions (or subjective values). This calls for multiple (and therefore extremely expensive) interviews. The BT method saves these inconveniences by transferring existing estimates of non-market values from one location to another. However, the authors have intentions to develop of a catalog of the benefits from WFD measures for water bodies in Bulgaria.

As a rule, the use value is assessed for each individual sector that benefits from improving the water quality. The assessment depends on the functional use of the water resources (drinking, bathing, fish/shellfish water) and the indirect damage caused by over-consumption of water. In Bulgaria's water basins, pollutants are mainly detrimental to the quality of drinking water. Regarding the benefits, the achievement of GES leads to reducing the cost of water treatment for bringing it to drinking standards and to reducing the costs related to solving the problem of water shortages in long periods of drought.

Estimates of the Benefits of Anti-Pollution Measures from Mining and Ore Processing.

Use benefits are assessed through the savings made. The achievement of GES saves the treatment of nutrient-contaminated water as well as the emergency response caused by water scarcity. The unit costs for denitrification and purification of water contaminated with organohalogen (bioremediation) were provided by the Basin Directorate of Plovdiv. The unit costs associated with emergency interventions are determined on the basis of available data in the region for the past 10 years.

Table 1.

Measures	Pressure	Units	Ave. Value
Non-use value			
Recovery value	Point	PP/household	8.11
Ecological value	Diffusive	PP/household	5.58
Use value			
Savings from drinking water treatment costs	Point	€/mc	0.09
Saving from drinking water treatment costs	Diffusive	€/mc	0.80
Saving from emergency interventions in case of drought costs	Qualitative	€/mc	0.79

Average estimates of the benefits for GES in the East Aegean region.

As mentioned above, the values for the Emilia-Romagna region (Galioto et al., 2013) have being adjusted to assess the non-use benefits of the measures against water pollution in the East Aegean region. Following Navrud and Ready (2007), the propensity to pay (PP) for Bulgaria is calculated after a correction that takes into account the income differences between East Aegean region and Emilia-Romagna region.

$$PP_B = PP_I\,(Y_B / Y_I)^{\beta}, \qquad (1)$$

where PP_B and PP_I represent the propensity to pay in Bulgaria and Italy, respectively; Y_B and Y_I are the income levels in the two countries, and β is the elasticity vs. the income of the demand for environmental goods. For the various eco-friendly goods, the elasticity typically has values less than one. For the new EU member states, the elasticity versus the income is 0.5 and this value is used in the present study. According to the World Bank (2015), in 2015 the GDP per capita based on purchasing power parity (PPP) is €15,731 for Bulgaria and €35,075 for Italy.

From formula (1) it follows that the transformation coefficient of Navrud and Ready $(Y_B / Y_I)^{\beta}$ of the Emilia-Romagna region for the East Aegean region is 0.67. Because of the lower ecological status in Bulgaria, the effectiveness of the measures is increased by 14%, or 0.14 (Mattheiß et al, 2012). Therefore, the coefficient of Navrud and Ready should be adjusted to 0.81.

As the object of this study is the pollution of water bodies from mining and ore processing, it is necessary to define the relative share of the measures taken as part of the total benefits. Following Younger (2001), we will assume that 67% of the deteriorated water quality in Europe's mining regions is due to point sources of pollution and 33% to diffuse pollutants. Therefore, it is reasonable to assume that in the mining regions of Bulgaria, the annual benefits of the measures against the point sources of pollution from mining enterprises are 67% of the total benefits.

The East Aegean region, which has 2,250,000 residents, has 48 underground water bodies with a total annual water extraction of 250 million cubic meters. From this data, a hypothetical average groundwater body can be defined which provides drinking water to 46,875 residents and has a water extraction of 5.2 million cubic meters.

The total annual extraction of drinking water (underground and surface) in the region is 270 million cubic meters. Before being directed to the distribution network, 11.61 million cubic meters of them are purified from nitrogen and phosphorus, 1.35 million cubic meters are purified from organohalogens, and an average of 1.62 million cubic meters of water per year are provided in cases of drought. Considering the use benefits of 1 cubic meter, the total value for the whole area is €10,689,300 per year (11.61 * 0.8 + 1.35 * 0.09 + 1.62 * 0.79 million).

The non-use benefits are calculated as the number of households is multiplied by €13.69. (See Table 1.) Assuming that the average household is composed of 2.3 people, it follows that the number of households in the area is 978,261 (2,250,000/2.3). Thus, the area-wide estimate of the non-use benefits is €13,392,393 (978,261 * 13.69).

The total value of the benefits is €24,081,693 per year (13,392,393 + 10,689,300) and per capita benefits are €10.7 per year (24,081,693/2,250,000).

If the body of water is in a mining region, 67% of these values are due to the measures against point pollution by mining enterprises. In other words, €16,134,734 total and €7.17 per capita are the annual benefits of these measures.

Following the recommendations that the cost-benefit ratio must exceed 1.2, the last calculations also show that the cost of implementing measures against point pollution from mining companies should not exceed €6 per year per resident and €281,250 per year for all residents using drinking water from this body of water.

Assuming that operating costs are 10% of the total costs (Mattheiß et all, 2012), from the last amount it follows that for the implementation of the measures for the conditional water body an investment of €253,125 and an operating cost of €28,125 per year are needed. For the 30-year period at a 5% cost of capital rate NPV of investment is €4,085,709.

Conclusions.

The presented methodology provides an economic assessment of the implementation of Directive 2009/90/EC in the period 2009-2015 and identifies the cases of deterioration (Article 4 (4), (5) & (7)) in the East Aegean Region. Based on the obtained results, additional measures could be taken in the period 2022-2027.

The methodology has been tested for contamination from mining and ore processing, but it is suitable for all water bodies and aggregates as well as for the entire region. The employment of administrative boundaries of the area stimulates the efforts of the local administrations to look for financing of the necessary measures.

The reference point in the methodology is the target water status in the area. Once the sources of pressure have been identified, local stakeholders are consulted about the possible measures for each form of pressure. Then, an analysis of cost minimization is carried out, which makes it possible to choose the most effective set of measures and the levels of activation of individual measures.

The main benefits of the methodology are the simplicity, the logical transition between the different steps, and the ease of practical use.

Limitations.

For benefit assessment, the changes in water status affecting the use costs and the changes that are associated with the non-use cost are analyzed. The available information is employed to estimate the use cost in terms of savings on drinking water treatment and emergency drought interventions. In the analysis of the non-use values, some secondary effects for the economy and society are omitted, which is a deficiency of all methodologies of this type.

The non-use benefits are determined through the benefit transfer (BT) method. We assess the value for recreation and the value of water quality. Due to the limitations of this method, sensitivity analysis is recommended. The analysis would allow determining the limits in which the actual recovery cost varies. Despite this additional information, in general, the way the benefits are determined needs significant improvements and further on-site research.

Another problem for carrying out analysis is the uncertainty. Uncertainty is generated by technical and economic factors. The incorrect estimates of these factors influence both the final outcome of the evaluation and the choice of intervention option. The consequences of uncertainty can be estimated with higher accuracy through more precise models such as stochastic models, Monte Carlo simulation, and Bayes models.

As any other methodology the one presented here depends on the data used and the time constraints. The credibility of the economic analysis could certainly be enhanced by a more detailed study of the local conditions.

From a practical point of view, the methodology can be improved by reproducing a more categorical two-step approach, which involves prior identification of the problem areas and subsequent detailed analysis of some of them (Vecherkov et al., 2017). For researchers and managers it would be useful to develop a model of the reciprocal dependence between pressure and water quality/quantity to help making more accurate assessments of the measures concerning water quality.

REFERENCES

1. Berbel, J. & Expósito, A. (2018). Economic challenges for the EU Water Framework Directive reform and implementation, *European Planning Studies* 26(1), 20–34.
2. Berbel, J., Martin-Ortega, J., & Mesa, P. (2011). A cost-effectiveness analysis of water-saving measures for the water framework directive: The case of the Guadalquivir River Basin in Southern Spain. *Water Resource Management*, 25, 630–640.
3. Borkey, P. (2006). Keeping water safe to drink. *Organisation for Economic Co-operation & Development*. Paris, France, pp. 1–8.
4. Brouwer, R. (2008). The potential role of stated preference methods in the Water Framework Directive to assess disproportionate costs. *Journal of Environmental Planning Management*, 51, 597–614.
5. European Commission. (2020). *The Habitat Directive*. Retrieved on February 4, 2020 from https://ec.europa.eu/environment/nature/legislation/habitatsdirective/index_en.htm
6. European Commission. (2020). *The Nitrates Directive*. Retrieved on February 4, 2020 from https://ec.europa.eu/environment/water/water-nitrates/index_en.html
7. European Commission. (2020). *Rural Development.* Retrieved on February 4, 2020 from https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy/rural-development_en
8. European Commission. (2000). Directive 2000/60/EC (Water Framework Directive). *Official of the European Communities*. Brussels, Belgium.
9. European Commission. (2003). Common implementation strategy for the water framework directive. In *Guidance Document on Exemptions to the Environmental Objectives*. Guidance Document No. 20 (pp. 1–49). Luxembourg, Luxembourg: European Commission.
10. European Commission. (2006). In *Guidance on the Methodology for Carrying out Cost-Benefit Analysis*. The New Programming Period 2007–2013. Methodological Working Document No. 4. DG Regional Policy. (pp. 1–23) EC: Brussels, Belgium.
11. European Commission. (2009). Directive 2009/90/EC (Quality Assurance Quality Control Directive). *Official of the European Communities*. Brussels, Belgium.
12. Galioto, F., Marconi, V., Raggi, M., & Viaggi, D. (2013). An assessment of disproportionate costs in WFD: The experience of Emilia-Romagna. *Water*, 5(4), 1967–1995.
13. Goswami, K. B. & Bisht, P. S. (2017). The Role of Water Resources in Socio-Economic Development. *International Journal for Research in Applied Science & Engineering Technology* 5(XII), 1669–1674.
14. Huijbregts, M., Thissen, U., Guinee, J., vande Meent, D., & Ragas, A. (2000). Priority assessment of toxic substances in life cycle assessment. Part I: Calculation of toxicity potentials for 181 substances with the nested multi-media fate, exposure and effects model USES-LCA. *Chemosphere* 41(4), 541–573.

15. Interwies, E., Gorlach, B., Strosser, P., Ozdemiroglu, E., & Brouwer, R. (2005). *The case for valuation studies in the water framework directive*. (pp. 1–97) Scotland and Northern Ireland Forum for Environmental Research: Edinburgh, UK.
16. Jensen, C., Jacobsen, B., Olsen, S., Dubgaard, A., & Hasler, B. (2013). A practical CBA-based screening procedure for identification of river basins where the costs of fulfilling the WFD requirements may be disproportionate - Applied to the case of Denmark. *Journal of Environmental Economics and Policy*, 2, 164-200.
17. Laurans, Y. (2006). Implementing cost-effectiveness analysis: Perspectives based on recent French pilot studies. In *Proceedings of Vortrag auf der Messe Wasser*, Berlin, Germany, 5 April 2006.
18. Martin-Ortega, J., Skuras, D., Perni, A., Holen, S., & Psaltopoulos, D. (2013). The disproportionality principle in the WFD: How to actually apply it? In Economics of Water Management in Agriculture; Bournaris, T., Berbel, J., Manos, B., Viaggi, D. Eds.; Science Publishers: Enfield, NH, USA.
19. Mattheiß V., De Paoli G., & Strosser P. (2012). Comparative study of pressures and measures in the major river basin management plans in the EU, Task 4 b: Costs & Benefits of WFD implementation. Retrieved on September 30, 2019 from https://ec.europa.eu/environment/archives/water/implrep2007/pdf/EU pressures and measures_Task_4b_Final report.pdf.
20. Macgregor C. & Warren, C. (2016). Evaluating the impacts of Nitrate Vulnerable Zones on the environment and farmers' practices: a Scottish case study. *Scottish Geographical Journal*, 132 (1), 1–20.
21. Navrud, S. & Ready, R. (Eds.) (2007). *Environmental Value Transfer: Issues and Methods.* London, UK: Springer.
22. Postle, M., Fenn, T., Foottit, A., & Salado, R. (2004). CEA and Developing a Methodology for Assessing Disproportionate Costs. In *Final Report for Department for Environment, Food and Rural Affairs (Defra)*. (pp. 1–72) Welsh Assembly Government (WAG), Scottish Executive (SE) and Department of the Environment in Northern Ireland (DOENI); Risk & Policy Analysts Limited (RPA): Norfolk, UK.
23. Stemplewski, J., Krull, D., Wermter, P., Nafo, I.I., Palm, N. and Lange, C. (2008). Integrative socio-economic planning of measures in the context of the water framework directive. *Water and Environment Journal*, 22, 250–257.
24. Van Soesbergen, A. (2008) Assessing the cost-effectiveness of pollution abatement measures in industry. *WEMPA* Working Paper-10.
25. Vecherkov, I., Yanev, N., & Anastasova, Y. (2017). Software tools for business intelligence. *Journal of Mining and Geological Sciences*, 60 (IV) 40–45.
26. Von Schiller D., Acuña, V., Aristi, I., Arroita, M. & et al. (2017). River ecosystem processes: a synthesis of approaches, criteria of use and sensitivity to environmental stressors, *Science of The Total Environment,* 596-597, 465–480.
27. Voulvoulis N., Arpon, K., & Giakoumis, T. (2017). The EU Water Framework Directive: from great expectations to problems with implementation, *Science of The Total Environment*, 575, 358–366.
28. Ward, F. (2009). Economics in integrated water management. *Environmental Modeling Software*, 24, 948–958.
29. World Bank. (2015). *The World Bank Annual Report 2015*. Washington, DC. Retrieved on September 30, 2019 fromhttps://openknowledge.worldbank.org/handle/10986/22550.
30. Younger, P. L. (2001). Mine water pollution in Scotland: Nature, extent and preventative strategies. The Science of the Total Environment, 265, 309–332.