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Sustainability and Water Supply Governance

A Literature Review on Regional Water Governance, Multi-Criteria Decision Analysis, Cost-Benefit Analysis and Sustainability Assessments

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SUMMARY

This literature review is part of a PhD project funded by the Swedish Research Council Formas with support from RISE Research Institutes of Sweden and the City of Gothenburg, performed within the centre for drinking water research (DRICKS). The research project aims to develop a decision support model for sustainability assessments of regional water supply interventions and cooperations. The decision support model is planned to be performed through a combination of multi-criteria decision analysis (MCDA) and cost-benefit analysis (CBA). In the process of developing the model, national and international studies on regional water governance, as well as on applications of MCDA, CBA, sustainability assessments, sustainability criteria and economic valuation techniques within water supply management were reviewed.

The MCDA approach is often used for complex decision problems with large amount of information and when several, possibly contradicting, views needs to be considered in a coherent way. It can, for example, be used to rank alternative interventions, find the unacceptable alternatives, and identify alternatives that need more detailed assessments. MCDA provides a means for integrating quantitative, semi-quantitative and qualitative information concerning alternative interventions. It allows for comparison between objectives and can be used for integrating social, economic and environmental analyses into comprehensive sustainability assessments.

CBA can be used to measure the economic profitability of alternative interventions. The method relies on the anthropocentric foundation of welfare economics in which benefits are defined as increases in human wellbeing and costs are defined as reductions in human wellbeing. Welfare economics is based on the assumptions that each individual is the best judge of his or her wellbeing at a given situation. Individuals' wellbeing depends on market goods and services as well as non-market goods and services, such as health and environmental quality. An intervention is considered economically profitable when its total benefits are larger than its total costs.

Both MCDA and CBA have been used in several applications in the water sector and numerous evaluation criteria have been proposed to assess the sustainability of alternative interventions. This review: (1) gives an overview on literature on regional cooperation in the water sector; (2) provides a general description of the decision-support techniques MCDA and CBA; and (3) presents an overview of applications of sustainability assessments and the use of MCDA and CBA as decision- support in the water sector.

CONTENT

1 INTRODUCTION

This literature review is part of a PhD project funded by the Swedish Research Council Formas with support from RISE Research Institutes of Sweden, Gothenburg Region and the City of Gothenburg, performed within the centre for drinking water research (DRICKS). The research project aims to develop a decision support model for sustainability assessments of regional water supply interventions and cooperations. The decision support model is planned to be performed through a combination of multi-criteria decision analysis (MCDA) and cost-benefit analysis (CBA).

In the process of developing the model, national and international studies on regional water governance, as well as on applications of MCDA, CBA, sustainability assessments, sustainability criteria and economic valuation techniques within water supply management were reviewed.

1.1 Background

Water supply provision has traditionally been a municipal responsibility (Kurki et al., 2016). But with a growing focus on how to best finance and implement water supply improvements to address the ever increasing challenges, e.g. from demographic and climate changes, a more diverse governance is emerging in the water sector (Palaniappan et al., 2007). Within this shift, the inter-municipal, regional level stands out as increasingly important for water governance (Lieberherr, 2011; Schmidt, 2014). This regionalization with cooperating municipalities can take various forms, from bilateral agreements to formations of regional companies and alliances (Kurki et al., 2016). But they all call for rescaling governance and adapting to a collaborative decision-making process to solve the common challenges (Lieberherr, 2011).

In Sweden, the responsibility for providing water supply lies on the 290 municipalities (SFS, 2006:412). About 65 percent of them operate the water supply within their municipality. Remaining municipalities operate the supply in various forms of inter-municipal cooperations including inter-municipal agreements, municipal alliances, joint committees and municipal companies (SOU 2016:32, 2016). In 2013, the Swedish government decided to investigate the public drinking water sector with the aim of identifying current and potential challenges for a safe drinking water supply, and if necessary propose appropriate measures. The inquiry points at several challenges for the Swedish water providers, including an ageing infrastructure; a continuous population growth in the larger cities; a depopulation of the countryside; as well as climate changes with higher average temperatures, increased precipitation, changed patterns for drainage and evaporation, rising sea levels, changed land and water use and predicted increase in chemical and microbiological health hazards. In addition, several municipalities are facing limited financial and personnel resources. To uphold a safe and reliable water supply, the inquiry recommended a further regionalization of the Swedish water sector including extended regional planning and coordination as well as increased inter-municipal cooperations (SOU 2016:32, 2016).

Regional cooperations can generate benefits in several ways, but there are also challenges associated with them. Currently, regional cooperations in the water sector take place in several countries and states in Europe, the United States, the Middle East, and North Africa. However, the topic is fairly under-researched and advantages and disadvantages not fully understood (Frone, 2008; Kurki et al., 2016). In addition, water governance in itself can be inherently complex and it is connected to several areas critical for a sustainable development such as health, environment, energy, agriculture and spatial planning (OECD, 2015). Inter-municipal governance, comprising decisions on large scale regional interventions such as decisions on (de)centralizations and on the cooperations themselves, can therefore have big effects on an array of social, environmental and economic aspects. Hence, decision-makers are faced with intricate decision situations when managing our future water supply, not only concerning what to do but also at which level of government.

This literature review was performed to form a basis for research on the effects of regionalization of drinking water supply in Sweden and for developing decision support methods to evaluate the sustainability of reginal water supply interventions.

1.2 Scope and Objectives

The aim of this literature study is three-fold: (1) to give an overview on literature on regional cooperation in the water sector; (2) provide a description of the decision-support methods MCDA and CBA; and (3) present an overview of literature regarding sustainability assessments and the use of MCDA and CBA as decision- support in the water sector.

2 REGIONAL COOPERATION IN THE WATER SECTOR

The most common form of regional cooperation in the Swedish water sector is inter-municipal agreements. These agreements can be reached on almost all kinds of water cooperation, such as joint production and source water use, and the responsibility of one municipality to provide water services to other municipalities. Another form of cooperation is joint committees. The joint committee is comprised in one of the cooperative municipalities' organizations, but it is not a separate legal entity. Each municipality is still responsible for the issues administrated by the joint committee. Yet another form is municipal alliance. The municipal alliance is a public entity that is independent of its member municipalities. The alliance becomes responsible of the issues handed over from the members. And finally, municipalities may also form joint companies. The undertakings given to the company is governed by ownership directives, and a board is responsible for and governs the operations (SOU 2016:32, 2016).

The American Water Works Association (AWWA) encourages regional collaboration in the American water sector and believes it can help the utilities provide safe and reliable water services in a sustainable way. They highlight potential benefits such as knowledge sharing, increased efficiency, minimized capital expenditure and enhanced source protection (AWWA, 2015). As indicated by Frone (2008), the main drivers for regionalization of utilities are typically the potentials of increased efficiency through economies of scale, access to water resources, integrated water resource management, enhanced professional capacity, access to finance and private sector participation and cost sharing between higher- and lower-cost service areas, Figure 1.

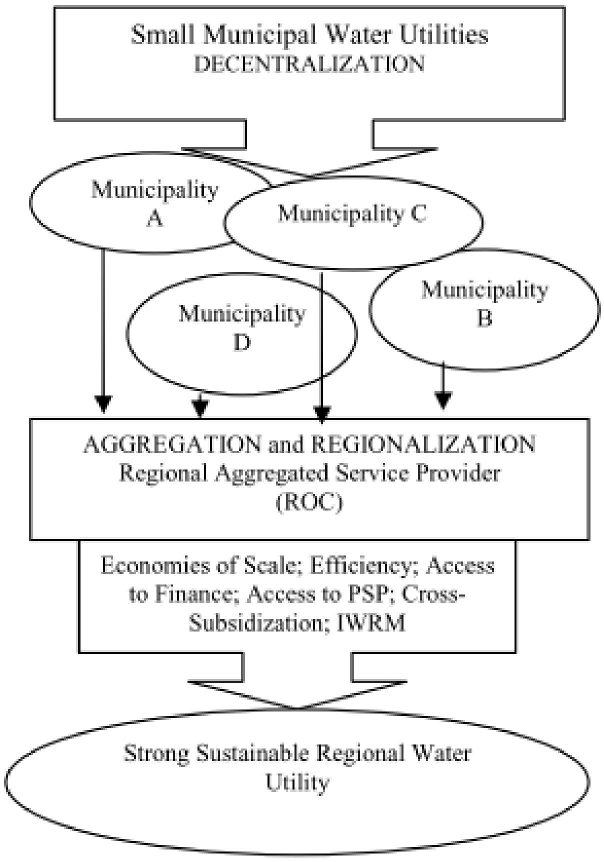

Figure 1 *The process of aggregation and regionalization of water utilities (Frone, 2008)*.

However, Frone (2008) also argues that since the regionalization process many times is seen as too complex and the benefits are not clearly understood, regionalization of water utilities tends to be held back.

Potential advantages and constraints of regionalization are summarized by Frone (2008) in Table 1 and by SOU 2016:32 (2016) in Table 2. Some of the effects and drivers of regionalization found in literature are described further in the text below.

Table 1 Potential benefits and constraints of regionalization (Frone, 2008).

Administrative aggregation and regionalization of water service providers	
Potential benefits	*Potential constraints*
Economies of scale in procurement and support functions; economies of scale in designing works for neighboring towns	Existing installations may limit potential for efficiency gains as they cannot be redesigned; resistance from labor against staff reductions
Better and easier access to water resources in water scarce areas	Lack of incentives to share water; sharing of water access would lead to tariff increase for water-rich municipalities
More integrated approach to water resources management	Administrative boundaries are often not aligned with river basin boundaries; conflicts and lack of coordination between water users
Enhanced professional capacity through transfer of management, technical know-how and expertise	Lack of local recognition of a need for support and potentially higher costs from external support; distance between population centers
Access to banking finance and international donors	Higher risk for municipalities due to joint liabilities for the loans
Access to private sector participation; can be combined with economies of scale to dramatically improve efficiency of operations	Participation of the private sector for the provision of utilities may generate popular and political resistance
Cost sharing between high- and low-cost service areas	Resistance of communities with lower costs to subsidize those with higher costs

Table 2 Pros and cons with municipal and regional responsibility of water supply in Sweden (SOU 2016:32, 2016).

Aspect of water supply	Municipality advantages	Municipality disadvantages	Regional advantages	Regional disadvantages
Planning	Links to other municipality plans	Missing regional perspective	Links to a regional developmental responsibility	Comprehensive task
Financing	Local and participatory	Vulnerable in small municipalities, high taxes	Economies of scale and a more robust size of tax-payers	Difficult for consumers to participate and have influence
Competence provision	-	Difficulties in small municipalities	Economies of scale, facilitates strategic work	New experiences may need to built up
Operation	Local knowledge	Vulnerable in small municipalities	Economies of scale, can cope with the future	New experiences may need to built up
Backup systems and redundancy	-	Inter-municipal cooperations is often a pre-requisite	Economies of scale, flexibility	-
Emergency preparedness	Local knowledge, principle of subsidiarity, participation, responsibility	Consumers in small municipalities are exposed	Economies of scale, links to other regional responsibilities, e.g. health	-

2.1 Economies of scale

One of the major drivers of regionalization, or inter-municipal cooperations, is to generate the effect of economies of scale, i.e. the cost advantage that may arise of an increased production. There are at least two kinds of scale economies in water supply systems. Capital equipment is the one most usually referred to, but there are also scale economies in ordinary business operations, i.e. billing, purchasing and computer systems, and in secondary treatment and testing operations (Shih et al., 2006).

A number of studies have been investigating scale (dis)economies in the water sector, and there is generally a consensus that the water industry has important economies of scale up to a certain output level after which diseconomies of scale appear (Carvalho & Marques, 2016; González-Gómez & García-Rubio, 2008; Saal et al., 2013). Further, several studies have focused on trying to identify an optimal size of service provision. The optimal scale however is found to vary between countries and over time (Nauges & van den Berg, 2008).

In the process of examining scale economies regarding water supply efficiency and productivity a variety of techniques, e.g. econometrics, stochastic frontier techniques, data envelopment analysis, and partial and total factor productivity measures, have been used. So far, the most frequently used method to evaluate efficiency in the water sector has been the econometric approach to estimate cost functions (Abbott & Cohen, 2009). Following are some examples of scale economy studies. For good overviews of additional studies, see also (Abbott & Cohen, 2009; Martins & Fortunato, 2016).

Garcia and Alban (2001) used a multi-product variable cost function to assess the benefit of joint production for 55 water utilities in the French Bordeaux region. They found profitability being highest when a water district was made up of up to 5 local communities.

However, the degree of economies of scale decreased when there were more than two local communities, and there were non-significant diseconomies of scale for water districts larger than 5 local communities. The results also indicated that it was less profitable to merge communities with low population densities, such as semi-urban or rural.

Mizutani and Urakami (2001) tried to find the optimal utility size with respect to minimum average costs by employing total cost models to 112 Japanese water supply utilities. Their results revealed an optimal utility which produced 261 million m^3, had a network length of 1,221 km and a population of about 766,000. Sauer (2005) analyzed the cost structure water suppliers in the rural parts of Germany and found that more than 90% of them were not producing at the optimum point of minimum average costs. The optimal firm size was found to be on average about three times larger than the existing ones. Houtsma (2003) found significant economies of scale when examining water charge data from 459 Californian cities and service areas served by 349 water providers. The average charge levels dropped for communities with population sizes over 10,000, and a further drop was observed for communities with more than 125,000 inhabitants.

Torres and Morrison Paul (2006) analyzed the cost structure of 255 US water utilities. Their estimates revealed that a 1 % higher volume of production, given a constant number of customers and size of service area, on average resulted in cost increases of 0.33 % for small and 0.61 % for large utilities. If the higher output involved a proportional increase in number of customers there were still some cost savings for small utilities. But for the larger utilities the increased costs from more customers counterbalanced economies of volume. If the service areas were also proportionately enlarged, there were significant diseconomies of size for medium-large to large utilities. Overall, their results indicated that merging small utilities could generate cost efficiencies depending on the increase of the network, but merging already large utilities without corresponding increases in output density is not expected to be cost effective.

Shih et al. (2006) used data sets from Community Water System Surveys of 595 American water supply systems to evaluate economies of scale. Their result showed that smaller systems had higher unit production costs across all production inputs: capital, labor, materials, energy, outside services and other costs. Doubling a system's production would lower unit costs by 10 to 30 %. If small systems merged into a larger system, the smaller system's scale could double several times and result in gains of 50 % or more.

2.2 Access to common water resources

Another driver for regionalization is the potential to share unevenly spaced resources. This can be particularly obvious in water scarce areas or areas with insufficient water quality. A predicted shortage was for example one of the drivers which led to the establishment of 10 Regional Water Authorities in England and Wales in 1974, replacing more than 150 water supply systems and 1,300 sewerage agencies (Okun, 1975). Water scarcity in the coastal zones was also a main driver when regional wholesale water companies were formed in Finland (Kurki et al., 2016).

Sharing water resources can also lead to the sharing of costs depending on if tariffs are balanced between low and high cost service areas. This may however also be an obstacle to regionalization, as some municipalities are unwilling to merge with more expensive areas (Frone, 2008).

2.3 Stronger governance skills and enhanced professional capacity

Access to sufficient and right skills is often one of the drivers for regionalization in the water sector. Small municipalities usually have enough personnel to carry out routine activities but are often short of staff to perform high skilled activities such as system planning and design, advanced maintenance and financial management (Frone, 2008; Schmidt, 2014). In a Swedish study, Thomasson (2015) interviewed water utility employees and politicians regarding experience of operating water and wastewater within small and medium-sized municipalities. Many of the challenges in the smaller municipalities were associated with the lack of competence provision. The small municipalities often had a high staff turnover and a working environment that demanded high personnel flexibility and many lonely workhours. The lack of personnel made the small organizations vulnerable to new and unexpected situations.

Several studies argue that forming larger, regional organizations increase the opportunities to employ and retain highly skilled personnel and by that enhance the organization's professionalism (Frone, 2008; Kurki et al., 2016; Lieberherr, 2011). A larger organization is often seen as a more attractive employer due to its career opportunities and due to its own identity or brand, which often is separated from the municipalities'. Exchange of experience within the organization also tends to increase when forming a larger organization and the employees can focus and develop more competence within their area of expertise (Thomasson, 2013). A larger organization can also facilitate pooling of resources between the municipalities and thereby be more cost efficient (Lieberherr, 2011). There is, however, a risk of losing local knowledge when forming a larger joint organization (Kurki et al., 2016).

2.4 Increased autonomy and decreased legitimacy

Inter-municipal decision-making may involve separating political decision-making from operational and management decisions, i.e. autonomization. Since that means that direct voter input becomes lower, autonomization is argued to undermine democratic structures by weakening the democratic legitimacy (Lieberherr, 2011). Autonomization is related to the replacement of vertical government structures with horizontal ones, and therefore the organizational autonomy varies with the different forms of inter-municipal cooperations (Kurki et al., 2016).

Inter-municipal agreements are expected to have the lowest degree of autonomy among the different forms of cooperations since it is operated through municipal utilities by political decision making. Within inter-municipal alliances and companies on the other hand, new organizational entities are formed and the degree of autonomy increases. An inter-municipal company is argued to have a higher degree of autonomy than an alliance (Kurki et al., 2016).

Lieberherr (2011) evaluated how regionalization had affected the performance of water governance in terms of legitimacy, efficiency and effectiveness at the largest utility in Switzerland, in Zurich. The utility had gone through an autonomization process, and become a semi-autonomous public enterprise. There was a positive correlation between autonomization and the utility's performance in terms of 1) clarifying roles and responsibilities, 2) an increase in professional management with more strategic planning and flexibility than before, 3) improved internal interactions in terms of adjustment flexibility and 4) increasing sustainable practices. Yet there was a negative relationship between autonomization and transparency as the public sphere had less oversight and control. The contract municipalities had no decision-rights and hence, this regional form weakened the legitimacy and the direct democratic influence.

When analyzing autonomy and legitimacy in inter-municipal water cooperations in Finland, Kurki et al. (2016) found that the decision-making process was more efficient and less bureaucratic in the more autonomous organizations such as joint companies. The decisions in joint companies could be made without political pressures and party politics. However, citizens recognized the problem of legitimacy and that the decision-making in a municipal water company could move too far away from democratic structures. Water professionals, managers, and authorities on the other hand desired even more autonomy to avoid political debates inside the decision-making process.

3 COST-BENEFIT ANALYSIS

Cost-benefit analysis (CBA) has for long been used to evaluate effects of projects and interventions in a wide range of areas (Johansson & Kriström, 2016). CBA relies on the anthropocentric foundation of neoclassical welfare economics in which benefits are defined as increases in human wellbeing and costs are defined as reductions in human wellbeing. Welfare economics is based on the assumptions that each individual is the best judge of his or her wellbeing at a given situation. Individuals' wellbeing depends on market goods and services as well as non-market goods and services, such as health, environmental quality etc. (Freeman et al., 2014). A project is considered economically profitable when its total benefits are larger than its total costs. The society in this meaning is the sum of individuals, i.e. the aggregated willingness to pay (WTP) for benefits and willingness to accept (WTA) compensation for losses. WTP and WTA should not, according to welfare economic theory, deviate much. However, usually there are quite large deviations where WTA exceeds WTP by far (Pearce et al., 2006).

The concept of total value, or total economic value (TEV), is often used in environmental economics and refers to that ecosystem services not only generate direct and indirect use values and option use values, but also non-use values, such as existence, altruistic and bequest values (Freeman et al., 2014). The TEV is hence the sum of all these values, see Figure 2 for descriptions.

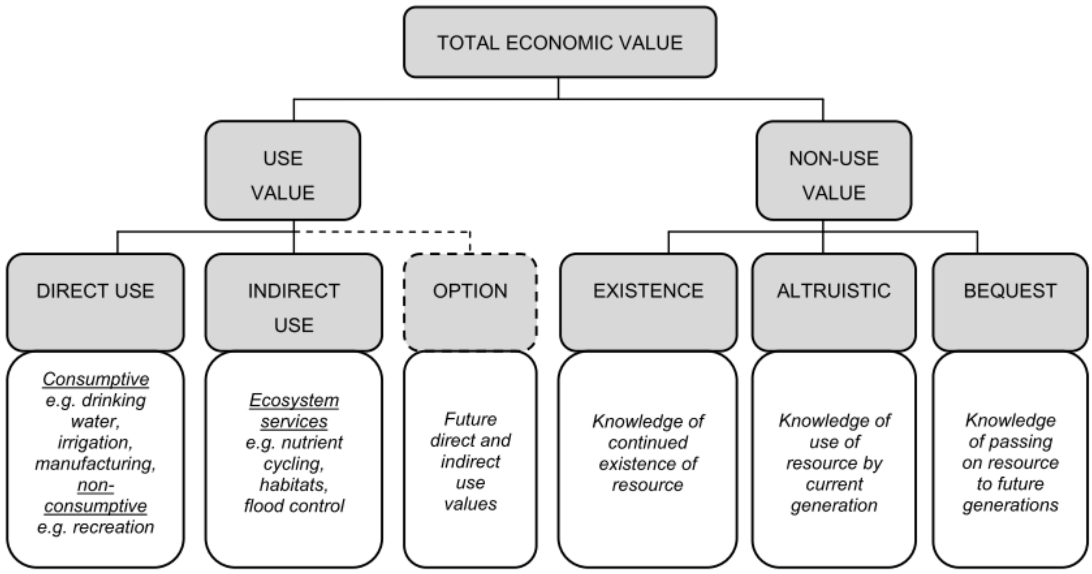

Figure 2 Total economic value of water (CCME, 2010).

There are many economic valuation methods based on welfare theory to estimate these values, Figure 3. Market values on goods and services can for example be used to calculate changes in consumer and producer surpluses to estimate effects on individuals and companies respectively (Kinell & Söderqvist, 2011). Estimations of individuals' values of non-marketed goods and services, often called shadow prices, are usually somewhat more complicated; see further in section 3.2.

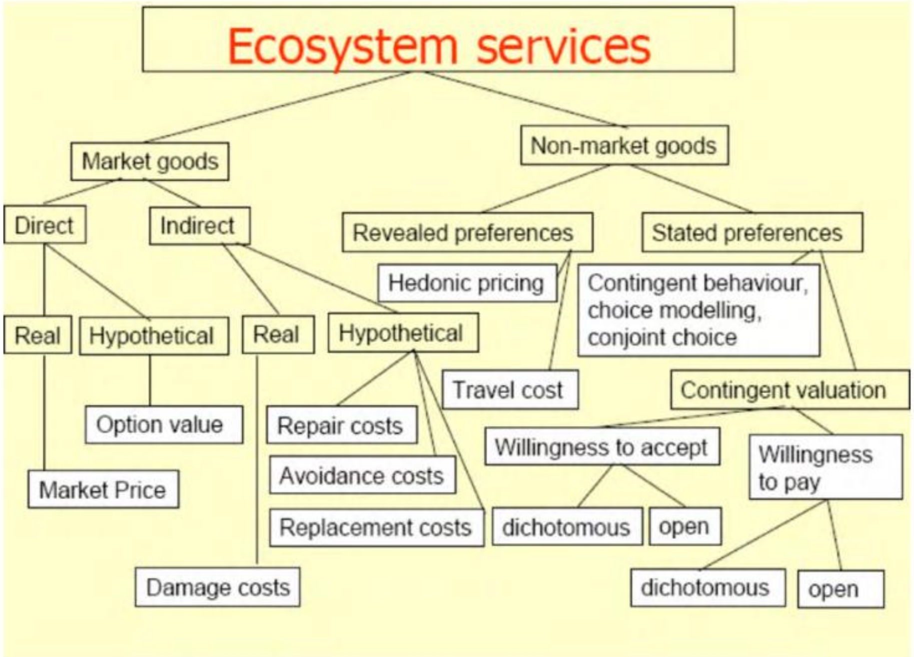

Figure 3 *Economic valuation approaches clustered by objects and methods (Spangenberg & Settele, 2010).*

The aggregation of consequences over time to estimate a present monetary value requires that the costs and benefits are discounted using specified discount rates. The discount rates can be interpreted as the minimum rate of return required to make an investment economically profitable to implement (Gollier, 2011). The decision metric is the net present value (NPV), which is the sum of discounted benefits minus the sum of discounted costs. An investment or project is economically profitable if the NPV is positive. The choice of discount rate illustrates how we value e.g. fairness between generations, and environmental resources versus capital resources etc. For investments with distant benefits, such as climate change mitigations, a low discount rate implies that we are more interested and willing to pay for distant benefits and hence improve the welfare of future generations. More investments will receive positive NPVs with low discount rates than with high discount rates. Hence, a greater portion of our wealth will be invested rather than consumed. There is an extensive literature on the subject of discount rates. Some have suggested declining discount rates to increase the weight devoted to the welfare of future generations (Gollier et al., 2008). In the case of climate change, Nordhaus (2008) argues that a 5 % discount rate would be efficient whereas Stern (2006) used an average discount rate of 1.4%. in the stern Review of Climate Change. The choice of discount rate has ethical and moral aspects that are important to be aware of.

The NPV is calculated as:

$$NPV_a = \sum_{t=0}^{T} \frac{1}{(1+r_t)^t} \left[B_{a,t}\right] - \sum_{t=0}^{T} \frac{1}{(1+r_t)^t} \left[C_{a,t}\right]$$

where a is the alternative intervention, t is the time when benefit or cost occur, T is the time horizon, r_t is the discount rate at time t, C are the costs and B are the benefits in relation to the reference alternative.

CBA is considered a valuable decision support technique because it considers costs and benefits to *all* individuals in the society for which the CBA is carried out; it uses a familiar measurement scale (money) to display the impacts on society; and the valuation techniques are based on people's actual preferences. But there are several critiques of CBA as well.

CBA is for example criticized for relying too much on Kaldor-Hicks compensation, i.e. those that are made better off could hypothetically compensate those that are made worse off so that a Pareto improvement could (but does not have to) be achieved. CBA is also being criticized for the same reason it is valued, i.e. allowing individuals' preferences to be the main decisive factor in informing public decisions (Pearce et al., 2006).

3.1 The steps of CBA

There are several steps an analyst needs to conduct to perform a full CBA, see Figure 4. The first steps typically answer questions regarding: which problem is supposed to be solved; what is the aim with the analysis; which time horizon should the analysis cover; whose consequences are to include; which time preferences should the analysis account for through discounting; what is the reference alternative (i.e. the alternative against which the solutions will be compared); and what alternative solutions are there.

Once the alternative solutions are established, their different consequences in relation to the reference alternative needs to be identified. This is done by means of various forms of expertise, e.g. health experts, limnologists, biologists etc. Before moving on with the most time-consuming part of the analysis, a check point assessment is preferably made to check whether the identified impacts indicate that some alternative solutions need to be adjusted.

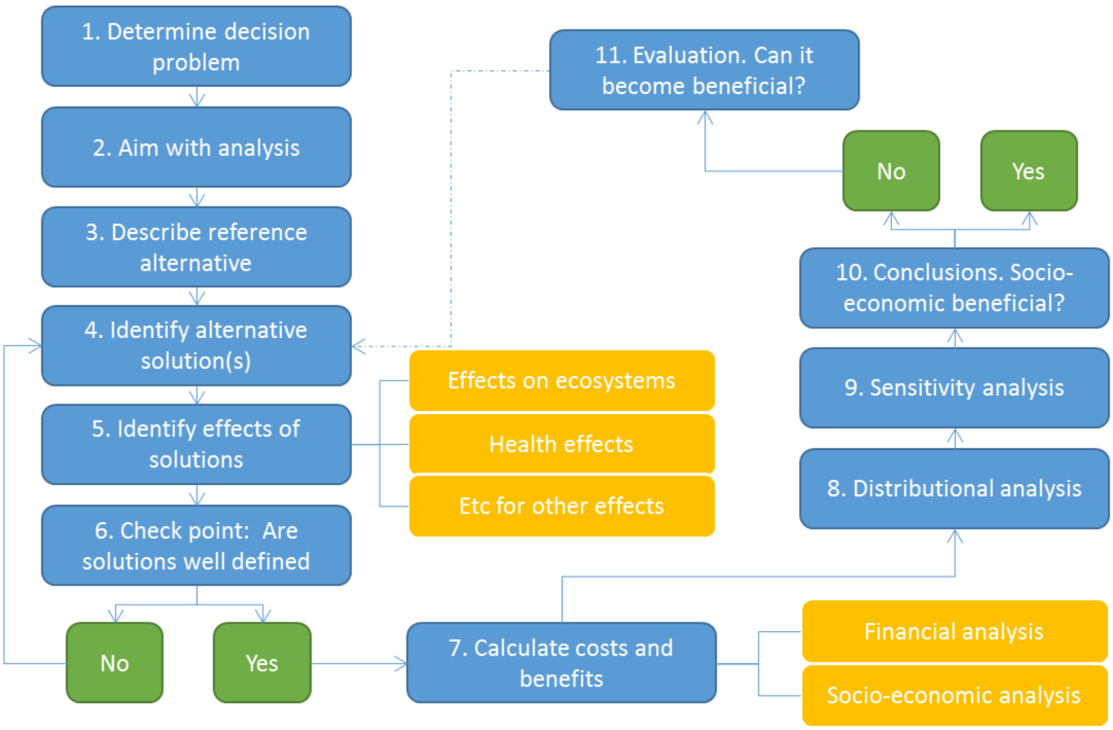

Figure 4 CBA step by step. Adapted from Kriström and Bonta Bergman (2014).

The identified costs and benefits, i.e. the consequences that positively and negatively affect the wellbeing of individuals and companies, can then be calculated. This is, as far as possible, done in terms of monetary units. Given that the costs and benefits rarely are known with certainty, the probabilistic outcomes, and uncertainties, i.e. when probabilities are not known, should also be calculated and included in the analysis (Pearce et al., 2006).

A distributional analysis is then performed to find how the consequences are divided between different groups in society, after which sensitivity analysis is conducted to determine how the CBA outcome is affected by changed conditions and to describe the degree of uncertainty in the results.

3.2 Overview of valuation methods

This section gives and overview of the main economic valuation methods applied in water management, see Table 3. Primary study methods, i.e. methods that generate original valuations, are grouped in: market price methods; production input methods; revealed preference methods; and stated preference methods. There are also secondary study methods, which make use of other existing valuation studies and transfer the value to the decision-making context.

Table 3 Overview of economic valuation methods (CCME, 2010).

Valuation method	**Scope – Component of TEV**	**Scope – types of goods and services**
Market pricing methods	Use value (direct and indirect)	Market goods and services and market substitutes (for non-market goods and services) Direct use value: mostly limited to water as a commodity (e.g. the spending on bottled water as a proxy for the value of drinkable public supply) or the contribution of water to marketed products (e.g. agriculture, forestry, fisheries, manufacturing, power generation) Indirect use value: estimating avoided damage (e.g. from flooding) or marketed substitutes/replacements (e.g. cost of water treatment) or tangible impacts (e.g. cost of illness)
Production input methods (e.g. production function)	Use value (direct and indirect)	Market goods and services Use value: Limited to the role of water as an input to production processes (e.g. the effect of water quality on agriculture).
Revealed preference methods		
Hedonic pricing (e.g. hedonic property pricing)	Use value (direct and indirect)	Non-market goods and services Use value: The contribution of water to environmental amenity that can be observed from markets (e.g. property market).
Travel cost method	Use value (direct and indirect)	Non-market goods and services Use value: The contribution of water to recreation activities that is revealed by the travel costs incurred by recreation users.
Multi-site recreation demand models	Use value (direct and indirect)	Non-market goods and services Use value: The contribution of water to recreation activities that is revealed by the choice decisions (i.e. whether to visit a specific site or not) and travel costs incurred by recreation users.
Stated preference methods		
Contingent valuation	TEV (use and non-use value)	Non-market goods and services TEV: The contribution of water to most non-market goods and services can be captured by contingent valuation.
Choice modeling (e.g. choice experiment)	TEV (use and non-use value)	Non-market goods and services TEV: The contribution of water to most non-market goods and services can be captured by choice modeling approaches.
Benefits transfer		
Unit value transfer / function transfer	TEV (use and non-use value), depending on evidence used	All of the above depending on the type of study from which evidence is sourced.

3.2.1 Market price methods

Market price methods use costs of goods and services which can be directly observed on an actual market. The defensive behavior method and damage cost method are examples of market price methods which assess the WTP by measuring costs for avoiding some negative effect. In the defensive behavior method, WTP is derived from measuring individuals' expenditures to reduce the negative effect, e.g. consumer's expenditure on water bottles to avoid polluted tap water. The defensive behavior method assumes that a rational person will take defensive behaviors if the value of the avoided damage exceeds the cost of the defensive action (Dickie, 2003). In damage cost methods, WTP is estimated by measuring the resource costs incurred by the negative change, including both direct and indirect costs. Direct costs are for example costs of medical visits due to a polluted drinking water (Yong & Loomis, 2014). Indirect costs reflect the opportunity costs of reduced productivity or profit due to the contamination. There are two main differences between the methods: the defensive behavior method measures behavior changes, whereas the damage cost method assumes that there is no behavioral change or at least that it is ineffective; and the defensive behavior method estimates an economic value like WTP, while the damage cost method does not (Dickie, 2003).

The replacement cost method and the substitute cost method are related methods based on the cost of replacing or substituting certain benefits or ecosystem services, such as costs for replacing a raw water resource. These methods, as well as the damage cost method, are based on costs to estimate benefits rather than on what a person is actually willing to pay for that particular benefit. This is based on an assumption that if we spend money on a replacement of some benefit, then that benefit must be worth at least what we paid for replacing it.

3.2.2 Production input methods

Production input methods assess the use value of an environmental resource by its input in production processes, e.g. assessing changes in drinking water production as a result of changes in source water quality (CCME, 2010).

3.2.3 Revealed preference methods

In revealed preference methods, individuals' expenditure choices on market goods are used to assess their WTP to related non-market goods or services. That is, if expenditures vary with the level of the non-market good or service, e.g. an ecosystem service, a valuation can be derived for that ecosystem service. There are however several conditions that must be met to perform a revealed preference analysis, e.g. the changes in expenditure are actual responses on changes in the non-market good or service and not reactions to other variables. Two common types of revealed preference methods are the travel cost method and the hedonic property value method. The travel cost method uses changes in individuals' visits and trips to derive a demand function and calculate consumers' surplus or WTP. The hedonic property value method uses differences in property pricing to assess individuals' values on for example environmental quality (Yong & Loomis, 2014).

3.2.4 Stated preference methods

When related market values are difficult to find, stated preference methods can be used to ask individuals about their WTP or WTA for specific changes in the environmental quality (DCLG, 2009). Two commonly used stated preference methods are the contingent valuation method and choice experiment method. In the contingent valuation method, individuals are asked directly what they would be willing to pay for some positive (environmental) change. In choice experiment s, individuals are presented with consequences and costs of alternative interventions and are asked to rank the interventions or choose the most preferred one. By means of statistical analysis, their WTP for different interventions can then be derived (Yong & Loomis, 2014).

3.2.5 Benefit transfer

The benefit transfer approach makes use of previously performed valuation studies and then transfers the economic values to the area for which a valuation is required. Two common methods are unit value transfer and function transfer. In unit value transfer the estimated WTP from the previous study is directly applied to the area of interest, e.g. in SEK/capita for an improved water quality. The function transfer approach is somewhat more complicated, taking information such as economic and demographic characteristics from the previous study into account to adjust the WTP when transferring the valuation (CCME, 2010).

3.3 Applications of valuation methods

3.3.1 Value of reducing waterborne health risks

The defensive behavior and the damage cost methods are the two approaches normally used to assess benefits of reduced water related health risks (Yong & Loomis, 2014). Dickie (2003) gives a thorough review of both defensive behavior and damage costs in relation to illness.

Abrahams et al. (2000) studied the averting expenditures to avoid illness in response to contamination risks in drinking water by examining choices between bottled, filtered tap, and unfiltered tap water. They incorporated non-health related water quality (taste, odor, and appearance) in the model to assess the possibility of added utility to health benefits from the averting behavior. Their results showed that averting cost estimates using bottled water may overstate the purely health-related benefits.

Harrington et al. (1991) studied both damage costs and defensive behavior in a giardiasis waterborne outbreak affecting several thousand people in Pennsylvania 1983-1984. Their hypothesis was that the WTP for avoiding acute illness was equal to medical costs and lost earnings.

The Swedish National Institute of Economic Research used an aggregation of damage costs to exemplify how to estimate changes in waterborne health risks. Assessed WTPs to avoid a day with symptoms common to gastrointestinal infections (nausea, headache, cramps and stomach ache and diarrhea) were combined with average medical costs of gastrointestinal infections and costs of direct and indirect productivity losses (Johansson & Forslund, 2009).

Disability-adjusted life years (DALYs) and quality-adjusted life-years (QALYs) are two other commonly used metrics used in health related valuations (Bergion, 2017; Sassi, 2006; WHO, 2016). QALY focuses on how the quality of life changes whereas DALY focuses on the functional status, i.e. how impaired a person's functional ability is compared to full functional ability. Hultkrantz and Svensson (2012) found a WTP per QALY in Sweden of about 1.2 million SEK by dividing median values of statistical life (VSL) from 27 published VSL estimates with the discounted quality-adjusted expected life-expectancy using a discount rate of 3%. Svensson et al. (2015) used data on reimbursement decisions on new pharmaceuticals in Sweden and found the lowest cost per QALY of declined reimbursements was 0.7 million SEK whereas the highest cost per QALY of approved reimbursements was 1.22 million SEK.

3.3.2 Value of residential water

Residential water values can be measured either as at-source values, i.e. the derived demand for untreated source waters in a water resource, or as at-site values, i.e. the WTP at the point of use i.e. in households. The at-site value captures however not only the value of water, but also the value of extracting, treating, transporting and storing the water. Most studies have used econometric techniques to analyze secondary residential demand data to derive at-site values. From these at-site values, it is possible to calculate at-source values by deducting values for extracting, treating, transporting and storing the water (Yong & Loomis, 2014).

The residential water demand function can be represented by the demand curve, Figure 5, or as:

$$Q_W = Q_W(P_W, P_a, Y, Z)$$

where Q_W is the individual's water use in a given time period, P_W is the price of water, P_a is the price of alternative source, Y is the individual's income, and Z represents factors such as climate and preferences etc.

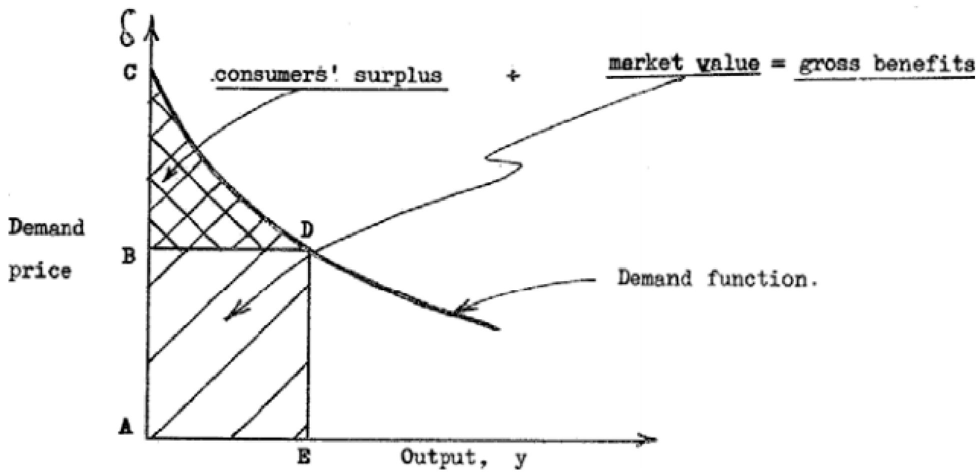

Figure 5 Demand function of WTP for water at different quantities (Bear et al., 1964; Harou et al., 2009).

The demand curve presents the consumer's WTP for varying quantities of water. The y-axis is the unit water price or marginal WTP, and the x-axis is the available water quantity. The area under the demand curve is the market value (ABDE) and the consumer surplus (BCD). The sum of market value and consumer surplus is the gross benefits of residential water delivery (Harou et al., 2009).

3.3.3 Value of water quality

Groundwater and surface water resources deliver several ecosystem services; see examples of groundwater ecosystem services in Figure 6. Examples of studies focusing on estimating the economic benefits of improving or maintaining the water quality with respect to drinking water are presented here.

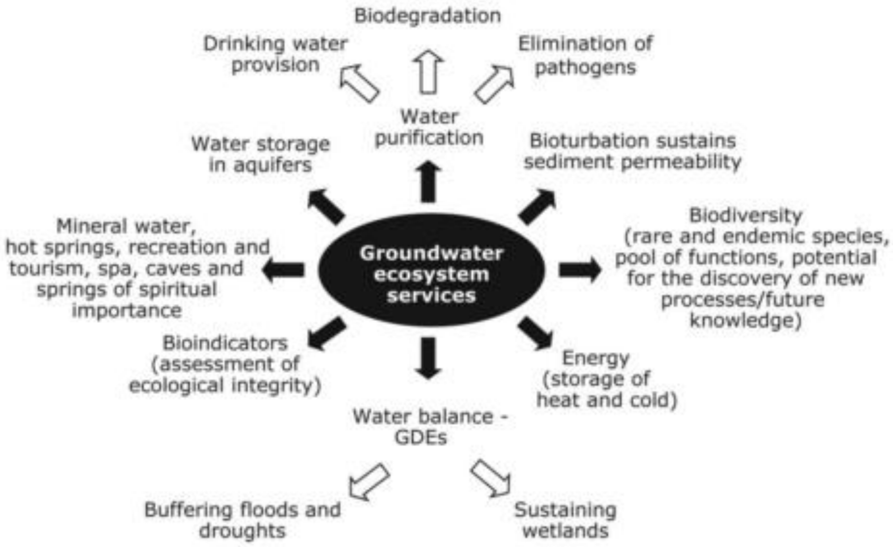

Figure 6 Groundwater ecosystem services (Griebler & Avramov, 2015).

Göransson (2008) estimated the value of clean water in the areas surrounding Kristianstad, Sweden, which has one of Northern Europe's largest groundwater resources. The value was calculated using replacement costs for the municipal water supply based on a scenario with groundwater pollution. Three methods were considered to deal with the polluted groundwater: treatment of groundwater; use of surface water; and replacement with other groundwater resources. The costs to replace the public drinking water supply using these methods were estimated to 50, 200 and 500 million SEK, respectively.

ten Brink et al. (2011) estimated benefits of Natura 2000 networks as protection for drinking water resources in terms of treatment and provision. For the four cities of Berlin, Vienna, Oslo and Munich, the protected areas were estimated to lead to benefits between €15 and €45 per capita per year. In the Aquamoney project, the willingness to pay to protect a large groundwater body in France against quantitative overuse was estimated through contingent valuation to around 40 € per household and year (Rinaudo, 2008).

In the Drastrup project in Denmark, the value of a clean groundwater for water supply was calculated by means of remediation costs for pesticides and nitrates in the water resource. If no remediation occurred, the groundwater was assumed to be unusable for water supply within 10 years. Sanitation costs were estimated to 1.41 DKR /m^3, which was put in relation to a 2.2 Mm^3 extraction for water supply per year. With the infinite time horizon and discount rate of 3%, the net present value of clean groundwater was estimated to SEK 80 million DKR (Dubgaard, 2003).

Hasler et al. (2005) used the choice experiment method to assess the benefits of groundwater protection compared to treatment of polluted groundwater for drinking water supply in Denmark. The willingness to pay for a naturally clean groundwater was assessed to around DKK 1900 per year and household compared to a willingness to pay for treated groundwater of about 900 DKK per household and year.

Elsin et al. (2010) used two benefit transfer approaches, value transfer and function transfer, to estimate cost reductions for drinking water production due to source water quality. Production costs and source water quality data from eight water treatment plants in the Neuse River Basin in North Carolina were used. Turbidity in source water was used as a proxy for water quality. A 30% improvement in water quality was estimated to generate a mean net present value \$2.7 million to \$16.6 million in cost reduction for the entire Neuse Basin over a 30-year period.

Löfmarck and Svensson (2014) used the Simpler method to assess economic values of clean water in the drinking water source waters Lake Vomb and Lake Mälaren in Sweden. The method was used to calculate the added value produced by actors around the water resources. The total value produced around Lake Vomb was estimated to about 1.6 billion SEK per year and to about 127 billion SEK per year around Lake Mälaren.

See also following references for more studies on valuations of water resources (Barton et al., 2011; Barton et al., 2009; Hasler et al., 2009; Hasselström et al., 2014; Johnston et al., 2003; Kettunen et al., 2012; Koundouri et al., 2013; Morrison, 2010; Poe et al., 2000; Söderqvist et al., 2014; Van Houtven et al., 2007).

3.3.4 Value of water supply reliability

Consumers' value of drinking water reliability is not captured in a conventional water demand function, since it assumes full reliability. In developing countries households' WTP for water reliability have been assessed through investments in home storage tanks, illegal extractions from distribution systems, and investments in wells (Yong & Loomis, 2014).

There is however no large empirical literature on the value of water reliability in developed countries (Griffin & Mjelde, 2000). Some studies have focused on assessing WTP during droughts. Koss and Khawajab (2001) used the contingent valuation method to ask consumers in California of their WTP for avoiding water shortages of a certain frequency. Their results indicated that consumers WTP ranged from \$12 to avoid a 10% shortage every ten years to around \$17/month to avoid a 50% reduction every 20 years, see Table 4.

Table 4 Mean monthly WTP (additional \$/month) (Koss & Khawajab, 2001).

Shortage (% reduction from full service)	Frequency (occurrences/years)				
	1/30	1/20	1/10	1/5	1/3
10			\$11.67	\$12.00	\$12.14
20	\$11.71	\$12.39	\$13.08		
30	\$13.13	\$13.84	\$14.56		
40	\$14.61	\$15.35	\$16.10		
50	\$16.15	\$16.92			

Buck et al. (2016) evaluated welfare losses from urban water supply disruptions by calculating shortage losses for Californian water utilities using water rates and utility-specific price elasticities. Their results indicated an average welfare loss of \$1,458 to \$3,426 per acre-foot of shortage (\$1.18 – \$2.78 per m^3), for a 10% and 30% supply disruption respectively. The WTP to avoid supply disruptions ranged from \$60 to \$600 per household depending on the duration and location of the disruption.

In a Swedish study, Törneke and Engman (2009) used interviews with representatives from different sectors in society to assess potential economic costs of a total water supply outage in two fictive municipalities, of 20,000 and 60,000 inhabitants respectively. The outage was assumed to last for 48 hours corresponding to the repair time of a severe pipe failure, the most common cause of delivery failure in Sweden. According to the interviews, the preparedness for adapting various activities to a water outage was very small. The total economic costs were estimated to 7 and 80 million SEK respectively in the municipalities. The large difference was due to major costs incurring in industry, health care and district heating in the larger municipality.

In the research area of natural disasters however, several studies have focused on assessing the economic values of water supply shortages and total outages. A variety of methods have been used to study the impacts of water supply disruptions at different levels of shortage and duration, including surveys, field observations, analysis of secondary data, and computational models (Chang, 2016; Rose et al., 2012).

Disruptions have been shown to cause different impacts across businesses and economic sectors. To estimate these differences the Applied Technology Council (ATC, 1991) used expert elicitation methods to derive so called importance factors, which corresponds to the value added lost for each economic sector due to a disruption. Chang et al. (2002) developed empirically based resilience factors, partly calibrated by data from the 1994 Northridge and 1995 Kobe earthquakes, to estimate direct financial losses of various water outage duration periods. The resiliency factors were defined as the remaining percentage output that an industry could still produce in the event of total water outage.

Different approaches have been used to estimate residential welfare losses, i.e. willingness to pay to pay (WTP) to avoid a water supply disruption of certain duration, e.g. contingent valuation (Barakat and Chamberlin, 1994; Griffin & Mjelde, 2000; Howe et al., 1994), mathematical programming, and integration of estimated demand curves (Brozović et al., 2007). Brozović et al. (2007) proposed methodologies for estimating economic impacts on both businesses and residential users of water supply disruptions. The method to assess the residential welfare loss was adapted from (Jenkins et al., 2003) using integration of the consumers demand curve for water services calibrated to local water prices and quantity data. The method to assess impact on businesses assumed that the only economic effect on businesses of water outage or shortage is lost revenue and that business output can be immediately adapted to existing water provision. Marginal losses were assumed to increase up to a certain water availability limit where the business activity was shut down. Water shortage beyond shutdown was assumed to cause no additional economic impact since the business activity had ceased.

The current standard cost estimations used by the US Federal Emergency Management Agency (FEMA, 2011) of loss of water service are based on Brozović et al. (2007) method for estimating the economic impact on residential consumers, and the water importance factors developed by ATC (1991) for estimating the impact on businesses.

4 MULTI-CRITERIA DECISION ANALYSIS

The MCDA approach is often used for complex decision problems with large amount of information and when several, possibly contradicting, views needs to be considered in a coherent way. The aim is to support decision-makers in such decision-situations. There is typically not one single optimal solution and it is therefore necessary to make use of the decision-maker’s preferences to distinguish between the alternatives. MCDA can for example be used to rank alternative interventions, find the unacceptable alternatives, and identify alternatives that need more detailed assessments. MCDA provides a means for integrating quantitative, semi-quantitative and qualitative information concerning alternative interventions, and it allows for comparison and tradeoffs between objectives (Lindhe et al., 2013; Rosén et al., 2015).

In the MCDA approach, large emphasis is placed on the judgement of the decision-making team and involved stakeholders to establish the objectives and criteria, to assess the relative importance between the criteria, and to decide whether trade-offs between criteria are allowed. There are a limited number of non-compensatory techniques to be used if trade-offs are not acceptable. If trade-offs are tolerated, a number of different MCDA techniques can be used to aggregate each alternative’s performance across the criteria.

Some key features often highlighted with MCDA are summarized in Figure 7.

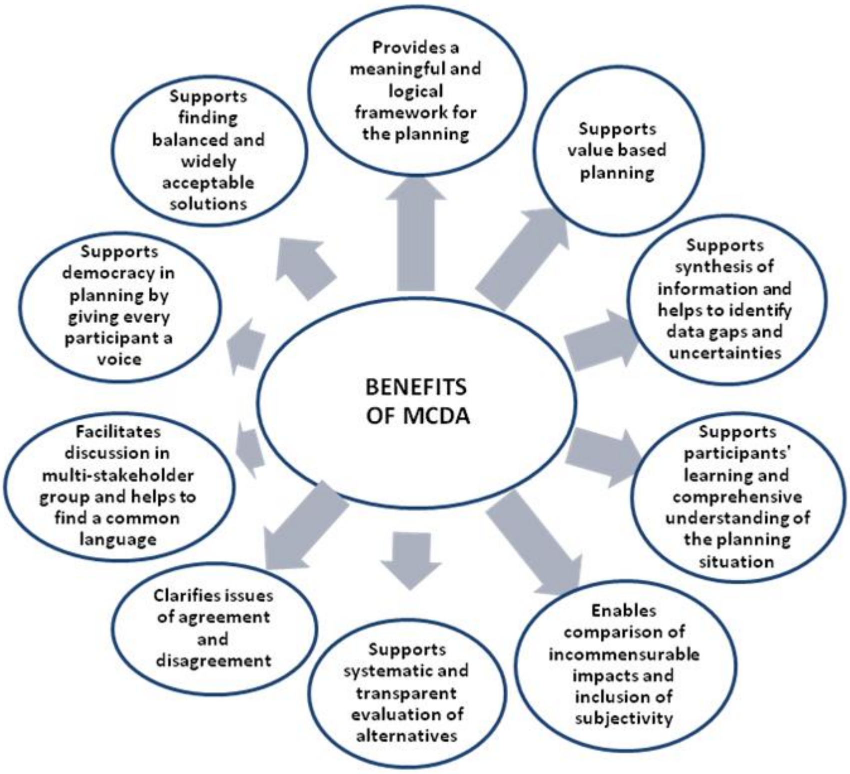

Figure 7 Key features of MCDA (Marttunen, 2010).

There are however critiques of the MCDA approach as well. One critique concerns the fact that preferences are normally elicited from a relatively small group of decision-makers and stakeholders, and not, as in CBA, aggregating preferences of all individuals in the society. Hanley (2001) even argues that this lack of broader inclusiveness can make decisions made with aid of MCDA fail on the ground of representativeness and democracy. MCDA should therefore be implemented so that the relevant groups in society are included and represented as widely as possible, and that the assessments are not only based on expert elicitations. This is particularly important when it comes to people's values and perceptions, as well as many social aspects.

4.1 The steps of MCDA

A brief description of the steps normally included in a MCDA are presented here (DCLG, 2009).

1. Establish the decision context.
   - Establish aims of the MCDA, and identify decision makers and other key players.
   - Design the socio-technical system for conducting the MCDA.
   - Consider the context of the appraisal.
2. Identify the options to be appraised.
3. Identify objectives and criteria.
   - Identify criteria for assessing the consequences of each option.
   - Organize the criteria by clustering them under high-level and lower-level objectives in a hierarchy.
4. 'Scoring'. Assess the expected performance of each option against the criteria. Then assess the value associated with the consequences of each option for each criterion.
   - Describe the consequences of the options.
   - Score the options on the criteria.
   - Check the consistency of the scores on each criterion.
5. 'Weighting'. Assign weights for each of the criterion to reflect their relative importance to the decision.
6. Combine the weights and scores for each option to derive an overall value.
   - Calculate overall weighted scores at each level in the hierarchy.
   - Calculate overall weighted scores.
7. Examine the results.
8. Sensitivity analysis.
   - Conduct a sensitivity analysis: do other preferences or weights affect the overall ordering of the options?
   - Look at the advantage and disadvantages of selected options, and compare pairs of options.
   - Create possible new options that might be better than those evaluated.

The first two steps focus on determining the decision context, objectives, and stakeholders, as well as which alternative solutions that might meet the goals and objectives. Once that is settled, the evaluation criteria need to be determined. The criteria are measurable objectives serving as performance measures in an MCDA. Hence, they need to be operational so that an expert judgement or a data measure can state how well an alternative perform in relation to a specific criterion. The number of criteria should be kept low, while still providing as complete foundation as possible for a well-informed decision. Apart from being operational and complete, the criteria also need to be mutual preference independent, i.e. the judged performance of one alternative on one criterion is independent of its judged performance on another criterion, and they need to be set up to avoid double counting (DCLG, 2009).

To score the alternatives performance against the criteria, the criteria need some sort of performance scales. A common way is to value the scores on a global interval scale from e.g. 0 to 100 where 0 represents the worst possible performance in such a decision problem, and 100 representing the best possible performance. Another way is to measure the criteria on a local scale, where the best performing alternative is given 100 scores and the worst performing alternative is given 0 scores for a certain criterion (Monat, 2009). Different MCDA approaches are described briefly below.

The scores can be assigned to the alternatives in three different ways: by using a value function to transform a measurement of the specific criterion to a score; by using expert opinions and judgements to assess the alternatives performance, i.e. direct rating; or by pairwise assessments by experts on how each alternative perform relative to the other alternatives.

Each criterion is then assigned a weight, reflecting that criterion's relative importance to the other criteria. There are several methods to elicit the weights from stakeholders and decision-makers. One method called swing weighting is based on comparisons of differences on the preference scales between the different criteria.

The weights and scores are then combined for each alternative solution. The most commonly used method is to calculate the weighted average of scores. The results can hence show the most preferred alternative as well as a ranking between the alternatives. A sensitivity analysis is then used to assess how the ranking is affected by different weighing and scoring.

4.2 Overview of methods

There are several different approaches to solve MCDA problems. Greco et al. (2004) and Slowinski et al. (2002) suggested a grouping of the approaches based on: the utility based theory, i.e. methods which aggregate the information into a unique parameter (performance aggregation based approaches); the outranking relation theory, i.e. methods based on pairwise comparisons (preference aggregation based approaches); and the decision rule theory, i.e. methods which derive a preference model based on classifications or comparisons of decision examples (Cinelli et al., 2014). A brief description of three MCDA approaches is provided here.

4.1.1 Multi attribute utility theory (MAUT)

MAUT is a performance aggregation based approach, in which utility (or value) functions and weights are elicited for each criterion. Those are then aggregated to derive a unique synthesized criterion. Several aggregation techniques are available, of which the linear additive aggregation (Keeney & Raiffa, 1993) is the most commonly applied (de Montis et al., 2005).

4.1.2 Analytical hierarchy process (AHP)

The analytical hierarchy process is a performance aggregation based approach, in which the decision problem is broken down into a hierarchy of evaluation criteria (value tree), see Figure 8. The decision-makers can then systematically evaluate the alternatives' performance on the criteria (scoring) and the criteria importance (weighting), by pair-wise comparisons with respect to their impact on elements above them in the hierarchy (Belton & Stewart, 2002). Different aggregation techniques can be used to calculate the overall performances of the alternatives (Ossadnik et al., 2015).

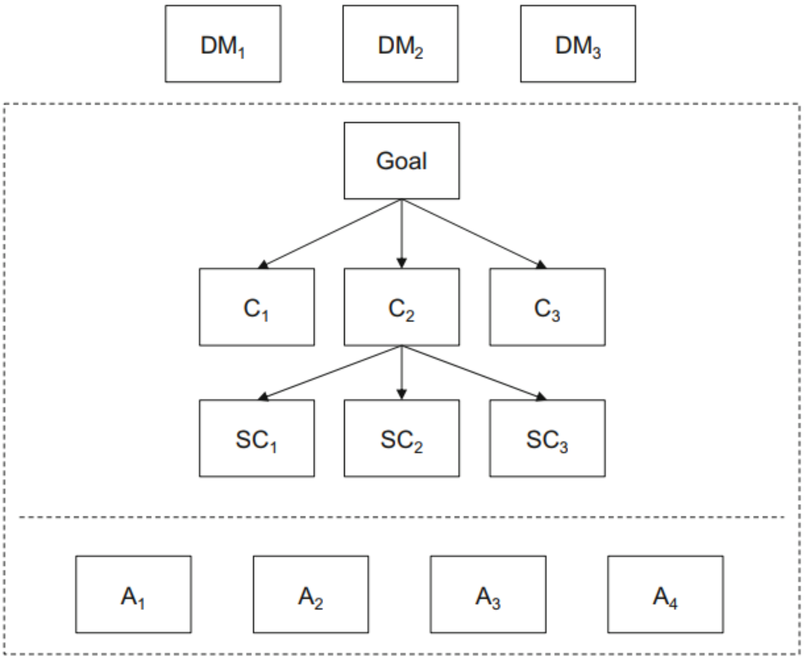

Figure 8 The structure of the AHP model, in which DM are the decision-makers, C are the criteria in the decision hierarchy, and SC are the operationalized sub-criteria for evaluations of alternatives A (Ossadnik et al., 2015).

4.1.3 Elimination and choice expressing the reality (ELECTRE)

ELECTRE are preference aggregation based methods, grounded on pair-wise comparisons of alternatives. The ELECTRE methods are also referred to as outranking methods since the aim is to evaluate whether an alternative is at least as good as another alternative (Cinelli et al., 2014). The ELECTRE methods consist of two main parts: the construction of outranking relations based on concordance and disconcordance indexes; and recommendations based the outranking relations. The ELECTRE methods can be based on choosing, ranking or sorting. The four preference situations handled by ELECTRE are organized by indifference, strict preference, weak preference and incomparability (Figueira et al., 2013).

5 SUSTAINABILITY ASSESSMENTS

5.1 Sustainability definitions

There is a wide range of definitions of sustainability and sustainable development. One of the most widely used is that of the Brundtland Report, in which it is defined as a *development that meets the needs of the present without compromising the ability of future generations to meet their own needs* (WCED, 1987). The International Union for the Conservation of Nature defined sustainability as the *development that improves the quality of human life while living within the carrying capacity of supporting ecosystems* (IUCN, 1991). Although there are many definitions of sustainability, nearly all contain some perception of the future (Loucks, 1997) and that human society and economy are intimately connected to the natural environment (Caradonna, 2014). The sustainability concept as put forward in the Brundtland Report has an anthropocentric point of departure, which basically deals with fairness between generations.

In the scope of water services, Gleick (2000) defined a sustainable water use as *the use of water that supports the ability of human society to endure and flourish into the indefinite future without undermining the integrity of the hydrological cycle or the ecological systems that depend on it*. ASCE and UNESCO (1998) proposed the following definition of sustainability for water resource systems *those water resource systems designed and managed to fully contribute to the objectives of society, now and in the future, while maintaining their ecological, environmental and hydrological integrity*. And the TRUST project defined that *Sustainability in urban water cycle services (UWCS) is met when the quality of assets and governance of the services is sufficient to actively secure the water sector's needed contributions to urban social, environmental and economic development in a way that meets the needs of the present without compromising the ability of future generations to meet their own needs* (Brattebø et al., 2013).

Generally, sustainable development is most often associated with issues related to the economic, social and environmental dimensions, which has become known as the Triple Bottom Line (TBL) (Mihelcic et al., 2003), see examples of sustainability models in Figure 9. The TBL was first mentioned by Elkington (1997) to consider financial, social and environmental effects in the decision-making process.

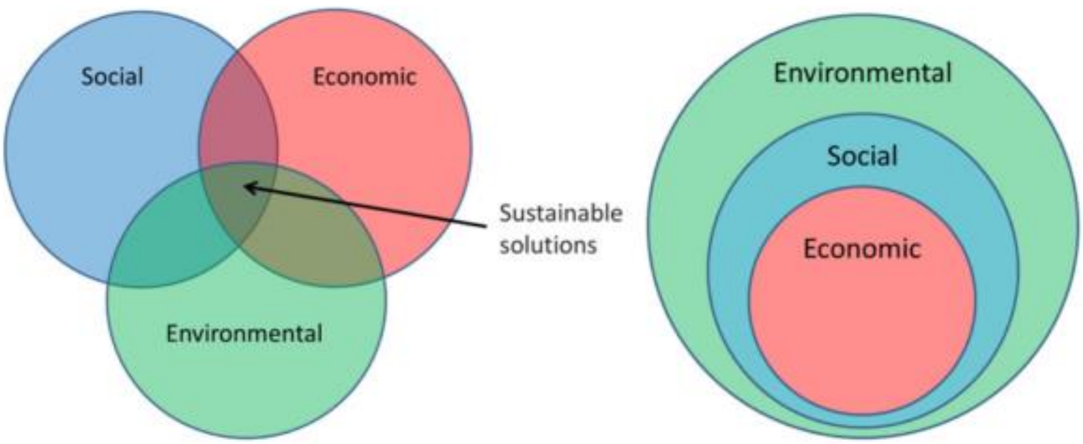

Figure 9 Two sustainability models based on the TBL, "Venn diagram" model (left) and "Bull's eye" model (right) (Rosén et al., 2015).

Based on the concepts of sustainability, different guidelines and frameworks have been developed to assess the sustainability of organizations, e.g. Global Reporting Initiative (GRI), Principles for Responsible Investment, and International Integrated Reporting Council (Pryor, 2016). Even though frameworks such as GRI has become important for water utilities to manage their impact on sustainability (Marques et al., 2015), the guidelines are rarely used by water planners because of lack of applicability and availability of data (Rathnayaka et al., 2016).

5.2 Sustainability approaches in water supply management

A significant number of studies have focused on evaluating the sustainability of water supply interventions using different evaluation methods such as MCDA (Rygaard et al., 2014; Sapkota et al., 2015), CBA (Mukheibir & Mitchell, 2011), life cycle assessments (Lundin & Morrison, 2002), and optimization techniques (Lim et al., 2010). Lai et al. (2008) summoned strengths and weaknesses of four approaches used in sustainability assessments in the water sector; see Table 5 for their assessments of CBA, MCA, and Integrated assessment (IA).

Table 5 Common features of four integrated approaches used in sustainability assessments. From Lai et al. (2008).

Method	Philosophy	Strengths	Weaknesses	Stakeholder engagement	Common features
CBA	The costs and benefits of an option are converted into monetary terms. Comparison is made on relative costs/benefits. It also involves discounting these values over the life time of the system into present values based on some predetermined discount rate to reflect the way humans value their goods	CBA has proven to be useful because of the one single aggregated result obtained which helps to clarify and provide information about the costs and benefits of alternatives, highlighting the tradeoffs.	There is a high level of ambiguity and uncertainty in translating value judgment into monetary values. Some of these impacts cannot be priced according to market values.	CBA is itself an analytical tool and not a framework for incorporating public participation, but part of the process of conducting CBA, social perception and values may involve public consultation.	-A tool for quantifying externalities or intangible matters -Highlights tradeoffs -Single cost criterion approach
MCA	As a decision analysis tool, Multi-Criteria Analysis (MCA) is a structured approach with a set of procedures to follow for aiding decision-making when dealing with more than a single criterion	It provides a structured framework to deal with multi-criteria problems which are often very complex issues.	It requires considerable cognitive effort from decision makers to make judgments, especially in setting up preference models.	MCA generally requires a problem structuring method that encompasses the process of stakeholder engagement.	-An integrative structured framework -Multiple criterion approach -Variant branches exist -Various approaches to conduct integration
IA	IA is structured process of dealing with complex issues, using	It is useful for framing issues and as a communication	It is a relatively new structured discipline. It is	As various scientific disciplines and stakeholders	-An interactive, transparent

	knowledge from various scientific disciplines and/or stakeholders, such that integrated insights are made available to decision makers.	tool between scientists and decision makers. IA operates on a variety of levels and scales, and thus diverse methods can be used not limited to technical modelling.	still mostly qualitative in nature without a robust model.	are drawn together in the process of IA, the process itself can become itself a stakeholder engagement framework.	framework -A process enriched by public participation -Linking of research to policy -An iterative, adaptive approach -Recognize missing knowledge

5.3 Sustainability criteria

The incorporation of sustainability in water supply decision-making requires assessments of the social, economic and environmental consequences of alternative interventions. This, in turn, requires sustainability criteria to assess whether the alternative intervention is likely to move the system towards or away from sustainability (Foxon et al., 2002), see Figure 10 for hierarchy among criteria.

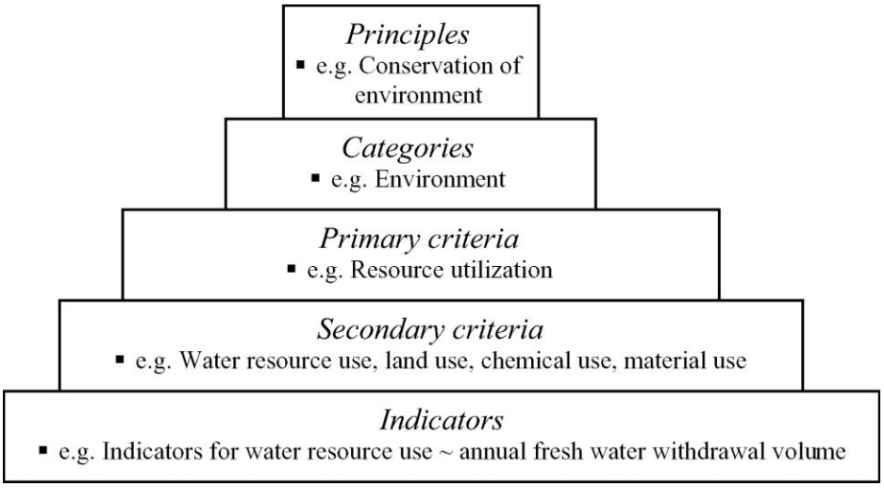

*Figure 10 Hierarchy of sustainability criteria (Lai et al., 2008)***.**

Some studies, e.g. Ashley et al. (2004), argue that environmental, social and economic dimensions are insufficient to evaluate sustainability for water services. Hence, several different sustainability dimensions and evaluation criteria have been proposed for the water sector. Rathnayaka et al. (2016) reviewed sustainability evaluation criteria used in water supply and demand management studies between 2000 and 2016, see Table 6. Rathnayaka et al. (2016) concludes that the environmental, social and economic dimensions are still the main categories for evaluating sustainability, of which social sustainability is given the least attention in the literature. They also recognize that cost externalities are not common in the water sustainability literature.

Table 6 *Evaluation criteria utilized in literature to assess sustainability of water supply and demand management options. From Rathnayaka et al. (2016).*

	Objectives	**Evaluation criteria**
Environmental criteria	River and waterbody health	Quality of waste water produced and their impacts (contribution to acidification and eutrophication, effects on flora and fauna) Quantity of wastewater produced Storm water runoff
	Maintain river, local creaks, and wetlands	Effect on environmental flow and surface water Freshwater/portable water saved Effects on groundwater level and pattern (ground water infiltration, recharge, and depletion)
	Protect land ecosystem	Effects on fauna and flora/biodiversity Effects on habitats and protected natural habitat area Land cover change effects (e.g. habitats affected) Solid waste quantity and quality (e.g. sludge)
	Protect atmospheric ecosystem	Greenhouse gas and other emissions Photochemical oxidant formation Other pollutants (e.g. dust, noise)
	Efficient resource use	Energy use and recovery Ability to use renewable energy source(s) Fresh water use Land use Materials for construction Chemical use Reuse and recycling of resources
Social criteria	Ability to meet user acceptance	User acceptance in terms of water quality Willingness to accept demand management options Acceptance of increase/decrease in water bill User awareness and involvement
	Ability to meet community acceptance	Recreational values (visual amenity) Impacts on urban heat island effect Provision of educational opportunities Small scale flood mitigation benefits Odor/pests—any other negative impacts on the local community Number of jobs it creates
	Health and hygiene	Safety (number of incidents/accidents) Risk of infections (number of outbreaks/people affected) Risk of other health hazards (presence of carcinogenic compounds in influent water) Exposure to toxic components (Cd, Hg, Pb) in operation
	Political approval	Project duration (e.g. design and construction phase) Management/institutional effectiveness and efficiency Uncertainty of volume, timing, cost, approval, and delivery State of readiness (availability of institution, documents, policy) Ability to meet environmental or other regulations
Economic criteria	Total direct cost	Capital cost Maintenance cost Operational cost including energy and other costs Disposal cost Cost of water distribution-construction, maintenance, and operation Cost of water storage—construction, maintenance, and operation
	Total indirect cost	Value of hydropower/energy and other byproducts, such as fertilizer
Risk-based criteria	Reliability	Probability of supply shortfalls (chance of not meeting the expected production)
	Vulnerability	Magnitude of failure
	Resilience	Failure duration or how quickly system returns to its satisfactory state after a failure

	Robustness	Ability to perform satisfactorily under a range of system changes (e.g. climate)
Functional criteria	Flexibility of the option	End-uses it can fit Flexibility in scaling Capacity/Yield Potential for growth
	Construction flexibility	Challenges with management of site (presence of contaminated soil and underground services) Ability to blend with available supplies/infrastructure
	Operational and maintenance flexibility	Ease of maintenance including monitoring frequency based on water quality and quantity Technical knowledge needed in handling the system
	Durability	Life span of the water supply infrastructure/option
	Interactions between the system components	Effects on sewer distribution network such as sewer blockage, odor, and corrosion Effects on drainage distribution network Effects on water supply network (e.g. size of pipe)

5.4 Applications of sustainability assessments in the water sector

As discussed earlier, several different evaluation methods and evaluation criteria have been used in sustainability assessments in the water sector. Some examples of frameworks, guidelines, tools and methods used in the water sector are presented here.

One of the earlier research studies on sustainability in the water sector was the Swedish research program “Sustainable Urban Water Management”, see schematic description of their systems analysis in Figure 11. A set of sustainability criteria covering health and hygiene, social and cultural aspects, environmental aspects, economy and technical aspects was developed, along with a set of suggested indicators for the criteria (Hellström et al., 2000).

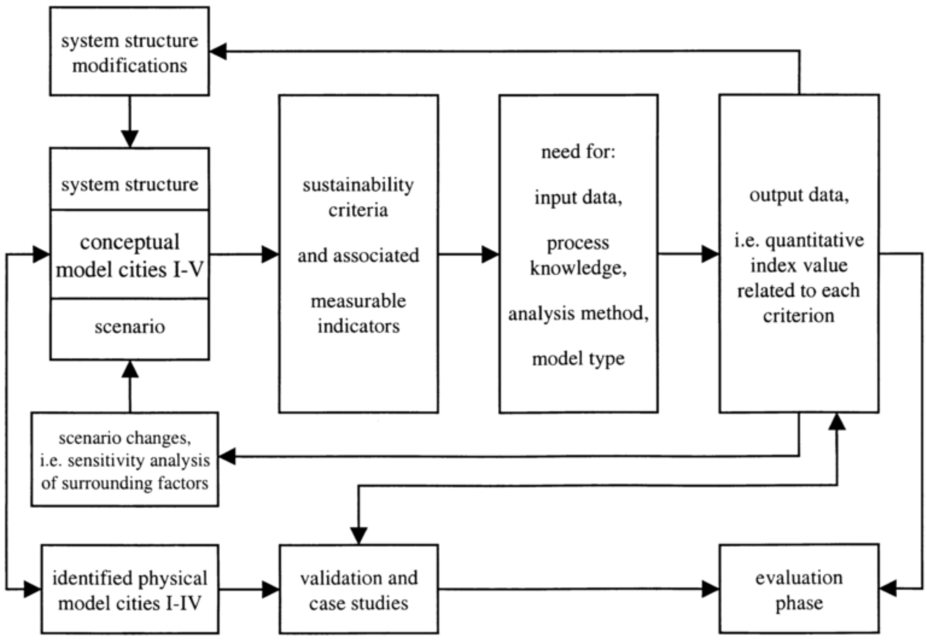

Figure 11 *Schematic description of the systems analysis work procedure within “Sustainable Urban Water Management” (Hellström et al., 2000).*

Mitchell (2006) discussed the concept of integrated urban water management (IUWM) as a way towards sustainable urban water services. According to the author, the principles of IUWM can be summarized as follows: 1) Consider all parts of the water cycle, natural and constructed, surface and subsurface, recognizing them as an integrated system; 2) Consider all requirements for water, both anthropogenic and ecological; 3) Consider the local context, accounting for environmental, social, cultural, and economic perspectives; 4) Include all stakeholders in planning and decision-making processes; 5) Strive for sustainability, aiming to balance environmental, social, and economic needs in the short, medium, and long term.

On a regional level, Beh et al. (2011) proposed a framework, based on multi-objective optimization, to sequence different regional water supply interventions. The aim was to find an optimal mix of water supply alternatives, as well as to find when these projects should be implemented, while taking sustainability, long term planning horizons and future uncertainties into account, Figure 12.

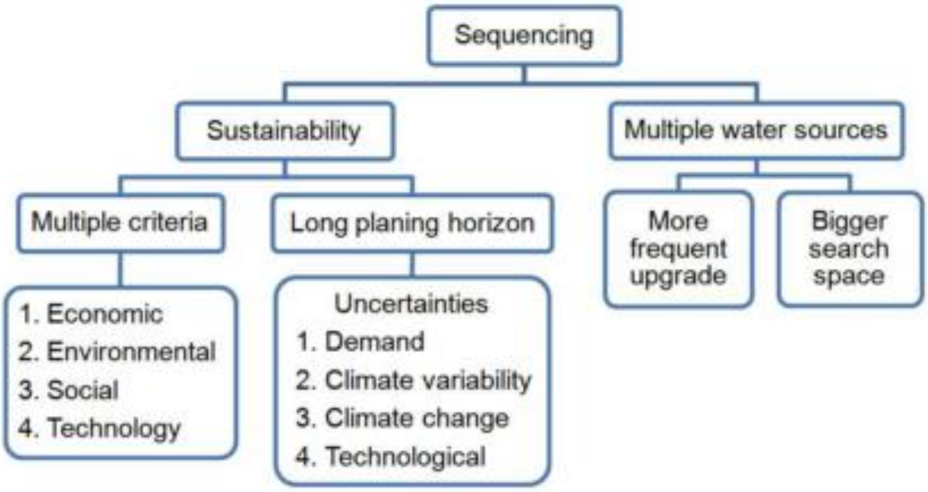

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Figure 12 Criteria involved in the sequencing of sustainable water supply projects (Beh et al., 2011).

The KWR Watercycle Research Institute and Deltares developed an indicator approach called City Blueprint to assess the sustainability in the urban water cycle (van Leeuwen et al., 2012). The approach used 24 indicators in the eight categories: water sequrity, water quality, drinking water, sanitation, infrastructure, climate robustness, biodiversity and attractiveness, and governance.

In the UK, Water UK (2011) developed a set of indicators to measure the water utilities sustainability progress. Starting in 2000 they produce an annual sustainability report for the water industry based on these indicators. The research project Sustainable Water industry Asset Resource Decisions (SWARD) developed a set of decision support processes for the UK water service providers to incorporate sustainability into their decision-making procedures. Suggested sustainability principles, criteria, indicators and processes could be applied at both an overall corporate strategic level and at an application level. A Guidebook was also produced that for the utilities and its stakeholders to go through the processes (Ashley et al., 2003; Ashley et al., 2004; Butler et al., 2003).

The EU project TRUST developed a framework and sustainability criteria to support water authorities and utilities in Europe in formulating and implementing appropriate urban water policies (Brattebø et al., 2013), Figure 13. The TRUST self-assessment tool is based on a simple scoring system. A three-level assessment scheme (green, yellow and red) was designed to indicate readiness for the 2040 challenge (Alegre et al., 2012). Marques et al. (2015) later proposed a MCDA-model to measure the sustainability of urban water services using criteria adapted from the TRUST project.

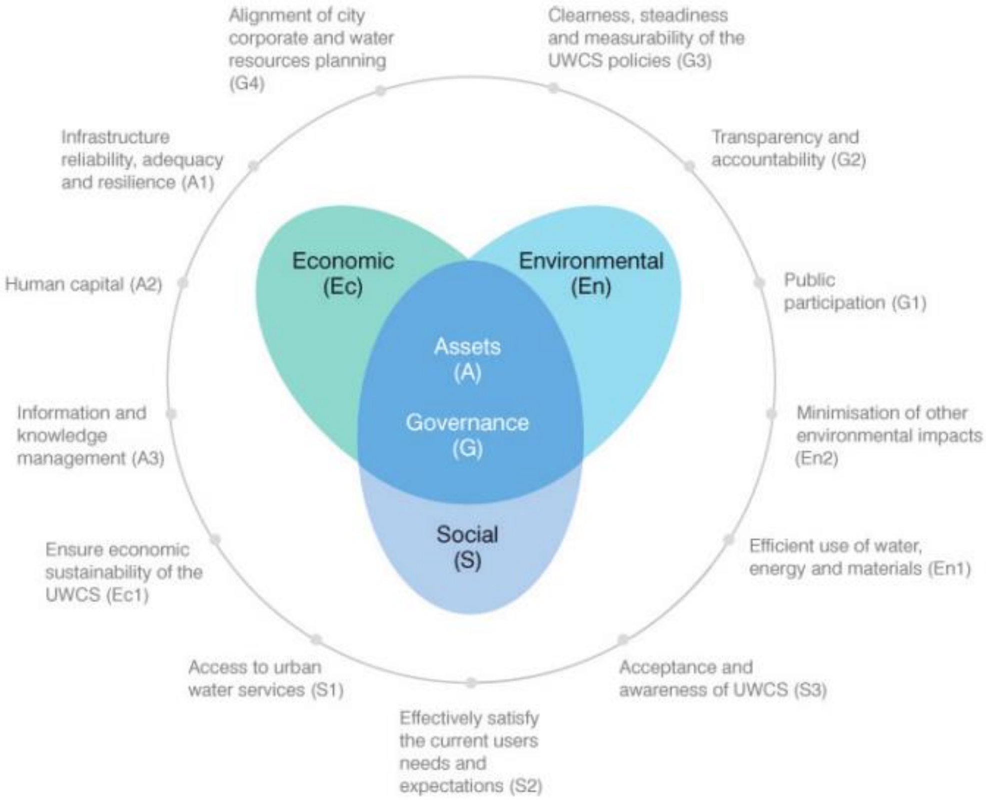

*Figure 13 TRUST approach to sustainability assessment (Brattebø et al., 2013)***.**

Liner and deMonsabert (2011) developed a method based on goal programming and multi criteria analysis to evaluate sustainability of water supply alternatives. The method was demonstrated for a California utility, using their water supply master plan to define trade-offs between environmental, social and economy goals.

Rygaard et al. (2014) used a combination of quantitative and qualitative methods, i.a. lifecycle assessment, freshwater impact methods, quantitative microbial risk assessment (QMRA), in a multi criteria assessment of four alternative concepts for a secondary water supply for parts of Copenhagen, Denmark. The alternative concepts were: 1) slightly polluted groundwater for use in toilets and laundry, 2) desalinated brackish water for use in toilets, laundry, and dishwashers, 3) desalinated brackish water for all uses, including drinking water, and 4) local reclamation of rain and gray water for use in toilets and laundry. The concepts were evaluated for their technical feasibility, economy, health risks, public acceptance, and environmental sustainability.

Finally, on a more general water and watershed level, the California Water Sustainability Indicators Framework lists 120 sustainability indicators corresponding to their 7 sustainability goals: Sustainable Water Management- Manage and make decisions about water in a way that integrates water availability, environmental conditions, and community well-being for future generations; Improve Water Supply Reliability- Improve water supply reliability to meet human needs, reduce energy demand, and restore and maintain aquatic ecosystems and processes; Contribute to Social and Ecological Benefits from Water Management- Improve beneficial uses and reduce impacts associated with water management; Increase Quality of Water - Improve quality of drinking water, irrigation water, and in-stream flows to protect human and environmental health; Safeguard Environmental Health - Protect and enhance environmental conditions by improving watershed, floodplain, and aquatic condition and processes; Integrate Flood Management Activities- Integrate flood risk management with other water and land management and restoration activities; Improve Adaptive Decision Making - Employ adaptive decision-making, especially in light of uncertainties, that support integrated regional water management and flood management systems (Shilling, 2013).

6 DISCUSSION

Several local water utilities in Sweden, Europe and elsewhere are forming inter-municipal cooperations as a way to handle current and future challenges such as climate change, social development, demographic alteration, and increasing requirements of managing source water protection, backup systems, infrastructure renewals, emergency preparedness and efficient production and distribution. As a result, a new regional level of water governance has emerged.

The research literature is full of studies focusing on assessing a few separate effects that can arise when water utilities grow or merge, particularly within the area of economies of scale. However, there are a limited number of decision aids focusing on assessing the formations of inter-municipal companies and alliances and the lighter forms of inter-municipal cooperations. There are also few decision aids focusing on assessing the large scale, inter-municipal policies and interventions that the regional decision-makers are faced with.

There is an international consensus that sustainability needs to be addressed in water supply planning, design and decision-making. The way to achieve a more sustainable water sector however differs between countries and jurisdiction (Rathnayaka et al., 2016). In the process of developing a decision aid for regional cooperations and interventions, national and international research literature was searched for decision support methods suitable for assessing sustainability.

A sustainability assessment method needs among other things to be transparent, valid and holistic (Brattebø et al., 2013). It also needs to be inclusive and allow public and stakeholder participation, which has been recognized as essential for good public policy (UNECE, 1998). Due to long asset and infrastructure life times, strategic decision-making also needs to consider uncertainties and trade-offs in future context conditions (Störmer et al., 2009).

Numerous studies have proposed MCDA for evaluating sustainability of water supply interventions, see for example Lai et al. (2008) and Scholten et al. (2015). MCDA provides a means for structured and transparent evaluations of alternatives and is suitable as decision aid in complex decision situations with several and conflicting objectives. It can be used to assess both weak and strong sustainability depending on choice of compensatory or non-compensatory techniques. A weak sustainability assessment allows for compensations of sustainability criteria or sustainability domains, as opposed to strong sustainability in which the criteria or domains are complimentary, but not interchangeable (Hopwood et al., 2005; Rosén et al., 2015).

MCDA meets several of the above-mentioned requirements on sustainability assessment methods. However, MCDA has no collectively used method to for incorporating time dependency and long term consequences for MCDA criteria (DCLG, 2009; Montibeller & Franco, 2011). Methods used in literature include e.g. applying discounting in a similar way as in CBA and use of several different MCDA models at a time, each method comprehending its own difficulties and limitations (Montibeller & Franco, 2011).

Moreover, even though MCDA often involves criteria valued in monetary terms (DCLG, 2009), cost externalities are not commonly included in sustainability assessments (Rathnayaka et al., 2016). By combining MCDA with CBA, valuations based on welfare economics of private costs and benefits as well as externalities can be included in the sustainability assessments, and the possibility arises to assess economic profitability in addition to sustainability.

7 CONCLUSIONS

The following main conclusions were drawn from the literature review:

- A growing number of water utilities in Sweden, Europe and elsewhere initiate various forms of inter-municipal cooperations, resulting in a new regional level of water governance.
- There is an international consensus that sustainability needs to be addressed in water supply planning, design and decision-making.
- A sustainability assessment method needs to be transparent, valid and holistic, and it needs to allow public and stakeholder participation.
- Several studies have proposed MCDA and evaluation criteria for sustainability assessments in the water sector.
- Few studies have focused on assessing the sustainability of the formations of inter-municipal cooperations or the large scale, inter-municipal policies and interventions that regional decision-makers are faced with.
- By combining MCDA with CBA, valuations based on welfare economics can be included in the sustainability assessments, and economic profitability can be assessed in addition to sustainability.

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