University of Economics, Prague

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# *ASSESSMENT OF DISPROPORTIONATE COSTS IN ENVIRONMENTAL POLICY WITH A SPECIAL FOCUS ON WATER MANAGEMENT*

**Doctoral Dissertation**

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Prague, Juni 2018

## Statement of originality

I declare that I have made my dissertation thesis titled "Assessment of Disproportionate Costs in Environmental Policy with a Special Focus on Water Management" on my own, only based on the bibliography and references quoted.

## Prohlášení

Prohlašuji, že jsem disertační práci s názvem „Assessment of Disproportionate Costs in Environmental Policy with a Special Focus on Water Management" vypracoval samostatně, pouze na základě uvedených pramenů a literatury.

Jan Macháč

Prague, 20 February 2018

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# Abstract

The dissertation deals with application of economic approaches in water management. The thesis focuses on the principle of disproportionate costs and the EU Water Framework Directive (Directive 2000/60 EC). The Directive has a major impact on water management and national economy and contains numerous requirements, including "good status" of all water bodies. However, achieving this environmental target is connected with large investments, often beyond polluters' capabilities. In justified cases, member states may request an exemption. Disproportionate costs of meeting the Water Framework Directive requirements can be used as a reason for extending the deadline for achieving the "good status" or reducing the desired goal. Nevertheless, the Directive grants a relatively high level of discretion relating to the definition of the cost proportionality. The Directive implies the need for application of economic analysis. The objective of the thesis is to discuss the different approaches used abroad, to test the methods in the Czech Republic's context and to provide recommendations for application of cost disproportionality analysis. The thesis puts a special emphasis on complementary methods that can be applied in planning in water management from the economic point of view to achieve the "good status". The dissertation is designed as a cumulative thesis composed of accepted articles/article in the review process in international journals with impact factor/in peer-reviewed international conference proceedings indexed in the Web of Science. The results of the five scholarly articles show that cost-benefit analysis seems to be an appropriate approach to cost disproportionality analysis, which can be combined with other methods such as cost-effectiveness analysis, Bayesian networks or game theory. However, calculation of benefits and costs of measures brings several methodological complications and uncertainties. Application of economic approaches may contribute to efficient meeting of environmental policy goals in water management. The recommendation is therefore to combine the methods to avoid shortcomings of individual methods. A combination of methods leads to a comprehensive approach to water management, which can better protect limited resources.

**Keywords:** Cost proportionality, water management, EU Water Framework Directive, cost-benefit analysis, new Leipzig approach, cumulative dissertation thesis

**JEL Classification**: G180, Q530, Q580, D140

# Abstrakt

Disertační práce se zabývá aplikací ekonomických přístupů v rámci vodního managementu. Práce se zaměřuje na princip přiměřenosti nákladů a Rámcovou směrnici EU pro vodní politiku (směrnice 2000/60 ES). Tato směrnice má významný dopad na vodní management a národní ekonomiky, ustanovuje řadu požadavků včetně dosažení „dobrého stavu“ všech vodních útvarů. Nicméně dosažení tohoto environmentálního cíle je spojeno s velkými investicemi, které často převyšují možnosti znečišťovatelů. V odůvodněných případech mohou členské státy požádat o výjimku. Nepřiměřené náklady na dosažení požadavků směrnice mohou být použity jako důvod pro prodloužení termínu pro dosažení „dobrého stavu“ nebo pro zmírnění požadovaného cíle. Rámcová směrnice poskytuje relativně vysokou míru volnosti, pokud jde o definici nákladové přiměřenosti. Směrnice vyžaduje nutnost aplikace ekonomické analýzy. Cílem práce je diskutovat různé přístupy užívané v zahraničí, otestovat tyto metody v České republice a poskytnout doporučení pro aplikaci výjimky na základě nákladové nepřiměřenosti. Práce klade zvláštní důraz na doplňkové metody, které lze z ekonomického hlediska aplikovat při plánování ve vodním hospodářství pro dosažení „dobrého stavu“. Disertační práce je koncipovaná jako kumulativní práce složená z přijatých článků/článku v recenzním řízení v mezinárodních časopisech s impakt faktorem/v mezinárodních recenzovaných konferenčních sbornících indexovaných na Web of Science. Výsledky pěti vědeckých článků ukázaly, že analýza nákladů a přínosů se zdá být vhodným přístupem pro analýzu nákladové nepřiměřenosti, která může být kombinována s dalšími metodami, jako je analýza efektivity nákladové efektivnosti, Bayesovské sítě nebo teorie her. Výpočet přínosů a nákladů opatření však přináší řadu metodických komplikací a nejistot. Použití ekonomických přístupů může přispět k efektivnímu plnění cílů environmentální politiky ve vodním hospodářství. Doporučení proto spočívá v kombinaci metod, aby se předešlo nedostatkům jednotlivých metod. Kombinace metod také vede ke komplexnímu přístupu ve vodním managementu, který lépe chrání omezené zdroje.

**Klíčová slova**: Přiměřenost nákladů, vodní management, Rámcová směrnice EU pro vodní politiku, analýza nákladů a přínosů, new Leipzig Approach, kumulativní disertační práce

**JEL klasifikace**: G180, Q530, Q580, D140

# Contents

# Introduction

More than 18 years ago, the Water Framework Directive (WFD) was created in response to growing demand for clean water and an integrated approach to water body management across the EU member states. Along with the air regulation, the WFD belongs among the most ambitious legislative projects of the European Union in the field of environmental policy. As Chave (2001) says, for example, its great significance is not only due to the length of preparation of the Directive, which took more than 10 years, but also its comprehensive view of water management, inclusion of all waters (inland surface waters, groundwater, brackish and coastal waters), an integrated attitude to the environment, and support to sustainable water uses. Despite the high expectations, the targets are not yet fulfilled due to problems of implementation. Voulvoulis et al. (2017) mentioned the most important problems. The main one is non-acceptance of the integrated thinking, which is "*a pre-requisite to effective WFD implementation*" (Voulvoulis et al., 2017). There is also a lack of understanding of the basic (core) principles among the various stakeholders, countries, entities, etc.

The primary environmental goals of the WFD include provision of protection, improvement of status and restoration of all water bodies, aiming at achieving their "good status" by the year 2015; furthermore, it may, in justified cases, extend this deadline to 2021 or 2027. In the dissertation, emphasis is placed on surface water bodies. "Good status" of a surface water body refers to such a state where its ecological and chemical conditions are at least "good". The conditions of water bodies, including requirements for good environmental (ecological) and chemical status, are further defined in the WFD in more detail based on qualitative values and partial indicators and in environmental quality standards (e.g., Directive 2008/105/EC). The achieving of the "good status" is connected with the "one out, all out" rule, which makes it difficult to achieve "good status".[1]

The existence of the WFD and its implementation has had a major impact on the water management costs of all the EU member states. The Directive came up with a harmonization of planning cycles and periods among EU Member States. It introduced six-year cycles. The first period was from 2009, with a view to achieving "good status" by 2015. The second period runs from the end of 2015 to 2021, the third between 2021 and

[1] If part of a water body fails on any one of the criteria, it will fail to achieve "good status".

2027. After each period, the actual state (achieving of “good status”) is reviewed. The binding targets of the Directive are very ambitious in relation to a large portion of water bodies and their current condition. In particular, achieving their “good status” thus may significantly increase monetary requirements of member states’ authorities in charge for implementation of required water management measures, including potential social impacts on the populations, e.g., due to increased sewerage or water charges. Under certain conditions, however, the WFD sets exemptions that may be applied to justify non-achievement of good water body status. These exemptions always have to be based on at least one provision of an applicable article of the WFD[2]. Deadlines for improving water body status and achieving “good status” may be extended for purposes of gradual achievement of goals based on technical feasibility, natural conditions or disproportionality of costs.

An exemption based on disproportionality of costs has a substantial use potential for water management authorities, since achievement of good water body status is, in many cases, associated with implementation of numerous measures at high costs, which may outweigh the potential benefits considerably in extreme cases. However, the application of an exemption from achieving “good status” based on disproportionality of costs is limited in member states, chiefly due to the non-existence of European/national methodologies or

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[2] As is apparent from Article 4 of the WFD, there are five possible exemptions to justify the failure to achieve the “good status” of a water body. These exemptions may take both a short and a long-term horizon and always have to be based on at least one provision (exemption) under the relevant WFD article: (i) designation of heavily modified or artificial water bodies to maintain useful functions provided by the water body (Article 4.3); (ii) extension of the deadline to achieve the “good status” of the water body (“good status” must be achieved by 2021, 2027 or as soon as natural conditions permit) (Article 4.4); (iii) achievement of less stringent objectives for meeting environmental and socio-economic needs (Article 4.5); (iv) temporary deterioration in the case of natural causes or force majeure (Article 4.6); (v) a new modification of the physical characteristics of surface waters or underground waters as a result of new sustainable human development activities (Article 4.7).

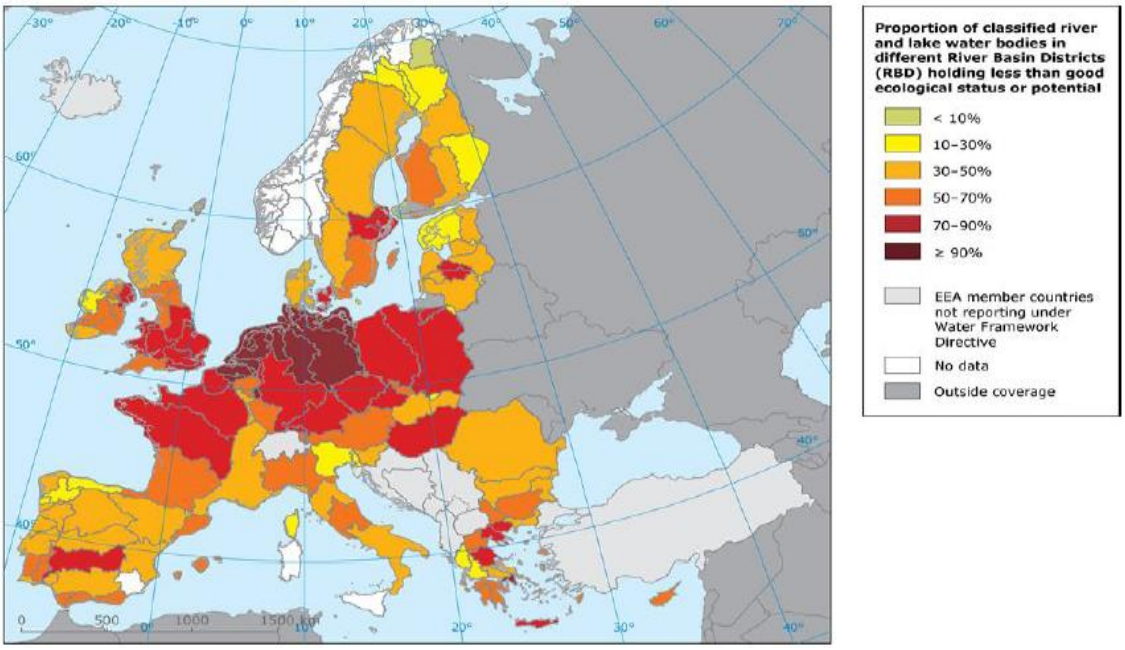
Common to all these exemption are the strict conditions for their use, as well as the inclusion of the justification for an exemption in the River Basin Management Plan. Each exemption has to be reviewed in six-year cycles. It follows from paragraphs 8 and 9 of Article 4 of the WFD that the exemption must not permanently obstruct or exclude the achievement of environmental objectives in other water bodies and that, despite the application of the exemption, at least the level of protection of the water body resulting from other applicable legislation has to be attained.

difficult interpretation of the exemption and also due to methodology complications. The WFD grants a relatively high level of discretion regarding the interpretation of the term "disproportionate costs", and does not specify its application when justifying an exemption from achieving good water body status. There is no detailed description of how the assessment should be done. Drafting suitable methodologies and procedures for assessing cost proportionality has therefore recently become a challenge and subject matter of debate among the professional public across the EU member states.

Until 2015, the importance of exemptions due to disproportionality was only secondary: they enable extension of deadlines for achievement of "good status". In the event of failure to achieve "good status" within the first planning period, another type of exemption was applied. Not all exemptions can be used in all planning periods. The option to extend the period for achieving "good status" is limited to no more than two consecutive updates of catchment area plans, i.e., until 2027, with the exception of cases where objective natural conditions do not permit achievement of the environmental target. After 2027, any non-achievement of "good status" will have to be justified before the European Commission. The justification of non-achievement of good water body status due to disproportionate costs can be expected to gain substantial importance.

Based on the European Commission's report published in 2012 (European Commission, 2012), only 43% of the surface water bodies were in good ecological status in 2009, and the report estimates that only 53% of the water bodies should achieve good ecological status by 2015 (European Commission, 2012, p. 174). Figure 1 graphically represents the relative numbers of surface water bodies (lakes and rivers) that do not achieve good ecological status or potential in EU countries.

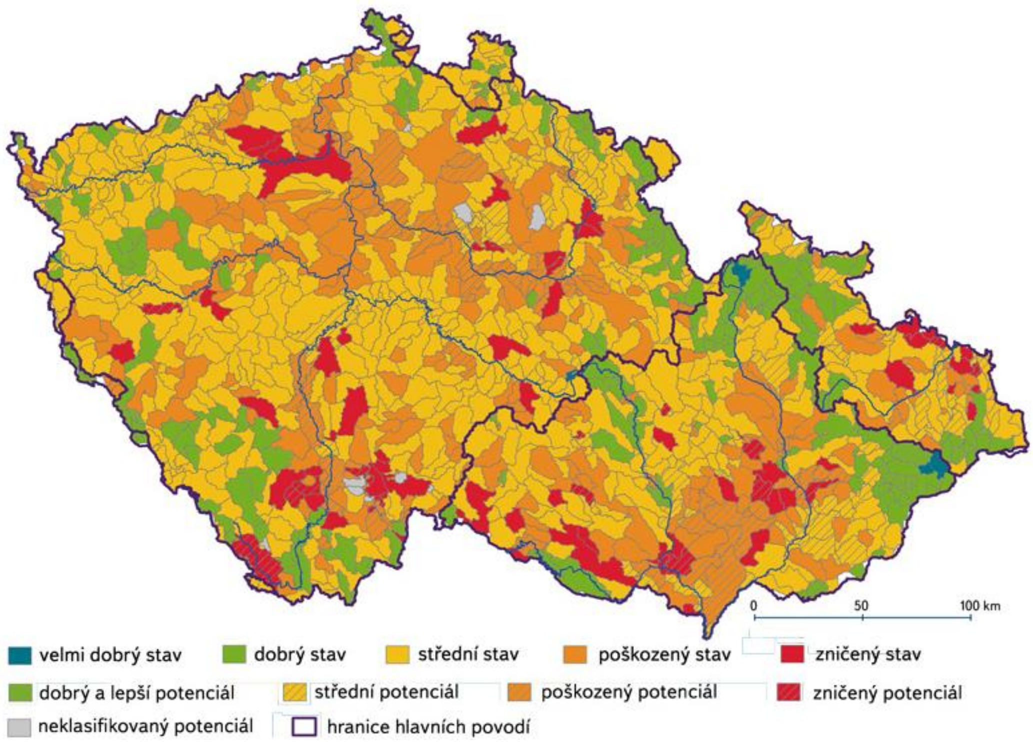
**Figure 1: Water bodies not showing good ecological status or potential**




Source: European Environment Agency (2015)

The current situation in the Czech Republic is very similar to the situation in the whole EU (Fig. 2). Most water bodies are classified with moderate or worse ecological status (poor or bad).

**Figure 2: Current state of achieving the "good status" in the Czech Republic**




Source: Vyskoč et al. (2017)

According to the report (European Commission, 2012), member states preferred extension of the deadline for achievement of "good status" as a justification of exemption in the first planning cycle. In the case of surface water bodies, exemptions for extension were used in more than 70% by good ecological status and almost 90% by achieving good chemical status. Only 8% of the exemptions will be justified by disproportionate costs. In another 30% of the cases, disproportionality was used in combination with other reasons such as technical feasibility or unfavourable natural conditions. In the Czech Republic, only exemptions due to technical feasibility and natural conditions were applied. Görlach et Pielen (2007) give a number of reasons why the principle of proportionality was not applied. In addition to lack of data, they mention non-existence of national methodologies and ambiguity of the exemption interpretation. The difficulty of practical application and interpretation of proportionality under the WFD are confirmed by consultations with selected river basin managers in the Czech Republic and Germany. The non-existence of suitable national methodological support as a tool for assessing proportionality of costs presents the concerned entities with prohibitively high time and monetary costs of performing a separate proportionality analysis.

So far, the only Czech analysis of cost disproportionality in relation to the WFD has been carried out for the catchment of the Orlík reservoir (Vojáček et al., 2013). The study was performed as part of the Refresh international project, the aim of which was "*to develop a framework that would enable water managers to design cost-effective restoration programmes for freshwater ecosystems*" (Refresh, 2010). The catchment of the Orlík reservoir was used as one of the pilot areas in partner countries. The catchment faces strong eutrophication caused by phosphorus inflow. The objectives of the case study included: (i) a hydrological study; (ii) selection of most cost-effective measures; (iii) a disproportionality analysis. The second and third objectives are relevant in the context of this thesis.

The results of the disproportionality analysis have shown that implementation of the most cost-effective measures to reduce the eutrophication in the catchment would bring a net social loss. It means that the costs (CZK 15 billion) exceed the benefits (CZK 2 billion) over the period under review. From an economic point of view, this means that improving water quality would lead to a decline in society's well-being, because the costs of this improvement are higher than the benefits. Given the differences between the costs and benefits, it would probably be possible to apply for an exemption, and to justify the exemption.

The analysis also showed the extreme necessity of carrying out similar types of analyses in river catchments especially before implementing measures to reduce water pollution (e.g., by phosphorus). According to Vojáček et Macháč (2015), performance of economic analyses such as cost-effectiveness analysis can lead to achievement of savings in implementation of measures in the Czech Republic. To prove this, it is possible to compare the effects and costs of measures already implemented in the catchment (by 2015) and measures from the cost-effective scenario from the analysis. Based on available data, the phosphorus inflow was reduced by about approximately 22 tonnes / year with the costs of CZK 465 million per year. Based on the results of the cost-effectiveness analysis, it is possible to achieve the "good status" due to implementation of a set of measures by a reduction of 114 tonnes / year at a cost of CZK 602 million per year. That means the annual cost effectiveness of the implemented measures was CZK 21 million per reduced tonne of phosphorus versus CZK 5.3 million in the case of the cost-effective scenario based on the analysis of Vojáček et al. (2013).

## Proportionality of costs from the economic perspective

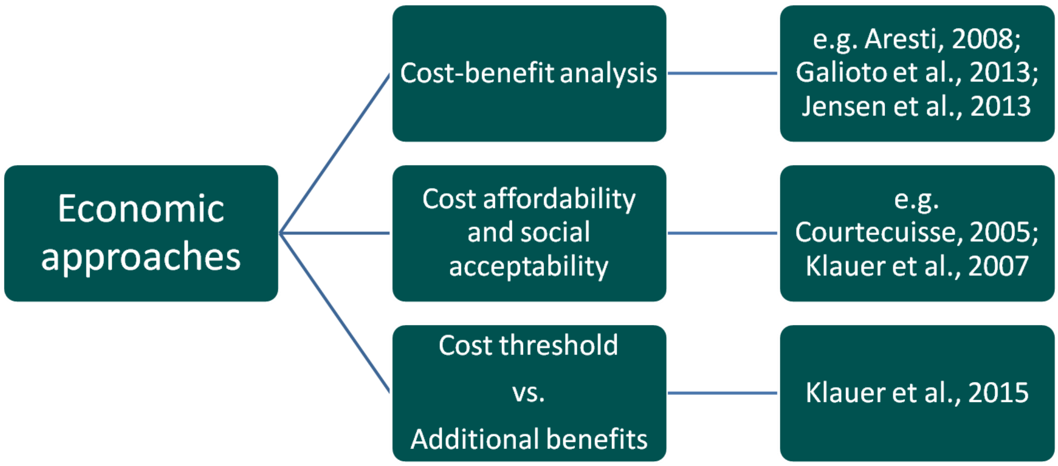
The proportionality principle has been highlighted in various acts of law, regulations and methodologies world-wide (in the European Union, the term has been used not only in the WFD, but also in relation to requirements on quality of regulation as such, e.g., the European Commission's "Smart Regulation" draft; European Commission, 2006). The principle is fundamental in the regulatory impact assessment (RIA) process as well.

In spite of the high frequency of occurrence of this principle, the economics literature does not pay so much attention to it; it may use the term but does not offer a precise definition. In connection with European regulation, the term is defined by, e.g., Veinla (2004), who emphasises the necessity of assessing regulation based on appropriateness and adequacy to the problem, necessity of intervention and its effectiveness. The requirements stated by Veinla (2004) largely match the interpretation of WFD application according to WATECO (2003). Proportionate measures are such that bring an improvement of status the implementation costs of which do not significantly exceed the benefits, and the most cost-effective combination of measures is required for implementation and evaluation. As stated, for example, by De Nocker et al. (2007), WATECO (2003) and the European

Commission (2009), the threshold of “disproportionality” is set by a public authority in charge of the issue in question (based on an administrative decision).

The fact that the WFD itself does not define the concept of disproportionality has led to different interpretations and approaches used to assess the proportionality. In the past 17 years since the Framework Directive entered into force, a number of national and international projects and pilot studies have been carried out that included proportionality assessment. Brouwer (2004) and Klauer et al. (2007) distinguish between two different fundamental attitudes to evaluating cost proportionality: (i) proportionality of society-wide costs and (ii) proportionality of costs of the individual entities arising from implementation of measures. Due to the response of the European Commission, a new (third) approach was developed. Nowadays we can divide the approaches into three groups (Figure 3) according to the type of method used for the assessment. These approaches are described below.

**Figure 3: Different approaches to disproportionality analysis**




Source: Own analysis

### Assessment using cost-benefit analysis

This approach is based on neoclassical and environmental economics. In this case, proportionality is assessed according to society well-being, or by comparison of all costs and benefits. If the total benefits exceed the total costs, the achievement of “good status” is considered to be proportional. Cost-benefit analysis or its modifications are the most commonly used technique (e.g., Hanley et Black, 2006; Aresti, 2008; Brouwer, 2009; Jensen et al., 2013; Galioto et al., 2013; Vojáček et al., 2013).

The process is usually divided into several steps. For example, Jensen et al. (2013) use seven steps:

1. definition of geographical scope of the analysis;
2. identification of status quo of water bodies;
3. estimate of benefits from achieving “good status”;
4. estimate of costs of achieving “good status”;
5. calculation of social profit (net social benefit);
6. sensitivity analysis;
7. final results and recommendations.

The first two steps are used to identify the local conditions and the gap between the current state and the “good status”. Steps 3 and 4 are crucial for proportionality assessment. First of all, costs and benefits are identified and then quantified. Monetisation of the costs and benefits is necessary for the comparison. However, not all of them can be monetised. In this case, such costs and benefits are considered in the last step.

When estimating the costs and benefits, the monetisation should be based on primary pricing studies for the area to avoid any distortion and to achieve the highest possible accuracy. Another possibility is to use the benefit transfer method. This method is recommended in the case of lack of data/primary studies on benefit/cost assessment (e.g., Klauer et al., 2007; Galioto et al., 2013). Benefit transfer is also less time and money-intensive than the application of primary valuation methods. However, the application of this method is conditioned by the existence of at least one suitable primary pricing study for the geographic scale chosen.

If no cost-effectiveness analysis of suitable measures has been made previously as a part of developing river basin plans, it is advisable to proceed to its elaboration at this point within step 4; the measures considered should be cost-effective to prevent ineffective use of public funds (e.g., De Nocker et al., 2007; Aresti, 2008; European Commission, 2009; Galioto, et al., 2013).

The comparison of costs and benefits can be made using several methods. Mostly the net social benefits are set in the form of net present value or in the form of net annualised value (Jacobsen, 2005). In this case, we obtain the net benefits from the total benefits minus total costs. Alternatively, we can use the benefit-cost ratio.

To avoid the uncertainty connected with monetisation of costs and benefits, sensitivity analysis is usually applied before the final results are determined.

Although this method is very common in environmental economics, it has a number of opponents who exclude the possibility of expressing the benefits in monetary terms. Klauer et al. (2007), Ammermüller et al. (2008) and Klauer et al. (2015) criticise this method as very inefficient due to the lack of primary data and high time and money demands of processing primary studies.

**Assessment according to cost affordability and social acceptability**

The second group covers all the methods assessing the proportionality of individual entities' costs. Both the financial availability and the financial burden to these entities due to implementation of measures are evaluated. This approach is used very often for assessment of cost affordability of drinking water, e.g., Courtecuisse (2005). According to this method, household budgets are compared with the price of water as part of the water and sewerage charges, in which costs of measure implementation are reflected. If the identified costs exceed the socially acceptable threshold of 2% of the annual household expenditures, then the costs of achieving "good status" transferred to the water price can be labelled as disproportionate. This approach turns out to be impractical for poorer areas (areas with low wages), where the costs would tend to be disproportionate, leading to non-implementation of any measures.

A number of other criteria that fall into this group have been gradually defined (e.g., Brouwer, 2004; Laurans, 2006; Klauer et al., 2007). A slightly different approach is chosen by Laurans (2006) and Klauer et al. (2007), who make a comparison of the costs of measure implementation with the expenditures on water management made in the area so far. If the costs do not exceed the expenditures so far augmented by 20% (Klauer et al., 2007), they are proportionate costs and an application for an exemption or further analyses are out of the question. Klauer et al. (2007) proposed a system of rules/criteria, which are used to assess the proportionality. Klauer et al. (2007) created a long list of possible criteria, which have been analysed with respect to requirements of the WFD and to practical use. However, at this stage of the study, a number of criteria have been found to be completely inappropriate and contravening the Framework Directive. The criteria have been further defined and applied in practical cases in the area of excess phosphorus in water and the throughput of aquatic bodies for aquatic organisms, where their suitability and functionality have been verified. In testing, it has been revealed that some of the criteria could be difficult to apply because of the date unavailability.

All measures that have passed the testing criteria have been decomposed into three stages of the assessment. A comparison of previous expenditures was one of the rules. Within each stage, measures which are not eligible for an exemption due to unreasonable costs are excluded. The rest moves on to the next stage. If the measure has passed all the criteria, cost-benefit analysis will be performed in the last step. The outcome of the CBA serves as the final economic argument for cost disproportionality analysis. After the justification of cost disproportionality, the process continues with applying an exemption.

Later, the European Commission rejected all the different methods based on cost affordability. Water quality improvement has an impact in the form of positive externalities. In this case, the investor is mostly the only net payer of the measures. However, the whole society is the recipient of the benefits. From the European Commission's point of view, the transfer of money between the investor and the final user of the benefits generated from the implementation of the measures has to be considered.

**Assessment based on cost threshold and additional benefits**

Due to rejection of the previous approach by the European Commission, a German researcher has developed a new method known as the "new Leipzig approach", which combined both approaches mentioned above. The approach is based on multi-criteria analysis (MCA) (Klauer et al., 2017). There are four inputs to the MCA: (i) current costs associated with achieving the "good status"; (ii) public expenditures made in the past; (iii) additional benefits generated by the improvement of water quality; and (iv) distance between the current state and "good status".

The cost disproportionality is set based on a comparison of a cost threshold and current costs associated with measures. The cost threshold is derived from past expenditures, the distance to the target and the additional benefits. This method is described in more detail in Chapter 1.

## Dissertation structure and goals

As the title of the dissertation indicates, the thesis deals with assessment of disproportionate costs in water management. A special emphasis is placed on approaches applied abroad. Although the WFD has been in force for 18 years and there are already a number of different methodologies abroad, there has still been no consensus on the approach. While in some countries such as Germany, exemptions for cost disproportionality have already been applied, in the Czech Republic, exemptions for other reasons are currently preferred. Non-use of the exemption due to disproportionality of costs follows from Czech water management experts' low awareness of appropriate methods.

The primary objective of the dissertation is to analyse how to assess disproportionality in the conditions of the Czech Republic and how to combine the methods to justify the exemptions. To meet the primary objective of the thesis, the goals listed below have been set, which are reflected in each chapter. These goals are as follows:

i) Compare both main approaches (based on cost-benefit analysis and on multi-criteria analysis) to provide economic and policy recommendations for water management. See Chapter 1.
ii) Discuss application of the cost affordability and social acceptability approach in water management outside the exemption application. See Chapters 2 and 3.
iii) Discuss possible limitations and challenges of different approaches. See Chapters 1, 2, 4 and 5.
iv) Demonstrate the significance of practical application of economic analysis in water management. See Chapters 1, 2, 4, 5 and partly 3.

Each chapter contains a methodological part, which presents the methods for achieving the primary objective and the goals. A description of the Czech methodology and challenges connected with the disproportionality analysis will be presented below in this chapter.

The present thesis is composed as a so-called cumulative dissertation thesis. Two of the five standalone articles were accepted in journals with an impact factor in 2017 and published in 2018. Two other papers were published in peer-reviewed international conference proceedings indexed in the Web of Science database in 2016-2017. The last one is in a review process for an impact factor journal at the time of writing this thesis. The articles were written on the basis of scientific projects and in collaboration with scientists from other countries.

The topic of the thesis is closely related to national project no. TD020352 "Cost-appropriateness evaluation of ensuring a good status of water bodies" supported by Technology Agency of the Czech Republic, in which the author of this thesis participated as a researcher. The results of the project were further developed by the author in cooperation with Dr. Katja Sigel (Department of Economics, Helmholtz Centre for Environmental Research – UFZ, Germany) in the form of testing the "new Leipzig approach" in the Czech Republic. The paper from Chapter 3 was written as part of the project CROSSFLOODS: Cross Border Flood Risk Management in cooperation with Dr. Thomas Hartmann from Utrecht University (Faculty of Geosciences).

The articles comprising this thesis are the work of a team of authors, with the dissertation author's average contribution to the articles being 60% (the authorial contribution to each of the articles is shown in % in each chapter). With the exception of one article, the author of this thesis is the first author of the paper.

The first article "*Assessment of disproportionate costs according to the WFD: Comparison of applications of two approaches in the catchment of the Stanovice reservoir (Czech Republic)"* is focused on comparison of two different approaches based on monetary cost-benefit analysis (a common approach in the EU) and on the cost threshold (the German approach). In addition to a comparison of the theoretical concepts and applications of approaches in practice, a case study of common applications of both approaches was prepared. The small catchment of the Stanovice reservoir was selected as an appropriate pilot area, where it is possible to achieve a good state. Strengths and weaknesses of both methods were identified based on the application. The main problem with the application of the German approach was the availability of the necessary data for evaluation. The article contains clear recommendations for water management, how to assess the disproportionality and how to combine both approaches. In the case of the German approach, it was the first application outside Germany and one of the first overall. The unique aspect is the confrontation of both approaches for the same water body. However, the results can be generalized only to a limited extent because both approaches were applied together in a single catchment.

While the above article concentrates on the application of cost proportionality at the water body level to justify a possible exemption (micro level), the second article "*How much extra will households pay for environmental improvement? Impacts of water and sewerage legislation in preparation on incomes of the poorest households in the South Bohemian*

*Region"* is based on the application of the social and price affordability approach in a regulatory impact assessment (macro level). First of all, costs of meeting the new proposed legislative requirements connected with the WFD requirements are valuated with respect to local aspects. The original analysis was carried out for the whole Czech Republic, but the article focuses on the South Bohemian Region only. Based on a literature review, criteria for social affordability were set as the burden on the lowest-income households. The quantified costs were included in the water price in each city taking into account the local conditions. The key indicator was the share of water expenditures of the poorest households in their entire expenditures. The article shows the local disparities resulting from the different conditions of the water supply networks and the existing wastewater treatment facilities in the cities. Currently the OECD criterion (4% share of water expenditures in the total household expenditures) is met by the lowest-income households. The criterion will be exceeded by 0.5-1% in the case of adoption of the proposed legislative changes. These results are in contrast to the study of the Ministry of Labour and Social Affairs, which comes with a significantly lower impact. The study is based only on average values for the whole country. As a result, there is a strong recommendation to carry out impact studies taking into account local specificities. Although the criterion is exceeded, it is not easy to reject the proposed changes from my point of view, because "*the price should always reflect the rarity of the good; only thus can responsible behaviour of consumers be achieved*" (Macháč et Zemková, 2017).

The reason for refusing approaches based on criteria was the inappropriate distribution of costs and benefits in society. The third article "*Negotiating land for flood risk management: upstream-downstream in the light of economic game theory*" deals with a possible change in redistribution of costs and benefits. The article differs methodologically from the two previous articles. It is based on game theory, which was applied to flood protection issues. The article is based on the fact that it is not possible to distribute costs and benefits appropriately using the CBA. CBA can be used only to assess measures/projects. On the issue of floods, the Coase theorem can be applied. There is a non-reciprocal relationship (Coase, 1960) between upstream and downstream. Game theory allows inclusion of stakeholder negotiation and modelling of stakeholders' behaviour in different scenarios. The article deals with two players, who represent upstream and downstream. Each player has two options for behaviour, thus the game is 2x2. The probability that a flood occurs plays an important role. Depending on changing scenarios (affecting the payoff), the nature of the game also changes (Prisoner's Dilemma, Deadlock, Stag-Hunt, and Chicken). In each of the games (scenarios), the negotiation leads to achievement of a better situation which is connected with a change of cost and benefit distribution.

Economic analyses are associated with a number of uncertainties that are addressed by more or less robust tools. In cost-benefit analysis, the problem is solved using sensitivity analyses, which test the effect of changing the individual input variables on the result. Alternatively, the sensitivity is tested in the form of scenarios (optimistic, pessimistic, etc.); this approach was used in the first article. The fourth article "*Using Bayesian Networks to Assess Effectiveness of Phosphorus Abatement Measures Under the WFD*" is focused on another way of dealing with uncertainties using Bayesian networks. Unlike a standard CBA, Bayesian analysis does not solve the uncertainties associated with valuing costs and benefits. The key question here is whether the proposed measures will be sufficient to achieve the effect (e.g., to achieve the "good status" through phosphorus reduction). The Bayesian networks work with the probability of achieving the target based on measures already implemented. The concept of Bayesian networks is introduced briefly and further implemented on a case study of the Stanovice water reservoir (the same catchment as in the first paper). To implement this approach, first it was necessary to collect sufficient data about the effectiveness of individual measures. For example, more than 100 operators of sewage treatment plants were contacted. In a number of cases (e.g., Barton et al., 2008; Moe et al., 2016), it was shown that the probability of achieving the "good status" is very low. Due to the frequency of measures and availability of data, 5 of the 8 types of measures were analysed. The results can be used to prioritise the measure implementation or to change the set of measures to achieve a higher probability.

The last article "*Appropriateness of cost-effectiveness analysis in water management: A comparison of cost evaluations in small and large catchment areas*" is focused on application of cost-effectiveness analysis in catchments as a tool for cost reduction. This type of analysis is used as an important step in cost proportionality assessment. According to the requirements of WFD, only the most cost-effective measures can be taken into account in the process of disproportionality assessment. A cost-effectiveness analysis should be carried out. In the practice, the proposed measures only cover the target, but they do not exceed it. Cost optimisation in these cases makes no sense, as all measures need to be implemented. However, it would usually be possible to propose more measures. In that case, a CEA should be carried out. The article compares implementation of CEA in large and small catchments. The catchment of the Orlík reservoir was chosen as a large catchment (it covers 1/7 of the area of the Czech Republic). It faces strong eutrophication caused by phosphorus inflow like the catchment of the Stanovice reservoir (a small catchment). In both cases, CEA was an appropriate tool for selection of cost-effective combinations of measures for reservoirs or other water bodies.

## Requirements arising from the WFD and other EU recommendations

The issue of designing a suitable, sufficiently transparent and practically utilisable methodology for assessing cost proportionality is very complex and difficult. Although the Water Framework Directive does not set out clearly defined rules for assessing proportionality and thus leaves a lot of room for member states to design their own assessment processes, there are numerous requirements for proportionality assessment in the Directive itself and its accompanying documents. According to the recommendations of the European Commission and the methodologies accompanying the Directive (European Commission, 2009; De Nocker et al., 2007; WATECO, 2003) the "disproportionality" threshold is set by a public authority competent for the matter. Nevertheless, the proportionality threshold has to be supported with an economic analysis of costs and benefits. It is clear from the logic of the Framework Directive that proportionality assessment only makes sense where a cost-effective combination measures for achieving the target is identified. Therefore, the application of an exemption requires the performance of a cost-effectiveness analysis (CEA; Articles 4, Para. 4, 5 and 7 of the WFD). It is a question whether a CEA shall be carried out separately for each pollutant/group of pollutants that can be reduced using a certain type of measure, or for the water body as such – i.e., all the pollutants and benefits associated with them in the next step.

It also follows from Articles 4, Para. 4, 5 and 7 of the WFD that wherever an exemption is applied due to disproportionate costs, all the measures that are not disproportionate should be implemented so that the best possible water quality status can be achieved. Therefore, the exemption does not mean a priori that no measures are implemented: only that part of them that is disproportionate to the costs is not implemented. This leads to the setting of a less strict goal, which corresponds to such conditions of the water body where all the measures that are feasible and not disproportionately costly have been implemented. When setting less strict goals, this must not lead to a worsening in the other qualitative elements to a status defined by the worse affected quality element, and the present potential for water quality improvement must not be disregarded either.

It thus follows indirectly from the essence of the matter that the analysis for a water body cannot include all the pollutants together. It therefore has to proceed by groups of pollutants that can be solved by means of "certain measures" (presumably together). Thus, for instance, elimination of phosphorus in the water reservoir A is disproportionately costly, but it may be proportionately costly to reduce emissions of undissolved organic solids in the same reservoir. It appears to be necessary for an easier proportionality assessment to carry out cost-benefit analysis and comparison in limit/incremental values. Thus, if the analysis shows that it is disproportionate to reduce the concentration of a certain pollutant to 5 mg/l, it has to be assessed whether a different concentration (e.g., 6 mg/l) is proportionate.

Another requirement of the Directive restricts application of exemptions to measures that have to be implemented in order to achieve good water quality and that have been defined before the adoption of the Framework Directive. Exemptions cannot be applied to these measures; an exemption is only possible in case the country has agreed one in its accession treaty. This point applies to both old EU countries and those that acceded between 2004 and 2007. Such outstanding commitments can thus not be included among the costs assessed and have to be omitted from the analysis.

In addition, the European Commission (2009) defines general principles for application of the term "disproportionate costs". With respect to the uncertainties concerning assessment of benefits and costs, the disproportionality threshold should not simply start at the point where the costs exceed the benefits. This point where the costs exceed the benefits should be sufficiently obvious and should show a high degree of reliability. The costs and the benefits should be assessed not only qualitatively but also quantitatively. Moreover, the Directive permits inclusion in the disproportionality assessment of evaluation of the ability to pay by those who should bear the costs. However, if the affordability principle is applied, it has to be proven that no other relevant alternative financing mechanisms are available. The economic analysis has to include sufficiently detailed information in order to assess the most cost-effective combinations of measures for achieving "good status" that are to be integrated in the schemes of measures for the catchment area.

Furthermore, the European Commission (2009) makes numerous requirements on watercourse managers. Above all, it requires public involvement and transparency of the whole process by means of informing the public about the reasons for applying the exemption. Catchment area management plans should state the reasons for extending the deadlines and setting less strict environmental criteria, a summary of measures that should gradually bring the water bodies into the required environmental status, and the expected timetable of implementation of measures for the event of delays or postponement of some measures. Moreover, the plans should contain criteria for applying exemptions and a summary of the assessment and calculation processes and methods.

Moreover, the European Commission (2009) is aware of the complexities on the benefit side, and therefore recommends a pragmatic approach. The assessment should lead to completeness and comprehensiveness. The benefits have to be at least estimated, and a qualitative assessment will suffice for the less important ones. In this context, benefit transfer is mentioned as a tool applicable to transfer of values.

Benefits from achieving “good status” can be of both a market and non-market nature. The European Commission (2009) methodology lists examples of benefits that can be taken into account when assessing proportionality of costs required to achieve targets:

a) benefits due to better protection and increasing quality of aquatic ecosystems and biodiversity;

b) benefits for human health (associated, e.g., with drinking water quality, aquatic ecosystems as sources of food, swimming);

c) benefits for water users due to reduced costs (associated, e.g., with reduced costs of water pre-treatment);

d) benefits from more efficient water body management based on the polluter pays principle (associated, e.g., with performance of cost-effective measures, appropriately set pricing policy);

e) benefits from promotion of cost-effective water body management, e.g., when adopting other European legislation (such as IPPC);

f) benefits from implementing an integrated approach to catchment area management;

g) benefits from increased attractiveness of water bodies (e.g., for visitors, tourists, water sports operators) and increased non-utility values and associated non-market benefits;

h) benefits from mitigated impacts of climate change and safe water supplies;

i) benefits due to implementation of a conflict resolution mechanism and balancing of different interests of different water users;

j) benefits from sustainable water use and generation of new jobs (e.g., in ecotourism, fishing, environmental technologies and nature protection).

De Nocker et al. (2007) specify four basic categories of benefits that should be assessed in connection with the WFD; they are avoided costs for treatment of drinking water, reduction of disposal costs for contaminated dredging material, more and better opportunities for informal recreation (walking, cycling) and water sports, and improved health and living environments. Surprisingly, the basic categories exclude ecosystem services, which are currently a highly promoted environmental protection concept in the EU. They are mentioned as secondary benefits along with biodiversity. De Nocker et al. (2007) point out that although the Directive as such requires application of the cost-effectiveness method and describes the method in the methodologies quite in detail, it does not require comprehensive assessment and monetary expression of all the benefits and does not deal with them in any detail.

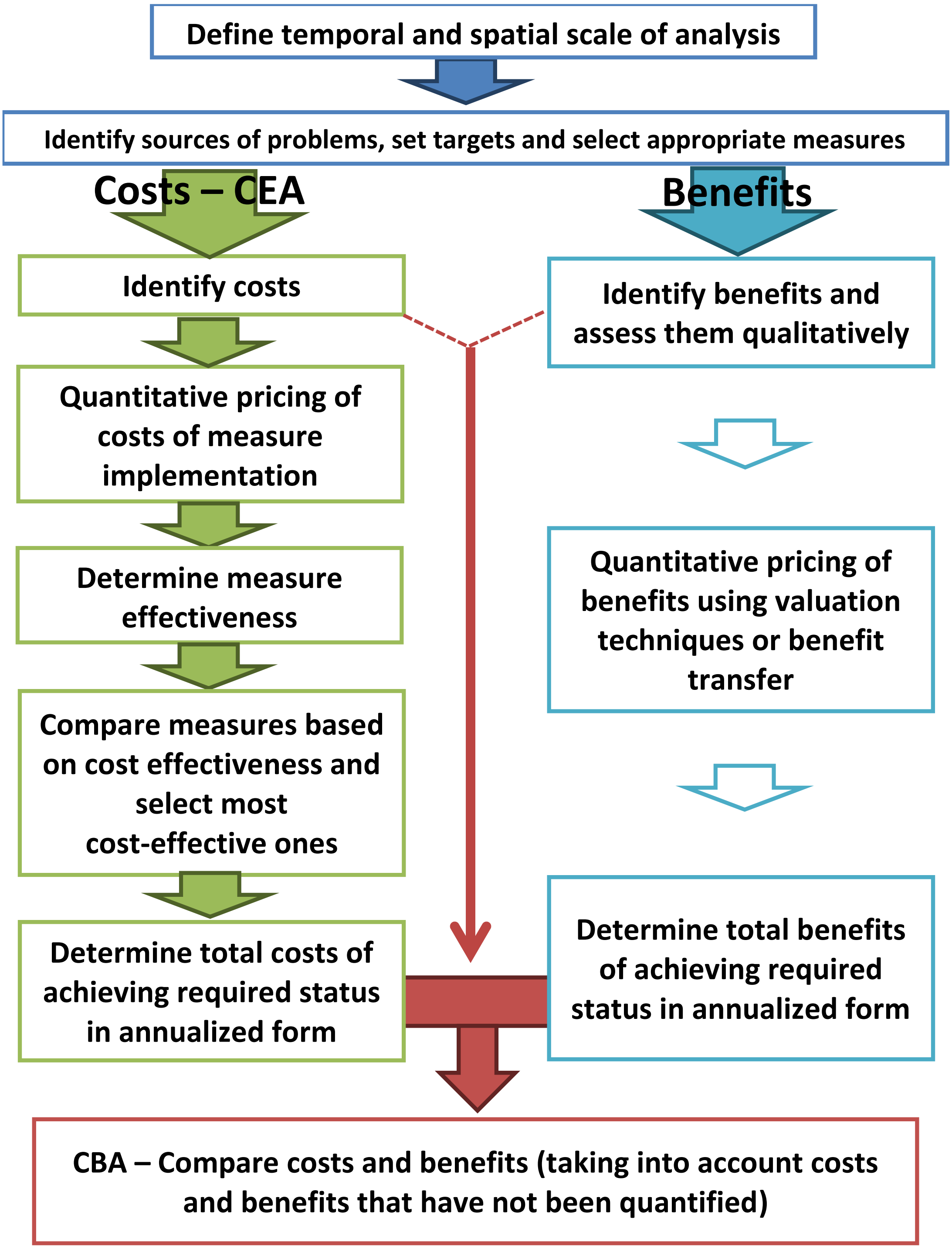
## Procedure for assessing cost proportionality

A procedure for assessing proportionality was designed based on an extensive literature review. The approach was proposed with reference to experience abroad and requirements of the Water Framework Directive and other EU documents. The assessment procedure was further expanded and certified by the Ministry of Agriculture (Slavíková et al., 2015). The author of this thesis was a co-author of the certified Czech methodology.

The procedure is divided into several steps that duplicate the division made by, e.g., De Nocker et al. (2007), Jensen et al. (2013), Vojáček et al. (2013) and Whitehead et al. (2013). The primary prerequisite for assessing proportionality of costs is the existence and technical feasibility of measures to achieve “good status”. If it is not possible to achieve the “good status”, another exemption due to technical feasibility should be applied. At the same time, it has to be determined in which parameters the water body does not achieve “good status”. If the indicator (parameter) should be achieved under any legislation other than the Water Framework Directive (for example before the adoption of the WFD), an exemption application is not possible or only very limited (an exemption may only be applied for the difference between the current status and what should have been met under pre-existent legislation compared to the “good status”).

As Fig. 4 shows, the assessment starts with a description of the problem (the cause of not achieving “good status”), an analysis of the distance to the target, identification of possible measures and their effects on the achievement of the “good status”.

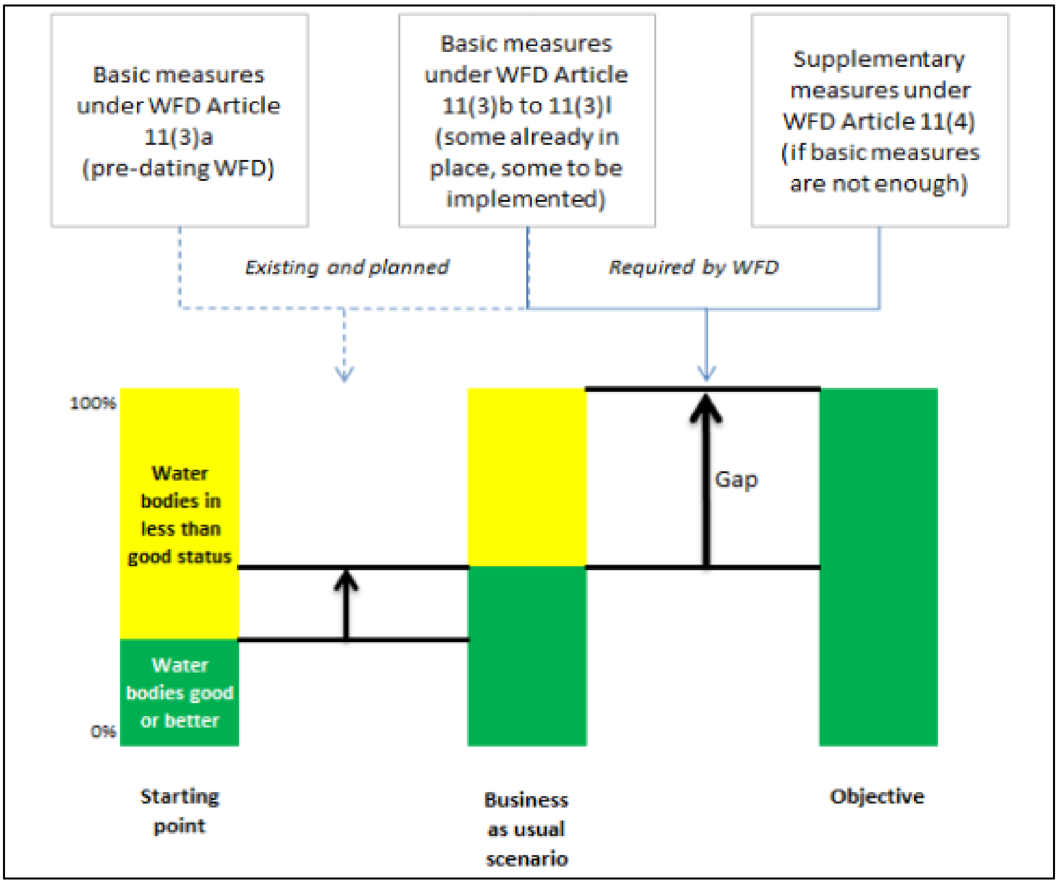
**Figure 4: Different approaches to disproportionality analysis**




Source: Own analysis

Analysis of the distance to the target is a very important step for the application for an exemption due to disproportionate costs. According to the European Commission (2015), the gap between "good status" and current state is defined as a part of the distance to the target. All the requirements (existing and planned measures) that entered into force before the Water Framework Directive may not have been included in the cost disproportionality analysis. In the gap analysis, one important question is answered: Has the water body not achieved these parameters even under other legislation outside the Framework Directive (national legislation)? As you can see in Fig. 5, all the measures that are based on requirements outside the WFD (pre-dating WFD) are excluded from further analysis together with associated costs and benefits.

**Figure 5: Analysis of the distance to the target**




Source: European Commission (2015)

The identification of the current state and parameters of water quality ("good status") for a given water body is based on the identification and assessment of the status of surface and ground water pursuant to Section 21 of Act no. 254/2001 Coll. and relevant monitoring programmes.

The second phase of the cost disproportionality assessment includes an economic analysis of costs and benefits of the defined measures. The costs and benefits are assessed separately. The analyses are very similar. First the costs and benefits are identified. Based on the results of the qualitative analysis, a quantitative analysis is performed.

To meet the EU requirements, the cost-effectiveness analysis is applied to find the most cost-effective combination of measures (to achieve "good status" with minimum possible costs). The analysis has to take into account the risks and uncertainties and specify other costs that have not been quantified. It is appropriate to divide the costs into several categories such as investment, operating and administrative costs, lost income and any other indirect costs. Monetisation is most often made according to the market prices, measures already implemented and various catalogues.

For the valuation and comparability of the costs and benefits, it is also advisable to apply the annualised cost method, as used, e.g., by Galioto et al. (2013), Georgopoulou et al. (2017). This method takes into account the real value of money and the opportunity to investment funds elsewhere (Jacobsen, 2005). "*Unlike the better-known net present value calculation method, which tries to express future costs and benefits using net present value, the annualised cost method attempts to transform the known present costs and benefits to a future flow of the same values based on annual costs, which when cumulated match the known present value.*" (Macháč et al., 2016, p. 2) The lifetime of measures is reflected using the annualisation method. Annualised costs are calculated for each of the measure components with different lifetime.

The most important categories of benefits were identified based on a literature analysis (e.g., Jensen et al., 2013; Vojáček et al., 2013; Galioto et al., 2013; De Nocker, 2007). The three following groups of benefits should be included in the assessment as a basis:

i) cultural ecosystem services (recreational and aesthetic benefits);
ii) water purification (savings of costs of water treatment – benefits for water and sewage utility companies);
iii) benefits from other ecosystem services (soil erosion, flood control, water retention, etc.)

The benefits are also monetised. From the economic point of view, improvement of water body status brings a situation where consumers realise higher utility, which is the benefit from improved water body status. The benefit assessment can apply one of a number of qualitative/quantitative valuation methods for environmental goods. Due to the great monetary demand of primary studies, the benefit transfer method can be applied for the purpose of the methodology (it is used to transfer values from existing studies to another study with similar features and context).

According to the literature review and existing studies, the first two categories of benefits appear to be the most important. These benefits should also be quantified by the analyser with priority. The other benefits, especially some ecosystem services, for the quantification of which there are no suitable data or the quantification/valuation of which would be charged with a high degree of uncertainty or disproportionately costly (such as requiring a socio-economic survey), have to be described at least qualitatively.

A sensitivity analysis should be an integral part of a disproportionality analysis; it tests the effect of individual variables on the result. Vojáček et al. (2013) used sensitivity analysis in the form of scenario analysis. In addition to the baseline scenario, an optimistic and pessimistic scenario is developed. The methodology of Slavíková et al. (2015) also included this procedure.

Finally, the costs and the benefits are compared and the proportionality of achieving “good status” is assessed. In case the benefits exceed the costs considerably, a refusal of the exemption is more likely. In case the assessment finds costs disproportionate to the benefits, the exemption cannot be applied for directly: the highest possible less strict target that is cost-proportionate has to be found. The exemption is applied only to the difference between that status and “good status”. When comparing the costs and the benefits, non-monetised costs and benefits also have to be taken into account (e.g., benefits arising from improvement of some ecosystem services).

The Czech approach by Slavíková et al. (2015) was tested before the certification. Due to the fact that the methodology was completed and certified at the turn of the first and second cycles (December 2015), it was not possible to apply the method to the planning of exemptions for the second cycle. The methodology is particularly important for the third cycle and after the year 2027, when application of some other exemptions (reasons) is very limited. The Czech methodology was used, e.g., by Sweco Hydroprojekt (2017) to assess the measures for sub-catchment plans in the Vltava river catchment. The methodology has also been applied outside water management in the economic assessment of adaptation measures in cities (e.g., Macháč et al., 2018a).

## Methodological complications

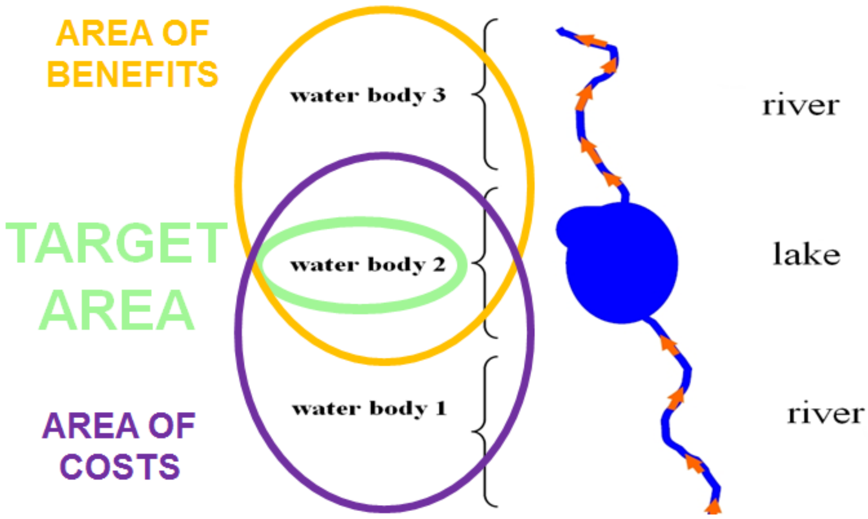
Application of the above procedure based on cost-benefit analysis is connected with numerous methodological complications, which have to be tackled. They include primarily defining the scale of the analysis, defining effectiveness of measures, coping with the risks when quantifying the costs and benefits, the issue of suitability of the benefit transfer method, the method of applying the cost-effectiveness analysis, including ranking of measures, and more. This sub-chapter focuses on the cost side of the methodology, specifically on defining the scale and the method of determining the most cost-effective combination of measures with respect to the given goal.

### Defining the scales of the analysis

A definition of the scale for carrying out the cost disproportionality assessment belongs among the initial steps. The exemption has to be applied for at the water body level. Based on individual consideration, the author of the proportionality analysis decides about the specific water body for which the cost proportionality will be assessed. A larger area (Fig. 6) has to be considered for the actual assessment and comparison of the costs and benefits. In terms of the costs, the entire sub-catchment located upstream of the water body has to be considered, because the achievement of "good status" is affected by all the potential measures having an effect on the water body in question, and the most cost-effective measures have to be selected for implementation. In terms of the benefits, one has to consider the fact that improvement in individual indicators may benefit not only the water body for which the measure is implemented but, to a certain degree, also water bodies further downstream in the catchment area, where additional benefits may be produced and should be included in the analysis on the benefit side. As part of the application for the exemption, the analyser should make a description of the affected water bodies and catchment areas. It has to include, above all, a geographical localisation of the water body, its primary and secondary purposes and functions, and an assessment of the local and wider significance of the water body in the catchment area. This initial step of the proposed methodology is intended for a general introduction to the scale of the analysis and an initial definition of the potential significance of the water body being assessed and the possible anthropogenic and other factors affecting the water quality components and pollutant concentrations in the surrounding area.

In relation to the benefits, considerable uncertainty is introduced by modelling the distance boundary within which the measures adopted for the water body have a significant effect on other water bodies downstream in the catchment area. Large uncertainties associated with defining the boundaries for benefits are evident in the study of reducing eutrophication in the Orlík reservoir (Vojáček et al., 2013), for example, where benefits for other water bodies (Slapy reservoir) were only considered qualitatively.

**Figure 6: Target water body and surrounding water bodies with an impact on the costs and benefits**




Source: Own construction

No less important than the spatial scale is the time aspect. The cost disproportionality analysis has to define the time frame for which the costs and benefits will be included. Various time frames are used abroad. For example, Martin-Ortega et al. (2013) based their time frame selection on the Water Framework Directive milestones, and used the year 2027 (a period of 14 years) for expressing the costs and benefits. Klauer et al. (2015) use the same time frame until the year 2027. The medium term of 20 years appears to be a suitable scale, as it is used the most often in similar analyses and studies worldwide. Alternatively, this time frame can be adjusted depending on the lifetime of the investment measures considered. It is absolutely essential for adequate assessment that the benefits of the measures are fully manifested in the time period used. The time scale therefore must not be so short as to only include the costs while the benefits will only occur after the end of the selected time frame. Conversely, too long periods are burdened with considerable uncertainty, which is due to both developments in the period as such and effects of global climate change and other factors. Alternatively, the concept of annualised costs and benefits can be used (recommended for application in the Czech Republic based on Slavíková et al., 2015).

A well-chosen scale has an influence on the definition of the pollution sources (reasons and originators of non-achievement of “good status”), and determination of the effectiveness of measures. The source analysis is based on catchment area monitoring. In addition to point sources, we need to consider nonpoint (diffuse) sources as well, and the contribution of each source to the non-achievement of “good status” has to be defined. The distance and time aspects have to be taken into account as well. Depending on the significance of the different sources, the analysis may focus in more detail on a specific category of sources. The time aspect has to be considered due to the often uneven contribution of the source to the total pollution caused by seasonal fluctuations associated with a particular time of year, etc. The source analysis should therefore be based on annual monitoring at the least. Considerable fluctuations may have a fundamental influence of achievement of “good status”. Additionally, the source analysis has to take into account the spatial aspect, i.e., the distance of the source from the area (water body) in which “good status” is not achieved. Notably, chemicals are naturally retained within a catchment area. Therefore, if we need to reduce phosphorus at the entry to the reservoir B, for instance, by 5 kg, it is necessary to consider the natural retention of the source located upstream of the reservoir B, i.e., to eliminate a proportionally larger quantity. If the natural retention is 20%, then the phosphorus has to be reduced by 6.25 kg at the source in order to achieve the target; this is manifested as a 5 kg reduction at the entry to the reservoir. This natural retention is often disregarded in analyses, leading to a considerable distortion and over-estimation of the effect of measures.

**Selection of the most cost-effective combination of measures using CEA**

Identification and definition of specific applications of measures and qualification of costs of their implementation are followed by their monetisation. As part of assessment of the costs and subsequent cost-effectiveness analysis, one has to cope with the different time aspects (lifetime of measures), different composition of the costs (e.g., comparison of measures based purely on investment costs with ones with predominant operating costs), combining of measures, etc. These complications are best tackled using the annualised cost concept. Using the annualised cost method, a known value of present costs is transferred to a future flow of the same costs based on annual costs, which correspond to the known present value when cumulated (Jacobsen, 2005). For the purposes of the cost-effectiveness analysis, it is advisable to break the costs down into total investment costs (sometimes also referred as acquisition or one-off costs), annual operating (periodically recurring) costs, and other costs (such as administrative costs, lost profits).

This is followed by a quantification of costs based on customary (market) prices; these data can be made more accurate by predicting future prices. This is the standard way of pricing both investment and operating costs. The general recommendation is to take into consideration the widest possible range of potential and known costs associated with the measure. Cellini et Kee (2010) state that all the costs cannot be estimated due to uncertainty, which is why an effort has to be made to identify those that are expected to have the greatest effect. Possible sources of data for the pricing are similar measures already implemented with similar parameters, expert studies, catalogues of measures or a market survey in the form of a non-binding request with contractors/implementers of measures. If such data are unavailable, experts can be consulted or educated estimates used. In many cases during the pricing, it is necessary to break costs down into an array of component costs that are priced differently depending on their structure (e.g., using a choice survey). The cost calculation has to make sure that no costs are accounted for twice. This is often the case when using costs from different catalogues and studies and expanding them with own data. Local conditions are an important factor: they may have a large effect on the implementation of selected measures.

After that, the lifetime of each measure is determined. It often happens that a measure consists of multiple goods with different lifetimes. In that case, the lifetimes shall be expressed separately for the different parts of the measure and the corresponding costs. Again, data on lifetimes are based on catalogues, projects already implemented, or technical documentation for existing measures of the same type.

The monetary expression of costs uses the cost annualisation method mentioned above. First, the present value of costs of the measure is determined (Equation 1), or of the component parts of the measure with different lifetimes.

**Equation 1: Present value of costs**

$$PV = \sum \frac{C_t}{(1+i)^t}$$

Source: Own construction

Where: PV – present value of costs
$C_t$ – total costs in the year t
i – discount rate
t – year of cost occurrence

Then, the annualised costs for each component are calculated (Equation 2). The sum of the component annualised costs related to a certain lifetime yields the total annualised costs of the measure.

**Equation 2: Annualisation of costs**

$$AC = PV \times \frac{i \times (1+i)^l}{(1+i)^l - 1}$$

Source: Own construction

Where: AC – total annual costs in the annualised form
PV – present value of costs
i – discount rate
l – expected lifetime of the measure

Once the annualised costs required for implementation of the different measures are calculated and the resulting effect expressed in physical units (e.g., amount of phosphorus reduced in kg) is known, we proceed to determining the cost-to-effect ratio, based on which the measures can be compared. The indicator is shown in Equation 3. The equation yields the costs of achieving a unit of the effect (e.g., elimination of 1 kg of phosphorus).

**Equation 3: Cost-to-effect ratio**

$$Relative\ cost\ effectiveness\ indicator = \frac{\begin{matrix} total\ annualised \\ costs\ of\ measure\ (AC) \end{matrix}}{\begin{matrix} annual\ effect \\ of\ measure \end{matrix}}$$

Source: Macháč (2014).

After determining the ratios for all the measures, we can rank the measures by the ratio and select the least costly measures per unit of output that can be used to achieve the required target.

Methodologically speaking, the CEA may assume many forms. In the field of water management, we come across both simple linear (optimisation) models used for smaller areas and models using mathematical programming, suitable for larger areas, including natural conditions (Macháč, 2014). Two different forms of output exist in optimisation CEA models. The EPA (1995) distinguishes between the cost minimisation and benefit maximisation approaches. When minimising costs, the principal goal of the measure is to achieve an effect with the least possible costs. This approach is applied predominantly. It is used, e.g., by Yang et al. (2005), van Soesbergen et al. (2007), Martin-Ortega et al. (2013) and Vojáček et al. (2013). Conversely, the benefit maximisation approach aims at achieving maximum possible level of output (effect) using a predetermined budgetary constraint; it is most commonly used for restoration measures. The benefit maximisation approach has been used, e.g., by Ancev et al. (2008) and Azzaino et al. (2002). Balana et al. (2013) describe both approaches in the mathematical form. Equation 4 shows the procedure for cost minimisation; Equation 5 shows the benefit maximisation.

**Equation 4: CEA method in the form of cost minimisation**

$$Min. \sum C_i(e_i) \text{ given that } \sum e_i \geq \mathcal{R},$$

Source: Balana et al. (2013).

Where:
$C_i$– cost function for i-th measure
$e_i$ – size of effect of measure expressed in units
R – required level of resulting effect expressed in number of units, thus the environmental target

**Equation 5: CEA method in the form of benefit maximisation**

$$Max. \sum e_i \text{ given that } \sum C_i(e_i) \leq B,$$

Source: Balana et al. (2013).

Where:
$e_i$ – size of effect of measure for i-th measure expressed in units
$C_i$ – cost function for i-th measure
B – predetermined maximum costs

The basic algorithm is the same for both forms. The measures are ranked by the relative indicator (effectiveness indicator). In the case of cost minimisation, the effect of measures is added cumulatively depending on their ranking. When the required size of the effect is achieved, all the instruments included comprise the most cost-effective way of solving the given problem. In the case of benefit maximisation, the effects are cumulated. The optimal selection is given by the cost ceiling.

The application of the optimisation CEA method may be complicated by the actual nature of the measures being considered. In many cases, the various categories of measures proposed for implementation in the same area may affect one another. In extreme cases, they may be substitutes, with application of one measure ruling out the application of another. For example, arable land cannot be simultaneously afforested and subjected to a change in the tillage technique. Van Soesbergen et al. (2007) gives more examples of possible connections among measures. Implementation of some measures may be conditioned by adoption of others. The summed size of the effects may be different when combining different measures than when implementing them separately.

The problem of mutually exclusive measures is tackled in practice by calculating the cost effectiveness separately for each measure and then including in the CEA only the most cost-effective one of a group of mutually exclusive measures. This makes use of the basic ranking algorithm. However, it may happen in the case of different effects on water quality that a measure is promoted into the analysis that is the most effective but whose total effect is lower than that of some of the excluded measures.

Concerning the sequence of measures (e.g., the measure B cannot be implemented without the measure A, so that the measure A has to be implemented first), the general recommendation (e.g., by van Soesbergen et al., 2007) is that sequential measures not be treated separately but integrated with the base ones. Thus, the choice is between the base measure (measure A) only and a combination of the base and the successive one (measures A+B). But again, this brings back the problem where the measure A and the combination of A+B may be mutually exclusive. Oversimplification may lead to selection of a sub-optimal combination of measures.

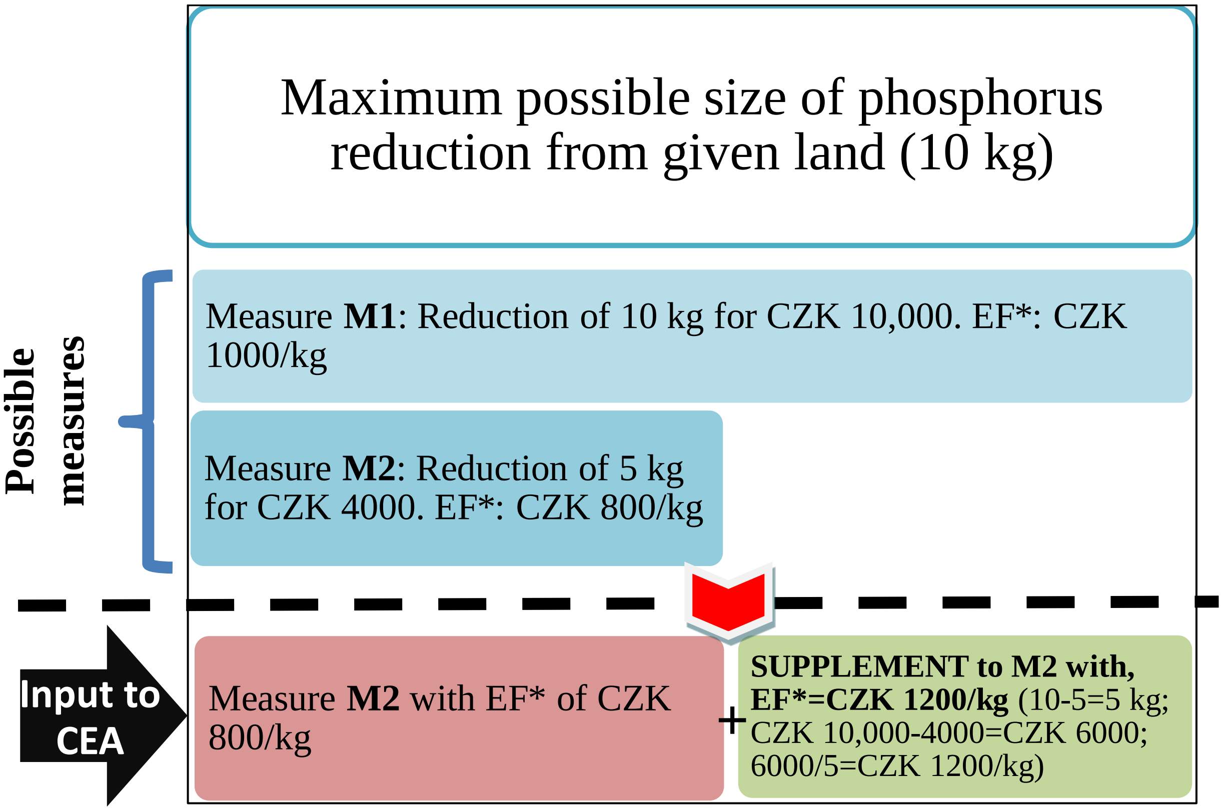
In both cases (mutually exclusive measures and sequential measures), a more complex algorithm can be used which will choose based on the target effect which of the possible measures is appropriate with respect to other measures. This dynamising process is based on the creation of all possible combinations of measures including both mutually exclusive and sequential measures. It has to be borne in mind when making the combinations that mostly the total effect is not a simple arithmetic sum of effects. Synergies among the measures have to be accounted for and the resulting effect has to be adjusted accordingly, determining a new rate of costs per unit of effect. The combinations thus created are ranked for the area by their cost-to-effect ratios, yielding the most cost-optimal combination. Up to here, the procedure follows the basic ranking algorithm.

If the sequential measures problem is resolved, the size of the effect of different combinations with respect to other measures in the water body has to be reflected. In the first step, we can eliminate all combinations with an effect lower than the most cost-effective combination in the group of mutually exclusive combinations of measures. After that, we can eliminate all other measures with an effect lower than the second-most cost-effective combination. I proceed analogously to eliminate measures for the third-best, and so on. This yields a list of combinations with decreasing effectiveness of measures (or growing costs per unit of effect) and size of the effect. For these remaining measures, determine the differences between their effect and the total annual costs. These differences can be viewed as supplements specifying how much has to be expended additionally to achieve a greater effect in the area compared to the most cost-effective measure. Afterwards, determine the cost-to-effect ratio for these supplements as per Equation 3 and feed them into the CEA process. The selection process is called dynamic CEA, and was used in articles 1, 4 and 5.

This procedure is best demonstrated on an example of two hypothetical measures. The first one (measure 1) reduces 10 kg of a pollutant for CZK 10,000; the other one (measure 2) only reduces 5 kg for CZK 4000. Their effectiveness ratios are CZK 1000/kg and CZK 800/kg, respectively. Measure 2 appears more expedient at first sight, but if measures on other land are around CZK 2000/kg, then it is more expedient to implement measure 1 on this land because it is still more advantageous and leads to more effect than the measures on other land.

If the above situation occurred, measure 2 would be included in the CEA as the most cost-effective measure. The difference between the costs and effects of measures 1 and 2 would be used to make a supplement to measure 2 reducing 5 kg for CZK 6000, with an effectiveness of CZK 1200/kg of phosphorus. If the supplementary measure is used in the CEA, the algorithm finally eliminates measure 2 and assumes implementation of measure 1. This procedure is shown in Figure 7. If there are more measures in the pre-selection, the same procedure can be applied to the others as well, only more supplements would be defined, connected to one another.

**Figure 7: Ranking algorithm diagram using Dynamic CEA**




*EF = effectiveness of measure

Source: Own construction

Supplements as described above then enter the overall CEA along with the most cost-effective combination of measures. In case the supplement is included in the list of cost-effective measures, the originally included measure (combination) has to be changed into the measure connected with the supplement.

**Benefit valuation**

At this point, it is necessary to emphasize that economic benefits are perceived as anthropogenic. Thus, benefits are tied to individuals and are derived from expressing a particular value by people (for example, by accepting a price or declaring willingness to pay for a given service). This economic concept is significantly different from the concepts of values in the natural sciences where the ecosystem or its parts can be attributed value independently of human attitudes. All other non-anthropogenic values are thus considered to be values that go beyond the level of human perception and knowledge, and thus remain unchanged in our approach (in monetary terms). A part of anthropogenic benefits does not go through the market, which needs to be addressed by choosing the appropriate valuation method.

In addition to the main benefits (e.g., sediment, phosphorus, nitrogen retention), indirect benefits are considered in the context of ecosystem services concepts. Thus, benefits from services provided by the ecosystem which have an indirect impact on humans (e.g., savings of costs of water treatment, water retention in the landscape and prevention of floods or droughts, erosion reduction, air pollution, $CO_2$ absorption, effects on aesthetic value) should be included. Therefore, the three basic categories (mentioned earlier) were defined in the certified Czech methodology (Slavíková et al., 2015).

A wide spectrum of methods can be used to monetize (valuate) the benefits. However, each category of benefits is specific, and it is necessary to choose the appropriate method with respect to available input data. Table 1 contains a basic set of methods that can be used to value the benefits mentioned above. The benefits are divided according to the concept of ecosystem services.

**Table 1: Classification of benefits and methods of valuation**

| Benefit provided | Valuation method |
|---|---|
| | **Regulating ecosystem services** |
| **Quantity of surface water and groundwater** | Market price method, Avoided cost method |
| **Flood risk reduction** | Avoided damage cost |
| **Water quality** | Avoided cost method, Substitute cost method |
| **Noise** | Substitute cost method |
| **Air quality** | Substitute cost method, Market price method |
| **Soil erosion** | Substitute cost method |
| **Microclimate** | Market price method, Substitute cost method |
| **$CO_2$ reduction** | Substitute cost method, Market price method |
| | **Cultural ecosystem services** |
| **Recreational utilities** | Travel cost method |
| **Aesthetic value** | Stated preferences method: Choice experiment, Willingness to pay |

| Benefit provided | Valuation method |
|---|---|
| | **Provisioning ecosystem services** |
| **Biomass production** | Market price method |
| **Crop production** | Market price method |
| | **Others** |
| **Value increase of adjacent properties** | Hedonic pricing – determination of market price of the attribute (park) using regression models |

Source: Macháč et al. (2018b - In press).

For the valuation, it is necessary to have data on individual indicators such as current value of real estate, numbers of visitors, quantity of water abstraction for drinking or other purposes, or other bio-physical indicators. An alternative method to conventional methods is benefit transfer, usable for taking over results of other studies (used, e.g., by Vojáček et al., 2013). This method is used predominantly where primary data are not available or where their acquisition would be disproportionately expensive (which was also the case of the Orlík reservoir study). More accurate values can be achieved using meta-analysis, which makes it possible to convert values to reflect local aspects (e.g., EFTEC, 2010).

Due to the lack of primary data and the difficulty of some valuation methods (time and cost demand), it is advisable to create a database of values that can be used for valuation and thus for analysis of disproportionality. This procedure was chosen by Jensen et al. (2013), who carried out detailed analyses in one river catchment, and transferred the values to other river catchments using a simple unit value transfer.

Creation of a detailed database can be one of the next steps in application for exemptions due disproportionate costs. When combining multiple valuation methods, it is crucial to avoid double counting of costs and/or benefits.

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# 1. Assessment of disproportionate costs according to the WFD: Comparison of applications of two approaches in the catchment of the Stanovice reservoir (Czech Republic)

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## 1.1 Introduction

With ever increasing concerns about water quality in Europe, the EU has implemented the Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy (2000; WFD). To improve water quality standards across Europe, a “good status” was introduced in the WFD. Annex V of the WFD describes (rather vaguely) the “good status” as a state of only a slight departure from the biological community, which would be expected in conditions of minimal anthropogenic impact. The “good status” is composed of two separate parts – an ecological status and a chemical status, which are both determined by several indicators. All water bodies within EU member states were expected to achieve this status by 2015. However, since the requirements are stringent and a "one out, all out" rule applies, there are numerous water bodies that failed to do so before the deadline.

This does not necessarily mean breaking the regulation. It is possible to apply for an exemption and extend the deadline until 2021/2027 or even mitigate the target as suggested in Article 5 of the WFD. There are three justifiable reasons for not achieving the “good status” listed in Article 4. One of them is technical feasibility, which allows to reach the target only gradually and not within the timescale. It is also justified not to achieve the “good status” when natural conditions do not allow for a timely improvement. The third reason is associated with disproportionate costs. In many cases, implementation of necessary measures may be too costly within a short timeframe. The application of an exemption based on disproportionate costs needs to be supported by an economic analysis, which concludes that measure implementation is disproportionate as benefits generated by meeting the "good status" are not large enough to outweigh associated costs. Unfortunately, the WFD does not specify how large the gap needs to be for the exemption to be approved (e.g., Nocker et al., 2007; Martin-Ortega, 2012; Jensen et al., 2013). As a result of the regulation, demand for cost proportionality analysis in water management has increased dramatically. Numerous methodologies have been created to assess the issue. These can be categorized into three distinct groups – analysis based on affordability, cost-benefit analysis (CBA) that uses monetization, and cost-benefit analysis that uses threshold and criteria (criterial CBA). As these approaches are quite similar, we will refer to CBA that uses monetization as to "monetary CBA".

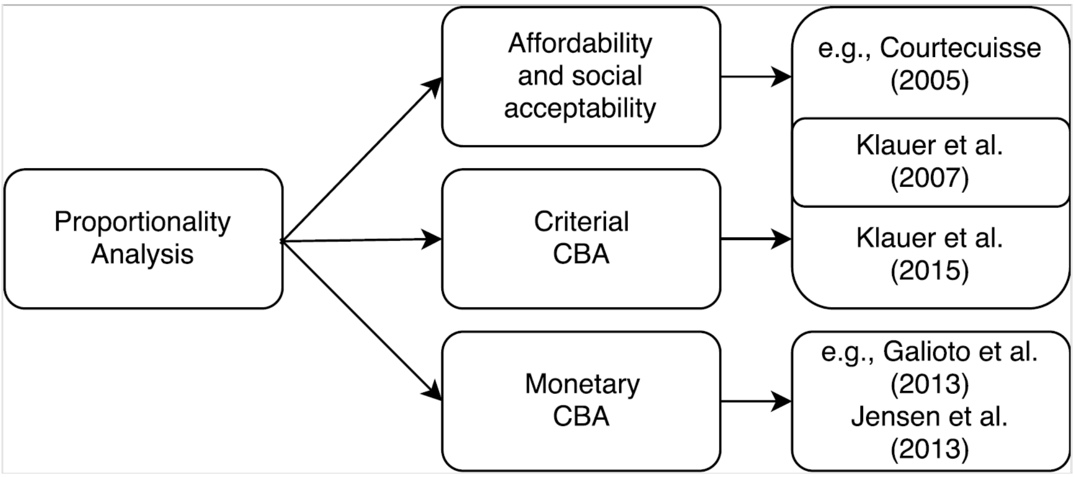
The purpose of this paper is to compare two of the existing approaches: monetary CBA and criterial CBA. This is done using numerous examples of application of these approaches. We focus on differences among individual methodologies as we later compare two of the methodologies directly. The official Czech methodology (Slavíková et al., 2015) and the "New Leipzig approach" (Klauer et al., 2015, 2016) were chosen as representatives of the two approaches. Both methodologies were recently used to evaluate cost proportionality of achieving the "good status" at the Stanovice water reservoir. Therefore, we can evaluate how they perform when used under identical conditions.

The paper is structured as follows. We skip introduction to the monetary CBA as it is a common tool for evaluation and we focus on its application in proportionality analysis under the WFD. This is followed by a short description of the Czech methodology. The "New Leipzig approach" is described in more detail as it has not been applied in many cases yet. A theoretical comparison is made prior to the case study. Results for both methodologies are followed by a discussion of weak and strong points of each approach.

## 1.2 Approaches

As mentioned above, countless methodologies have been developed to evaluate cost proportionality of achieving the "good status" (e.g., Courtecuisse, 2005; Klauer et al., 2007, 2015; Vinten et al., 2012; Galioto et al., 2013; Jensen et al., 2013; Slavíková et al., 2015). These methodologies can be sorted into three main groups of approaches as shown in Figure 1.

**Figure 1: Overview of possible approaches**




Source: Authors

The first approach takes into account affordability and social acceptability of achieving the "good status". It estimates the financial impact on affected groups (e.g., comparison of household budgets with water supply and sewerage charges in Courtecuisse, 2005). The second approach is based on CBA, but avoids monetizing benefits generated by achieving the "good status". Instead, criteria are used to form a cost threshold. This approach is an alternative to the third approach – monetary CBA – which monetizes both costs and benefits and is the most widely represented group in the EU (based on Klauer et al., 2015).

### 1.2.1 Monetary CBA

The WFD does not set many rules about the form of the economic analysis. Monetary CBA is often used, because it is relatively straightforward and widely recognized as a useful tool in proportionality analysis. As a result, many slight variations of monetary CBA can be found in literature (e.g., Hanley et Black, 2006; Brouwer, 2009; Vinten et al., 2012; Galioto et al., 2013; Jensen et al., 2013; Martin-Ortega et al., 2013; Vojáček et al., 2014; Feuillette et al., 2016).

To be more specific, Jensen et al. (2013) developed a methodology to assess proportionality in Danish catchment areas. It is based on a preliminary comparison of costs and benefits in each area. It uses rough estimates to decide whether it is necessary to perform a thorough analysis. This applies to the catchments where costs exceed or are similar to generated benefits. Authors recommend using the most cost-effective measures and primary pricing studies for valuation of benefits, but admit benefit transfer is less costly and defend using the method where primary data are not available. Jensen et al.

(2013) apply the methodology to 23 catchment areas in Denmark to find 5 areas where costs of measure implementation seem to be disproportionate (after sensitivity analysis). The authors extrapolate data about willingness to pay (WTP) from one of the catchment areas and adjust them based on a number of households living in the analysed area.

Galioto et al. (2013) follow steps similar to those of Jensen et al. (2013), but specify their methodology by adding 4 equations that describe relationships among variables. Also, they identify costs and benefits separately for each water body. The proposed measures can affect each other and the same holds for individual water bodies. Therefore, it is necessary to identify these relations and analyse cost proportionality for both the water body and the catchment. The methodology is very detailed when identifying all the different types of costs. It includes investment and operating costs, decreased profits as a result of measure implementation, social costs in the form of taxes and other indirect costs. The authors point out that a comprehensive analysis of benefits is disproportionately costly and for non-use benefits they only consider a situation in which the "good status" is achieved. As in many other methodologies, benefit transfer was used to value associated benefits.

In Scotland, Vinten et al. (2012) propose a methodology that is quite similar to the Danish one. Costs of implementing the most cost-effective measures are compared with benefits, which are estimated using a choice experiment. Hanley et Black (2006) divide the analysis into a "micro" and "macro" level. The "micro" analysis aims at studying an impact on a single sector (e.g., lost agricultural yields are compared with benefits generated thanks to improved environmental status of relevant water bodies). The "macro" level aggregates costs of various industries and, after discounting, makes a comparison with nationwide benefits. Whenever costs exceed benefits, the implementation is considered disproportionate.

Recently, Feuillette et al. (2016) used a French guidance on exemption to assess cost proportionality under the WFD in France. The guidelines recommend performing a financial capacity test, which preselected over 700 water bodies where monetary CBAs were carried out. Given uncertainties, costs are considered proportionate if benefits from achieving the "good status" cover at least 80% of them. Benefit transfer was often used because of the large amount of analyses. A spreadsheet with benefit values was created, which allows users to enter several parameters about a particular water body and receive a range of possible benefits. As the water bodies fall under multiple river basins, some differences in the monetary CBAs may be found (e.g., use of the spreadsheet, choice of population). The authors also point out that implementation costs are often overestimated, while generated benefits are usually underestimated, which justifies the 0.8 ratio used to assess the proportionality.

This chapter shows that each methodology used to evaluate implementation of the WFD is unique and that it is possible to learn from both strengths and weaknesses. Table 1 shows the main attributes of selected methodologies from various European countries. The Czech methodology described in the next part combines various elements from European methodologies and adds a few new ideas.

**Table 1: Comparison of European methodologies**

| Country | Authors | Scale of proportionality analysis | Method | Estimation of costs | Estimation of benefits | Recommendation |
|---|---|---|---|---|---|---|
| **Denmark** | Jensen et al. (2013) | Catchment area | Cost-benefit analysis | Cost-effectiveness analysis | Benefit transfer of simple mean value from existing study for one of Denmark's catchment areas | More detailed cost-benefit analysis for catchments that appear to be disproportionate |
| **France** | Feuille-tte et al. (2016) | Seven main river basins | Cost-benefit analysis | - | Benefit transfer from existing available literature | When benefits are less than 80% of costs, it is considered that costs are disproportionate |
| **Italy** | Galioto et al. (2013) | Regional scale, further subdivided into specific categories | Cost-benefit analysis | Cost-effectiveness analysis | Benefit transfer from existing available literature | Interactions between measures and pressures and interactions among water bodies need to be identified and taken into account |
| **Scotland** | Hanley et Black (2006) | River and national level | Cost-benefit analysis | | Benefit transfer from existing available literature | CBA appears to be an appropriate means for both the microeconomic and macroeconomic analysis level |
| | Aresti (2008) | Agriculture sector | Cost-benefit analysis, financial affordability test | Cost-effectiveness analysis including farm viability and affordability assessment | Benefit transfer, choice experiment | As a first step: assessment of the financial impact on individual water users and of the level to which the cost of achieving "good status" may jeopardize their financial viability or sustainability |
| | Vinten et al. (2012) | Water body (lake) | Cost-benefit analysis | Cost-effectiveness analysis | Choice experiment | Combination of Cost-effectiveness analysis and choice experiment appears to be an appropriate tool |

Source: Own construction

### 1.2.2 Czech methodology

The certified Czech methodology (Slavíková et al., 2015) uses monetary CBA to evaluate proportionality of achieving the "good status". The analysis is performed at a basin level and for a specific substance. It is necessary to estimate the amount of the substance inflow that needs to be prevented each year in order to achieve the "good status". The inflow reduction is specified in terms of kilograms per year. Important steps of the methodology are described below.

Similar to most of the European methodologies, the Czech methodology requires implementation of the most efficient measures. The first step is to identify all the available measures and associated costs. These are expressed in annual form, which is innovative to the WFD proportionality analysis. The detailed annualization process is described in the methodology. This solution makes all the measures directly comparable and is viewed by the authors as more useful than the concept of net present value. Once computed, the costs of each measure are divided by the amount of the substance they reduce and the measures are ranked based on their efficiency.

The selection process itself recognizes that the measures may affect one another as suggested by, e.g., Galioto et al. (2013). Some of the measures are mutually exclusive (designed for the same area), while others enter the selection process as part of a bundle. Additionally, the selection of mutually exclusive measures is improved in the sense that a more efficient measure may be rejected in favour of a less efficient one with a higher absolute impact, which means fewer measures are needed overall. The resulting measures may not be the most cost-effective ones according to the ranking, but they achieve the reduction with lowest possible costs.

The second part of the analysis covers monetization of generated benefits. This is a standard procedure in monetary CBA, but the Czech methodology clearly specifies the categories that should be included in the analysis – recreational benefits, lower costs of drinking water treatment and improved ecosystem services. Slavíková et al. (2015) recommend primary valuation methods (revealed preferences, stated preferences) and provide a guidance on how to proceed. However, they agree with the methodologies mentioned above in that it is often too costly to collect primary data, and in such cases, encourage the use of benefit transfer. Benefits in the monetary form are also annualized to be directly comparable with costs, which is the last step of the analysis. However, sometimes it is impossible to monetize all the identified costs and benefits, and in such cases, it is important to bear in mind that the final values are likely underestimated.

Additionally, sensitivity analysis should be performed, because especially evaluation of benefits is problematic and faces many uncertainties. Optimistic and pessimistic scenarios should be considered beyond the original setting.

Based on the comparison, a recommendation for exemption is made. If total benefits exceed total costs, an exemption based on disproportionate costs is going to be rejected.

### 1.2.3 New Leipzig approach

An alternative to monetary CBA has been developed in Germany. Some other studies preceded the current German methodology, the "New Leipzig approach". The first one (Klauer et al., 2007) analysed all the possible ways of assessing proportionality. The authors came up with several options. Most of them were based on social affordability or affordability for the private owner of the land, which is suitable for implementation of any measures. These approaches were rejected by the European Commission. The reason for the rejection was that most of the measures were in the form of positive externalities, which means the payer (investor of the measure) is mostly not the bearer of all the benefits. From the European Commission's point of view, the transfer of money between the investor and the final user of the benefits generated from the implementation of the measures must be considered. Therefore, it was necessary to find new ways of assessing proportionality. The direct predecessor of the "New Leipzig approach" was the methodology developed by Ammermüller et al. (2008). The idea was to apply a methodology which can compare costs and benefits without monetizing the benefits.

Ammermüller et al. (2008) defined the logic of the "New Leipzig approach". The current costs associated with achieving the "good status" may increase in comparison with the public expenditures made in the past and additional benefits generated by implementation of the measures. Ammermüller et al. (2008) developed a rather complicated methodology, which is based on multi-criteria analysis. The additional benefits are defined in the form of criteria.

Klauer et al. (2015) modified this method slightly, trying to make it more efficient. The "New Leipzig approach" was applied to seven water bodies in Rhineland-Palatinate (on a tributary to the Rhine river) (Klauer et al., 2016) and later also to other 164 surface water bodies (Klauer et al., 2017). The catchment of Rhineland-Palatinate faces two major problems in the form of river morphology and eutrophication. Additional measures were designed to deal with these problems. The whole process is divided into a number of steps. First, there are pre-steps 0-1 and 0-2, in which a water body for evaluation is identified and nationwide expenditures on water protection are determined. Only expenditures prior to 2009 are included, since that corresponds to the start of the first WFD planning cycle (Klauer et al., 2016).

Step 1 determines costs (investment and operational) of achieving the "good status" by 2027. While the total value of the investment is taken into account (EUR 72,548,300 for Rhineland-Palatinate), operating costs are included only for the 2015-2027 period (EUR 6,792,396).

Step 2 is designed to determine a water body's cost threshold and is divided into several sub-steps. In 2-1, annual public expenditures on the water body are calculated. Average past expenditures in water protection in the selected country are divided by the country's total area and multiplied by the catchment's total area. Annual average past public expenditure for the catchment area of the seven water bodies in Rhineland-Palatinate were EUR 9,533,250. Costs for the other 164 water bodies were estimated in a similar way (Klauer et al., 2017).

The final number is then used to determine allowed additional spending, which also depends on the distance to the "good status". This objective distance is determined in step 2-2 and is based on several criteria[3]. Each category is evaluated on a scale of 0-3 (0 = the best state = "good status") and the total distance is determined by averaging over all the categories. 2.12 is the average distance to target (objective distance) in the case of Rhineland-Palatinate. Achieving the "good status" is also expected to generate additional benefits. These are assessed in step 2-3. Just like in the previous case, there are multiple categories to evaluate. Expert judgement was used to assign a value between 0 and 3 to each category (3 = highest benefits) in Rhineland-Palatinate[4] and for the other 164 water bodies.

[3] Macrophytes/Phytobenthos, Macroinvertebrates, Phytoplankton, Fish, Environmental quality standards

[4] Ecology and nature protection (3); Freshwater provision and treatment (0); Flood protection (2); Soil protection (2); Tourism and recreation (3)

The average of these values is then used to calculate an effort factor in step 2-4, together with a cost threshold. The effort factor determines by how much spending on the particular water body is allowed to increase (see Equation 1).

**Equation 1: The effort factor**

$$Effort\ factor = \frac{2}{18} * Objective\ distance + \frac{1}{18} * Average\ additional\ benefits$$

Source: Klauer et al. (2015)

The effort factor varies between 0 and 0.5. In the case of the seven water bodies the result was 0.35, meaning a 35% increase in spending can be justified in this area. Multiplying the effort factor by annual expenditures on the water body, the maximum additional yearly spending on the water body is determined. To compute the cost threshold, additional annual spending is multiplied by the number of years remaining until the deadline.

In the case of Rhineland-Palatinate, EUR 59.5 million represent the amount of money that can be spent on top of the current spending to achieve the "good status" at the analysed water bodies.

However, spending the money might prove to be disproportionate. Step 3 compares costs of implementing the selected measures from step 1 (EUR 79.3 million) with the threshold (EUR 59.5 million). If the threshold is not exceeded, the measures should be implemented. If the opposite holds, it is reasonable to apply for an exemption, which may or may not be approved (no clear definition of cost proportionality in the WFD). Based on the result, the cost disproportionality is confirmed in the case of Rhineland-Palatinate. Therefore, it is justifiable to set less stringent environmental objectives for the seven water bodies.

### 1.2.4 **Comparison of the methodologies**

It is clear from the description above that comparison of all the methodologies is rather difficult. In further analysis, we focus only on the Czech methodology and the "New Leipzig approach".

Both selected methodologies use CEA in the measure selection process, which means identical measures may be implemented when applied to the same water body. However, they assess proportionality in very different ways. Each methodology identifies certain categories of possible benefits generated by achieving the "good status", but these are assessed differently. Benefits play a major role in the Czech methodology as they have the same status as costs, which they are directly compared with. Therefore, benefits are monetized and are crucial for the final decision. On the contrary, the "New Leipzig approach" evaluates predetermined categories of benefits on a discontinuous scale. Those benefits play only a minor role in the final recommendation as costs of measure implementation are compared with the cost threshold and not the benefits themselves. Benefits can increase the allowed expenditures, but do not have a major impact on the final spending. Moreover, the German methodology is entirely based on the equations described above, meaning all analyses look the same and are comparable. In contrast, the Czech methodology is more flexible and deals with uncertainty to some extent to remain robust. A comparison of key characteristics is presented in Table 2.

**Table 2: Comparison of the methodologies**

| Characteristics | Czech methodology | New Leipzig approach |
|---|---|---|
| Based on | Monetary cost-benefit analysis | Cost-benefit analysis & criteria (costs vs. cost threshold) |
| Measure selection | Cost-effectiveness analysis | Cost-effectiveness analysis |
| Benefits | Market prices, benefit transfer, WTP/WTA | Based on a scale |
| Uncertainty | Sensitivity analysis | Not tackled |
| Costs compared with | Benefits | Cost threshold |

Source: Own construction

## 1.3 Case study of Stanovice catchment

Both methodologies were tested on the same catchment in the Czech Republic (Macháč et al., 2015a, 2016). The Stanovice water reservoir is situated in North-West Bohemia near Karlovy Vary (Figure 2). Local conditions around Stanovice are homogenous and together with two inflows, the area covers 92 km$^2$. Povodí Ohře (2014) states that the main purpose of the reservoir is supplying drinking water for the Karlovy Vary region. Minor functions include electricity generation, fishery and flood protection for Karlovy Vary.

**Figure 2: Location of Stanovice water reservoir**


Source: Macháč et al. (2015a)

According to Povodí Ohře (2009), Stanovice reservoir currently fails to achieve the "good status" required by the WFD. The water quality is unsatisfactory mainly as a result of anthropogenic effects in the catchment area such as population and agriculture. Specifically, excessive phosphorus inflows are responsible for most of the damage. The whole area is subject to cyanobacterial growth in the summer months. T. G. Masaryk Water Research Institute estimates that an annual reduction of 60-200 kg of dissolved phosphorus is necessary to achieve the "good status". Phosphorus contamination is divided evenly between point sources (wastewater) and diffused sources (mainly agriculture).

243 unique measures were identified to reduce phosphorus inflows into Stanovice. Together, the measures can reduce 344.6 kg of phosphorus each year. Both measures on point sources[5] and agricultural measures[6] enter the analysis.

[5] Construction and renovation of wastewater treatment plants, sewer systems, dead-end and accumulation cesspits, retention wetlands, biological reservoirs, domestic wastewater treatment plants, intensification of the treatment process at wastewater treatment plants.

[6] Building of broad-base terraces, grassing of sloping areas, changing of crop rotation, leaving crop residue, introduction of no-tillage methods.

### 1.3.1 Results: Czech methodology

The Czech methodology was applied to the Stanovice reservoir by Macháč et al. (2016). Costs of achieving the “good status” were previously estimated by Macháč et al. (2015b) using a dynamic CEA. All the possible measures mentioned above were ranked based on their efficiency. The most efficient combination of measures (99 in total, 62% of reduction comes from measures on point sources) that meets the 200-kg threshold enters the analysis. The annualized costs of the selected measures were calculated at EUR 42,200.

Macháč et al. (2016) follow the benefit categories established in the Czech methodology and evaluate them separately. Recreational benefits are usually the most important category. However, swimming is prohibited in the reservoir, which means only aesthetic benefits are generated. Benefit transfer was used to evaluate gains for local people and incoming tourists. Authors assume that all people from the closest neighbourhood and only a part of tourists are recipients of these benefits. Only municipalities within 5 km of the reservoir were taken into consideration, despite many other municipalities located nearby. The full value of aesthetic benefits based on the benefit transfer from Corrigan et al. (2009) was used only for the nearest village. The total benefits for other residents in the defined area and part of the tourists visiting the area were calculated based on the lowest value of recreational benefits from Vojáček et al. (2014). The lowest value was chosen mainly because swimming is prohibited in the Stanovice reservoir, which means a large portion of recreational benefits cannot be generated. According to the paper, the better quality of water increases the recreational benefits by EUR 1.9 per man-day (one-day visit). The total value of the recreational benefits is EUR 76,050 per year.

Another important source of benefits is reduction in costs of drinking water treatment. Based on the amount of water treated, average operating costs of the Březová water treatment station and estimated potential savings, benefits of achieving the “good status” were calculated. Based on consultation with the T. G. Masaryk Water Research Institute and the study by Pretty et al. (2003), 10 percent of treatment costs can be saved if eutrophication is not present. Therefore, the savings of EUR 207,407 can be expected.

Together, these two groups represent a gain of EUR 282,758 a year. This number is not final as authors were not able to evaluate several groups of benefits, the increase in property values near the reservoir after water quality improvement being the most important one. Unfortunately, no such data are available. Additionally, minor benefits are generated by ecosystems, specifically flood protection, soil protection and higher biodiversity. Because of this omission, the true benefits of achieving the "good status" are likely higher than EUR 282,758.

Comparing the costs of measure implementation with the generated benefits, there are net gains for society of EUR 240,558 a year. This finding was later confirmed by a sensitivity analysis, which was applied in the form of scenario analysis. Beside the base scenario, optimistic and pessimistic scenarios were defined. These scenarios differed in their levels of costs and benefits. In the pessimistic scenario we worked with a presumption that the costs are underestimated and benefits overestimated (no cost savings from drinking water treatment and significantly lower aesthetic benefits). The net social benefits are EUR 262,963 in the case of the optimistic scenario. Even in the pessimistic scenario, there are some net benefits associated with achieving the "good status" (EUR 300). Therefore, the exemption is not justifiable and the measures should be implemented.

### 1.3.2 **Results: New Leipzig approach**

The German approach was tested on the Stanovice reservoir by Macháč et al. (2015a). The time series of Environmental Protection Investment[7] (Czech Statistical Office, 2015) was used to determine prior spending on water protection. The data from 1994 to 2009 were adjusted for inflation and averaged out to give an annual spending of EUR 527 million. Using this figure in the equation, the past annual spending on Stanovice was determined – EUR 614,889. Despite struggles with data availability (half of the values are unknown), the authors determined the objective distance to be 0.2 (average of all 3 water bodies). The authors estimate that achieving the "good status" is associated with total additional benefits of 1.4, most of them generated in freshwater provision and treatment and soil protection. Plugging this into (1), the effort factor of 0.1 was determined, meaning it is cost-proportionate to spend an additional EUR 61,492 on the Stanovice catchment each year. Starting in 2009 and using 2027 as the last deadline, the time period lasts for 18 years. Therefore, an overall allowed spending within the timeframe is EUR 1,106,853.

The estimate by Macháč et al. (2015b) was used once again to assess the costs of achieving the "good status". Therefore, EUR 42,200 is the yearly costs of meeting the target. Although the time period starts in 2009, no measures were implemented prior to 2016. It

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[7] Expenditures on sewage disposal and water management, and cultivation are considered.

is therefore reasonable to accumulate the costs for the twelve remaining years only, yielding EUR 506,400.

As stated above, achieving the "good status" at the Stanovice water reservoir is possible with EUR 506,400. However, according to the "New Leipzig approach", it is proportionate to spend over EUR 1.1 million to reach the goal by 2027. The policy recommendation is to implement the selected measures, because achieving the "good status" is cost-proportionate and an exemption would be denied.

## 1.4 Discussion

Results from both studies show that the methodologies are not in contradiction. They both recommend implementing the measures to achieve the "good status" as the costs seem to be proportionate. Also, the gap between the costs and benefits/allowed increase in spending is quite large. However, comparison on more water bodies is required for a thorough conclusion. Here we discuss the most significant differences between the methodologies based on the two case studies that were carried out on the same water body.

First, the "New Leipzig approach" uses past public expenditures to determine a cost threshold. As the authors admit, this is questionable and has been criticised for "comparing apples with oranges" (Klauer et al., 2017), since the expenditures do not necessarily relate to the WFD objectives and may vary significantly for different regions. Therefore, large projects with no impact on water quality might skew the results. This is often true for countries that went through a transformation process and spent money excessively in the following years. It is confirmed by Czech data: the real expenditures were higher in the early 1990s than in recent years. Also, events such as floods tend to increase public expenditures on water protection with no real impact on water quality (large increase in 2003 following massive floods in 2002). Unless these distortions are accounted for, the method may give biased results.

Another possibly problematic area of the "New Leipzig approach" is evaluation of benefits. While monetizing different benefits in the Czech methodology is certainly challenging, assigning an integer value to all preselected groups in the "New Leipzig approach" might be even trickier. Experts from the T. G. Masaryk Water Research Institute were not able to determine how large the generated benefits from achieving the "good status" are on a 0-3 scale. They were unsure where the line between the values is and what conditions need to be met in order to improve the rating. The results might be distorted, because benefits are a necessary part of the most crucial equation of the whole approach. With an inaccurate effort factor, additional allowed spending is affected and consequently a wrong conclusion about proportionality may be made. Moreover, there is no reason to give all the objective distance determinants the same importance. However, benefits do not play such a significant role as in the Czech methodology.

The German approach relies heavily on very specific data. In the Czech Republic, many required observations are not available, which makes application of the approach more difficult. In the case study, half of the indicators were unknown and had to be estimated. This, however, is not a flaw of the methodology. Its purpose is to test cost proportionality in Germany, where the data are generally available. Similarly, the Czech methodology was constructed to fit the local conditions.

Among other positives, the "New Leipzig approach" has an advantage of not being too time-intensive. While the Czech methodology relies on a thorough analysis, the German approach can be done very quickly with the right data. This is true primarily for evaluation of benefits. While the "New Leipzig approach" does not require monetizing, it is a crucial and the most time-consuming aspect of the Czech methodology. Therefore, the "New Leipzig approach" may serve as a great tool for a preliminary analysis to identify water bodies with a high probability of disproportionate costs as suggested by, e.g., Aresti et al. (2008) or Jensen et al. (2013). There are a lot of water bodies that are likely not to achieve the "good status" in Germany and the Czech Republic. As Klauer et al. (2017) acknowledge, candidates for the exemption should still be considered case by case in a more detailed analysis.

Evaluating benefits may also be viewed as a weak point of the Czech approach. Monetizing benefits may be inaccurate, since there are no market prices to rely on. Estimates of WTP are not appropriate substitutes for market prices, especially when benefit transfer is used. The method saves a lot of time and effort, but may be a source of inaccuracy. Using a value from a different country and catchment can never be precise. This obstacle is partly compensated for by performing a sensitivity analysis, which makes the results somewhat robust.

## 1.5 Conclusion

As the comparison shows, both methodologies have their place in proportionality analysis, each having its stronger and weaker sides. Based on the results of the case study, both approaches are applicable in the Czech Republic. However, there are some recommendations to be made by authors based on their experience with both approaches. The German methodology seems to be a good choice for a preliminary analysis. If the result is clear, it makes little sense to waste more resources on monetary CBA, as the results tend to be similar. However, if the result is close, thorough monetary CBA is a good option to increase our confidence about the outcome.

There are also areas in which the methodologies can be improved. The German methodology should be more specific when defining what public expenditures to use to determine the country's past spending. It should also be more flexible when identifying the objective distance and benefits generated by achieving the "good status". The Czech methodology should avoid using benefit transfer unless it is necessary, and in such a case, make it more accurate. One possible way is to perform a primary valuation of benefits on multiple catchments and create a benefits catalogue with monetary values, which would be used in a similar way as the spreadsheet introduced by French Ministry of Environment and used by Feuillette et al. (2016). Some primary data are available, e.g., recreational benefits used by Vojáček et al. (2014). Another suggestion is including uncertainty about measure effectiveness, costs and benefits in the analysis.

To conclude, the Czech methodology seems to be more suitable for the Czech conditions, which is no surprise as it was created to be used in the Czech Republic. We acknowledge that performing a full monetary CBA is demanding and it might be reasonable to develop a preliminary analysis to identify problematic water bodies. As it turned out, both approaches estimate cost proportionality reasonably well, although there are still areas for improvement.

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# 2. How much extra will households pay for environmental improvement? Impacts of water and sewerage legislation in preparation on incomes of the poorest households in the South Bohemian Region

MACHÁČ, J.; ZEMKOVÁ, L. 2017. How much extra will households pay for environmental improvement? Impacts of water and sewerage legislation in preparation on incomes of the poorest households in the South Bohemian Region. In MATĚJOVÁ, L. (ed.) *Proceedings of the 21st International Conference Current Trends in Public Sector Research 2017*. Brno: Masaryk University, pp. 305-312. ISSN 2336-1239. ISBN 978-80-210-8448-3. (indexed in the Thomson Reuters Web of Science)

Share of J. Macháč: 60%

## 2.1 Introduction

Water is considered a fundamental and necessary good for human well-being, and is tightly connected to development of societies. Sustainability development studies pay considerable attention to it (Martinas et al., 2016). There is a global pressure on environmental and water quality improvement. In this connection, the EU has adopted the Water Framework Directive, setting clear goals and directions in the area of water management and environmental protection. As part of the efforts to meet the requirements of the Directive, overall regulation of the water utility sector has been in preparation in the recent months, and there has been a considerable increase in legislative requirements on water and sewerage utilities, primarily in the area of water quality improvement and assurance of reliable and sustainable water supplies. Achievement of the environmental goals is associated with significant economic and social impacts.

The legislative process includes an assessment of proportionality of the regulation using a regulatory impact assessment process. The legislative changes in preparation involve both an assessment of impacts of each amendment separately and an aggregate assessment of the conceptual design for regulation in the water utility sector; the Government has commissioned the Ministry of Labour and Social Affairs to assess the social acceptability of changes to water prices. According to the study and forecast, the expenditures should increase the most in low-income families with children, namely up to 2.76% of the total family expenditures in 2020 (Ministerstvo práce a sociálních věcí ČR, 2015). However, no comprehensive assessment of the legislative changes in progress; only separate changes in preparation are assessed.

The paper analyses the impacts on expenditures of the lowest-income households associated with increased fees for surface/groundwater consumption, restricted utilisation of WWTP sludge on farmland (Ministerstvo životního prostředí ČR, 2016a), and definition of emission limits in wastewater. The paper is based on a complete study (Macháč et al., 2016), where the impacts were analysed using a regional principle built on micro models, compiled for selected municipalities of different regions in which the water and sewerage charges are the highest. For these municipalities, we then calculated additional costs associated with the meeting of legislative changes newly approved and currently in preparation. The social acceptability of water prices is mostly perceived from the perspective of a whole country, but this approach does not correctly document the actual impacts on household expenditures due to considerable regional and local disparities. This paper therefore applies the "bottom-up" approach, i.e., one that is based on a reflection of situations in different regions. The South Bohemian Region was selected as the case for the purposes of this paper, as the local impacts differ the most in it. The situation in four cities of this region was used to model the impact. The analysis is based primarily on existing and approved RIAs (Ministerstvo životního prostředí ČR, 2015; Ministerstvo životního prostředí ČR, 2016a; Ministerstvo životního prostředí ČR, 2016b), which contain expected impacts on businesses and expected partial cost increases.

The following chapter deals with the issue of socially acceptable price, which is often recognised as one of the rules of proportionality. Moreover, it briefly introduces the legislative changes in preparation, including the data used in the models described in the final part of the chapter. The third chapter presents the model results and their discussion. A discussion of the significance of impact modelling at the local level is part of the conclusion.

## 2.2 Material and Methods

### 2.2.1 Social acceptability of water price as a proportionality indicator

The proportionality of regulation is often discussed in the area of water policy and the water and sewerage sector. The concept of social acceptability of the water price is often used as one of the primary indicators (Courtecuisse, 2005). Thus, social acceptability of the water price influences the water and sewerage charges calculations in many countries

(Chan, 2015). The indicator is also used in the study by the Ministry of Labour and Social Affairs, the first one to assess the impacts of the legislation in preparation. It concerns the share of water expenditures (water and sewerage charges) in the total household expenditures. However, the rate of this share is not uniform according to different sources. The World Bank defines the acceptable share of water expenditures in proportion to the household income as 3-5%; the UN assumes a 3% threshold, and the OECD uses 4% (Martinas et al., 2016). For OP ENV projects in the Czech Republic, the social acceptability threshold is defined as 2%, but it is respected generally. The proportion is most commonly bound to the average household incomes in the country. Absolute quantification can be done across instead of relative figures. According to the State Environmental Fund (Státní fond životního prostředí České republiky, 2015), the socially acceptable water and sewerage fee price for 2016 is set at CZK 144.40/$m^3$ (valid for Prague as the maximum in the CR); the minimum is valid for the Moravian-Silesian Region, being CZK 93.93/$m^3$.

The social acceptability of the water price involves comparison of calculations from many countries (e.g., Chile - Molinos-Senante et al., 2016), but the different quality of water supplied is not taken into account. In the CR, water is regarded as very good quality, often achieving parameters of baby water, but the quality in some other countries matches that of utility water and is not intended for drinking purposes. Therefore, the social acceptability of the water price should not be compared only from a purely economic point of view; correct comparison should take into account additional parameters such as chemical composition of water, its origin, need for purification, etc.

Generally speaking, the terms proportionality/acceptability are used increasingly often in recent years, not only when determining water prices. The notion of acceptability occurs in many other areas too, such as photovoltaics (Vojáček at al., 2015), power industry and housing.

### 2.2.2 **Data sources - Legislation in preparation**

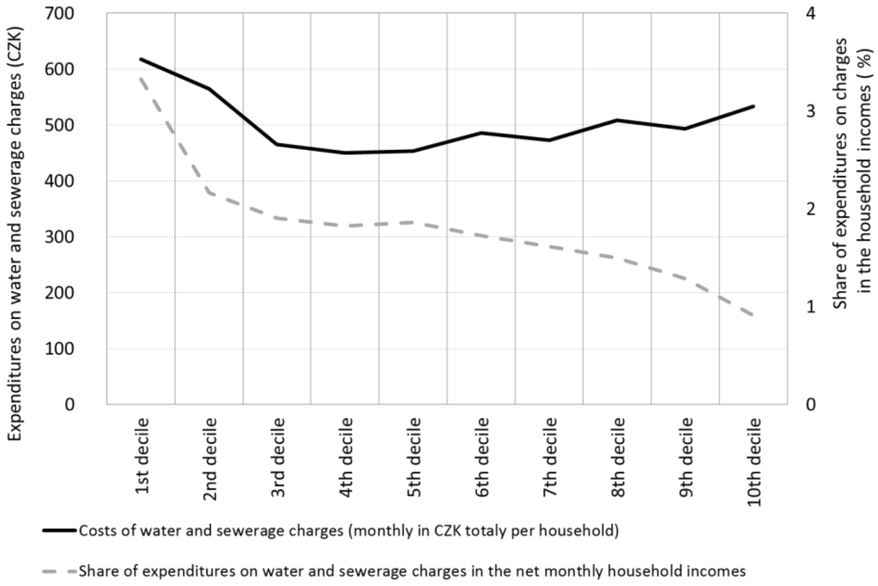
Nearly all newly emerging legislation is based on the Directive of the European Parliament and of the Council No. 2000/60/EC of 23 October 2000 establishing a framework for Community action in the field of water policy. However, regulation in the water utility sector is connected with numerous other areas, such as the impact of changes in waste management and a number of other regulations indirectly associated with the issue. In accordance with the EU law, the CR has adopted a Government Regulation on indicators and values of permissible surface water and wastewater pollution, requirements of permits for wastewater discharge into surface water and sewerage, and on sensitive areas (Ministersvo životního prostředí ČR, 2015). The Ministry of the Environment (MoE) has developed regulatory impact assessments (RIA) for the individual changes to the Water Act, on which this paper is based.

The proportion of household expenditures on water and sewerage charges is published by the Czech Statistical Office; water and sewerage charges are published by respective utilities on their websites, and the other data are collected by the Ministry of Agriculture as part of Selected Data on Property and Operating Records of Water and Sewerage Networks (VUME, VUPE). We made the micro models described below based on these publicly available data and own designs.

### 2.2.3 **Model structure and creation**

As part of the determination of the current burden on the lowest-income households, we first made an analysis of the current situation based on data on household incomes and expenditures in 2015 (Český statistický úřad, 2016). For the purpose of processing of impacts with respect to the lowest decile in the distribution of household incomes, we calculated with incomes up to CZK 6000 and up to CZK 8000 (net income per person in the household); these are amounts falling within the first decile; the income up to CZK 8000 is often in the second decile, resulting in an assessment of impacts on the first and second deciles with incomes of CZK 6000 and CZK 8000. Chart 1 shows the absolute costs of water and sewerage charges per different household income deciles at the CR level, and the share of expenditures on water and sewerage charges in the net monthly household incomes. It is clear from the data that the poorest households have the absolutely highest expenditures on water per person. The absolutely lowest expenditures on water and sewerage charges are in the 4$^{th}$ decile. The amount of expenditures continues growing in the other deciles. The costs of water and sewerage charges represent the heaviest burden for the poorest households. In the first decile, the costs of water and sewerage charges in 2015 were more than 3% of the net monthly household incomes; these expenditures were more than 2% in the second decile and less than 2% in the other deciles. However, data at the local level are not available in the same classification as the national data. Notably, they lack a division of costs by more detailed characteristics such as income deciles. The expenditures on water in the lowest household decile in each region could be determined using expert estimates and an array of data at the regional level (e.g., average expenditures and water consumption in the selected region).

**Figure 1: Household costs of water and sewerage charges by income decile in the CR in 2015**




Source: Authors using (Český statistický úřad, 2016)

The analysis of impacts of new legislation is based on micro models, made by selecting 4-5 municipalities in each of the 13 regions in the Czech Republic with the highest water and sewerage charges and including Prague as a whole. The impacts are thus calculated based on reflection of local disparities. In the South Bohemian Region, the impacts were modelled for the cities of Český Krumlov, Prachatice, Strakonice and Tábor. In total, more than 206 thousand equivalent inhabitants of the South Bohemian Region are covered with regard to wastewater treatment. The comparison uses data on household expenditures and incomes in 2015 in the South Bohemian Region and water and sewerage charges in 2015. The household expenditures were derived from average expenditures in the region and from the distribution of the expenditures by the income decile in the Czech Republic due to the lack of detailed data, The forecast calculations only include the impact of the legislative changes, and exclude any other impacts (e.g., energy price trends), and disregard the population income trend due to highly uncertain forecasts, potential oscillations of the economic cycle, primarily with respect to the lowest household decile as per income, on which these oscillations do not have a marked effect in the long run.

The impacts are reflected in the 2015 prices as an increase in each year, depending on the expected year coming into effect. We assume a reflection of 10% margin in the costs of investment and increased charges and 15% VAT for water and sewerage charges. The impact models are made so that the municipality either uses groundwater sources or surface water sources. In this connection, we assume a loss in networks and process consumption in water purification of 25.8%, meaning that 1 $m^3$ of water invoiced requires a consumption of 1.35 $m^3$ of raw water. According to the current legislation plans, the increase in groundwater consumption charges can be expected gradually from 2017. Besides, the impact assessment assumes implementation of charges for wastewater discharge from 2017. The restricted use of WWTP sludge on farmland can be expected from 2019. The stricter emission limits for wastewater discharges is expected from 2021.

The impacts were analysed using micro models separately for each city. First of all, the impact of legislation on current water and sewerage charges was calculated. Stricter emission limits were reflected through the growth in operating and/or investment costs for individual WWTP according to the current state and emission. This increase in costs connected with WWTP together with other newly established or increased charges (e.g., charges for wastewater discharge) forms the overall increase in costs and thus in water and sewerage charges. In the next step, the total increase of water and sewerage charges was reflected in the expenditure of poorest households (first decile).

Due to the increasing charges for groundwater consumption, increased charges for surface water consumption have to be considered as well. Each River Basin organisation raises its prices on an annual basis. The study also has to consider increasing prices of surface water consumption. Based on the price growth so far, we made a forecast of price trends until 2023 using a sliding average of the amounts for four previous years. The estimated amounts of charges for Povodí Vltavy excluding VAT are shown in Table 1. Including losses in networks, process consumption, profit margin and VAT, the end price will increase by CZK 0.12-1.1/$m^3$ in the different years compared to the 2015 prices, assuming 100% use of surface water.

**Table 1: Forecast of surface water consumption price increase in 2017-2023 (CZK/m3 excl. VAT)**

| Year | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 |
|---|---|---|---|---|---|---|---|
| Value | 3.77 | 3.85 | 3.93 | 4.01 | 4.09 | 4.18 | 4.27 |

Source: Authors

As concerns groundwater, the consumption charge should increase from CZK 2/$m^3$ up to the final CZK 6/$m^3$, as of from 1 January 2022. This increase will be reflected in the end price of water as an increase by CZK 1.70 in 2017, CZK 3.41 in 2019, CZK 5.11 as of

2021, and CZK 6.82 as of 2022. In municipalities with predominant groundwater consumption, there will thus be a significantly greater price increase compared to those using primarily surface water.

The impacts of the increased charges for wastewater discharge to surface water and stricter emission limits on permissible pollution have been determined based on data on current emissions, technologies and extra costs distributed as depreciation of any investment associated with attainment of required concentrations. In the South Bohemian Region, the adoption of the emission limits proposed by the MoE (Ministerstvo životního prostředí, 2016b) would require investment in reduction of total phosphorus emissions (Ptot) in all the four cities. Tábor would require the biggest investment; according to current concentrations, it would not meet emissions of total nitrogen (Ntot) and ammonia nitrogen (N-NH4+) either. The calculation included depreciation of long-term tangible assets in depreciation class 5, i.e., depreciation over 30 years, the operator/owner's profit margin, and the VAT on sewerage charges. The increase in sewerage charges with average investment in light of various emission concentration proposals is shown in Table 2.

**Table 2: Impacts on sewerage charges from year of effect of decree (CZK/m3) compared to 2015**

| City | Total expected average capital costs (CZK) | Cost in depreciation (CZK) | Cost of sewerage charge (CZK/m3), incl. margin and VAT |
|---|---|---|---|
| Český Krumlov | 33,000,750 | 1,100,025 | 1.02 |
| Prachatice | 14,152,050 | 471.735 | 2.08 |
| Strakonice | 15,220,350 | 507.345 | 0.42 |
| Tábor | 404,942,750 | 13,498,092 | 12.51 |

Source: Authors

## 2.3 Results and Discussion

The individual increases to charges should take place gradually and the changes, or impacts on prices, are therefore assumed in 2017, 2019, 2021 and 2023. These increases are compared against 2015, which is considered the initial year before the change. The four cities studied in the South Bohemian Region will see increases in the water and sewerage charges of CZK 3.63-5.82 in 2017, and CZK 10.06-22.15 in 2023. Table 3 shows the total increase in the water and sewerage charges in each city and year.

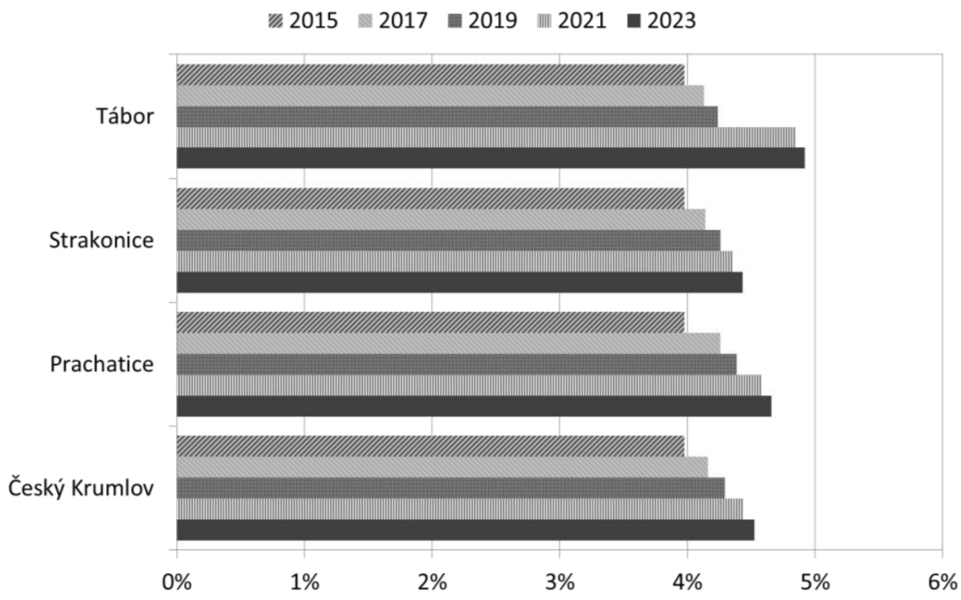
**Table 3: Water and sewerage charge increase (CZK/m3) in each year compared to 2015**

| City/year | 2017 | 2019 | 2021 | 2023 |
|---|---|---|---|---|
| Český Krumlov | 3.63 | 6.23 | 8.96 | 10.66 |
| Prachatice | 5.82 | 8.42 | 12.21 | 13.91 |
| Strakonice | 3.63 | 6.23 | 8.35 | 10.06 |
| Tábor | 3.63 | 6.23 | 20.45 | 22.15 |

Source: Authors

The increase in the water and sewerage charges due to the legislation changes can be further expressed as an increase in the share of expenditures on water in total household expenditures. The chart below shows the gradual increase in the share of costs of water and sewerage charges per household with incomes of CZK 6000 per person, i.e., households included in the first decile. Starting from 2017, we expect the implementation of a charge for groundwater consumption, a charge for surface water consumption, and a charge for wastewater discharge. Starting from 2019, we can expect an increase in the price associated with the restriction on use of wastewater treatment plant sludge on farmland. The stricter emission limits for wastewater discharges can be expected from 2021. Chart 2 shows the cost increases in the four cities of the South Bohemian Region – Tábor, Strakonice, Prachatice and Český Krumlov, selected as the cities with highest water prices in the Region. The most striking increase in the share in expenditures is seen in Tábor between 2019 and 2021; this area will thus be the most affected by the stricter emission limits, requiring an extensive adjustment to the wastewater treatment plant connected with massive capital investment costs. The impacts of the necessary adjustments will be perceptible for the following 30 years due to the depreciation period, i.e., at least for the depreciation duration. According to the analysis, the share of expenditures among the lowest-income households will not exceed 5% in any of the cities. Tábor is the closest to that threshold, with the expenditures being 4.92% of the incomes of households in the first decile. The share of expenditures on water is almost 0.5% lower in Český Krumlov and Strakonice.

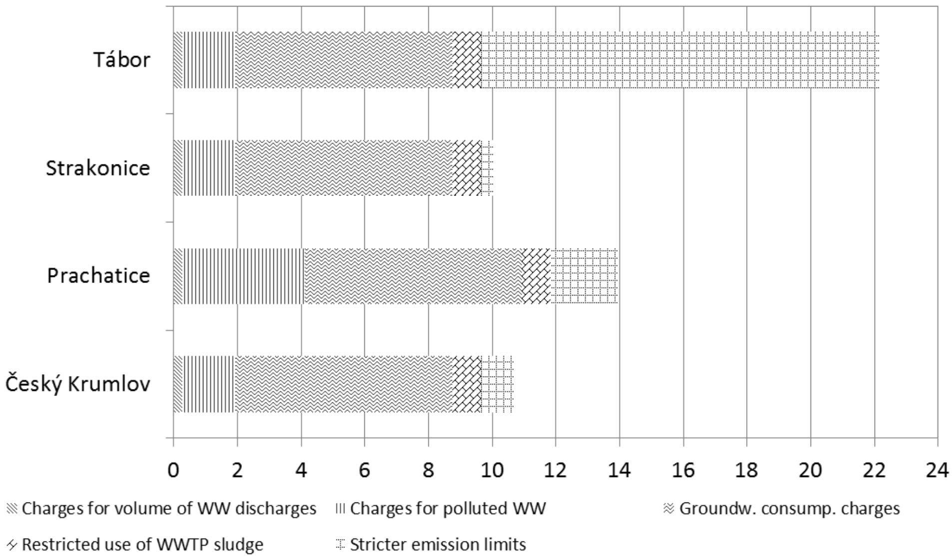
**Figure 2: Share of costs per household with incomes of CZK 6000 per person (%) for selected cities**




Source: Authors

The total increase in the water and sewerage charges should therefore be perceptible in 2023, after all the charges and legislative changes are implemented. Chart 3 below shows the individual items (charges) that will be reflected in the total water and sewerage charges in the four selected cities of the South Bohemian Region. In most of the cities, the increase in the water and sewerage charges will be associated mostly with the increase charge for groundwater consumption (from CZK 6.82/$m^3$); in the case of Tábor, it will be the investment associated with meeting emission limits for the WWTP, amounting to CZK 12.51/$m^3$. The component changes are recorded in CZK per $m^3$, including 15% VAT. On the whole, households in Tábor will pay CZK 22/$m^3$ more in 2023 compared to 2015. The other cities in the South Bohemian Region will not be affected much by the stricter limits on discharges, and the increase in the water and sewerage charges will not be as noticeable there.

**Figure 3: Expected increase in water and sewerage charges in 2023 (CZK per m3, incl. VAT)**




Source: Authors

The results presented above assume the maximum impact. Due to data availability, it was impossible to determine the exact amount of impact of the increase in the charges for surface/groundwater consumption. Municipalities using surface water as their source will not face such a high impact of the price increase. The decision on setting the emission limits will play a major role in the overall impact. If the stricter limits proposed by the Ministry of the Environment are implemented, the impact on the water and sewerage charges would be much more significant due to the failure to comply with the planned technical parameters in the majority of cases. The study assumed the average of the current proposals; the compliance is thus a question mark with a big influence on the final impacts.

## 2.4 Conclusion

The objective of the paper was to assess the impacts of legislative changes in preparation on water and sewerage charges, or on the expenditures of the lowest household decile at the local level. The analysis of the local impacts showed that the poorest households have a noticeably higher relative share of household expenditures on water than maintained by the study of the Ministry of Labour and Social Affairs. Compared to the average expenditures of 2.8% of the total expenditures in 2020 among low-income families quoted by the Ministry, the above expenditures exceed in 4% of the total expenditures of the lowest-income households in the various cities of the South Bohemian Region in 2019. The difference is partly due to including different impacts on the water prices. Besides the increase in the charge for surface/groundwater consumption, the paper also assumes other impacts, including the need for investment in wastewater treatment plants due to stricter limits on quality of water discharged.

With respect to the decision-making process, the assessment of proportionality and acceptability has to evaluate all the impacts in aggregate, with an emphasis on the lowest-income households. Due to the significant differences, it is advisable in the event of significant legislative changes to carry out an assessment of impacts at not only the national level but also the local level on at least a sample of municipalities depending on data availability. Significant differences in the water management area can be observed even within the same Region due to the different infrastructures and technical equipment of treatment plants. On the example of the South Bohemian Region, the impact on the expenditures on water differs by 0.5% of the total expenditures. Compared to the results from other municipalities, the impacts can be regarded as average with the exception of the city of Tábor. In the event of implementation of all the legislative changes, the study identified the highest increase in the water and sewerage charges in Prague, where the expenditures on water and sewerage charges would be up to 8% of the total expenditures in the lowest-income households in 2023. On the other hand, the comprehensive model identified the lowest share of expenditures on water in 2023 (3.5%) for Hradec Králové.

Based on the above results, it offers itself for discussion whether it is economically justifiable to increase the water and sewerage charges with respect to environmental impacts. Besides the direct financial effects, the assessment of the social benefits would require inclusion of additional non-financial and indirect utilities that the legislation in preparation and implementation of measures will entail. From a purely economic point of view, the price should always reflect the rarity of the good; only thus can responsible behaviour of consumers be achieved.

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# 3. Negotiating land for flood risk management: upstream-downstream in the light of economic game theory

MACHÁČ, J.; HARTMANN, T.; JÍLKOVÁ, J. 2018. Negotiating land for flood risk management – Upstream-downstream in the light of economic game theory. *Journal of Flood Risk Management*. Vol. 11(1), pp. 66-75. DOI: http://dx.doi.org/10.1111/jfr3.12317 (IF: 3.121)

Share of J. Macháč: 60%

## 3.1 Introduction

Flood protection with dikes alone is not an acceptable solution for increasing risks of river floods. In many cases the use of dikes is perfectly feasible but when used in isolation they are not the best solution. Two alternative options are: Retain floods upstream or adapt land uses downstream (resilient cities). Retention and resilience cannot substitute traditional flood protection by dikes entirely, but their value for reducing flood risk has been acknowledged in the academic debate (Hartmann, 2012) and politics (Directive 2007/60/EC). The main challenge is to implement them on land in appropriate upstream-downstream relations. Usually, flood storage options are implemented on 'cheap' land, for example, where the upstream land has a low value use such as grassland (e.g., polders on the Havel river in Germany).

The situation is more complicated if the land suitable for upstream retention is a valuable area for land uses that would be affected by occasional flooding (i.e., agriculture, settlements, etc.). But also adapting the downstream land uses to flooding can be very expensive. So, the question is on how to decide under such competitive situations between using the potential upstream (i.e., valuable areas) or adapting downstream land uses (which would cost a lot of money) (Scherer, 1990).

The trade-off between upstream-downstream has been addressed in the scholarly debate and in practice for more than 200 years (Jüpner, 2017). Most of the literature on upstream-downstream trade-offs address either technical aspects of emissions or pollutions (e.g., Groll et al., 2015), cross-border aspects (e.g., Bracken et al., 2016), or pursues a catchment perspective on governance (e.g., Rouillard et al., 2015). Fewer scholars have addressed the relationship between upstream and downstream from the aspect of land use planning and trade-off (Scherer, 1990; Hartmann, 2011; Rouillard et al., 2015; Thaler et al., 2016). Economic observations on flooding and the relation between upstream and downstream are also rare (i.e., White, 1936; Lind, 1967). Chang looked for tradable flood mitigation permits, asking the question how upstream and downstream parties can be encouraged to collaborate (Chang, 2008). The economic approach to the evaluation of costs and benefits connected with floods and flood protection is described in many papers and manuals. Among the most comprehensive ranks a new 'Multi-Coloured Manual' (Penning-Rowsell et al., 2014), which provides assessment techniques for flood risk management costs and benefits including useful data for the practical assessment, a manual of the cost benefit analysis (CBA), indirect benefits, limitations and complications of CBA, to guide decision-making etc. Specific instruments such as control-command and market based instruments and its application in flood risk management have also been discussed (Filatova, 2014). Although in all current environmental and flood protection planning process, there is a strong emphasis on stakeholder and public engagement, there is almost no discussion on opportunities to utilise negotiation between different groups of stakeholders/upstream-downstream.

In this contribution, the focus lies on the question in which scenarios do negotiations between upstream and downstream lead to what patterns of negotiations. Therefore, different scenarios are analysed with game theory. However, the paper does not provide an answer for upstream-downstream agreements, but develops and discusses the approach of game theory for those cases. It is thus a methodological contribution.

## 3.2 Traditional economic methods in water management

The most common and politically feasible approach is based on neoclassical environmental economics. It uses the cost-effectiveness analysis (CEA) that ranges possible measures depending on their effectiveness and is used for cost minimisation and the CBA that compares the measures at their costs and benefits (WATECO, 2003, Penning-Rowsell et al., 2014). Such methods are regularly applied to justify flood protection measures. However, these approaches are associated with considerable uncertainty especially in the part of the determination of benefits and economic effects (Laurans, 2006; Jensen et al., 2013). Also, the conflict over water issues is not only about costs and benefits, but 'arises from social and political aspects' (Madani, 2010, p. 255). The possibility of including stakeholder's negotiation or other dynamic elements is very limited in CBA. The negotiation could be integrated in CBA as a result of optimisation or in form of scenarios.

The solution could be either to combine CBA and CEA with institutional analysis (Ostrom et al., 1999), or to use a method for solving multi-criterial and multi-decision-maker problems such as multi-criterion decision analysis (Elshorbagy, 2006).

Ronald Coase explains that economists have in the past often followed the argument of Pigouvian theory of externalities (Pigou, 1920), who thought the question in terms of which a company A inflicts damage on a company B (i.e., polluter-pays-principle). So, restrictions are proposed to restrain company A (Coase, 1960). But who is A and who is B in our case of an upstream and a downstream party? Coase avoids blaming one party as a polluter; rather he emphasises that 'we are dealing with a problem of a reciprocal nature' (Coase, 1960, p. 1). He regards externalities as a situation of two rival opportunities.

Based on the limitations of CBA and the difficulties of applying the polluter-pays-principle, other schemes need to be found to distribute costs and allocate flood risk management measures. Game theory provides a simple method to combine CBA as an input of costs and benefits with the approach of Coase (Coase Theorem) to also discuss and display the outcomes of different scenarios (Bennett et al., 1998; Madani, 2010; Delille and Pereau, 2014). At the general level, game theory is used to identify and interpret behaviour and interaction of different parties who behave strategically. Game theory is used in location problems in planning or in sharing of natural resources or in reduction of emissions (Basaran, 2005). Floods are suited for the application of game theory because they are predominantly economic disasters; the economic values involved create incentives to look for negotiated agreements. With respect to flooding, Delille and Pereau (2014) used game theory to model the bargaining between agents over the building a seawall. Bennett et al. (1998) used the game theory to justify the international agreements and cooperation. Thus, game theory is an appropriate method that allows the researcher to experiment with certain arrangements and break with existing paradigms between upstream and downstream to predict its outcomes (Cooter and Ulen, 2004).

## 3.3 Game theory for flooding

Each game requires players, the strategies of each player, and pay-offs for each player for each strategy (Cooter and Ulen, 2004). Players in a game strive for maximising their utility and income, so that the outcome depends on choice of all groups (Luce and Raiffa, 1957). Each player pursues their interests with regard to the strategies of other players; such a situation can lead to equilibrium(s).

The size of pay-offs for a player depends on each player's behaviour (implementation of flood protection measures versus nonimplementation), his location (upstream versus downstream) and legal liability to pay for damages (Delille and Pereau, 2014). There are a couple of complexities for the game:

- Flood situations usually result from accumulated behaviour of many individuals.
- The land (both up- and downstream) might not be owned by locals, but people or organisations remote from the catchment.
- Asymmetric relationships in catchments (not individuals, but rather complex groups) need to agree on solutions.
- Identifying the beneficiary from an upstream perspective is fairly easy (except in Deltas), but for the downstream there are usually many upstream actors.

Leaving those complexities aside, it is assumed in the following game that there are only two players – one upstream and one downstream. For the following example, it is appropriate to imagine the players as two cities along a river. In initial situation, new housing projects are planned in floodplains. For the simplicity, only two types of behaviour of each part are considered. This creates a 2×2 game. Within the game, 'Upstream' can retain or accelerate floods, 'Downstream' can adapt to flooding or ignore it. This is possible if the players are empowered and able to decide upon the use of the land and if each player encompasses a cohesive area (Hartmann, 2011). Each party acts individually, simultaneously (in the basic scenario) and for its own account. It will be discussed later, if and how the results can be applied to more complicated situations.

The remaining part of the paper discusses different scenarios of arrangements between upstream and downstream. Modelling is done using various types of behaviour including negotiation and conditions (different property values of upstream-downstream, different costs of adaptation). With regard to the flood risks, it makes sense to discuss not only the nature of the game (simultaneous versus sequential; cooperative versus non-cooperative etc.), but also the setting (such as a change of legal liability for damages) and influence of length of the period (increasing occurrence of floods). This leads to a change of the nature of game. The scenarios are demonstrated on practical/hypothetical examples.

### 3.3.1 Choosing games

Based on analysis of approaches and applications of game theory, for example Bardhan (1993), Dombrowsky (2007), Madani (2010), and Hartmann (2011), in water management investigations mainly used 2×2 games. Usually every player had two options for behaviour, each of them led to a different result based on the behaviour of the second player. There are three basic types of nonsequential games which are used in water economics (2×2 games): the Prisoner's Dilemma, the Stag-Hunt, and the Chicken game (e.g., Madani, 2010). These games differ in player's strategies (existence of dominant strategy, Nash equilibrium, and Pareto-optimal outcome). The results of the games are influenced by (non-)cooperation. The basis of the Chicken game is a conflict situation in which both players have the same goal. In the event that both meet this goal, the utility of both players decreases rapidly (Colman, 1995). Usually it happens that one of the players succumb to pressure and becomes a coward or a 'chicken'. The interest of the players is also to choose the opposite option than their opponent. The other two types of the game (the Prisoner's Dilemma and the Stag-Hunt) are similar. In the Stag-Hunt game (coordination game), the interest of each player is to do exactly the same as the other player (Skyrms, 2001). In this type of the game, there are two Nash equilibria. This differs from the prisoner's dilemma (Rapoport et al., 1970), which has only one Nash equilibrium, because in this game dominant strategy exists, that means each player prefers constantly certain behaviour. A less frequently mentioned and used game is Deadlock. There are also two dominant strategies similar to the Prisoner's Dilemma, but the equilibrium represents at once the Pareto optimality, which means that there are no possibilities to make any one individual better off without making at least one individual worse off. In the Prisoner's Dilemma, the allocation in Nash equilibrium can be changed to a different that makes at least one individual better off without making any other individual worse off.

It is also possible to apply a series of sequential games. In sequential games, one of the players starts and the other reacts with his behaviour according the maximisation of his pay-off. In relation upstream-downstream, it cannot be clearly determined who would be the player, who chooses their action before the others and who is the second one, who gets some information of the first's choice. In scenarios where there is the dominant strategy by both players, there is no difference between simultaneous and sequential game. Applications games with more players or strategies or as a sequential game are offered as a further possible extension of this article. When designing a laboratory experiment it is also necessary to include this sequential type of game to test and to observe the differences in results.

Depending on the pay-off distribution, different games can be applied to the issue of flooding. Those games can be created or fostered by manipulating the pay-off matrix, for example, by introducing certain liabilities, property right assignments, or assessment criteria (e.g., appraising residential areas more valuable than, e.g., agriculture, or vice versa). Examples for such agreements are payments for ecosystem services (PES) (Kerr, 2002) or tradable development rights (TDR). Those are normative and political decisions, just depending on the distribution.

### 3.3.2 **Setting of the game**

As mentioned above, there are four types of behaviour. Each player has to decide between two options regarding the use of their part of the floodplains, whereas it is assumed that both players profit from housing projects in their own floodplain. Upstream can either build up new housing projects with high dikes to prevent the houses (i.e., 'accelerate' the flood) or withdraw from building in flood–prone areas and instead provide retention volume for the sake of Downstream ('retain' strategy). 'Accelerate' – ultimately will lead to increasing water levels downstream. Downstream chooses between realising housing projects in floodplains disregarding the threat of inundation ('ignore' strategy) or implement a risk-adapted construction for the buildings ('adapt' strategy).

If both decide to profit most individually (both players maximise their utility), the other party will not be taken into account. This condition means, if Downstream 'ignores' the flood, the housing area looks the same, regardless whether Upstream retains or accelerates the flood. However, Downstream must consider the cumulative probability $P$ of an extreme flood over a predetermined period (whereas the probability also includes a possibility that for example the 'one in hundred-year flood' occurs more often – it is in the end just a statistical probability, but this shall be discussed elsewhere). Moreover, $P$ represents a likelihood of a flood over a predetermined period, not certainty. One could imagine that in reality there is a probability of a flood for each short period, and players need to work with a cumulative probability. Flooding is a random event. The odds of occurring are independent of past occurrences (Cooley, 2006, p. 105). Downstream, therefore, is interested in the probability of an event in a period of y years (depending on the investment calculations for the housing project). If x represents the probability of a flood in a certain year, then, $(1-x)$ is the chance that this event will not take place in a given year. The odds that an event will not occur in two successive years would be $(1-x)(1-x) = (1-x)^2$. So, if $(1-x)^y$ is less than $P$, Downstream has an incentive to 'ignore'. For the centennial flood, this would mean: $(1–0.01)^y = P$. A critical length of a 'no-flood' period can be computed based on the probability assigned. According to the result, the period within which the necessary profit needs to be generated is determined. However, the outcome is not certain and decisions are made based on expected pay-offs. Also, Downstream has to consider that in the end, this is gambling with probabilities.

In the short term, the 'ignore'-strategy is very attractive, but in the long run, the cumulative probability of flooding on one or more occasions increases (precisely: the probability that a flood does not occur for a long period decreases). The longer a project needs to be profitable, the higher is the chance of a flood within the project lifetime. But the 'ignore'-strategy is often applied in practice. Housing areas, industrial areas, and further flood-sensitive land uses are often located downstream to other high-value uses, which are protected by dikes, and thus accelerate the wave: 'urban waterfronts' along the rivers, financed by credit institutes, and promoted with slogans like 'Living near the River' are typical examples. In the long-term of our simple example, the collective benefit of such allocations is zero. A flood would reduce the profit for Downstream. If Upstream decides to 'accelerate' the flood, he does not regard the effects on the Downstream. Rational individual behaviour is able to produce the most individual gain. Upstream is able to realise housing project in the whole floodplain if high embankments protect these areas. Downstream, on the other hand, gains the most if Upstream acts collectively rational despite Downstream acts individually rational. Then, cheap and extensive housing projects can be built.

A major challenge is to assess the pay-offs. Pay-offs consist of the benefits and costs of arrangements between upstream and downstream. Benefits of flood risk management are the avoided damage, whereas different definitions of damage exist (Berg, 1994); costs include opportunity costs connected with land uses of floodplains as well as costs for protection measures (investment and operational costs). The pay-offs are defined as the difference between benefits and costs and with respect to the probability of floods in the equilibrium for 'ignore/accelerate'. For simplification, all transaction costs are disregarded.

### 3.3.3 **The game: Accelerate/adapt, Retain/ignore, Retain/adapt**

Game theory distinguishes one shot games and repeated games. In case of flooding, it makes sense only to apply the principle of one shot games. Built development floodplains usually are planned to last for many years (over 50 years in the United Kingdom for commercial developments and 100 years for residential development, while the measure is usually considered permanent in the Czech Republic). For that reason, a decision binds the player for a long time. The situation may change if the players are willing to cooperate. Cooperative solutions are those which maximise the common pay-off of both players. Some of the pay-off structure prevents finding the cooperative equilibrium. The model of the game where the Nash equilibrium profit is less than it could be with the cooperation of players is called the Prisoner's Dilemma (Axelrod, 1984). Players in this game choose an action once and for all. Thus, a wise strategy is needed.

Possible players' behaviour can be combined, in case of floods four action profiles come into consideration: retain/ adapt, retain/ignore, accelerate/adapt, and accelerate/ignore. The combinations form a 2×2 matrix (Figure 1). The behaviour of the first player (Downstream) forms the rows of a matrix. The behaviour of upstream fills the columns of the matrix. Inside each matrix, there are two numbers, which represent pay-offs for each player depending on the behaviour of both players. The right one (capital letter) belongs to Upstream and the left one (lower case) to Downstream. Each player prefers higher pay-offs.

**Figure 1: Example of the game**

| | | **Upstream** | |
|---|---|---|---|
| | | Retain | Accelerate |
| **Downstream** | Adapt | **a** / **A** | **b** / **B** |
| | Ignore | **c** / **C** | **d x (1-*P*)** / **D** |

Source: Own construction

Based on definition of the players' behaviour, there is partial asymmetry regarding the dependencies, since the Upstream's actions fully affect downstream while Downstream's actions have no effect on Upstream. According to the conditions of scenarios the asymmetry will be changed.

**Figure 2: Flooding game payoffs**

| | | **Upstream** | | | |
|---|---|---|---|---|---|
| | | Retain | | Accelerate | |
| **Downstream** | Adapt | **6** | **4** | **6** | **9** |
| | Ignore | **10** | **4** | **10 x (1-*P*)** | **9** |

Source: Own construction

Figure 2 shows an arithmetic example of a pay-off matrix, considering the conditions above. The pay-offs are a monetary gain (e.g., millions of Euros). Just for the simplification of the comparability, simple values are assumed. Upstream gains 4 in case of ‘retain’ and 9 in case of ‘accelerate’. Downstream earns 6 in the case of ‘adapt’. Following the asymmetry mentioned above, the gain for Upstream is 9, if Upstream ‘accelerates’ the flood-wave, regardless whether the Downstream ‘adapts’ or ‘ignores’. On the contrary, for the same reason, a rational decision of Downstream in a short run leads to 10 in the combination ‘ignore/retain’ and $10*(1-P)$ in case of ‘ignore/accelerate’, but in a long run, Downstream’s profit decreases to a nearly 0 in the case of combination ‘ignore/accelerate’ (because $P$ becomes almost 1). The asymmetry explains why the particular maximum of 10/9 can be achieved by rational decisions, whereas only Downstream takes risk of losing pay-offs. Such situations can be observed in practice. In the Netherlands, there is discussion of designs for houses and even greenhouses that float on water and rise and fall as a flood passes. Thus, the Dutch try to gain as much as possible by a risk-adapted behaviour. Germany, France, and Switzerland are the upstream parties. Some similar cases can be found in other catchment, for example, in the Czech Republic in the catchment of rivers Berounka and Vltava. The following combinations of strategies can be played:

*Accelerate/adapt*

For Downstream, it would be most profitable if Upstream pursues ‘retain’. For Upstream, however, it is most tempting to act individually rational as well. Then, however, Downstream’s profits depend strongly on the time period that is considered in the economic assessment. So, if Upstream indeed ‘accelerates’, Downstream should ‘adapt’ in order to achieve at least a profit of six if he thinks that the flood is coming with probability of at least 0.4. The highest economic welfare of the whole catchment then achieves a pay-off of 15. In case of Germany, France, and Switzerland, retention takes place to some extent, but as a whole, these densely settled upstream parties accelerate flood waves and force the Dutch to adapt their housing projects (whereas it has to be admitted that the adaptive strategy of the Dutch is also owed to sea level rises, not only to river floods).

Accelerate/Ignore: If Upstream accelerates, the ‘ignore’ strategy pays off for Downstream if he views a probability of the flood as less than 0.4. In that case, namely, the payoff is $10 \times (1-0.4) = 6$, which is equal to the strategy ‘adapt’. Downstream then becomes indifferent between the two strategies and strictly prefers ‘ignore’ if he thinks the probability is below 0.4.

*Retain/ignore*

The combination ‘retain/ignore’ achieves the maximal gain for Downstream. However, this opportunity will dissatisfy Upstream, because he carries all the burdens and Downstream gets all benefit. This combination will only result in a situation with a very strong downstream party, which has the opportunity to control or at least influence Upstream extraordinarily. Probably, Upstream and Downstream are within the same administrative borders, and the decision power is with the downstream party. Within the arithmetic example, this combination reaches the second best collective gain, namely 14.

*Retain/adapt*

It achieves a common profit of 10. From the perspective of efficient allocation, this combination of strategies is not preferable. This is a consequence of implementing both retain and adapt measures when only one of them would be sufficient. It is a result of lack of cooperation.

Finally, in a theoretical world without liability or other legal framings, the combination ‘accelerate/adapt’ is predicted in long run (with probability of floods $P > 0.4$). The most probable strategies are highlighted in grey in the pay-off matrix. The combination ‘accelerate/ ignore’ emerges if short-term profits dominate decision-makers.

In short term with probability of floods lower than 0.4, the game is similar in structure to Deadlock. Both players have a dominant strategy. Upstream prefers ‘accelerate’- strategy and downstream ‘ignore’. It follows that equilibrium is located in the combination ‘accelerate/ignore’. This situation is in contrast to prisoner’s dilemma, because the equilibrium results in Pareto optimality. It is impossible to make one player better off without making the other one worse off.

In case of $P > 0.4$, Downstream loses the dominant strategy. Downstream prefers ‘ignore’ if upstream retains and ‘adapt’ if upstream accelerates. The dominant strategy of Upstream is maintained. The game has a Nash equilibrium ‘accelerate/adapt’ in pure strategy for given value of $P$. In this case, the game cannot be likened to any of the basic types.

## 3.4 Playing with different types of games

In our example above, we had only two parties, when introducing many more, almost every party is both an upstream and a downstream – so each has the incentive to ‘accelerate’ and ‘ignore’. Hartmann (2011) describes this situation as one of ‘clumsy floodplains’. The economically best result for the whole catchment is unlikely to happen. Starting from the flooding game, we modify the rules of the game to see how redistributions of gains and losses may generate an economic more efficient allocation in the catchment area.

### 3.4.1 Scenario 1: introducing upstream liability rule

Assume an authority decides against the reckless Upstream who is affecting Downstream by accelerating the flood. From now, Upstream has to compensate Downstream for the losses. The distribution in Figure 3 is the result (the right column changes). In the case ‘accelerate/adapt’, Downstream claims a pay-off of 10. The remaining five are for Upstream. In the case of ‘ignore/accelerate’, the liability takes away the risk from Downstream. The risk is transferred to Upstream, who has now to estimate the risk. In long-term, Upstream prefers ‘accelerate’ when Downstream plays ‘.adapt’ and ‘retain’ if Downstream prefers ‘ignore’. The liability has another implication: the compensation of Downstream’s losses through the liability rule deletes disadvantages of building in the floodplains. There is no economical reason for Downstream to reduce damage. The risk of flooding has no impact on allocation decisions, Downstream has an incentive to accumulate values, because Upstream takes the risk. Downstream has in this manner an incentive to waste resources, which is inefficient (Baumol and Oates, 1988). The asymmetry is changed. Now, the Downstream has an advantage. Based on the $P$, in case of high probability ($P > 0.56$) the situation leads to ignore/retain’ and in case of $P < 0.56$ to ignore/accelerate. The pay-offs in adapt/retain’ offers possibility of negotiation.

**Figure 3: Introducing Upstream-liability-rule**

| | | Upstream | | | |
|---|---|---|---|---|---|
| | | Retain | | Accelerate | |
| Downstream | Adapt | | **4** | | ~~9~~ **5** |
| | | **6** | | ~~6~~ **10** | |
| | Ignore | | **4** | | **9 x (1-*P* )** |
| | | **10** | | **10** ~~x (1-*P* )~~ | |

Source: Own construction

Ronald Coase describes the problem of indifference of players whether to be compensated for the losses or receiving income from certain goods (Coase, 1960, p. 15). This liability thus creates moral hazards.

**Figure 4: Introducing Upstream-liability-rule after the negotiation**

| | | Upstream | | | |
|---|---|---|---|---|---|
| | | Retain | | Accelerate | |
| Downstream | Adapt | | **4** | | ~~5~~ **4.5** |
| | | **6** | | ~~10~~ **10.5** | |
| | Ignore | | **4** | | **9 x (1-*P* )** |
| | | **10** | | **10** ~~x (1-*P* )~~ | |

Source: Own construction

In case of high value of *P* Upstream can offer a payment to Downstream for pursuing ‘adapt’ instead of ‘ignore’, and pursues himself the strategy ‘accelerate’ (Figure 4). Upstream could offer a payment of 0.5 of the original 5 to attract Downstream with the highest pay-off in the matrix (10.5) for ‘adapt’. This makes Downstream to play ‘adapt’ over ‘ignore’ when Upstream plays ‘accelerate’. Initial payment for Downstream was not sufficient. Therefore, Upstream needs to pay additional 0.5 to make the action ‘adapt’ attractive. Then, no damages happen and in sum, the catchment yields a benefit of 15. The most efficient allocation is achieved.

### 3.4.2 Scenario 2: introducing downstream liability rules

What possibilities does Downstream have without the liability rule? Ronald Coase shows that the allocation of resources will be the same. The allocation depends on the benefit and the costs of damage. If the benefit is bigger than the damage, the firm accepts costs of liability the victims are not able to pay the firm off (Coase, 1960).

**Figure 5: Downstream pays Upstream**

| | | **Upstream** | |
|---|---|---|---|
| | | Retain | Accelerate |
| **Downstream** | Adapt | **4**<br>**6** | **9**<br>**6** |
| | Ignore | ~~4~~ **9**<br>~~10~~ **5** | **9**<br>**10 x (1-*P*)** |

Source: Own construction

The pay-off matrix changes (Figure 5): Downstream would pay Upstream for pursuing the ‘retain’ strategy in order to stay in ‘ignore’. Upstream will only agree if he is at least not worst off with this option than with the other options. So Upstream agrees on every offer that assigns at least a pay-off of 9 for him. This implies that a pay-off of 5 remains in the combination ‘ignore/retain’ for Downstream, the payment is about 5. However, this is not a Nash equilibrium. Under these circumstances, Downstream has a dominant strategy ‘adapt’ in a long run. It earns 6 no matter what Upstream does. In this situation, Upstream prefers strategy ‘accelerate’, which brings him pay-off of 9. However, if Downstream considers only short-term profits and estimates $P$ lower than 0.4, there is no Nash equilibrium.

Outcomes in a long run are the same no matter who is responsible for flood protection. It corresponds with Coase’s theorem and Coase’s allocation neutrality (Coase, 1960).

Whether the conclusions of Ronald Coase are transferable to the Upstream-Downstream case, depends very much on the estimation of the *P*. A sustainable treatment of the situation, however, regarding long-term effects, and in long terms, *P* increases. Compared to the previous scenario, upstream achieves higher profits in equilibrium.

### 3.4.3 Scenario 3: valuable upstream

The case will be different if Upstream and Downstream are not equal in their abilities to gain profit from new housing projects. To realise the potential trade-offs, it needs to be demonstrated that the gains are great enough to make it worthwhile to overcome obstacles. The costs of flood damage mitigation in urban areas are (usually) high, whereas costs of flood mitigation measures in rural areas are (usually) relatively low. Imagine one party yields more land rent (because of better infrastructure, better marketing, better conditions for building etc.). How will the parties distribute gains and losses, which allocation results?

**Figure 6: Valuable Upstream**

| | | **Upstream** | |
|---|---|---|---|
| | | Retain | Accelerate |
| **Downstream** | Adapt | **6**; **2** | **6**; **10** |
| | Ignore | **10**; **2** | **10 x (1-*P*)**; **10** |

Source: Own construction

Figure 6 shows a situation of an Upstream, which yields more benefit from housing projects than Downstream. Upstream yields now a pay-off of 10 maximum in the ‘accelerate’ strategy; in the ‘retain’ strategy is able to achieve only 2 based on the higher value of the housing projects in Upstream. This is a result of higher opportunity costs. In this situation, Downstream has no bargaining power to convince Upstream not to play ‘accelerate’. In case of lower probability ($P < 0.4$) both players have a dominant strategy. Upstream prefers ‘accelerate’ and downstream ‘ignore’. This game corresponds to Deadlock with result ‘accelerate/ignore’. In long term, ($P > 0.4$) downstream loses the dominant strategy and would avoid the loss caused by floods. New Nash equilibrium is achieved in the combination of 'accelerate/adapt'. This situation is Pareto efficient with the highest possible social welfare pay-offs (16).

### 3.4.4 Scenario 4: valuable downstream

Vice versa, if Downstream yields higher pay-off from the housing projects, like in Figure 7, Downstream profits 11 maximum with the strategy 'ignore'; 'adapt' yields even less pay-off. Downstream can realise housing areas and negotiate with Upstream about the costs for the 'retain' strategy. Before starting negotiations, equilibrium is in case of lower probability (P < 0.64) in situation 'accelerate/ ignore' based on dominant strategies of both players and type of game Deadlock. In long run, new Nash equilibrium is located in 'accelerate/adapt'. The situation changes in case of negotiation. The highest social benefits is connected with the situation 'retain/ignore'. If the Downstream pays more than 5 (e.g., 6) to Upstream the equilibrium moves to the 'retain/ignore'. Both players reach the higher pay-offs than without negotiation. Both of them profit 1.

**Figure 7: Valuable Downstream**

| | | **Upstream** | |
|---|---|---|---|
| | | Retain | Accelerate |
| **Downstream** | Adapt | **4**<br>**4** | **9**<br>**4** |
| | Ignore | **4**<br>**11** | **9**<br>**11 x (1-*P*)** |

Source: Own construction

## 3.5 Discussion and conclusion

Based on the scenarios presented above, it is not possible to create a universal game solving all the supposed settings. Given that transaction costs are ignored and property rights are determined, the model confirms the allocation neutrality in negotiation. The original rights allocation affects only transfer of wealth (distributional aspect). Negotiation constitutes an important role in the issue of floods. In all cases there was a significant shift in the situation due to possibilities of negotiation. The total pay-off increased using negotiation. Within the modelling, it is possible to solve the situation within transfer of payment as a reduction of money which receives the recipient. In the context of the real world this problem should be solved as pressure to reduce transaction costs. In this regard, the State can contribute, for example through policy, by defining (property) rights and their enforcement.

Among the constraints of the discussed approach are that the probability of floods and risk perception can have a significant influence on the outcomes of the games. Also moral hazard or free-riding have not been considered.

Another important aspect of the above games is the assessment of the costs and benefits, because this is part of a political and normative process. This also incorporates the rather difficult aspect of potential benefits and costs as consequences of particular measures, that is, the question becomes difficult when asking if a party realise a certain benefit because of some measure or if an existing flood protection level inherently leads to certain benefits (White, 1936).

In the real world, where more than one upstream party might provide retention areas, the payments would be a matter of negotiations. We can derive the general conclusion: either find a less-valuable upstream, which you can convince by payments to retain floods, or offer a valuable downstream retention volume for an appropriate payment. In short: pay or swim! This of course, raises interesting issues regarding the notion of justice in flood risk management in practice (Thaler and Hartmann, 2016).

In any real major river catchment the removal of a small volume of storage for a single urban development (e.g., a few km$^2$), has barely measurable influence on downstream flood levels. In our game, we consider only one player in the upstream and one in the downstream for simplicity. Our assumption is a significant impact on downstream correspond to the cumulative impact of multiple floodplains in the upstream. In practice, from the point of view of cities in downstream it would be necessary to negotiate with more cities in the upstream to achieve significant influence.

So, what can we learn from the economic analysis of upstream-downstream relations in the flooding games presented above? The game theory can help to set effective incentives for flood management. Finally, game theory, as discussed above, can help to decide where to take action in catchments – upstream or downstream. The constraints discussed above show that game theory can only contribute one piece for decision. However, as floods have – at least in developed countries – predominantly financial damage (or damage that is relatively easy to monetise, as insurance communities show), this economic approach can be a valuable tool. Such games are not solely applicable to flood risk management, but also to similar problems which are based on arrangements and agreements between landowners within river basin areas. Notably the games for increasing water quality and reducing pollution are similar and in many cases solve the same problem of allocation of measures.

The above discussion excluded the complexity and institutional framing from real-world examples to illustrate basic principles underlying possible negotiations between upstream and downstream on land for flood risk management. To some extent these constraints the applicability, because institutions influence the game substantially, transactions costs are high and political issues of (i.e., across borders) change the setting. Nonetheless, the discussion above also reveals basic economic arguments underlying the layers of complexity that have been disregarded here. It makes explicit to discuss how flood risk management distorts or works with market mechanisms (as simplistic as they are). But the real advantage of using game theory for flooding is not depicting the costs and benefits and making informed decisions on allocation (actually other methods such as CBA might indeed be better suited for this); the benefit of game theory is that it enables experimenting with certain rules such as liabilities, responsibilities, and property right assignments. This is to understand (or even predict) outcomes of negotiations between upstream and downstream under certain regulatory regimes. This can ultimately contribute to better land and water governance for retention and resilience on a catchment scale. The approaches explored in this paper need to be further empirically tested on real-world examples and cases.

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# 4. Using Bayesian Networks to Assess Effectiveness of Phosphorus Abatement Measures Under the WFD

BRABEC, J; MACHÁČ, J.; JÍLKOVÁ, J. 2017 - in review. Using Bayesian Networks to Assess Effectiveness of Phosphorus Abatement Measures Under the WFD. *Hydrological Sciences Journal* (IF 2016: 2.222)

Share of J. Macháč: 40%

## 4.1 Introduction

There is only little doubt that eutrophication causes accelerated growth of harmful algal blooms, whether the nutrient enrichment comes from sewage or agricultural runoff (e.g., Anderson, Glibert & Burkholder, 2002). In a reaction to increasing concerns about water quality, European Commission introduced Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy (2000; Water Framework Directive, abbreviated to WFD). The directive introduced a "good status" and required all water bodies to achieve it by 2015. The "good status" ensures the water body departs only slightly from the biological community that would be expected under conditions of minimal anthropogenic impact, as described in Annex V of the WFD. The "good status" consists of ecological status and chemical status – each based on multiple criteria. The WFD applies the 'one-out, all-out' principle in assessing the "good status". Failing to satisfy just one of the indicators means the "good status" is not achieved. For this reason, the majority of European water bodies did not meet the required standard by 2015, mainly because of eutrophication. However, failing to reach the "good status" might not mean violating the regulation. Water bodies may apply for an exemption if, e.g., costs of achieving the "good status" exceed generated benefits (the WFD does not state how large the gap needs to be). The application must be supported by economic analysis of costs and benefits. If the exemption is approved, the deadline can be extended to 2021/2027 or a lower target can be set.

Numerous methodologies have been created in European countries to tackle cost proportionality of achieving the "good status" (e.g., Galioto, Marconi, Raggi & Viaggi, 2013; Klauer, Sigel, Schiller, Hagemann & Kern, 2015; Slavíková, Vojáček, Macháč, Hekrle & Ansorge, 2015). These approaches use cost-benefit analysis (CBA) or criteria to assess proportionality of measure implementation. Martin-Ortega et al. (2015) use hydro-chemical models to simulate effectiveness of phosphorus mitigation measures, but most of the time effectiveness of measures is simply assumed. Designed measures are expected to reduce exactly the desired amount of the problematic substance. Robustness is often tested by sensitivity analysis of costs and benefits, but testing of the total effect is rather scarce. Effectiveness of a certain measure type is predetermined and constant, regardless of local conditions. This can hardly be true as not all the measures are identical and are implemented under heterogeneous conditions. This means the reduction target might not be met and consequently, estimated benefits may be unrealistically high.

The main goal of this paper is to deal with uncertainty of measure effectiveness in the context of the disproportionality analysis and to make all the approaches based on CBA more accurate. The idea is to apply the concept of Bayesian networks prior to the economic analysis to find out whether the "good status" can be achieved using the selected measures. Unlike a standard CBA, Bayesian analysis does not automatically assume that the target "good status" in this case) will be met. Instead, Bayesian analysis reports probability of achieving the target, which should precede the CBA. Knowledge of this probability is important as the generated benefits would undoubtedly be lower if the target was not met. This approach increases our awareness about possible outcomes of measure implementation and may be used to make the CBA more precise. We demonstrate the approach on a case study of Stanovice water reservoir.

The paper consists of seven chapters and is structured as follows. In the next chapter, we shortly introduce the concept of Bayesian networks, then we provide some examples of their utilization in water management. The main part of the paper demonstrates the approach on a case study of Stanovice water reservoir, where phosphorus abatement measures are to be implemented to satisfy the WFD requirement of the "good status". The results are presented in the next chapter. Discussions and conclusions follow.

## 4.2 Bayesian Networks

Heckerman (2008) describes a Bayesian network as 'a graphical model for probabilistic relationships among a set of variables'. These are an integral part of influence diagrams, which also include utility nodes and decision nodes. He provides rigorous derivation of the approach, including distribution analysis, inference, or learning structure and parameters.

The Bayesian view of probability differs significantly from the classic approach. Heckerman (2016) describes the frequentist approach as analysing the probability of seeing specific data given a hypothesis – P(D|H). On the contrary, the Bayesian approach considers the probability of a hypothesis being true given the data – P(H|D). While classic probability is a true state of the world and is set, Bayesian probability is much closer to a person's degree of belief. However, it still satisfies the rules of probability and is not necessarily subjective. As shown by Briggs (1999), the Bayesian approach based on empirical data is the same as the frequentist approach that uses pooling available data, and Bayesian analysis based on uninformative prior distribution is similar to frequentist analysis based on observed data. Only when expert opinion is used to form a prior, Bayesian analysis may be viewed as subjective.

Heckerman (2008) shows that we can learn about parameters after defining variables, assigning priors and using Bayes' rule (Figure 1), which updates beliefs about the parameters of given data.

**Equation 1: Bayes' rule**

$$P(H|D) = \frac{P(H) * P(D|H)}{P(D)}$$

Source: Heckerman (2008)

All the terms in the equation above also depend on our belief. P(H|belief) is called a prior, P(D|H, belief) likelihood and P(H|D, belief) a posterior.

Bayesian networks are heavily used in different fields, e.g., medical diagnosis (Heckerman, Horvitz & Nathwani, 1992) or manufacturing control (Nadi, Agogino & Hodges, 1991). In water management, Bayesian networks can be used to implement uncertainty about effectiveness of measures of the same type. Probability of achieving the target is obtained as a result of this analysis. Outputs of such analysis are more robust and can be used by policy makers. Compared to a deterministic analysis, the results are probabilistic and do not provide a simple yes/no answer. Nevertheless, knowing whether the "good status" can be reached with the selected measures is crucial in proportionality analysis.

## 4.3 Application in water management

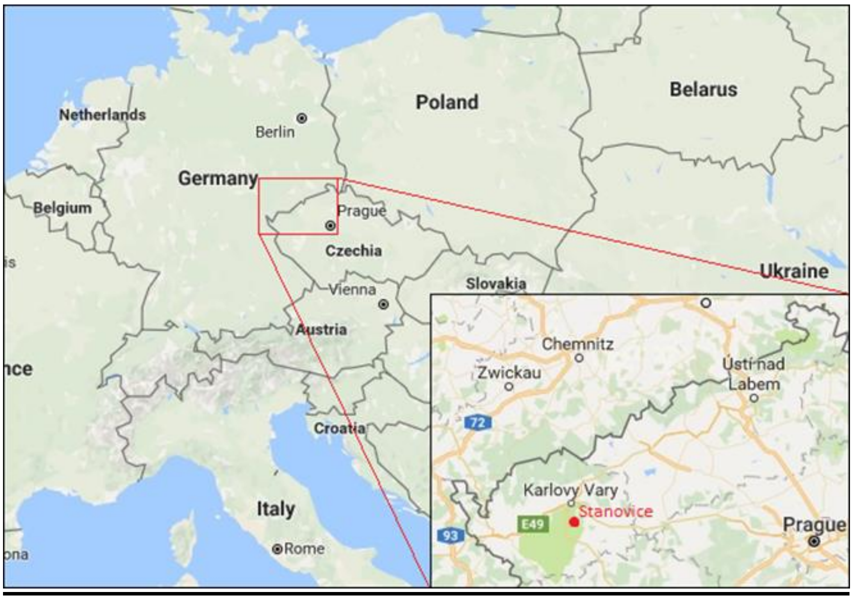
Bayesian networks are becoming increasingly popular in environmental sciences. Hamilton et al. (2007) use them to confirm the relationship between nutrient levels and algal bloom initiation. They find the probability goes up when nutrient inflows from land runoff and point sources increase. This approach provides a useful way of assessing probability of meeting a given goal, which is not restricted to phosphorus abatement. This section introduces previous utilization of Bayesian networks in water management.

Ames, Neilson, Stevens & Lall (2005) provide a guideline for authors who plan to use Bayesian analysis in watershed management. They focus on defining the problem as a graphical structure, which includes identification of management endpoints, alternatives, data sources, variables and their discretization. They show how to build conditional probability tables (based on observed data, a simulation model or expert judgement) and demonstrate the approach on phosphorus management in the East Canyon watershed in Utah. More examples of Bayesian networks application can be found in literature (e.g., Barton, Saloranta, Moe, Eggestad & Kuikka, 2008; Borsuk, Stow & Reckhow, 2004; Fernandes et al., 2012; Helle, Vanhatalo, Rahikainen, Mäntyniemi & Kuikka, 2012; Lehikoinen et al., 2014; Maldonado, Aguilera & Salmerón, 2016; Moe, Haande & Couture, 2016; Ticehurst, Newham, Rissik, Letcher & Jakeman, 2007). The introduction of the WFD is the main reason why the approach is becoming more popular in uncertainty analysis of water management options as all of the mentioned studies past 2010 are applied to European water bodies affected by the WFD. The papers present networks of various complexity, ranging from a very simple model used by Fernandes et al. (2012) to a complex network consisting of 35 nodes by Borsuk et al. (2004). Some papers focus solely on satisfying a given limit (e.g., Ames et al., 2005), while others model sustainability of several coastal lake-catchment systems (Ticehurst et al., 2007). Studies also differ in input data as Bayesian networks are not limited to empirical inputs. These are prevalent, but they are often complemented by use of expert judgement (e.g., Barton et al., 2008), regression or process-based models (e.g., Borsuk et al., 2004) or dynamic models such as MyLake (Moe et al., 2016). Collected data are discretized in most of the papers, but Maldonado et al. (2016) develop a regression-oriented Tree Augmented Naive Bayes, which can work with continuous variables and allows for dependence among explanatory variables. Authors present risk maps of exceeding the nitrogen limit. Results of the analyses show that achieving the "good status" will be very challenging in some water bodies. Barton et al. (2008) find that the chance of breaking the regulation is still 44% for phosphorus if all the measures are implemented (0% in the deterministic case) and the net benefits become negative when uncertainty is considered. Moe et al. (2016) conclude it is virtually impossible for Lake Vansjø to reach the "good status", even under the "best case" management. It is the result of physico-chemical criteria as the probability of meeting the phosphorus concentration limit is close to zero. Fernandes et al. (2012) focus on coastal waters and claim that reduction in nutrient loadings is needed as the most probable status in several criteria is moderate and the probability of satisfying chlorophyll-a is 9-59% depending on the area. Borsuk et al. (2004) also find the decrease in chlorophyll concentration is only minor when nitrogen inputs are halved. Lehikoinen et al. (2014) also focus on coastal waters and find it depends on how the "good status" is defined. It can be reached with reasonable probability when the 'averaging rule' is applied (Finnish approach), but it is almost impossible to do so when the 'one out-all out' rule applies. Ames et al. (2005) state the probability of not meeting the criterion can be lowered significantly, but implementation of the measures results in negative total benefits. The remaining European studies also find achievement of the "good status" challenging. Authors of the above studies also mention various advantages and drawbacks of the Bayesian approach (see Barton et al., 2008, for a thorough discussion). Borsuk et al. (2004) state that Bayesian networks more realistically represent current knowledge about the system and can be updated without distorting the rest of the model thanks to conditional independencies. On the other hand, the network is unable to represent system feedbacks because of one-way causal relationships in the model and it is hard to test whether the model specification is correct. Ames et al. (2005) appreciate potential use of expert judgement and Lehikoinen et al. (2014) stress that the approach is easy to use and that we can test the 'what if' type of questions. Fernandes et al. (2012) argue Bayesian networks are not suitable for environmental data, but it corresponds with WFD logic as we are interested in whether the limits are satisfied, not by how much. Like many other authors, they also mention a drawback of information loss because of discretization.

## 4.4 Case study: Stanovice water reservoir

Stanovice reservoir is situated in the Czech Republic next to the city of Karlovy Vary in North-West Bohemia as indicated in Figure 1. The catchment area covers 92 $km^2$. According to Povodí Ohře (http://www.poh.cz), its main purpose is to supply the Karlovy Vary area with drinking water. Minor functions include fishery, flood protection and electricity generation.

**Figure 1: Location of Stanovice**




Source: Own construction using Google Maps

Based on Povodí Ohře (2009), Stanovice fails to reach the "good status", mainly because of anthropogenic effects in the surrounding area (e.g., agriculture, population). The whole area suffers from cyanobacterial growths in summer as a result of excessive inflows of dissolved phosphorus. Point sources and diffuse sources contribute evenly to the total inflows. T. G. Masaryk Water Research Institute estimated that 60-200 kg of dissolved phosphorus need to be reduced annually to reach the "good status", and we work with the upper bound. A total of 243 possible measures in several categories were identified. Both measures on point sources[8] and agricultural measures are viable and described by Dostál and Krása (2014) and Ansorge and Drozd (2014). In total, the measures were designed to reduce 344.6 kg of dissolved phosphorus. Annualized costs and effects were estimated for each measure, and the measures were later ranked based on their cost-effectiveness. Authors use dynamic cost-effectiveness analysis (Macháč & Slavíková, 2016). Adding up, the selected measures are believed to reduce phosphorus inflows by 200.04 kg each year and ensure the "good status" is achieved. The most cost-effective combination of measures consists of broad-base terraces (BBT), vegetated filter strips (VFS), changes in crop rotation (CCR), leaving crop residue (LCR), no-tillage methods (NTM), WWTP and

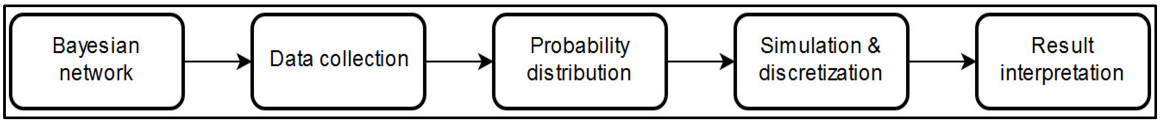
[8] Construction and renovation of wastewater treatment plants, sewerage systems, dead-end and accumulation cesspits, retention wetlands, biological reservoirs, domestic wastewater treatment plants, intensification of the treatment process at wastewater treatment plants.

sewerage system (SW) construction and clearance of a biological pond. The desired threshold can be reached by implementation of 99 measures with the total annualized costs of EUR 43,233. Five of the selected measures are on point sources (especially the construction of WWTPs). Despite a small number of measures on point sources, they contribute to the total reduction by 62%.

### 4.4.1 Methodology

The approach used in the paper is illustrated in Figure 2. The analysis begins with construction of the Bayesian network. It is important to decide what variables enter the model and to define causalities among them. This is followed by data collection. Most importantly, information about effectiveness of individual measures are collected and assigned to the measure types described above. The approach is heavily reliant on a large dataset, as probability distributions for each measure type are estimated in the next step. The distribution estimate is reliable only if a sufficient number of observations has been collected. If the dataset is large, it is possible to skip this step straight to discretization (e.g., Ames et al., 2005, use historical records to estimate probabilities of streamflow and phosphorus concentration). In this paper, not enough data for discretization were collected. Alternatively, it is possible to use the data to estimate a probability distribution for each measure type and simulate a large number of values, which are later discretized into intervals (e.g., Lehikoinen et al., 2014, use simulation models to learn about conditional probabilities and dependencies). It is preferred to use a small number of intervals, which can correspond to some set limits (reduction target in this case). Probabilities of falling into a specific interval then enter the previously designed Bayesian network. Given these attributes and dependencies among variables, software simulates many different scenarios and gives an outcome in terms of a probability that a scenario falls within the desired interval. The final step is to interpret the results, which is not trivial due to their probabilistic nature. It is important to consider how close to 100% one wants to get. Overall, the analysis is quite straightforward as we do not attempt to model the whole ecosystem and all indicators of the “good status”, but only reduction of the phosphorus that we know is present at a given source.

**Figure 2: Methodology**




Source: Own construction

### 4.4.2 **Data**

Data used in the analysis come from different sources. As stated above, a significant number of observations is required for the analysis. However, no data about the agricultural measures that would compare the situation prior to and after measure implementation are available in the Czech Republic. Therefore, results of empirical experiments from literature were used to collect enough observations to estimate probability distributions (e.g., Abu-Zreig, Rudra, Whiteley, Lalonde & Kaushik, 2003). Despite using the available literature, our samples for Changes in crop rotation and Broad-base terraces remain unsatisfactory and are not included in the analysis. As for WWTPs, municipalities where a WWTP was built or intensified recently were approached. Although the municipalities often differ in size from the ones where the measures will be applied, it gives us a decent idea about their effectiveness and whether they comply with their respective limits. A similar approach was used for construction of a WWTP and a sewerage system. Data collection about recently built WWTPs (last 6 years) was carried out to get a satisfactory dataset (the sample consists of more than 100 sewerage and WWTP construction projects, including reconstruction and intensification). Unfortunately, the last measure on point sources (biological ponds) is quite unique in the Czech Republic and there are no data available about its effectiveness. In such cases, expert judgement may be used to form a triangular distribution as suggested by Barton et al. (2008), or a uniform distribution can be used if the parameters cannot be estimated. However, we want to keep this case study as empirical as possible and we discard this measure. Table 1 presents characteristics of all the measure types. Only five of them enter the analysis (the ones in italic). It is much more difficult to achieve the “good status” with fewer measures and this omission is dealt with later by adjusting the reduction target.

**Table 1: Selected measures**

| Measure type | Number of selected measures | Reduction from selected measures (kg) | Expected effectiveness of selected measures |
|---|---|---|---|
| Broad-base terraces (BBT) | 4 | 6.03 | 70% |
| *Vegetated filter strips (VFS)* | 34 | 26.45 | 60-75% |
| Changes in crop rotation (CCR) | 16 | 11.79 | 50% |
| *Leaving crop residue (LCR)* | 13 | 5.67 | 30% |
| *No-tillage methods (NTM)* | 27 | 26.36 | 60% |
| *WWTP construction* | 3 | 64.44 | 100% |
| *WWTP + sewerage system (SW) construction* | 1 | 25.90 | 100% |
| Clearance of a biological pond | 1 | 33.40 | 100% |
| **Total** | **99** | **200.04** | |

Source: Own construction

The analysis begins with converting the collected data on effectiveness to absolute values in terms of phosphorus reduced. For agricultural measures, each observation (effectiveness) was multiplied by the total amount of phosphorus present at the area where corresponding measures from the same group are to be applied. In the case of WWTPs, data on targeted phosphorus concentration and real inflow and outflow concentrations were used to determine WWTP effectiveness. Higher (lower) amounts of phosphorus than expected can be reduced using each measure type. For agricultural measures, the real effectiveness needs to exceed (fall short of) the expected one indicated in Table 1. For WWTPs, phosphorus concentration in outflow water needs to be lower (higher) than the target. This gives us the observations in absolute values (kilograms reduced). To avoid problems with overrepresentation of one WWTP in the sample (especially from larger municipalities), 10 observations from each WWTP were chosen randomly. This issue is still present in the case of agricultural measures, but is less of a problem. Many more agricultural measures are planned and they will be implemented under slightly heterogeneous conditions. Moreover, the dataset is still rather sparse in some cases (no-tillage methods; leaving crop residue).

### 4.4.3 Analysis

The dataset was tested for outliers, which were excluded from the analysis. Using the remaining observations, a probability distribution was estimated for each measure type. We need to be careful as we have an insufficient number of observations for two of the variables. Table 2 summarizes the process, where abbreviations represent the different measure types (VFS – vegetated filter strips; NTM – no-tillage method; LCR – leaving crop residue; SW – sewerage system). While LCR, NTM, WWTP and WWTP+SW follow a normal distribution, VFS did not fit in any common distribution family and had to be transformed to follow a normal distribution. Based on their attributes, 1000 values were simulated for each measure type. As it is unlikely that any measure would lead to a worsening of the current state, all negative values were replaced with zeros. Similarly, if the observation suggested more than the total amount of present phosphorus was reduced, the maximum possible value was used instead. Descriptive statistics for the simulated values are also captured in Table 2.

**Table 2: Collected and simulated data (descriptive statistics)**

| Observation | VFS | NTM | LCR | WWTP | WWTP+SW |
|---|---|---|---|---|---|
| **Mean** | 28.60 | 26.16 | 8.27 | 69.97 | 27.88 |
| **Median** | 30.60 | 27.66 | 8.32 | 69.63 | 27.83 |
| **Minimum** | 6.03 | 2.63 | 1.13 | 51.18 | 8.72 |
| **Maximum** | 42.24 | 39.51 | 16.63 | 86.19 | 47.07 |
| **Standard deviation** | 9.06 | 8.84 | 4.55 | 8.09 | 8.39 |
| **Observations** | 68 | 19 | 23 | 65 | 77 |
| **Distribution type** | **Johnson transformation** | **Normal** | **Normal** | **Normal** | **Normal** |
| **Simulation** | **VFS** | **NTM** | **LCR** | **WWTP** | **SEWERAGE** |
| **Mean** | 28.71 | 26.07 | 8.20 | 69.59 | 27.81 |
| **Median** | 30.47 | 26.24 | 8.11 | 69.41 | 27.46 |
| **Minimum** | 0.00 | 0.00 | 0.00 | 43.30 | 0.93 |
| **Maximum** | 41.88 | 43.90 | 18.90 | 95.76 | 55.80 |
| **Standard deviation** | 8.92 | 8.9 | 4.44 | 8.24 | 8.33 |
| **Observations** | 1000 | 1000 | 1000 | 1000 | 1000 |

Source: Own construction

As discretization is common in Bayesian analysis (e.g., Ames et al., 2005), each measure type was discretized into four categories to make the analysis less complicated. It should be noted that this inevitably leads to information loss. Intervals of almost identical size were used within each category. In the cases of WWTP and WWTP+SW there was one outlier, which was included in the analysis, but was ignored for interval selection as it would distort the final gaps. Looking at histograms of the simulated data, most of the measures seem to be slightly more effective on average than previously expected. This holds especially for VFS and WWTP, but it may not be apparent from the summary of discretized variables shown in Table 3 as the intervals do not correspond with expected effectiveness.

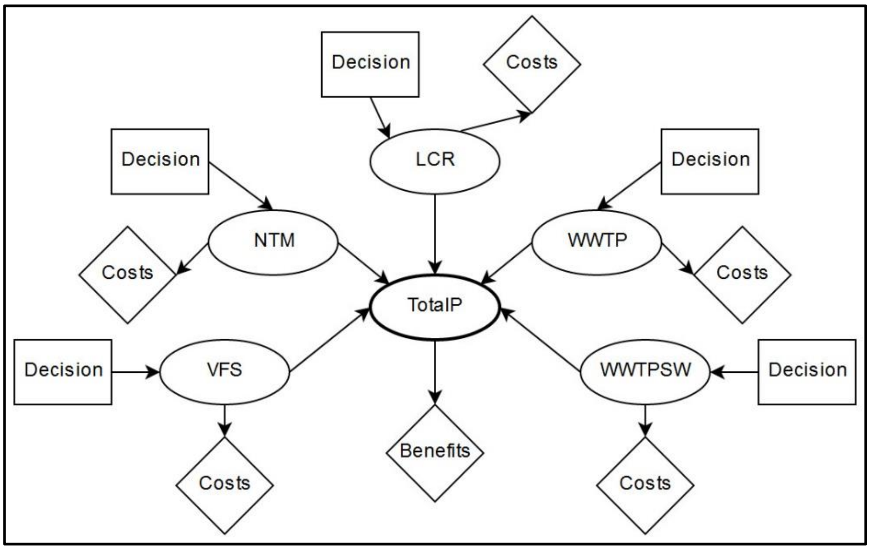
**Table 3: Probabilities of effectiveness intervals**

| | Minimum | Low | Medium | High |
|---|---|---|---|---|
| **VFS** | **[0; 10.5]** | **(10.5; 21]** | **(21; 31.5]** | **(30; 42]** |
| **Probability** | 0.049 | 0.133 | 0.358 | 0.46 |
| **NTM** | **[0; 11]** | **(11; 22]** | **(22; 33]** | **(33; 44]** |
| **Probability** | 0.045 | 0.280 | 0.449 | 0.226 |
| **LCR** | **[0; 4.75]** | **(4.75; 9.5]** | **(9.5; 14.25]** | **(14.25; 19]** |
| **Probability** | 0.229 | 0.391 | 0.28 | 0.1 |
| **WWTP** | **[46; 58.5]** | **(58.5; 71]** | **(71; 83.5]** | **(83.5; 96]** |
| **Probability** | 0.082 | 0.497 | 0.374 | 0.047 |
| **WWTP+SW** | **[0; 12.75]** | **(12.75; 25.5]** | **(25.5; 38.25]** | **(38.25; 51]** |
| **Probability** | 0.031 | 0.376 | 0.469 | 0.124 |

Source: Own construction

As mentioned previously, not enough data are available to evaluate the impact of broad-base terraces, changes in crop rotation and clearance of a biological pond. We can test what the probability of achieving the "good status" is, but this analysis does not reveal much as 51.22 kg of phosphorus are expected to be reduced by the omitted inputs. Alternatively, we can evaluate how effective the implemented measures would be compared to the expectations. To do so, only phosphorus that is expected to be reduced by the measure types entering the analysis was considered. This establishes an "adjusted good status" at 148.82 kg. A Bayesian network of the phosphorus abatement was designed based on dependencies among the variables. Figure 3 shows the final diagram structure. Ovals represent chance nodes, rectangles stand for decision nodes and rhombi for utility nodes. Arrows indicate causality between two variables. TotalP represents the variable of interest (probability of reducing the expected amount of phosphorus).

**Figure 3: Influence Diagram**

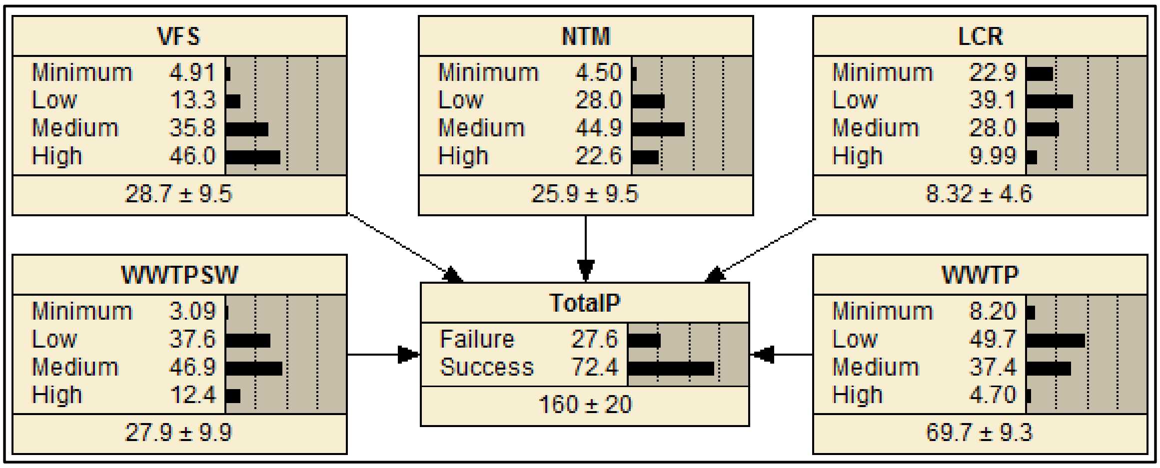



Source: Own construction

## 4.5 Results

Netica software was used to construct the network (decision and utility nodes were omitted for simplicity) and probabilities obtained from the discretization were entered into the model. Simulation results are presented in Figure 4.

**Figure 4: Bayesian network results**




Source: Own construction

TotalP was discretized into only two categories based on the reduction target. Failure means the required amount of phosphorus was not reduced, while Success indicates more than the expected amount was removed. Figure 4 shows the "adjusted good status" will be achieved with a probability of 72.4%, which suggests that breaking the regulation is quite unlikely. Modifying the "good status" turned out to be a good option as these measures would reduce the original 200-kg target only in 2% of cases. It is also possible to see how the probability of reaching the respective "adjusted good status" changes when we drop one variable at a time. While WWTP seems to be crucial to the reduction as the probability of Success decreases to 65% when the variable is dropped, it jumps up to 75.9% when NTM is left out, indicating the variable has a lower potential than the other ones.

## 4.6 Discussion

Bayesian networks provide a robust way of determining whether the "good status" will be achieved in a specific water body. There are both advantages and drawbacks to this approach. Besides the previously mentioned loss of information due to discretizing, this section discusses several findings we made while working with Bayesian networks.

First, one never gets a clear yes/no answer as the results are always probabilistic. The only way to be certain about the result is to aim for a 100% probability of success (or zero). However, this is not desirable as that would mean the goal is achieved even when all the measure types are least effective, meaning that resources are wasted in any other case. Therefore, careful consideration is needed when it comes to evaluating the results. Based on this, we see 72.4% as a reasonable number. Specific to the "good status" analysis is computation of costs and benefits. The costs stay the same as the measures are implemented just like in the case of deterministic analysis, but the "good status" is now achieved only with a certain probability. It is very unlikely the benefits grow linearly as some of them are probably reached only when the "good status" is achieved. This is not too problematic in the Stanovice case study as Macháč, Brabec & Slavíková (2016) estimate the benefits of reaching the "good status" (EUR 289,136) are significantly larger than the costs of measure implementation (EUR 43,233). However, if the results of the CBA are closer, benefits need to be revaluated.

The Bayesian approach is also more demanding on data availability. While the classic approach uses a fixed effectiveness for measures of the same type (often based on expert judgement), the Bayesian method requires more observations as a large sample is necessary for (ideally) representative discretizing or at least probability distribution estimation. Allowing effectiveness within the group of measures to vary gets us closer to reality. As the collected data show, similar measures do not always yield the same results. First, not all the suggested measures in one group have the same parameters and are therefore likely to have different effectiveness. Moreover, local conditions also affect measure effectiveness and we can hardly expect the fixed effectiveness to cover this variance. Expert judgement may also be used when the number of observations is unsatisfactory. This solution is not preferred, but is possibly less distortive than using just few observations as standard error is inversely related to the sample size.

As indicated above, Bayesian networks may be used to test how much particular group of measures contributes to achieving the target, while maintaining the robustness. This may be useful for measure selection. In the case study, the probability of meeting the criteria increases when NTM is dropped and its expected reduction is deducted from the target. This indicates NTM is less effective than other measures. Therefore, it may be efficient to go through the measure selection process again and handicap the ineffective measures. This method is more time-consuming (as the whole Bayesian approach), but may be economically desirable as fewer measures may be needed in the end. We recommend adjusting the selection process according to effectiveness of measures to prevent wasting resources on measures that might not be essential for achieving the target.

Bayesian networks are useful in assessing measure effectiveness, but they may be used to test uncertainty about costs and benefits as well. It might not make too much sense for costs of implementation as they are mostly based on market prices and the uncertainty is problematic mainly for large long-term investments (interest rate), but it can be useful for evaluation of benefits. The largest part of benefits in water management (aesthetic, recreational, etc.) are not traded on markets and other techniques need to be used to estimate them correctly. Integrating the evaluation of benefits into the analysis should lead to accurate results (although they will again be probabilistic).

## 4.7 Conclusion

This paper applies the concept of Bayesian networks to decision making in water management. It helps to determine whether the "good status" required by the WFD can be achieved with a set of selected measures. Bayesian networks represent an effective way of assessing uncertainty of effectiveness of implemented measures, which can also be extended to costs and benefits. The approach is presented on Stanovice water reservoir. Results of the case study show that the chosen measures eliminate the expected amount of phosphorus with a probability of 72.4%. Barton et al. (2008) find the phosphorus limits will be met with a probability of 56%. This number is not dramatically different from our result. However, they also find the agricultural measures dominate the model, which is contrary to our conclusions. We discovered some groups of measures are more effective than others, which indicates changes in the selection process might be desirable.

It is appropriate to incorporate the Bayesian approach as a first step of the proportionality analysis, which needs to be carried out if the water body applies for an exemption from achieving the "good status". If the results show that achieving the "good status" is unlikely, it makes little sense to continue with the proportionality analysis. Instead, the selection process should be revised to increase the probability of reducing the desired amount of a pollutant. The new CEA should increase the reduction target and penalize the least effective measure groups (based on probability distributions). The new set of measures should be tested using the Bayesian network and if a sufficient probability is achieved, it can be followed by an analysis of costs and benefits. The application for an exemption is also more likely to be successful when we are fairly certain that the measures are capable of achieving the "good status" as all the relevant costs enter the analysis.

Bayesian networks are still used rather sparsely as they are demanding on data and not straightforward to interpret (especially in relation to the WFD). However, we believe this approach is informative and should be used more heavily in water management. This paper offers an additional source of information for stakeholders who seek a more reliable way of assessing abatement measures in water management.

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# 5. Appropriateness of cost-effectiveness analysis in water management: A comparison of cost evaluations in small and large catchment areas

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## 5.1 Introduction

The Water Framework Directive (European Parliament, 2000), further referred to as the Directive, has introduced a comprehensive perspective of river basin management and institutionalised numerous analytical tools of socioeconomic research that have not been widely applied. Particularly, evaluation of measures proposed to reach "good status" of a water body is promoted within the economic analysis. Such evaluation usually calls for the application of standardised methods of environmental economics, such as cost-effectiveness analysis (CEA) and cost-benefit analysis (CBA), which should be complemented with more qualitative institutional analysis and a stakeholder consultation process (Martin-Ortega et al., 2013).

Specifically, the use of CEA is emphasised in order to judge the proportionality of costs of different types of measures. In justified cases (according to Article 4 of European Parliament, (2000)), it is possible to apply an exemption due disproportionate costs of measures (Görlach at al., 2007 and Martin-Ortega, 2012). However, the practical applicability of both CBA and CEA struggles with data availability and a large uncertainty of future costs of implementation and impacts (especially regarding the calculation of costs and benefits of planned water quality improvement measures). The data problem is aggravated when considering large catchment areas with complex environmental and social linkages.

The purpose of the paper is to show practically how the methodological challenges of CEA increase together with the territory in focus. We introduce two in-depth case-studies of the

Orlík reservoir (as a large catchment area) and the Stanovice reservoir (as a small one) undertaken in the Czech Republic in 2013–2015. The intention is to discuss how to interpret results produced with imperfect data under time and resource constrains.

The paper consists of three chapters. The first chapter summarises the existing applications of CEA and the basic steps in the analysis. The second chapter presents results of two complex studies of cost-effectiveness of measures in the Orlík and Stanovice reservoirs. The aim of that chapter is to show the different options and appropriate tools for processing studies based on a spatial scale or catchment size. The further appropriateness of CEA application is discussed in the conclusion.

## 5.2 Material and Methods

### 5.2.1 Existing applications of cost-effectiveness analysis

According to the current requirements of the Directive, CEA should be processed as part of River Basin Management Plans (RBMP). In practice, however, this process is often omitted and only one pre-selected set of measures is promoted. CEA is elaborated often only in connection with proportionality analysis (such as in the case of Jensen et al., 2013 and Galioto et al., 2013). Methodologically speaking, CEA may assume many forms. In the water area, we come across both simple linear (optimisation) models used for smaller areas and models using mathematical programming, suitable for larger areas including natural conditions. Two different forms of output exist in optimisation CEA models. The EPA (Environmental Protection Agency, 1995) distinguishes between the cost minimisation and benefit (effect) maximisation approaches. When minimising costs, the principal goal of the measure is to achieve a given effect with the least possible costs. This approach is applied predominantly. It is used, e.g., by Yang et al. (2005) and Martin-Ortega et al. (2013). Conversely, the maximisation approach aims at achieving maximum possible level of output (effect) using a predetermined budgetary constraint; it is most commonly used for restoration measures. The maximisation approach has been used, e.g., by Ancev et al. (2008) and Balana et al. (2013).

The basic procedure is common regardless of the variable optimised (costs or effect). In the first stage, as shown, e.g., by Whitehead et al. (2013), the costs of measures are identified depending on the nature of the environmental problem. Costs can be classified in many different ways. Most common direct costs of application represent the main focus in the whole CEA process. Generally, it is recommended to take into account the most of the potential and known costs. Cellini and Kee (2010) show that, for reasons of uncertainty, it is not possible to estimate all costs, therefore efforts should be made to identify and monetise those that are expected to have the greatest effect. According to Whitehead et al. (2013), the cost valuation should be based on the principle of opportunity costs; therefore, it should also include indirect costs such as social costs. According to Musgrave et al. (1989), the direct costs include wages and salaries, costs of equipment and materials, or administration with respect to the type of project; indirect costs are generated as an unintended result of application of measures. Indirect costs are often produced as a by-product, or multiplication and pouring action in areas other than originally intended. The analysis by Galioto et al. (2013) is connected with a maximum effort to include costs, ranging from additional costs (investment and operating costs of new measures) to costs arising from decreased profits due to having to implement measures (including the possibility of compensatory payments), social costs (additional taxation to finance measures) and other indirect costs (increase/decrease in other sources of emissions).

Costs can be quantified in different ways. They can be expressed as the total present costs related to a specific period, or as the annual costs in the form of average or annualised costs. In all cases, the costs of individual applications are divided by the effect (for example, amounts of phosphorus reduction in kg) and thus the ratio of cost-effectiveness is set. In the next step, the measures are ranked by their cost-effectiveness ratio. In the case of cost minimisation, the effect of measures is added cumulatively depending on their ranking. When the required size of the effect is cumulatively achieved, all the measures included comprise the most cost-effective way of solving the goal (given problem). As Van Soesbergen et al. (2007) note, this basic algorithm of measure ranking is associated with many complications given by the nature of the measures and the options for combining them. On the one hand, measures may be substitutes, with application of one measure ruling out the application of another. On the other hand, implementation of some measures may be conditioned by adoption of others. The summed size of the effects may be different when combining different measures than when implementing them separately. In the case of measure substitution and using the basic algorithm, the least cost-effective measures are eliminated from the process. With regard to other measures, it may be more convenient to implement a less cost-effective measure that achieves a higher effect (e.g., phosphorus reduction). In this case, it is necessary to apply a more complex algorithm, which is based on the creation of all possible combinations of measures, including both mutually exclusive and sequential measures.

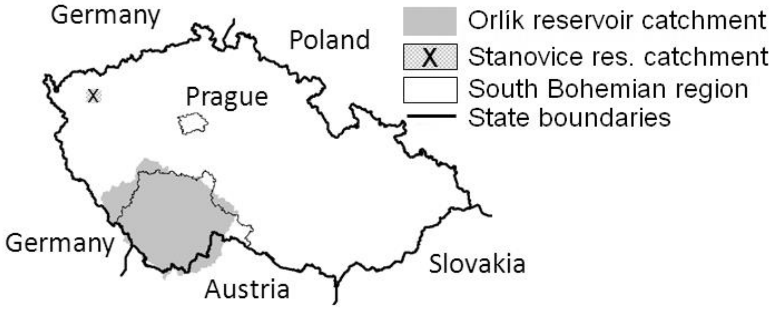
### 5.2.2 Introduction of case studies

Two complex studies of cost proportionality were carried out in the Czech Republic in the years 2013-2015. The first study was focused on the large catchment area of the Orlík reservoir as a part of the REFRESH international project. The subject of the second study was the small catchment of the Stanovice reservoir. The aim of both the studies was, among other things, to find the most cost-effective combination of measures to reduce phosphorus and thus to achieve "good status" in terms of the Directive. Both the reservoirs are affected by excessive water eutrophication.

The catchment of the Orlík reservoir is located in the south of the Czech Republic. The reservoir catchment area matches that of the Upper Vltava River and covers an area of 12,117 km$^2$ (representing 15.4% of the area of the Czech Republic) and consists of several sub-catchments. Each of the sub-catchments faces different conditions. The sub-catchments differ not only in their natural conditions but also in the size of cities, methods of wastewater disposal, etc. The reservoir itself is used primarily for power generation and recreational purposes (mainly swimming).

The Stanovice reservoir is situated in Western Bohemia, in the Karlovy Vary Region and falls within the catchment of the Lomnický brook. The catchment of the whole brook covers almost 92 km$^2$ (representing 0.001% of the area of the Czech Republic). The primary purpose of the Stanovice reservoir as specified by the manager Povodí Ohře, (2014), is supply of drinking water for the Karlovy Vary area, securing of minimum flow rates, and flood protection for Karlovy Vary. Secondary purposes of the water body include electricity generation and fishery, among others. Location of both catchment areas is shown in Figure 1.

**Figure 1: Map of Orlík and Stanovice reservoir catchment areas**



Source: Own construction using (Hejzlar et al., 2014)

The main sources of the phosphorus contamination in the Orlík reservoir are municipal wastewater discharged into the watercourses (55% of the phosphorus), intensive aquaculture in fishponds (22%) and agricultural activities in the catchment (11%). At present, fishponds covering a combined area of approx. 154 $km^2$ are managed for aquaculture in the Orlík reservoir catchment. There are only 16 small villages and a few ponds in the Stanovice reservoir catchment; therefore, the contribution of phosphorus is distributed evenly between point (municipal wastewater) and diffuse sources (agricultural activities). To prevent massive algal bloom in the Orlík reservoir in the summer, the amount of phosphorus from the identified sources in the catchment has to be reduced by 136 tonnes a year compared to the average for 2007-2009 (Vojáček et al., 2013). Measures are being implemented at present that will lead to a phosphorus reduction by 22 t annually. The inflow therefore has to be reduced by another 114 tonnes a year in the upcoming period. According to information from the T. G. Masaryk Water Research Institute (Výzkumný ústav vodohospodářský T. G. Masaryka), achievement of "good status" in the Stanovice reservoir requires a reduction of phosphorus inflow to the reservoir by 60-200 kg a year compared to the present status. The cost-effectiveness analysis of the Stanovice reservoir calculates with a reduction of 200 kg of phosphorus annually at the inflow to the reservoir.

In the Orlík reservoir catchment, 3,097 possible measures have been identified Macháč, (2014) to reduce phosphorus from all three groups of sources (wastewater, fisheries, agriculture); in the Stanovice area, 243 measures for wastewater and agriculture sources (Macháč et al., 2015). Measures relating to construction and renovation of wastewater treatment plants, sewer systems, dead-end and accumulation cesspits, retention wetlands, biological reservoirs and domestic wastewater treatment plants, and measures relating to intensification of the treatment process at wastewater treatment plants were proposed for the point sources. In order to reduce the phosphorus admission from fishponds it is necessary to change the management, which means notably (i) reducing the populations and thus the fish production, (ii) setting the fodder and fertiliser doses to levels that best correspond to the amount of phosphorus consumed in the fish production (zero balance). It offers two alternative methods to the present way of semi-intensive fish keeping: level-balance production or extensive fish keeping. Agricultural phosphorus inflow measures involve in case of Orlík reservoir catchment 4 types of measures (grassing of 20-metrewide strips along watercourses and reservoirs, grassing of sloping areas, no fertilisers in sloping areas, introduction of no-tillage methods) and in case of Stanovice reservoir 5 types of measures (building of broad-base terrace, grassing of sloping areas, changes of crop rotation, leaving crop residue, and introduction of no-tillage methods). Table 1 summarises the basic characteristics of both catchment areas.

**Table 1: Basic characteristics of the Orlík and Stanovice reservoir catchment areas**

| Characteristics | Orlík reservoir catchment | Stanovice reservoir catchment |
|---|---|---|
| Area | 12,117 km² | 92 km² |
| Location | South Bohemia | Western Bohemia |
| Natural and other conditions | Heterogeneous | Homogeneous |
| Reduction target | 114 t/year | 200 kg/year |
| Number of potential measures | 3,097 | 243 |
| Types of measures | Point, Fishery and Agricultural phosphorus inflow measures | Point and Agricultural phosphorus inflow measures |

Source: Authors

Identification and definition of specific applications of measures and qualification of costs of their implementation are followed by their monetisation based on expert studies, catalogues of measures or a market survey in the form of a non-binding request with contractors/implementers of measures. Annual cost was calculated using the annualised cost method. Known value of present investment, operating and other costs (such as administrative costs, lost profits) are transferred to a future flow of the same costs based on annual costs, which correspond to the known present value when cumulated. First, the present value of costs of the measure or the present value of the component parts of the measure with different lifetimes is determined. Then, the annualised costs for each component are calculated (Equation 1). The sum of the component annualised costs yields the total annualised costs of the measure.

**Equation 1: Annual costs in the annualised form**

$$AC = PV \times \frac{i \times (1+i)^l}{(1+i)^l - 1} \quad (1)$$

Where: AC – annual costs in the annualised form

PV – present value of costs

i – discount rate

l – expected lifetime of the measure

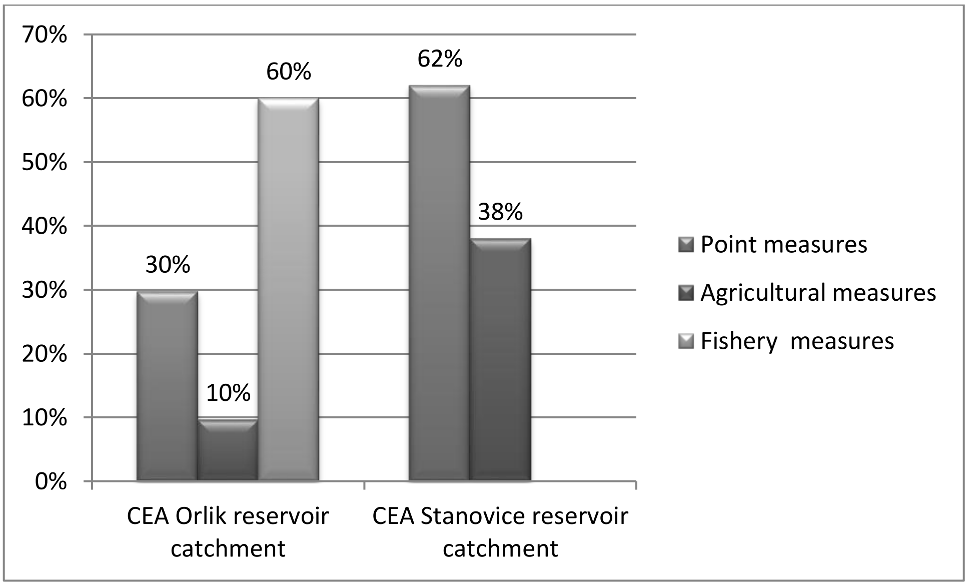
Source: Authors

The cost-effectiveness ratio (costs of eliminating 1 tonne of phosphorus) has been determined for the identified measures based on their efficiency and costs. In determining the costs, emphasis was placed on investment and operating costs and lost profits. It must be stressed here that the natural phosphorus retention capacity of the corresponding watercourse was taken into account, so the cost-effectiveness of the measure application expresses the ratio of costs and the phosphorus not discharged into the reservoirs. After we calculated unit costs per kg of phosphorus not discharged into the reservoirs, we could perform the final step of the analysis. The final step of the CEA was ranking the applications of measures by their cost-effectiveness ratios from the cheapest ones to the most expensive ones. A basic ranking algorithm was used in the case of the Orlík reservoir. If some measures were mutually exclusive, the less cost-effective measures were removed from the analysis. A more complex dynamic CEA optimisation process was applied to the Stanovice reservoir, based on combinatorics of all above mentioned measures and on formulation of supplementary measures. The introduction of a more complex algorithm in the case of Stanovice resulted in an increase in the maximum possible reduction to almost 72 kg/year.

## 5.3 Results and Discussion

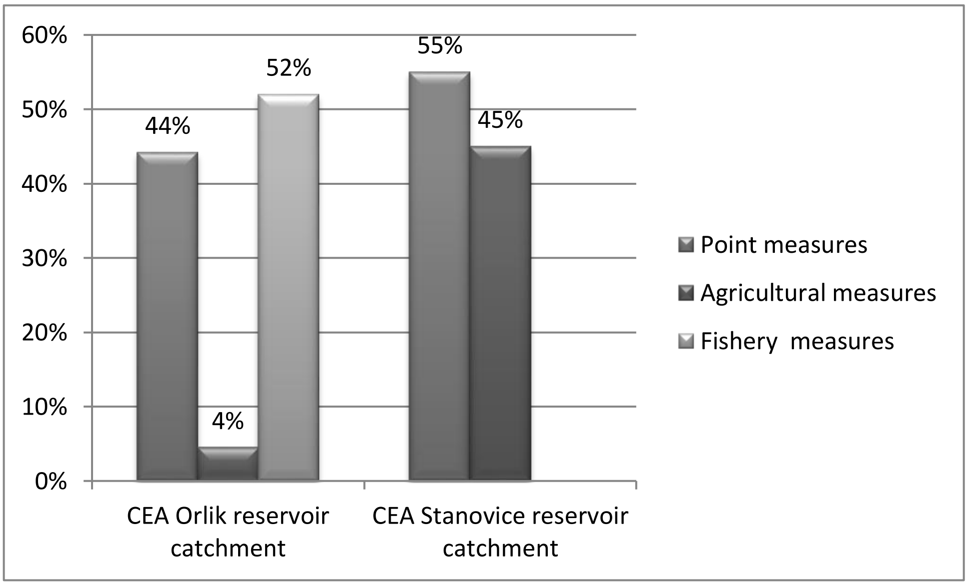
According to the CEA, the total annual costs of all the measures are CZK 602 million (EUR 23 million) for the Orlík reservoir and CZK 1.15 million (EUR 44,231) for the Stanovice reservoir. As shown in Figure 2, point measures formed a significant share of the phosphorus reduction measures in both the catchment areas. The greatest amount of phosphorus in the large catchment of the Orlík reservoir can be reduced in an optimal scenario using fishery measures. On the contrary, only a very low reduction is achieved by agricultural measures. Reduction of phosphorus from agriculture also has a significant impact in the small catchment, because the villages are too small and therefore the point measures are not so cost-effective. The significance of agricultural measures in the Stanovice reservoir catchment is more evident from the comparison of the cost categories of measures (see Figure 3).

**Figure 2: Comparison of the reduction ratio of individual categories of measures (100% = reduction target)**




Source: Own construction using (Vojáček et al., 2013) and (Macháče et al., 2015)

**Figure 3: Comparison of costs of individual categories of measures (100% = total costs in each catchment)**




Source: Own construction using (Vojáček et al., 2013) and (Macháče et al., 2015)

As a result, the CEA showed that annual costs derived from the phosphorus inflow reduction are significant. The total annual costs of achieving "good status" (reduced phosphorus inflow in the catchment) are influenced not only by the measure identification process (the categories of measures involved in the analysis), but also the method of ranking the measures. To obtain the most cost-effective combination of measures, it is suitable to apply a complex algorithm as in the case of the Stanovice Reservoir, where all possible combinations and supplements were created. Besides the possibility of achieving lower total costs, it is possible to achieve a higher maximum rate of reduction compared to the basic algorithm. The application of a complex algorithm is associated with greater time demand. It is necessary to determine all possible combinations of measures that are mutually exclusive. In the next step, supplements are specified. In the case of large catchment areas, hundreds of possible combinations of measures as well as related supplements can be identified. In case of the Orlík reservoir catchment, there are more than 3,000 possible measures. In this case, the number of combinations and supplements is estimated at more than one thousand. Therefore, the application of a complex algorithm rises feasibility issues. In many cases, the application of the basic algorithm can be regarded as sufficient. The question remains where it is still proportionate to use a complex algorithm and where not. Nevertheless, even in the case of large catchment areas significant financial savings can be achieved using the complex algorithm. In practical terms, it is much easier to implement several major measures with a relatively large effect than a number of small measures. From this point of view it makes sense to apply a complex algorithm in all cases where it is technically possible.

Theoretical definition of small and large catchment areas clashes in practice with the aspects of upstream-downstream. In the case of evaluation of cost-effectiveness in a water body/catchment that is not located in the upper reaches, it is necessary to take into account possible sources of pollution upstream. In these cases, it is necessary to extend the analysis with further measures upstream that will affect the evaluated area. This situation is already evident from the Orlík reservoir example, where measures were considered for the whole upstream area. It is necessary to take into account the natural pollution reduction in the catchment area.

The analysis also shows that there is no single recommendation on which groups of measures are generally the most cost-effective in the Czech Republic; local conditions always have to be taken into account, and measures assessed accordingly.

## 5.4 Conclusion

The purpose of the paper was to discuss the appropriateness of the CEA method for selection of suitable measures for reaching "good status" of a particular water body. Based on presented case studies, we have pointed out key methodological and practical obstacles to the method application, particularly considering large catchment areas.

In general, we can conclude that CEA is an appropriate tool for selection of cost-effective combinations of measures for reservoirs or other water bodies. Its rare application in water management, however, confirms the time and resource intensity of the method described above. A possible simplification for better applicability of CEA might include: (a) pre-selection of combinations of measures (to avoid considering all possible options), and (b) creation of artificial sub-catchments in order to decrease the complexity of measures and impacts. Both the steps are likely to simplify the numerical procedures, but also decrease the level of the maximum possible reduction.

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# General Conclusions

Application of economic and advanced economic methods (methods beyond calculation of costs) in water management is becoming more common. The Water Framework Directive (WFD) has an important influence on incorporation of economic approaches and methods into water management. Application of the proportionality concept in water management is one of the most discussed topics among scholars and professionals in water management. The primary object of the dissertation is to answer the key question "How to assess disproportionality in the conditions of the Czech Republic?" The thesis contains both a proposal of the appropriate methodology, which has been adopted by the Ministry of Agriculture, and other recommendations (e.g., how to combine different methods).

This thesis "Assessment of Disproportionate Costs in Environmental Policy with a Special Focus on Water Management" is conceived as a cumulative dissertation thesis. It consists of five scientific articles related to the problem of cost proportionality in water management. The individual papers deal with a comparison of existing approaches and methods, a proposal for a combination of methods and practical applications and pilot testing of different approaches and methods. The author of this thesis is the first author (in four cases) or the second author (in one case) of the papers. Four of these articles were published or accepted in impact factor journals / peer-reviewed international conference proceedings indexed in the Web of Science database in 2016-2018; one article was being peer-reviewed in an impact factor journal at the time of writing the thesis.

The EU's effort to improve water quality has led, in the case of the ambitious Water Framework Directive objectives, to the need to test the cost disproportionality of achieving "good status" in water bodies with respect to the high financial burden. The WFD pragmatically recognises that there may be cases where the costs outweigh the benefits. For this reason, when costs outweigh overall social benefits, the WFD offers the possibility to apply an exemption. However, there is no clear definition of how the assessment should be performed. The thesis summarises the basic approaches to assessing cost proportionality of achieving "good status" abroad. A Czech methodology was drawn up based on experience abroad. Until 2015, the importance of exemptions due to disproportionality was only secondary: it enables extension of deadlines for achievement of "good status". After 2027, any non-achievement of "good status" will have to be justified before the European Commission. The justification of non-achievement of "good status" in a water body due to disproportionate costs can be expected to gain substantial importance.

Economic theory and methods were applied in the individual chapters of the thesis. Three possible approaches are described in the Introduction and Chapter 1. The approaches differ in their scale, methods used, complexity, necessity of input data and time and financial demands. Chapter 1 aims to compare the methods based on cost-benefit analysis and the German "new Leipzig approach" based on multi-criteria analysis. For this purpose, a small catchment in the Czech Republic was chosen as a pilot area. An approach based on cost affordability and social acceptability was applied to the legislative changes proposed to achieve the "good status" in the Czech Republic in Chapter 2. An analysis of possible redistribution of costs and benefits to prevent disproportional costs in flood protection was the subject of Chapter 3.

Even through cost-benefit analysis (CBA) and multi-criteria analysis (MCA) are considered by many scholars to be complementary methods (e.g., Qureshi et Harrison, 2001), applications of these methods in disproportionality analysis differ significantly. According to Mysiak et al. (2005), MCA is very popular because it does not require monetary valuation of costs and benefits. Due to the wide range of non-use values related to benefits, it is significantly easier to apply MCA to compare different measures with different impacts (benefits). Since it is impossible to monetise all costs and benefits within the CBA, non-monetised costs and benefits should be included and taken into account in the last step of the CBA in qualitative form. In the context of the WFD, it is necessary to compare all social costs and benefits. In this case, there is a situation where we compare mostly investments and operating costs of newly designed measures with all types of benefits. The literature analysis showed a predominance of CBA method applications, which was accompanied in some cases with an initial review based on criteria evaluation. In terms of MCA, one of the criteria is costs (mostly expressed in monetary value) with other non-monetary criteria in a qualitative form. To assess the cost disproportionality, it is necessary to set a weight for each criterion, which allows a comparison of the bundle of costs with the bundle of benefits. Setting of weights is very subjective. From the author's point of view, it is difficult to say which aspect has a greater weight. As shown, for example by Vojáček et al. (2013), many benefits, such as recreational value, are non-linear. Setting of weights can easily lead to a distortion of results.

In this context, both approaches (based on a classical CBA and the "new Leipzig approach" based on MCA) face a certain degree of inaccuracy and uncertainty. The "new Leipzig approach" works with different criteria such as average previous expenditures per ha of catchment, investments and operating costs of newly designed measures, distance to "good status" for 5 characteristics (objective distance) and with 5 groups of additional benefits generated by achieving "good status". The calculation of the cost threshold is based on multiplying the distance to "good status" and average additional benefits by two constants (2/18 and 1/18), which are the weights. Some of the input data are unavailable for the Czech Republic (thus, some of the characteristic were not determined for the Stanovice catchment). The assessment of the expected improvement in the form of additional benefits on a scale from zero to three is also highly subjective. The total level of additional benefits depends on the evaluator. A new supporting methodology for rating the additional benefits would be required to avoid this type of distortion. Without a strict methodology for rating the criteria of additional benefits, the application of the "new Leipzig approach" is based on feelings rather than on reality.

Cost-benefit analysis is built upon neoclassical economics. CBA perceives all costs and benefits as anthropogenic. This method requires monetary valuation of both costs and benefits. Although there is a high uncertainty deriving from the input data used, the costs still represent the easier side of the analysis. This initial step – valuation of costs – is also necessary for the assessment based on the "new Leipzig approach". Significant complications are associated primarily with the benefit side, where it is difficult to monetise the part of the benefits connected with the potential effect of "good status" achievement.

Both approaches are suitable methods. In the author's opinion, these methods complement each other. A method based on criteria such as the "new Leipzig approach" can be used as an initial analysis (first step). The aim is to exclude water bodies where the existence of disproportionate costs is unlikely. The rest of the water bodies will be assessed using cost-benefit analysis. The parameters of multi-criteria analysis have to be recreated for the Czech conditions. The original parameters from Germany are inappropriate due to unavailability of data. In order to generalize the conclusions of the article on the comparison of Czech and German approaches regarding the possible application of the German approach in the Czech Republic, it would be necessary to carry out further pilot testing in another catchment. Whenever disproportionate costs are set according to the CBA (Czech methodology), it should be possible to achieve the same results using the German approach.

The last approach based on social acceptability of water prices was analysed and applied in practice in Chapter 2. In the case of water, there is a conflict between two basic principles: (i) the social principle based on the right to water, and (ii) the economic principle based on the value derived from rarity. Prices of (not only) environmental goods should reflect their rarity. However, in practice, most of the measures are paid from the state budget in the Czech Republic (money collected in the form of taxes). Part of the costs is not included in the drinking water price. This method is therefore suitable only for situations where the price reflects the implementation of measures. This situation has occurred in the case of newly drafted environmental legislation. In regulatory impact assessment, the social acceptability should be considered as one of the criteria. Households are one of the important affected subjects, but it should not be the only criterion. This approach is appropriate for impact assessment of new regulation or strategies at the national level. Its application at the level of water bodies/catchments is very limited. Achievement of "good status" cannot be confused with drinking water prices.

As is discussed in Chapter 3, the location of the water body plays an important role. The quality of water and also achieving of "good status" is very commonly strongly influenced by the upstream-downstream relations. The paper handles this relationship in the context of floods. Cooperation of both parties (upstream and downstream) leads to better results. Reallocation of flood protection measures is connected with payments from downstream to upstream. The situation with achievement of "good status" is almost the same. However, in most cases, reducing pollution in the same water body is not enough. To achieve "good status", it is also necessary to implement some measures upstream. In this case, the disproportionality analysis is made at the catchment level, which enables inclusion of the upstream-downstream relations. Nevertheless, it is appropriate to extend the analysis with further measures upstream.

Chapter 3 demonstrates that game theory is an appropriate tool for solving water issues in the context of upstream-downstream relations. Scenario analysis is one possible extension. Its practical application depends on the transaction costs and willingness to negotiate/cooperate. The awareness about the issue ("good status") influences the result. Mostly there are also more different stakeholders than only two players with different interests; therefore, negotiation is mostly connected with high transaction costs. If the measure it not required by law, the land owner requires compensation in the form of subsidies. In practice, fewer cost-effective measures with lower transaction costs are often preferred. In this respect, it would be appropriate to include transaction costs in the cost-benefit analysis as an essential part of the costs. This new category goes beyond the costs considered by the WFD. From the economic point of view, it is very difficult to estimate the transaction costs ex-ante. According to New Institutional Economics (e.g., Coase, 1960) and Free-market Environmentalism (e.g., Anderson et Leal, 2015), the government should take measures to reduce transaction costs. A reduction in transaction costs can be met in the current situation in the Czech Republic by way of reducing the transaction costs by, e.g., defining subsidy conditions for farmers. Lubell et Lippert (2011) state one of the problems – fragmentation of water management in many different sectors. Reduction in transaction costs needs to be addressed at the level of water governance.

Application of cost-benefit analysis is connected with many uncertainties. Most of the uncertainties are associated with cost and benefit valuation or possibly with the discount rate used. These types of uncertainties are solved using sensitive analysis, which is one of the last steps of CBA. One of the fundamental uncertainties is overlooked. In almost all cases, the achievement of "good status" with selected measures is taken for granted. The effectiveness of the measures dictates the number of measures that should be implemented, and therefore the costs. The assumption about the granted effectiveness may be wrong. Based on Chapter 4, Bayesian networks can be used to deal with the uncertainty of measure effectiveness in the context of disproportionality analysis.

The probability of achievement of "good status" is calculated based on an application of Bayesian networks. The result is the likelihood that "good status" will be achieved through the implementation of measures. It is based on previous measure implementations and simulations of results. This step makes the analysis much more robust and decreases the level of uncertainty. However, application of Bayesian networks is not a direct step of disproportionality analysis. The process should be applied generally in the context of measure planning and selection. Enough data are required to achieve the highest possible effect. Ex-post analysis of already implemented measures would give precision to the necessary input data for Bayesian networks. The lack of data was the reason why not all types of measures entered into the pilot analysis described in the article.

According to the Water Framework Directive, cost-effectiveness analysis (CEA) should be an integral part of disproportionality analysis. A comparison of applications of cost-effectiveness analysis in a small and a large catchment is the subject of the last chapter. In the WFD context, CEA is used in the cost minimisation form. In both cases, the catchments face eutrophication. Both cases differ in categories of measures which were taken into account. This is mainly due to different sources of phosphorus. In the case of the Orlík catchment, it originates not only from the usual sources (point sources in the form of wastewater from villages and diffused sources connected mainly with agricultural activities), but also from fishponds (due to intensive fish production). Application of CEA in both cases leads to selection of specific most cost-effective measures. The comparison of the results shows that it cannot be said in general that some measures are more or less cost-effective in the Czech Republic. The effectiveness of the measure always depends on local conditions. Therefore, it makes sense to carry out the cost-effectiveness analysis.

In the introduction, both types of CEA were presented. Usually the basic cost-effectiveness analysis is used. In that case, the CEA is carried out purely based on the cost-effect ratio. In many cases, however, it appears that the most effective measures are able to produce only a very small effect. Selection of these measures leads to exclusion of measures with higher effect, which are substitutes to the measures with lower effect. The optimisation principle then fails in this case. A new innovative process of selection of appropriate measures has been designed by the author of this thesis. It is called dynamic CEA. The process of dynamic CEA rejects the process of eliminating less cost-effective measures. Instead of doing so, the less cost-effective measures are also taken into account in the form of additional measures with cost-effect ratios based on additional costs/additional effects compared to the most cost-effective one. The selection of the most appropriate measures varies according to the desired goals. This method increases the spending efficiency in the implementation of environmental investments. Dynamic CEA can be transferred to other areas of environmental investments. It plays an important role wherever there are mutually exclusive measures. The basic algorithm is based on combinatorics and a comparison of the effect of each option.

Dynamic CEA was applied in the Stanovice catchment (Chapters 1, 4, 5). Application of this method in the area of the Orlík reservoir could also lead to further cost elimination. The difficulty of CEA implementation increases with the number of measures. It would be necessary to create software that would perform algorithms based on inputs. In the case of smaller catchments, this can be done using a combination of simple functions in MS Excel.

However, the application of CEA and the advanced dynamic CEA very often collides with political support and financial resources in the Czech Republic. Many measures are politically unacceptable. Another problem is connected with financing the implementation of measures. Most appropriate measures are located on private land. The landowners demand subsidies or other forms of monetary compensation. In practice, measures are not selected according to their cost-effectiveness (using CEA). Despite the requirement of the WFD, the utilisation of CEA is limited. However, performance of CEA is a necessary step in the context of justification of disproportionate costs.

As already mentioned, there are a number of limitations connected with disproportionality assessment. The Czech methodology (Slavíková et al., 2015) certified by the Ministry of Agriculture has partly solved the problem of non-existence of a clear definition of how the assessment should be carried out. From the economic point of view and with respect to local conditions, the optimum option for the proposed Czech methodology appears to be the recommended combination of (dynamic) cost-effectiveness analysis and modified cost-benefit analysis (CBA) using the method of benefit transfer. It consists of several successive steps. The methodology assumes an assessment of cost proportionality on the scale of a water body. Following an analysis of the present state of the water body and choice of the target "good status" indicator to be the subject of the proportionality analysis, the methodology assumes an analysis of costs of achieving "good status". The direct investment, operating and other costs have to be assessed. The cost and benefit assessment is followed by an analysis of market and non-market impacts brought to society by improvement of the water body status using the concept of ecosystem services provided by water bodies. The methodology assumes notably assessment of three categories of water body benefits. Monetary value should be expressed for recreational and aesthetic ecosystem services and benefits arising from treatment of raw water into drinking water.

Also the challenge with different lifetimes of measures has been solved by application of the concept of annualised costs. With regard to the durability of the solution, it does not make sense to apply the concept of net present value, which is based on setting a time frame.

The other challenges remain. Especially application of cost-benefit analysis is connected with the necessity to monetise the costs and benefits. In this thesis and also in the Czech methodology, the benefit transfer method is recommended for assessment of benefits. Its weakness is a frequent lack of domestic primary analyses and coping with local conditions. Nevertheless, it is impossible to carry out primary studies due to their time and money intensity. To make the results more accurate, it is necessary to monetise some of the benefits based on primary analyses in the Czech conditions in order to prevent inaccurate transfer of values from abroad. A compilation of a catalogue with benefit values could make the cost disproportionality assessment much easier. Justification of exemptions based on disproportionate costs could be applied more frequently.

Most of the measures implemented so far have not been selected based on their cost-effectiveness, because measures are currently prioritised by attributes other than according to their cost-effectiveness. Important challenges also include the distribution of measures in the catchment, distribution of costs among different groups of stakeholders in the catchment, and application of negotiations, which can lead to better results. Application of game theory in the context of upstream-downstream can help solve these tasks. Another possible improvement to water management can be achieved be implementation of payments for ecosystem services (e.g., Louda et Vojáček, 2017). Application of payments for ecosystem services can lead to the emergence of a market. This means generation of financial resources for appropriate measure implementation.

The author's original contribution to application of economic methods in water management and especially in the field of cost disproportionality is presented in this thesis in the form of five standalone scientific articles. The aim of these articles is to discuss different methods and to find appropriate tools for disproportionality assessment. Together with additional information in the Introduction to this thesis, the articles create a comprehensive view on the issue. Different approaches were analysed, tested and evaluated based on the meeting of WFD requirements, local conditions in the Czech Republic and from the economic point of view.

The relevance of this topic is evident not only from the academic debate (development of different approaches in the EU), but also in the context of expenditures on environmental protection. Achieving of "good status" thus may significantly increase monetary requirements of Czech authorities in charge of implementation of required water management measures. It significantly affects the national economy. The drawing of a methodology allows application/justification of exemptions; that means prevention of implementation of necessary measures which are too costly within a short timeframe. For this reason, application for an exemption due to disproportionate costs, and thus application of the Czech methodology, is expected in the third cycle (2021-2027) and then after the year 2027.

The main objective of this thesis, as well as the partial goals defined in the Introduction, can be regarded as met based on (i) the sub-chapters of the Introduction and (ii) the conclusions of each chapter (article). The General Conclusions provide a synthesis of the results and findings. However, it cannot be said that the issue of disproportionality assessment has been fully addressed at a general level. This thesis focused closely on water management. A similar debate is now opening up in the field of air protection. The current debate is primary about the proportionality of costs of achieving the emission levels of best available techniques (BAT).

In conclusion, practical application of the cost disproportionality concept in water management has a great potential for further implementation in other fields of environmental policy, and in economic policy in general. Its implementation can be regarded as protection against excessive regulation. If the newly designed legislation implements the cost proportionality concept, based on the experience gained so far with implementation of the WFD and application of exemptions (based on disproportionate costs), it will still be necessary to define the assessment method in detail.

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