



Assessment methods for disproportionate costs according to the Swedish Water Management Ordinance (2004:660)

An overview



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**Swedish Agency
for Marine and
Water Management**



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This report has been produced by the Swedish Agency for Marine and Water Management. The Agency is responsible for the content and conclusions of the report.

Author: Associate professor Tore Söderqvist, Anthesis AB.

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Foreword

Assessing disproportionate costs is an important part of deciding whether there are grounds for applying exemptions to the environmental objectives for a water body, according to the Swedish Water Management Ordinance (2004:660). These assessments require a good understanding of both costs and benefits. To facilitate this work and to support the application of HVMFS 2019:25 and the accompanying guidance, the Swedish Agency for Marine and Water Management (SwAM) has commissioned this assignment report. The purpose of the study has been to provide a basis for the development of a method to assess disproportionate costs in accordance with Chapter 4, sections 9 and 10 of the Swedish Water Management Ordinance. This method development is an important piece of the puzzle in SwAM:s guidance on the assessment of disproportionate costs according to the Swedish Water Management Ordinance. The report was written by Tore Söderqvist, Anthesis AB, and reviewed by investigators and legal officers at SwAM.

Gothenburg, 7 July 2022, Signild Nerheim

Swedish summary

Att bedöma orimliga kostnader för miljöåtgärder handlar om att väga samman delar som beskriver motstående intressen i beslutsfattande. För att göra sådana sammanvägningar finns en rad olika bedömningsmetoder tillgängliga. I den här rapporten ges en introduktion till de två metoder som är vanligast i litteraturen för att avväga kostnader och nyttor förknippade med miljöåtgärder respektive samhällsnyttiga verksamheter såsom vattenkraft: Kostnads-nyttoanalys (cost-benefit analysis, CBA) och multikriterieanalys (multi-criteria analysis, MCA). I rapporten finns även en beskrivning av den så kallade Leipzig-modellen.

Eftersom EU-gemensamma vägledningar anger att det är samhällsekonomiska kostnader och samhällsekonomiska nyttor som ska ligga till grund för bedömningen av huruvida kostnaderna är orimliga eller inte kan CBA anses vara ett förstahandsval för att bedöma orimliga kostnader. Detta eftersom själva syftet med en CBA är att identifiera samtliga positiva samhällsekonomiska konsekvenser (nyttor) och samtliga negativa samhällsekonomiska konsekvenser (kostnader) av ett projekt och jämföra dem med varandra. För att kostnaderna ska bedömas vara orimliga krävs att kostnaderna "påtagligt" överstiger nyttorna.

Nyttor och kostnader är lätt jämförbara om de är uttryckta i samma enhet, och i CBA finns en strävan efter att så långt möjligt uttrycka dem i monetära enheter. Ett praktiskt problem är att vissa nyttor och kostnader är svåra att monetarisera, bland annat på grund av bristande kunskap om deras samhällsekonomiska betydelse. Detta kan exempelvis handla om miljöeffekter som på goda grunder kan antas påverka människors välbefinnande genom ett förändrat tillhandahållande av olika ekosystemtjänster, men där det kan saknas tillräckligt god information om såväl sambandet mellan miljöeffekter och ekosystemtjänster som individers preferenser för olika ekosystemtjänster. Detta praktiska problem kan hanteras på flera olika sätt, bland annat genom att använda MCA som en kompletterande eller alternativ metod.

Rapporten presenterar och diskuterar kortfattat CBA och MCA, samt även några olika sätt att värdera ekosystemtjänster och andra svårvärderade nyttor och kostnader i monetära enheter: Marknadsdatametoder, scenariometoder, deliberativ värdering och värdeöverföring. När det gäller MCA finns en rad olika typer av MCA-metoder tillgängliga. Därför ges exemplifieringar av ett antal MCA-huvudmetoder: Linjära additiva metoder, multiattributmetoder, analytisk hierarkisk process, utsorteringsmetoder och icke-kompensationsmetoder.

Leipzig-modellen är ett tillvägagångssätt att bedöma orimliga kostnader som följer ett annat metodspår än CBA och MCA i och med att den har som utgångspunkt är att utnyttja information om tidigare satsningar på miljöåtgärder och utifrån sådan information beräkna en referenskostnad som ger ett tröskelvärde för vad som är orimliga kostnader. Tröskelvärdet tar vidare hänsyn till nyttan av miljöåtgärder genom icke-monetära expertbedömningar.

Summary

Assessing the disproportionate costs of environmental measures involves weighing up elements that describe competing interests in decision-making. To make such trade-offs, a range of different assessment methods are available. This report introduces the two methods most commonly used in the literature to weigh the costs and benefits associated with environmental measures and socially beneficial activities such as hydropower: *cost-benefit analysis* (CBA) and *multi-criteria analysis* (MCA). The report also describes the so-called Leipzig model.

As EU guidance indicates that economic costs and benefits should be the basis for assessing whether costs are disproportionate or not, CBA can be considered a first choice for assessing disproportionate costs. This is because the very purpose of a CBA is to identify all positive economic impacts (benefits) and all negative economic impacts (costs) of a project and compare them with each other. For the costs to be considered disproportionate, the margin by which costs exceed benefits should be appreciable and have a high level of confidence.

Benefits and costs are easily comparable if they are expressed in the same unit, and CBA endeavours to express them in monetary units as far as possible. A practical problem is that some benefits and costs are difficult to monetise, partly because of a lack of knowledge about their economic importance. This may be the case, for example, for environmental effects that can reasonably be assumed to affect people's well-being through changes in the provision of various ecosystem services, but where there may be insufficient information on both the relationship between environmental effects and ecosystem services and individuals' preferences for various ecosystem services. This practical problem can be addressed in several ways, including by using MCA as a complementary or alternative approach.

The report briefly presents and discusses CBA and MCA, as well as some different ways of valuing ecosystem services and other hard-to-value benefits and costs in monetary units: revealed preference methods, stated preference methods, deliberative valuation and value transfer. In the case of MCA, a range of different types of MCA methods are available. Therefore, examples are given of a number of main MCA methods: linear additive methods, multi-attribute methods, analytical hierarchical process, sorting methods and non-compensation methods.

The Leipzig model is an approach to assess disproportionate costs that follows a different methodological path from CBA and MCA in that it takes as its starting point information on past investments in environmental measures and uses this information to calculate a reference cost that provides a threshold for what constitutes disproportionate costs. The threshold further takes into account the benefits of environmental measures through non-monetary expert judgements.

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1 Introduction

Assessing disproportionate costs of environmental measures¹ involves weighing up elements that describe competing interests in decision-making. To make such trade-offs, a variety of assessment methods are available. Some of them can be complex and difficult to communicate, such as fuzzy logic, genetic algorithms or other methods usually categorised as AI methods. This overview of assessment methods ignores such methods and instead provides an introduction to the two methods that are most commonly used in the literature to weigh the costs and benefits associated with environmental measures and socially beneficial activities such as hydropower: *cost-benefit analysis* (CBA) and *multi-criteria analysis* (MCA). In addition to these two main methods, a method developed in Germany, the so-called Leipzig model, will also be presented. Both MCA and CBA can be seen as a group of different methods with different sub-methods.

Since the EU common guidelines² state that it is economic costs and economic benefits that should form the basis for the assessment of whether the costs are disproportionate or not, CBA can be considered a first choice for assessing disproportionate costs. This is because the very purpose of a CBA is to identify all positive economic impacts (benefits) and all negative economic impacts (costs) of a project and compare them with each other. For the costs to be considered disproportionate, the margin by which costs exceed benefits should be appreciable and have a high level of confidence.

Benefits and costs are easily comparable if they are expressed in the same unit, and CBA endeavours to express them in monetary units as far as possible. A practical problem is that some benefits and costs are difficult to monetise, partly because of a lack of knowledge about their economic importance. This may be the case, for example, for environmental effects that can reasonably be assumed to affect people's well-being through changes in the provision of various ecosystem services, but where there may be insufficient information on both the relationship between environmental effects and ecosystem services and individuals' preferences for various ecosystem services.

¹ Chapter 4, section 10(1) of the Water Management Ordinance (2004:660).

² The main guidance documents in this context are Guidance Document No. 1 (European Communities, 2003), No. 20 (European Communities, 2009) and No. 36 (European Communities, 2018).

This practical problem can be addressed in several ways, for example

1. The best available monetary estimate is included in the CBA, but with an estimate of the uncertainty in this estimate. However, the uncertainty may be so large that it is still difficult to assess whether a cost is disproportionate or not.
2. Given the information currently available, a deliberative monetary valuation is conducted, where relevant stakeholders agree on what monetary valuation might be reasonable. This deliberative valuation can be iterative, so that it is updated as more information becomes available. More on deliberative valuation in section 3.3.
3. The magnitude of benefits and costs that are not deemed monetisable is instead assessed in a different way, complementing the information provided by the monetised benefits and costs included in a CBA. Such a complementary assessment can be done using MCA. Subsequently, it would be necessary to find a condition for the relationship between monetised benefits and costs and non-monetised benefits and costs in order to assess whether the costs as a whole are disproportionate or not.
4. *All* benefits and costs are assessed using an MCA. This may involve translating monetised benefits and costs as well as non-monetised benefits and costs into a different unit to enable comparison. A condition is then needed for how high costs (expressed in this unit) need to be for them to be judged disproportionate in relation to the benefits (expressed in the same unit). For example, costs can be estimated on a negative scale (e.g. – 1 to – 10), while benefits can be estimated on a positive scale (e.g. 1 to 10). Each item is then weighted according to how much weight is given to each item. If the weighted and totalled costs exceed the weighted and totalled benefits, disproportionate costs may exist. For the costs to be considered disproportionate, the margin by which costs exceed benefits should be appreciable and have a high level of confidence.

Another type of complication is also conceivable, namely that impacts that are not strictly economic, and therefore not relevant to include as benefits or costs in a CBA, are nevertheless considered relevant for the assessment of disproportionate costs. An example is local employment effects. The magnitude of such impacts must therefore by definition be addressed outside a CBA, for example using an MCA.

The fact that CBA and MCA can play a central role as methods for the assessment of disproportionate costs creates a need for a general understanding of these methods. Therefore, this report provides an overview of the methods, CBA in Chapter 2 and MCA in Chapter 4. Chapter 3 deals with monetary valuation methods, which can be used to monetise benefits and costs and then include them in a CBA. The review concludes in Chapter 5 with a description of the so-called Leipzig model, which is an approach to assessing disproportionate costs that follows a different methodological track than CBA and MCA.

2 Cost-benefit analysis (CBA)

2.1 General information on CBA

A CBA involves examining the positive economic impacts (benefits) and negative economic impacts (costs) resulting from a project. Generally speaking, 'impacts' equals effects on people's well-being, and in principle, *all* impacts on people's well-being today and in the future should be included in a CBA. The analysis is thus done at the societal level, which is why a CBA is referred to as an assessment of profitability for society as a whole. For detailed descriptions of CBA, see for example Kriström and Bonta Bergman (2014), Johansson and Kriström (2016, 2018) and Boardman et al. (2018).

Economic welfare theory has developed economic measures of changes in people's well-being; they are based on individuals' preferences for things that matter for well-being, such as different goods and services, and aim to measure individuals' *willingness to pay* (WTP) or *willingness to accept compensation* (WTA) (Freeman et al., 2014; Johansson and Kriström, 2016). These economic metrics are usually measured in monetary units (money). Expressing consequences in money in this way ('monetisation') makes it possible to compare consequences with each other. Thus, although money is used as a unit of measurement, CBA aims to examine impacts on people's well-being, not impacts on money flows. It is therefore important not to confuse CBA with different types of financial analyses. Financial analyses also use monetary units and can provide information useful for a CBA, but financial analyses have a different purpose in that they examine how money flows are affected by a project, which is not necessarily the same as how people's well-being is affected. Examples of financial analyses are analyses of how the income and expenditure of a household, a company, a municipality or the government are affected by a project.

Through monetisation in a CBA, the economic profitability assessment can be made by calculating the net present value (NPV), which is basically calculated according to this equation:

$$NPV = \sum_{t=1}^T \frac{1}{(1+r)^{t-1}} (B_t - C_t)$$

The equation assumes that benefits and costs occur at the beginning of each time period. Other assumptions are possible which imply modifications to the formula above. The equation shows that the NPV resulting from a project is equal to the present value sum of benefits (B) minus costs (C) for all people affected by the project from the start of the project at the beginning of period 1 ($t = 1$) for as long as the impact occurs, i.e. until time period T . Comparisons of benefits and costs occurring at different points in time are made using the social discount rate r . Usually, time is counted in years. The benefits and costs of a project are calculated in comparison with the benefits and costs that arise in a reference alternative. If $NPV > 0$, the project is economically profitable.

Benefits and costs resulting from a project can also be described in simplified terms using Figure 1. Costs arise mainly because resources of various kinds (production factors such as labour, machinery, etc.) must be used for the project to be realised. The economic cost of this resource consumption is equal to the opportunity cost of the resources, i.e. what society loses by not being able to utilise the resources in their best alternative use. On the plus side, however, are the results of the project, which hopefully have a positive impact on people's well-being and thus bring benefits. The figure illustrates, among other things, that labour (job creation) is not a benefit in itself in a CBA. On the contrary, the use of productive labour in a project is a cost (although it is lower if the project employs unemployed people than if the project employs people who are

already in work), but the labour input can lead to the creation of something whose benefits exceed the costs.

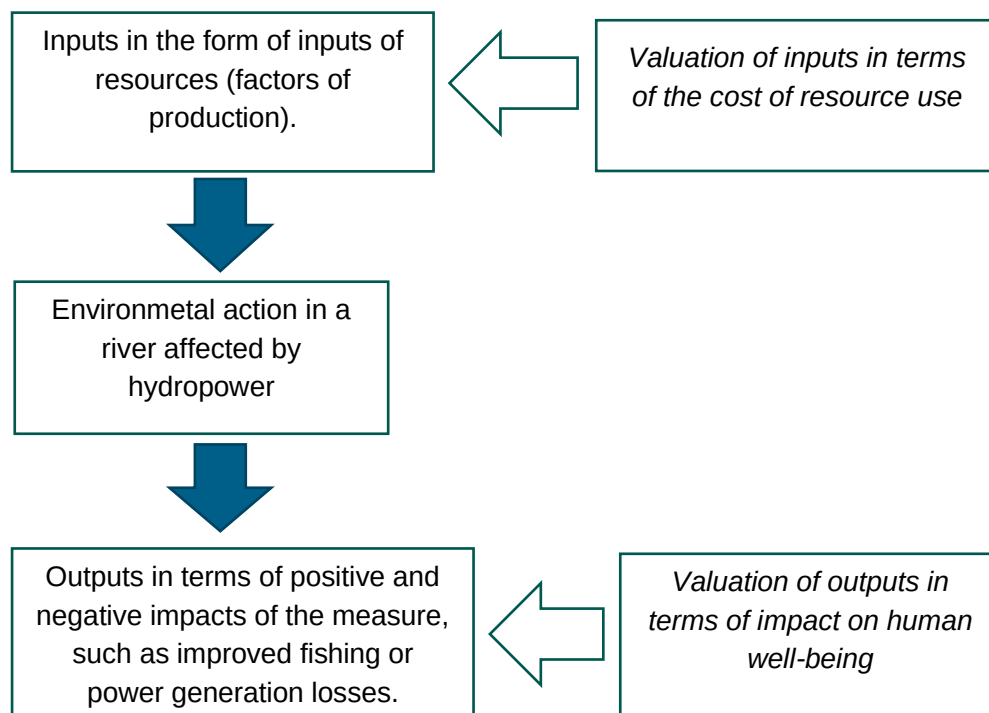


Figure 1 The cost-benefit analysis view of a project, for example a measure to improve water quality in a hydropower-affected river. After Söderqvist et al (2017, Figure 1).

If all the impacts of a project can be expressed in terms of benefits and costs in monetary terms, the net present value will provide an overall picture of the project's impacts. In practice, however, some benefits and costs are often difficult to monetise. This is often the case when a project affects goods and services that, while not traded in any market, are nevertheless important for human well-being. Many of the goods and services provided by nature, such as ecosystem services, fall into this category. In the next chapter, we outline the specific valuation methods available for valuing the benefits and costs resulting from a change in the availability of such non-market goods.

A CBA is usually conducted through a stepwise procedure. These steps can be described as follows (Kiström and Bonta Bergman, 2014):

1. Problem statement, which provides a background to the project being evaluated with CBA. For example, it explains the wider context of the project and the problem that the project is intended to help solve.
2. A statement of purpose, which specifies the aim of the project. Such clarification can help to identify possible conflicts with other societal problems.
3. Description and justification of the reference alternative, i.e. the baseline against which the project is to be compared.
4. Identification and description of the project in time and space.
5. Identification of the impacts of the project in relation to the impacts of the reference alternative. This is done, among others, with the help of different forms of expertise, such

as ecological expertise to identify impacts related to the availability of ecosystem services, medical expertise to identify health impacts, and so on.

6. Summarising the impact of the project.
7. Checkpoint. After the synthesis in step 6, an assessment can be made whether the impacts indicate whether the project is reasonably defined given the problem and purpose statements, or whether the project needs to be adjusted.
8. Calculation of the benefits and costs of the project, i.e. expressing the summarised impacts in monetary terms as far as possible.
9. Distributional analysis, i.e. studying how impacts are distributed among different groups in society.
10. Sensitivity analysis, i.e. studying how the CBA results are affected by changing the assumptions for the calculations and describing the degree of uncertainty in the results.
11. Conclusions. Summarise the results from steps 8-10 and refer back to the summary from step 6 to assess whether the impacts that could not be monetised affect the conclusions. Finally, assess the economic viability of the project.
12. If the project is judged to be unprofitable from an economic point of view, a final step may be to evaluate what changes in the project could possibly turn the loss into a profit.

2.2 Advantages and disadvantages of CBA

CBA is a controversial methodology that needs to be understood in terms of the assumptions on which it is based, including that valuations are based on individuals' preferences (Perman et al., 2011).

The following are often mentioned as advantages of CBA as a decision support tool:

- The methodology provides a structured and consistent basis on the advantages and disadvantages of a change for different groups in society.
- The monetary values used to weigh the relative importance of different impacts are based on citizens' preferences. In this way, CBA can be said to be democratic. It is possible to weight the willingness to pay of different groups to adjust the results for income differences.
- The methodology includes welfare changes³ for all citizens within the geographical scope of the study.
- CBA forces decision-makers to identify who stands to gain and who stands to lose from a change, and not just highlight certain groups or types of impacts. For example, a decision is not made solely from one perspective, e.g. not only the health impacts that could arise.
- All decisions have consequences that favour or disadvantage different individuals/groups. CBA makes the valuation of consequences explicit, by giving a monetary value to the consequences included in the analysis, rather than being implicit in the decision-making process. For example, if a decision-maker makes a decision based on qualitative information about five significant consequences of a measure, it is not explicitly stated how the decision-maker weighed the different consequences to arrive at his or her decision.
- The method values impacts in a common unit: money. Therefore, in the ideal case, when all significant costs and benefits are monetisable, it is possible to show whether the costs to society exceed the benefits of implementing a change compared to a reference alternative (e.g. business as usual).
- CBA applies discounting to enable the comparison of benefits and costs occurring at different points in time. Many countries use a discount rate that declines over time, so that

³ In this report, 'well-being' and 'welfare' are used synonymously.

the welfare of future generations is reasonably factored into the calculation (see, for example, Johansson and Kriström, 2016, Dasgupta, 2021).

A number of disadvantages of CBA are as follows:

- Many of the benefits we derive from nature are difficult to value in monetary terms and thus difficult to compare with monetised benefits and costs in a CBA. However, specific valuation methods have been developed (see next chapter) and studies that have used such methods are summarised in freely available databases (e.g. www.esvd.info).
- The method is based on an anthropocentric perspective, i.e. it is only to the extent that something in nature provides a benefit to humans that it has a value. In other words, only the instrumental values of nature for humans are relevant for a CBA. However, such values include, for example, altruistic values and existence values.
- In situations where not all impacts are monetised, the effect may be that the decision-maker implicitly weighs monetised impacts against non-monetised impacts to assess the overall economic profitability. If this is done only implicitly, it is not explicitly clear how the decision-maker has weighed different consequences against each other.
- The choice of discount rate can be controversial. For example, even a relatively low discount rate may result in consequences that occur relatively far in the future having a negligible present value.

The Swedish Government Public Inquiry 2013:68 (2013), which was tasked, among other things, with analysing measures and proposing methods and actions to better value ecosystem services, argued, with reference to TEEB (2010), that monetary valuation is less reliable or outright inappropriate in more complex situations involving a diversity of ecosystem services or different ethical beliefs about what values are possible or appropriate to express in monetary terms. The 'insurance value', which refers to the ability of ecosystems and biodiversity to deliver ecosystem services in times of change, can be very large for those ecosystem services that are difficult to replace and require many assumptions to make a monetary valuation. They argued that it is therefore important to note that the identification and mapping of key ecosystem services can provide the most reliable, relevant and cost-effective basis for an impact assessment of different alternative decisions, and that it is not always justified to additionally perform a quantitative estimation of benefits or monetary valuation. This is particularly the case where there is considerable uncertainty about how benefits can be estimated. They considered that estimating values in monetary terms is particularly relevant when externalities on biodiversity and ecosystem services risk being neglected in the absence of a monetary valuation, while the conflicts are relatively small regarding basic ethical assumptions about which values are possible or appropriate to express in monetary terms. In cases other than these, they argued that there is a case for expressing and visualising the value of biodiversity and ecosystem services in qualitative or non-monetary quantitative terms instead.

2.3 Example of CBA of an environmental measure

The FRAM-KLIV project developed an Excel-based tool for CBA of environmental measures in hydropower-affected rivers (Söderqvist et al., 2017).⁴ The tool is based on essentially the same step-by-step CBA procedure described above (Kriström and Bonta Bergman, 2014). The tool includes impact lists that include various ecosystem services. How the tool can be applied was illustrated using a cost-benefit assessment of environmental measures in the Mörrum River. The

⁴ The tool is available for download at <https://energiforsk.se/program/kraft-och-liv-i-vatten/verktyg-for-lonsamhetsbedomning-cba/>.

purpose of this assessment was to exemplify CBA as a method and not to produce a decision basis, but the example illustrated what additional information would be needed to produce a good decision basis.

The seven of Mörrumsån's 24 hydropower plants located downstream of Lake Åsnen were included in the study: Granö furthest upstream, then Fridafors upper, Fridafors lower, Ebbamåla, Hemsjö upper, Hemsjö lower and Marieberg furthest downstream. Mörrumsån has very high nature values, partly due to a rich fauna, including the presence of salmon and sea trout. Migration barriers at Fridafors lower, about 30 km upstream from the river's outlet to the sea, mean that there are no stocks of salmon and sea trout upstream of Fridafors lower.

The reference alternative was defined as the current situation regarding power generation and existing environmental measures such as fish ladders, diversions, minimum flows and fish stocking. The time horizon was set at 40 years.

Two project alternatives were studied:

1. Fishways will be created at Fridafors lower and Fridafors upper. A minimum flow is released in Granö old channel and in this channel biotope conservation measures are also carried out. In addition, it is assumed that there will be a dam removal at Marieberg, but not for another 40 years - this means that at this particular point, option 1 is identical to the reference option.
2. Same measures as in option 1, but in addition, the dam removal at Marieberg is assumed to take place today instead of in 40 years.

Remember that a CBA concerns the impacts (in terms of costs or benefits) resulting from a project *in relation to* the impacts arising from the reference alternative. Thus, the way the reference alternative is formulated has a crucial impact on the outcome of the CBA.

Table 1 summarises the impacts identified for the project alternatives, which are *different from* those of the reference alternative.

It was assumed that the environmental measures in this case lead to a negligible loss of regulating capacity. The reason was that the measures imply a loss of production by spilling water, but they do not affect the time of the year or day when electricity is produced, which could have been valued by the difference in price between day/night or seasons.

Following the identification of impacts, quantification and monetisation took place as far as possible. This process helped to highlight important knowledge gaps that need to be filled in order to be able to say something about the economic impacts. In this methodological illustration, assumptions were used to overcome the knowledge gaps in order to explicitly show the type of information that should be produced in order to perform a satisfactory CBA. Such assumptions were specifically highlighted in the report.

Table 1 summarises what was monetised and how this was done. In the third column of the table we have also included our rough assessment of the degree of uncertainty in the monetisation. The study used ranges to account for uncertainty to some extent, but the FRAM-KLIV tool also provides the option of specifying a probability distribution (for example, a triangular or lognormal distribution) for each benefit and cost. The selected probability distributions are then used in simulations to calculate an expected net present value, including a simulated probability distribution for the net present value.

Table 1 Identified impacts and their monetisation, and a rough assessment of the degree of uncertainty in the monetisation.

Identified impacts (g-I only occurs in project alternative 2)	Monetisation	Rough assessment of uncertainty in monetisation
a. Cost: Construction and maintenance costs of the environmental measures	Yes, based on previous investigations	Low
b. Cost: Losses in power generation	Yes, based on scenarios for future electricity price paths	Low
c. Cost: Increased emissions from coal-fired power, based on the assumption that reduced electricity generation is replaced by coal-fired power from Denmark. Due to the EU GHG emissions trading scheme, no increase in CO ₂ emissions was estimated, but there is an increase in emissions of pollutants such as NO _x .	Yes, based on estimates in the separate tool EcoSenseLE	High
d. Benefits: Increased fish stocks leading to increased recreational values for those who actually visit Mörrumsån and experience or utilise the fish resource (recreational fishermen)	Yes, based on quantification of effects on fish stocks and valuation study from Mörrumsån	Medium
e. Benefits: Increased fish stocks leading to increased non-use values for those who value the benefit of fish stocks but do not experience or utilise them in situ	Yes, based on effects on fish stocks and valuation study from Vindelälven	Very high
f. Benefit: Increased aesthetic values due to the fact that flowing water in dry channels is considered to provide a landscape that is perceived as more beautiful	Yes, based on assumptions about the number of affected households along the Mörrum river and valuation study from Ljusnan	High
g. Benefit: Increased sales of fishing licences, which benefits sport fishing interests in the area	Yes, based on the cost of fishing licences in Mörrumsån and an estimate from Emån of the relationship between increase in fish stocks and increase in the number of fishing days	High
h. Cost: Less flood protection due to changes in water flow	Yes, based on an assumption of decreasing values of riparian properties along the part of the Mörrum River affected by the dam removal at Marieberg. This was assumed to be a net effect of (h) and (k)	Very high
i. Cost: Deterioration of existing bathing and boating opportunities upstream of Marieberg.	No, due to lack of data. It was assumed that this cost is approximately equal to the benefit (j).	–
j. Benefit: Improved conditions for other types of water-related recreation than (i), such as white water rafting.	No, due to lack of data. See also (i).	–
k. Benefit: Better erosion protection due to change in water flow	Yes, see (h).	Very high
l. Benefits: Increase in river pearl mussel and other aquatic animal species as well as riparian vegetation, which may lead to an increase in non-use values	Yes, based on a hypothetical amount and an assumption that non-use values are limited to households in Karlshamn municipality	Very high

The aggregation of monetised costs and benefits into net present values resulted in a negative net present value for alternative 1, but a positive net present value for alternative 2. The magnitude of the net present values should not be taken literally, as this was a methodological study. However, the largest values on the benefits side were those linked to non-use values (impact (e) and (l) in Table 1) and thus to monetisations that can be considered to have a very high degree of uncertainty. Without these benefits, alternative 2 would have had a negative net present value.

Finally, some summarising observations:

- The structured procedure in a CBA aims to get as far as possible in terms of the magnitude of economic effects, i.e. how people are affected. This procedure stimulates the identification of key knowledge gaps in both scientific and economic data.
- A CBA tool of the type developed by FRAM-KLIV is only a calculation tool to calculate net present values. All results are therefore dependent on what the user puts in. There are no presets in the tool that have computational consequences, and so the calculations are transparent. All calculations could just as easily be done "by hand", unless the simulation option is utilised. With simulation, the calculations become more complex, but they are done based on the user's choice of probability distributions for benefits and costs.
- High uncertainty in monetisations is partly due to the lack of site-specific valuation studies. One way to reduce uncertainty is therefore to conduct more such studies.
- Estimates of non-use values can be expected to remain controversial, while they are likely to have a major impact on the outcome of a full CBA.
- The time horizon in the study was 40 years and a real social discount rate of 3% was used. Particularly in the case of longer time horizons and an uneven distribution of benefits and costs over time, the choice of the size of the discount rate can be of great importance for the size of the net present value and thus for the outcome of the CBA. As pointed out above, many countries have chosen to work with a falling discount rate over time in order to take this complication into account to some extent, but this has not yet been adopted in Sweden. It should also be emphasised that the complication of discounting is made explicit in a CBA, but the question of how consequences occurring at different points in time should be assessed is general, and is therefore also relevant in other assessment approaches, such as MCA.

3 Economic valuation of non-market goods and services

Methods for valuing goods and services that do not have a market price include *revealed preference* methods and *stated preference* methods. Only the latter are able to capture non-use values. These two types of valuation methods are described briefly below, see e.g. Freeman et al. (2014) for a detailed review. A common feature is that they aim to capture as representative information as possible about the preferences of the individuals concerned, for example by collecting data through surveys to representative samples of individuals or to web panels that are representatively composed. Deliberative valuation is another type of valuation method, but it relies on the participation of a smaller number of people and involves an interactive, deliberative process. Deliberative valuation is also described briefly below. Finally, we also briefly review *benefits transfer*, which is not a valuation method in its own right, but is used to generalise results from previous valuation studies.

3.1 Revealed preference methods

Revealed preference methods use information from related markets where market prices exist to estimate the price of a good that does not have a market price. They are thus based on information about actual market transactions, i.e. how individuals actually behave in some existing market. Revealed preference methods are limited to valuing use values. Non-use values (altruistic, bequest and existence values) can be valued using stated preference methods, see section 3.2:

- The property value approach (hedonic pricing): This approach uses the implicit information on how environmental factors are valued that can be provided by the behaviour of individuals in the property market. For example, the absence of noise can be valued by comparing the price of property in a noisy area with the price of property in another area that is very similar in all respects to the first area except that there is no noise. The difference in price can be used to estimate the cost of noise. Usually, the method is applied by collecting data on characteristics that describe the properties and using statistical analysis to isolate the influence of each characteristic on the property price.
- Travel cost method: This method estimates the recreational value of an area, or changes in individual characteristics of the area, based on information about individuals' behaviour in travel markets, i.e. based on individuals' travel costs including time spent.
- The production function method: The method is based on formulating a production function, which describes what can be produced by a market-priced good or service given inputs of various production factors, including inputs of the environment in the form of various ecosystem services, for example. The production function can then be used to examine how much a change in the supply of an ecosystem service affects the production of the market-priced good or service. The impact of the ecosystem service on production can then be valued using the market price.
- Damage cost method/avoidance cost method/defensive expenditure method - this method values a resource based on the loss avoided by conserving/protecting a resource or protecting against degradation of a resource. The value of climate change adaptation measures can be valued based on the estimated costs that would be incurred if climate change adaptation measures are not taken, for example due to increased flood risk. This valuation can sometimes be done using revealed preference methods, such as what the

property value method can tell us about reduced property values due to increased flood risks.

The replacement cost approach is another method that may be similar to revealed preference methods, but provides a different kind of information: it values a resource based on the cost of achieving the same benefit with a different solution. For example, the value of clean drinking water can be estimated at the treatment cost associated with drinking water production, investment in desalination plants to turn seawater into drinking water, and so on. The replacement cost method generally does not provide an accurate value of the benefit individuals associate with a good/service but only indicates the cost of an equivalent alternative. If the alternative has actually been acquired as a substitute, its cost can be seen as a lower bound on the value that individuals attribute to a good/service, but the method is often used for non-acquired alternatives and does not provide information on whether there is a willingness to pay for the alternative.

3.2 Stated preference methods

In order to monetise changes that include non-use values, stated preference methods are used. Two main methods are the following:

- Contingent valuation (CV) involves asking a usually representative sample of individuals about their willingness to pay (or sometimes about willingness to accept compensation) for a hypothetical change. The method has been widely criticised (some of the drawbacks of which are mentioned above under "drawbacks of CBA"), which has also led to extensive methodological development to address various methodological problems (OECD, 2018, page 87).
- Choice experiments (CE) involve asking a usually representative sample of individuals which combination of attributes they prefer, where different attributes are combined in different ways and at different levels (e.g. X number of fish stocks are enhanced, of which Y number are endangered species, at a cost of Z SEK via the tax bill). This way, the willingness to pay for the different attributes can be obtained. Based on the results, non-monetary trade-offs between different attributes can also be derived. The method is particularly suitable when the choice situation includes several dimensions and there is a trade-off between different attributes. Like the CV method, the method is criticised. One disadvantage of the method is that it can be too cognitively demanding for respondents if they have to consider many different combinations of attributes and levels. Another problem is that the values that respondents choose depend on how the choice situation is framed. For example, the choice of attributes and how they are described, as well as the levels and, for example, photos shown, can influence the choices respondents make.

Deliberative valuation is another method similar to stated preference methods, but with a slightly different theoretical basis. See more in the next section.

A major strength of stated preference methods is their ability to assess non-use values, which can often be expected to be significant especially in terms of the benefits of environmental measures in water bodies. However, the methods are controversial due to their hypothetical nature. This can lead to hypothetical bias, i.e. individuals tend to exaggerate their willingness to pay for good causes when asked about their willingness to pay without actually having to pay the amount they indicate. How such a bias can be reduced has been the subject of extensive research, and methods have been designed to reduce the problem, such as the respondent answering under oath, where they promise to answer honestly (Swedish Agency for Marine and Water Management, 2019). Another problem is so-called embedding, which refers to individuals

tending to value a minor change, such as improved water quality in a single watercourse, as highly as when asked what the individual is willing to pay for a major change, such as improving water quality in five watercourses where the first watercourse is included among the five. A further problem is that willingness to pay can be affected by how the decision situation is described, for example if photos are shown, so-called framing (Dasgupta, 2021).

Stated preference methods collect data on individuals' willingness to pay or compensation requirements through surveys or interviews. In such a situation, there is limited scope to describe the actual scenario that respondents are asked to evaluate and the way questions are asked may influence respondents' answers. This may mean, for example, that the evaluation risks being inaccurate due to the respondent not understanding or not absorbing information about what the scenario actually entails in terms of benefits or costs. Therefore, great care is needed in the design of stated preference studies, such as testing information and questions using focus groups and pilot studies. See, for example, Johnston et al (2017) for recommendations to follow when designing and conducting stated preference studies. Dasgupta (2021) argues that stated preference studies should be conducted with caution. If respondents do not have knowledge of the functional characteristics of the resources under evaluation, then the study may be questionable. He points out that in democracies, we entrust many decisions to elected representatives, and that the limitations of what stated preference studies can cover should be guided by that fact.

3.3 Deliberative valuation

Deliberation can be described as a process in which people deliberate, ponder, exchange opinions, evaluate evidence, reflect on issues of common interest, negotiate and try to persuade each other. Deliberation includes both consensual and adversarial communication processes (Fish et al., 2011).

Deliberative valuation is similar to stated preference methods in that it uses a scenario of, for example, environmental change as a starting point. However, instead of communicating the scenario to a representative sample of individuals and asking them about their preferences, deliberative valuation uses a process where a smaller group participates in the valuation in a deliberative manner. Among other things, the method is increasingly used to include values that may be difficult to include in individually based surveys, such as value pluralism, non-anthropocentric values or social justice (TEEB, 2010). Kenter (2016a) describes a case study comparing the results between deliberative valuation and a more traditional scenario method.

Kenter et al. (2016b) describe a six-step process that can be used in deliberative valuation:

- 1) the institutional context

The first step is to explain the context in which decision-making takes place and how deliberative valuation can contribute - purpose, structure, who participates and who does not, how the results can and cannot be used, and important boundaries, so that participants do not have false expectations of the process or important issues fall through the cracks.

- 2) transcendental values (overall life goals and guiding principles)

In the second step, the group deliberatively identifies the transcendental values of the participants, the groups they belong to, and their shared social and cultural values.

- 3) contextual beliefs, broader policy effects and systemic relationships

In a third step, the participants' own views on the specific issue to which the deliberative valuation relates are raised, including the consequences of different policy options, who is responsible, the roles of different institutions, the extent to which different actors are able to influence the situation. In this step, different methods to clarify the consequences of different

policy options can be used, such as MCA or choice/consequence matrices. Experts can be invited to explain the consequences of different options.

4) implications for transcendental values

Once participants have an understanding of the consequences of different policy options, they compare how well different policy options align with the transcendental values that were deemed most important. For example, if social justice and security were judged to be most important, to what extent different options increase or decrease these values. This can be done in connection with step 3 in group discussion and/or with different decision support methods.

5) norms and contextual values

The fifth step integrates the evidence from the previous steps to discuss and draw conclusions about norms and contextual values. A traditional economic valuation is then based on individual estimates of the benefits of different options, compare the valuation methods in sections 3.1 and 3.2. One can also frame the decision situation in terms of collective benefits and ask what should be done based on the knowledge we have about the consequences of different policy options on what we are valuing. Contextual values reflect the degree to which different value objects (e.g. different ecosystem services) contribute to the best outcome and thus the relative valuation or importance of these objects and options.

6) value indicators

Value indicators should reflect the contextual values. This can be done by individuals indicating their willingness to pay, or ranking choice options. It can also be developed as a collective process of discussion and negotiation to agree on a final outcome, a ranking, a vote on which option is best, or what society or individuals should pay for different options.

In deliberative valuation, it is important to have good facilitation, i.e. that those leading the discussions do so in a good way.

A disadvantage of deliberative valuation compared to the stated preference methods in section 3.2 is that it limits the number of people who can participate in the valuation, which means that the results may not be representative of the citizens affected by the changes.

3.4 Value transfer

Value transfer is about generalising results from the application of valuation methods in different ways. Suppose a valuation study has been carried out using primary data collection on the willingness to pay for an environmental change in a river in Norrbotten. Value transfer can then involve using the results of this primary study to value a similar environmental change in some other river in another part of Sweden. In the context of value transfer, we usually talk about transferring results from a study area (S, the area that the primary study concerned) to a policy area (P, the area or context to which we want to transfer the results).

Value transfer methods are usually divided into two main types: (1) Transfer of point estimates and (2) Transfer of functions (Rosenberger and Loomis, 2003). The former main type can in turn be divided into (1a) Transfer of a single point estimate and (1b) Transfer of averages of point estimates. For function transfer, there are (at least) three different possibilities: (2a) Function transfer based on single valuation study, (2b) Function transfer based on meta-analysis and (2c) Structural function transfer. Each of these ways of making a value transfer is briefly explained below. For more details, see, for example, Rosenberger and Loomis (2003), Johnston et al. (2015, 2021) and Krström and Bonta Bergman (2014).

Transfer of point estimates. For the transfer of point estimates, it is assumed in principle that the mean willingness to pay estimated for the study area (WTP_S) is equal to the mean willingness to pay for the policy area (WTP_P), but that occasional adjustments to WTP_S may be necessary for the generalisation to WTP_P to be valid. A common type of adjustment is to adjust for any differences in income, so that

$$WTP_P = WTP_S (y_P / y_S)^\varepsilon,$$

where y_P and y_S are the mean income level in the policy area and the study area respectively and ε is the income elasticity of willingness to pay. This reflects a case where WTP_P is calculated based on a single valuation study, which provides the estimate of WTP_S (value transfer method 1a). However, WTP_S could also be an average of WTP estimates from more than one valuation study applied to a similar environmental change (i.e. value transfer method 1b). In this case, y_S could also be an average of the mean income level of the different valuation studies. It may also be reasonable to take into account the likely variation between the results of the valuation studies by presenting an interval for WTP_P .

However, making value transfers using point estimates including single adjustments, for example for income, has obvious weaknesses. It is conceivable that the valued environmental change is not similar between S and P, and preferences may also differ between S and P, and so on. While a variety of one-off adjustments for such differences are conceivable, functional transfers are probably a more powerful way of dealing with many differences.

Function transfer assumes that it is possible to estimate from a single valuation study (value transfer method 2a) or from more than one valuation study (value transfer method 2b) a function for WTP_S , which explains how WTP_S varies with respect to characteristics of the environmental change (G) and characteristics of the individuals/households (H), and, in case 2b, also characteristics of the valuation methods used in the studies (M):

$$WTP_S = \alpha_S + \beta_S \mathbf{G}_S + \gamma_S \mathbf{H}_S + \delta_S \mathbf{M}_S + e,$$

where bold indicates vectors, α, β, γ , and δ are coefficients estimated for example by regression analysis and e is a random term. The estimated function is then used to calculate the WTP_P by replacing $\mathbf{G}_S$, $\mathbf{H}_S$ and $\mathbf{M}_S$ with values valid for the policy area, i.e. $\mathbf{G}_P$, $\mathbf{H}_P$ and $\mathbf{M}_P$.

For feature transfer to work, it is clear that there must be policy area data for the features that were included in the estimation of the WTP_S feature for the study area.

Value transfer methods 2a and 2b can be criticised for arbitrariness regarding which characteristics should be included in the function. Structural function transfer (2c) attempts to overcome this weakness by starting from a specification of a utility function, thus providing a foundation for value transfer based on economic theory. This is an (even) more sophisticated approach, which itself requires assumptions about, for example, the shape of the utility function.

4 Multi-criteria analysis (MCA)

4.1 General information on the MCA

Multi-criteria analysis (MCA) is a collective term for methods used to assess how different policy options perform with respect to one or more desirable objectives, where the objectives are described by a number of criteria. A standard content of an MCA is therefore a performance matrix, in which the rows describe the policy options and the columns describe the criteria, with each cell describing how each policy option performs in relation to each criterion. For example, performance can be described in terms of scores on scales from, for example, 0 to 100 or from –10 to +10. Often the MCA also includes a weighting of the criteria, where the weight defines how important each criterion is considered to be relative to the others. However, the design of the MCA can be very different depending on the MCA method used, see also section 4.3.

Furthermore, a fundamental feature of MCA is the formulation of objectives and criteria by stakeholders, as well as the estimation of the relative importance of different criteria. Participants may also have a central role in assessing the performance of policy options, although these may also be based on or described by 'objective' data (Department for Communities and Local Government (DCLG), 2009).

With regard to MCA in relation to CBA, DCLG (2009) points out that most CBA studies include impacts that are identified as relevant consequences, but are not monetised. In some cases they are seen as less important, and are reported alongside monetised costs and benefits. In some cases, however, there may be variables for which it is difficult to estimate monetary values, but which are nevertheless considered to be of high importance. In these situations, MCA techniques can be useful. For example, it is possible to measure the performance of some criteria in monetary terms and other criteria in non-monetary terms, for example by scoring.

The OECD believes that MCA can be a good complement to CBA, and in some cases used as an alternative to CBA. MCA can provide an alternative when monetisation is challenging and there are many non-monetised impacts that need to be included. The OECD points out that while the need to incorporate quantitative and qualitative evidence into analyses is emphasised in many countries' guidance, there is no methodology for doing so, with the exception of MCA. However, the OECD sees a risk that MCA is used as an excuse for not spending resources on quantifying and monetising impacts to the extent possible (which could contribute to better CBA analyses) and that MCA may be misunderstood as a simpler analysis than a CBA. They state that if MCA is to become one of the main methods of regulatory impact assessment, authorities need to build up expertise in the method. They state that a major advantage of MCA, compared with the alternative of only qualitatively describing certain advantages and disadvantages, is that it is transparent in the sense that it is clear on the basis of which criteria different alternatives have been assessed and how the criteria have been weighted (OECD 2009).

4.2 Advantages and disadvantages of MCA

A general strength of MCA is that it can bring structure, analysis and transparency to the types of decisions that do not lend themselves, in whole or in part, to the practical application of CBA. DCLG (2009) lists some advantages:

- The methodology is in principle open and explicit.
- The choice of objectives and criteria made by decision-makers is open to analysis and to change if deemed inappropriate.

- Scores and weights, when used, are explicit and developed according to established methods. They can also be cross-referenced to other sources of information on relative values and modified if necessary.
- Performance measurements can be carried out by experts, and do not necessarily need to be put in the hands of decision-makers.
- It can provide important communication within the decision-making group and sometimes, later, between the decision-makers and society at large.
- When scores and weights are used, the method provides a verification chain.

As examples of disadvantages, DCLG (2009) highlights that:

- As with cost-effectiveness analyses, an MCA does not necessarily identify a 'best' option that is consistent with improving the overall well-being of individuals. (However, this does not preclude an MCA from examining whether an action option increases the overall well-being of individuals compared to a baseline).
- The judgements in an MCA are not necessarily consistent with what is crucial for a CBA, i.e. the best possible evidence on individuals' preferences. In MCA, judgements may instead be based on expert judgements or the preferences of different actors or decision-makers. Thus, the results of an MCA rely to a lesser or greater extent on the subjective judgements of stakeholders or decision-makers. It is therefore crucial to be able to deal with subjective judgements in order to provide defensible decision support. This problem can be addressed through sensitivity analyses and measures to increase the robustness of the analysis. It also becomes crucial that the MCA is characterised by transparency and the ability to deal with different preferences among different actors, including experts, the public and stakeholders.

Rosén et al. (2009) further point out in their review of the MCA that:

- MCA methods can give the appearance of scientific rigour even if the criteria formulated are poorly chosen.
- There may be some arbitrariness in the methods, for example at what level an option should be considered acceptable or not.

A particular problem is that MCA can be perceived as confusing due to the fact that there are many different MCA methods and also many different variants of each method. This can be a problem because the methods can give different answers and it is not obvious which method is best for a particular application. Therefore, the next section presents a number of main MCA methods. The different methods have different strengths and weaknesses, but it is also important to be aware that they all require work on structuring the decision problem itself. In this context, Hajkowicz and Higgins (2008) compared the results of five different MCA methods on six decision problems related to water management. Their conclusion is that it is often much more important to focus MCA work on achieving a good structure of the decision problem, including the identification of policy options and the formulation of objectives and criteria, than on which MCA method is most suitable, although the choice of MCA method is also important.

4.3 Different MCA methods

4.3.1 General methodological overview

Many different MCA methods have been developed, see for example Vassoney et al. (2017) for an overview of the types of MCA methods that have been used in hydropower planning and management. Some of the main methods are summarised below, based on DCLG (2009) and

Rosén et al. (2009), but it is not easy to understand the methods from overviews of applications or brief descriptions. Therefore, examples of applications of a number of main methods are also given below.

Selecting the MCA method for a particular application can also be said to be a kind of multi-criteria analysis. DCLG (2009) recommends using the following criteria in assessing which MCA method is most appropriate to apply:

- internal consistency and logical soundness.
- transparency.
- ease of use.
- data needs that are not inconsistent with the size of the problem being analysed.
- realistic in terms of time and availability of skills required for the analysis process.
- the extent to which the method can contribute to an audit trail.
- access to software, where needed.

A further criterion for assessing whether measures are disproportionately costly is that the methodology needs to be designed to demonstrate that the costs of a measure exceed benefits with a margin. A method that simply ranks different options is not sufficient.

Linear additive methods. These methods are probably the most common MCA methods. It usually involves expressing the performance of policy options for different criteria in terms of scores. The scores are then weighted to produce a final score for each policy option in the form of a weighted sum, with a weight assigned to each criterion. The policy options can then be ranked according to the final scores. The method assumes that the criteria are independent of each other.

Multi-attribute methods are a family of methods within which linear additive methods can be said to be a special case. Multi-attribute methods can be said to consist of three building blocks: (1) a performance matrix, (2) procedures to determine whether the criteria used are independent of each other or not, and (3) methods to estimate the parameters of a mathematical function that can be used to describe how favourable an action alternative is, based on how well the various criteria are met. The methodology is accepted but relatively demanding and is therefore most useful in projects where the requirements are high and sufficient resources are available to hire the necessary specialists.

The analytical hierarchy process (AHP) is a linear additive method, but here each criterion is compared in pairs with each of the other criteria to provide a basis for weighting the different criteria. The comparison is made by giving decision-makers a series of questions to answer, pitting one criterion against another. The methodology assumes that people are more likely to make relative judgements, rather than absolute ones. In the full application of the AHP, pairwise comparisons of alternatives are also made with respect to the different criteria. The comparisons of criterion against criterion and option against option result in matrices and finding solutions to the problems requires complex matrix calculations. Specialised software is therefore used to assist in the application of the method. AHP is often perceived as relatively simple and easy to use by decision-makers, but the theoretical basis of the method has been questioned. One reason is that the ranking of options can be altered by the addition of a further criterion, although logically the new criterion should not affect the ranking.

Outranking methods. Outranking methods aim to identify those options that appear better than others, but not a specific best option. Outranking uses a form of pairwise comparison, where one option is ranked more favourable than another if enough criteria indicate that it is better (taking into account the weight of the criteria), provided that the option is not significantly worse with respect to any of the other criteria. An interesting feature of screening methods is that two options

can be categorised as "difficult to compare" if, for example, important information is missing. In such cases, the analysis can still be carried out, even though it is not clear which of the two alternatives is better, which can be an advantage in many decision-making situations where information is lacking. Another advantage of screening methods is that the methodology is fairly consistent with political aspects of decision-making, where alternatives that are bad in a certain but important respect are screened out. A weakness of outranking methods is that it is rather arbitrary how to define one option as better or worse than another. Two commonly used outranking methods are ELECTRE (Elimination and choice expressing the reality) and PROMETHEE (Preference Ranking Organisation Method for Enrichment Evaluation).

Non-compensation methods can be used for problems where a performance matrix has been developed but the decision maker is not willing to accept compensation between criteria. In other words, a well-fulfilled criterion is not allowed to compensate for another insufficiently fulfilled criterion. Such methods are effective in screening out alternatives that must fulfil absolute requirements. The methods could therefore be characterised as a type of outranking methods. In order to distinguish between alternatives in more detail, non-compensatory methods usually need to be complemented by more rank-based methods, such as a linear additive method or AHP. Non-compensatory methods are often based on the definition of thresholds for one or more of the criteria. A distinction is made between conjunctive and disjunctive models. In a conjunctive model, the alternatives that do not meet the thresholds for all criteria are excluded. Disjunctive models allow options to pass that meet the threshold level for at least one criterion. The conjunctive and disjunctive models thus act as filters. It is perfectly possible to use a combination of both types.

In the following sections, linear additive methods, multi-attribute methods, AHP and outranking methods are exemplified. Non-compensation methods are not specifically exemplified, but the illustration of linear additive methods points out how a non-compensation procedure can take place in the context of a linear additive method.

4.3.2 Example of an MCA using a linear additive approach: SCORE

We use the Excel-based MCA tool SCORE (Sustainable Choice Of REmediation) as an illustration of a linear additive MCA approach. While SCORE is a decision support tool for choosing the most sustainable course of action to remediate contaminated land, the method is generic and can be adapted to assess environmental measures in hydropower-affected watercourses through a different set of criteria. The method is well-publicised (Anderson et al., 2018; Rosén et al., 2015; Norrman et al., 2020; Söderqvist et al., 2015) and should be of particular interest due to the following characteristics:

- Economic assessment through CBA is integrated as part of the tool.
- The tool combines monetary measures (from CBA) with score measures.
- Uncertainty analysis is included.
- Possibility of non-compensatory MCA method included.

SCORE is a tool for sustainability assessment of remediation options for a given contaminated site, and is based on the idea that sustainability can be viewed from (at least) three different dimensions: environmental, social and economic sustainability. Figure 2 shows what is basically included in SCORE. The tool is based on the assumption that a number of alternative courses of action have been identified (1) and the aim is to help assess the extent to which different alternatives contribute to increased sustainability, relative to a reference alternative. Based on a predefined gross list of criteria, the criteria that may not be relevant for a specific location are sorted out (2), after which the options are assessed based on environmental, social and

economic sustainability (3-5). The criteria within the environmental and social dimensions are weighted, and in addition a weighting at the dimension level is performed (6). This is followed by a Monte Carlo simulation that takes into account the uncertainty specified by the user for the assessments of environmental, social and economic impacts (7), after which results are presented (8). The loop back to (1) illustrates that the results may lead to adjustments of the policy options.

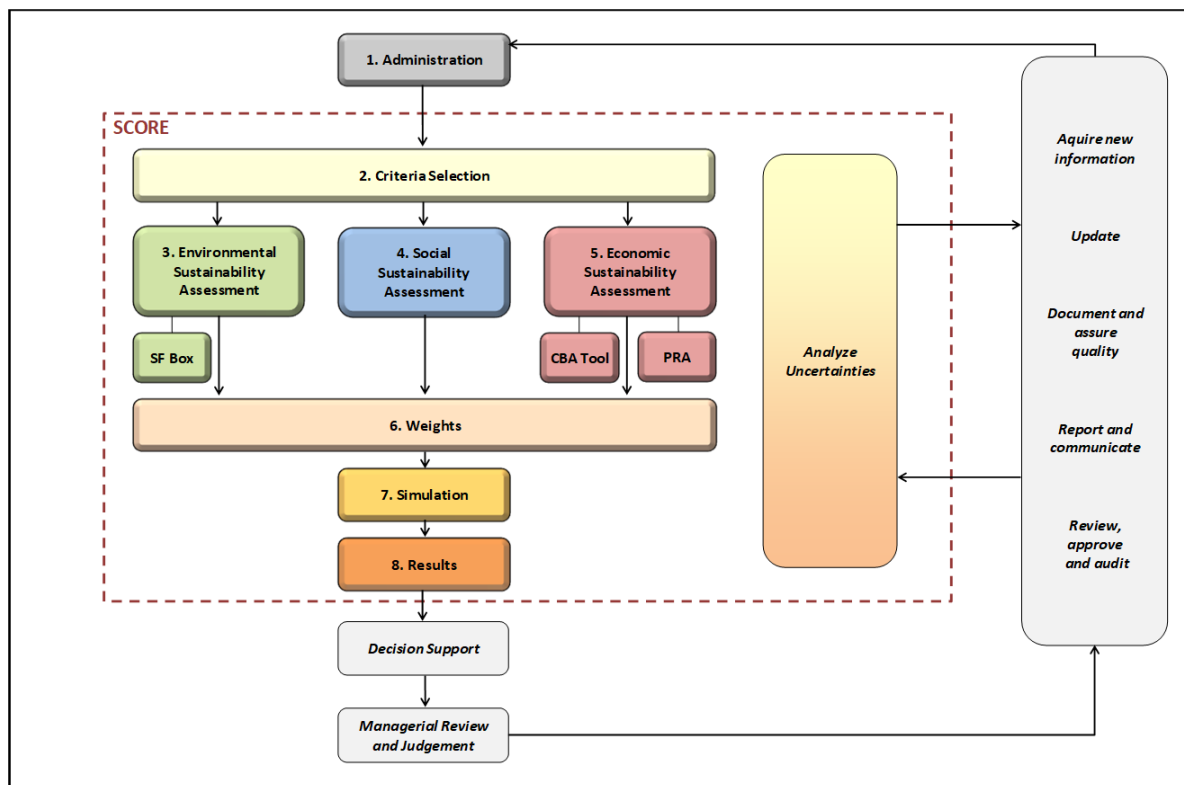


Figure 2 The basic workflow of the SCORE tool.

Economic sustainability is assessed in SCORE based on the economic profitability of the alternatives, which means that CBA is used for this assessment. This also implies that economic sustainability is assessed given the ethical starting points for CBA, which are usually described as anthropocentric consequential ethics based on individuals' preference satisfaction (Perman et al., 2011). Such an ethic involves a focus on the consequences of actions for people, and it is considered reasonable to weigh different consequences against each other and also to aggregate consequences for different people. However, there are many contrasts to such an ethic, and SCORE recognises that such contrasts provide complementary perspectives that together can provide a holistic assessment consistent with sustainability. One such contrast is a focus on human rights, where rights ethics can justify that every human being should have the right to good health, good education, work, and so on. In SCORE, the social dimension is used to capture such aspects. Another contrast is a focus on the rights of nature, where environmental ethics can justify that nature has a value in itself, not just an instrumental value for humans (which is incidentally in line with the preparatory work of the Swedish Environmental Code (Strömberg, 2016; Nordzell et al., 2017)). The environmental dimension is used in SCORE to capture such aspects.

In SCORE, for the economic dimension, there is only one criterion: economic profitability. In the social dimension, there are six criteria:

- Local environmental quality and amenity
- Cultural heritage
- Health and safety
- Justice
- Local participation
- Local acceptance

In the environmental dimension, there are eight criteria:

- Soil
- Flora and fauna
- Groundwater
- Surface water
- Sediment
- Air
- Non-renewable natural resources
- Non-recyclable waste

The researchers defined the criteria based on literature and discussions in focus groups and interviews. Within each dimension, it is important to avoid dependencies and overlaps between criteria, as this could lead to some impacts having a greater impact on the results of the analysis than intended. It is therefore intentional that the user *cannot* add criteria to SCORE, but can choose to exclude criteria that are not deemed relevant in a specific case. For example, the criterion "cultural heritage" can be excluded if there is no cultural heritage at a particular contaminated site.

The economic criterion of economic profitability is measured monetarily through benefits and costs in a CBA, while the criteria in the environmental and social dimensions are measured semi-quantitatively through scores. For each criterion, the user has to indicate a score on a scale from –10 to +10 depending on the user's judgement of whether the policy alternative would lead to a very positive effect on the criterion compared to the reference alternative (+6 to +10), positive effect (+1 to +5), no effect (0), negative effect (– 1 to – 5) or very negative effect (– 6 to – 10).⁵ As shown in Figure 3, the scoring was done in a workshop format, with discussions between stakeholders leading to an overall judgement on which score is the most likely.⁶

For each criterion, the scoring is guided by a table describing what is meant by 'very positive impact', 'positive impact', and so on, including examples and key questions to ask when assessing the score. These guidance tables are important to define the scale from –10 to +10 and to make the difference between different scores understandable and transparent. The

⁵ What is scored in SCORE is more precisely a number of sub-criteria within each criterion, but for simplicity we disregard this additional level here.

⁶ In addition, the user should assess the uncertainty in the score estimation and in the costs and benefits. Through applications of probability distributions and simulations, SCORE also calculates the uncertainty of all results, but we do not go into these details in this brief description.

SCORE tables are designed for impacts of remediation options, but their general layout is shown below.

Table 2 The general layout of the guidance tables provided for each environmental and social criterion in the SCORE tool.

1. Scoring guide (the effect is assessed in comparison to the reference alternative)				
<i>Very negative effect: -6 to -10</i>	<i>Negative effect: -1 to -5</i>	<i>No effect: 0</i>	<i>Positive effect: +1 to +5</i>	<i>Very positive effect: +6 to +10</i>
Verbal and general description of what a 'very negative effect' is.	Verbal and general description of what a 'negative effect' is.	Verbal and general description of what a 'no effect' is.	Verbal and general description of what a 'positive effect' is.	Verbal and general description of what a 'very positive effect' is.
Concrete examples of changes that would have a 'very negative effect'	Concrete examples of changes that would have a 'negative effect'	Concrete examples of changes that would have a 'no effect'	Concrete examples of changes that would have a 'positive effect'	Concrete examples of changes that would have a 'very positive effect'
Examples of indicators that can be used, including, where possible, specifications of which indicator values can be interpreted as a 'very negative impact'	Examples of indicators that can be used, including, where possible, specifications of which indicator values can be interpreted as a 'negative effect'	Examples of indicators that can be used, including, where possible, specifications of which indicator values can be interpreted as a 'no effect'	Examples of indicators that can be used, including, where possible, specifications of which indicator values can be interpreted as a 'positive effect'	Examples of indicators that can be used, including, where possible, specifications of which indicator values can be interpreted as a "very positive effect"
2. Key questions and sources of information. A number of key questions that the user should answer are listed. Suggestions are also provided for sources of information that may be needed to carry out the assessment.				
3. Explanation of how policy options may affect the criterion. Here examples are provided of how the criterion can be affected by different policy options, so that the user gets a good understanding of how the criterion can be affected.				
4. Examples of scoring. At least one example of scoring is provided, including a justification for the selected score.				

After scoring, each environmental and social criterion is weighted according to the user's judgement of its importance relative to the other criteria in the respective dimension. A numerical value (*I*) between 0 and 25 is used to measure the importance of the criterion, after which the weight is calculated as *I* for the criterion divided by the sum of *I* for all criteria in the dimension.⁷ A weighted index for each policy option (*i*) can then be calculated for the environmental (*H_E*) and social (*H_S*) dimensions:

$$H_{E,i} = \sum_j w_{E,j} K_{E,j} \text{ and } H_{S,i} = \sum_j w_{S,j} K_{S,j},$$

⁷ Example: If there were only two criteria in a given dimension and criterion 1 is given the value 5 and criterion 2 is given the value 15, the weight of criterion 1 would be equal to 5/(5+15)=0.25 and the weight of criterion 2 would be equal to 15/(5+15)=0.75.

where w_j is the weight of the j th criterion and K_j is the score of the j th criterion.⁸ The equivalent of H for the economic dimension is the net present value of each policy option (NPV_i).

It was mentioned above that SCORE includes the possibility of a non-compensatory approach. This consists in considering the results for each sustainability dimension separately and requiring that a policy option must result in $H_E > 0$, $H_S > 0$ and $NPV > 0$, i.e. lead to a positive result for all sustainability dimensions.

Where trade-offs between sustainability dimensions are considered reasonable, SCORE provides information in the form of a normalised weighted index for each policy option (H_i). This index is calculated according to the equation below, which includes a weighting at dimension level (W_E , W_{SC} and W_{NPV} , respectively), i.e. the dimensions can be assigned different weights in the aggregation. The index has a value between –100 and +100, where a positive value indicates that the policy option as a whole can be expected to contribute to sustainable development. It should be noted that the index is based on a relative ranking of the policy options. Note also that for the calculation of this index, it is not a problem that the impacts of the policy option have been measured differently in the three dimensions: semi-quantitatively with scores in the environmental and social dimensions respectively and monetarily in the economic dimension.

$$H_i = 100 \left[\frac{W_E \frac{H_{E,i}}{\max[\max(H_{E,1..N}); |\min(H_{E,1..N})|]} + W_{SC} \frac{H_{S,i}}{\max[\max(H_{S,1..N}); |\min(H_{S,1..N})|]}}{W_{NPV} \frac{NPV_i}{\max[\max(NPV_{1..N}); |\min(NPV_{1..N})|]}} \right]$$

SCORE illustrates the different steps of an MCA that characterise the use of a linear additive method, and also how an element of non-compensatory MCA method can come into play. It is particularly important to note the following:

- Scoring is often used in MCA. In order to avoid arbitrary and opaque scoring, it is important to have a definition of the scoring scale, so that it is clear why a particular policy option is given a certain score. For SCORE, guidance tables were developed for this purpose, see above.
- SCORE as a whole aims to provide an overall sustainability assessment that takes into account more than economic considerations. The tool is designed so that the economic profitability is assessed in the CBA part of the tool. However, it should be noted that it is entirely conceivable to have an alternative design where the MCA as a whole aims to assess only economic considerations, but where the costs and benefits that are not currently deemed possible to monetise are removed from the CBA and form separate criteria. These criteria could be scored and then included in an overall assessment using the same type of normalised weighted index as above. Another possible route to take is to translate monetised impacts into scores and thus gain comparability with non-monetised impacts. This approach was used by Barton et al (2020), which we use below as an example of the application of multi-attribute methods.

⁸ A common problem with the use of scoring scales is that it can be questioned whether equidistance in the scoring scale applies, i.e. whether the distance between e.g. scores 1 and 2 is the same as the distance between scores 6 and 7.

4.3.3 Example of an MCA using a multi-attribute approach: Barton et al.

An example of an MCA using a type of multi-attribute method is a study by Barton et al. (2020), whose application is the assessment of different policy options regarding environmental measures in the Mandalselva in Väst-Agder in southernmost Norway, a well-known salmon river. In relation to SCORE, there are both methodological similarities and differences. One similarity is that assessments of uncertainty are included in the analysis; Barton et al. use a so-called Bayesian network. This is based on the specification of conditional probabilities, i.e. the probability of an event given that another event has occurred. The structure of the network makes it possible, for example, to calculate the benefits of the action alternatives for a particular actor, given the preferences that a particular actor has stated for different action alternatives, see below. Hugin software was used to conduct the analysis (www.hugin.com).

The study used the following four criteria. As in the case of SCORE, the number of criteria was thus given at the outset.

1. Costs, which included costs for environmental measures and revenue losses due to hydropower production losses.
2. Smolt production.
3. Fishing opportunities, which refers to the existence of suitable places for sport fishing.
4. Aesthetics, which is about how the river's flow is perceived in terms of beauty.

A number of different policy options with different impacts on these criteria were identified (symbolised by the rectangle at the top of Figure 4). The options varied with respect to flow, dam removal and spawning gravel placement. A number of models were used to predict the impact of the policy options on power generation and smolt production. Impacts on fishing opportunities were described based on a classification of meso-habitats in the river and expert judgement, and impacts on aesthetics were described using photomontages and expert judgement. All impacts were assessed in relation to a reference alternative.

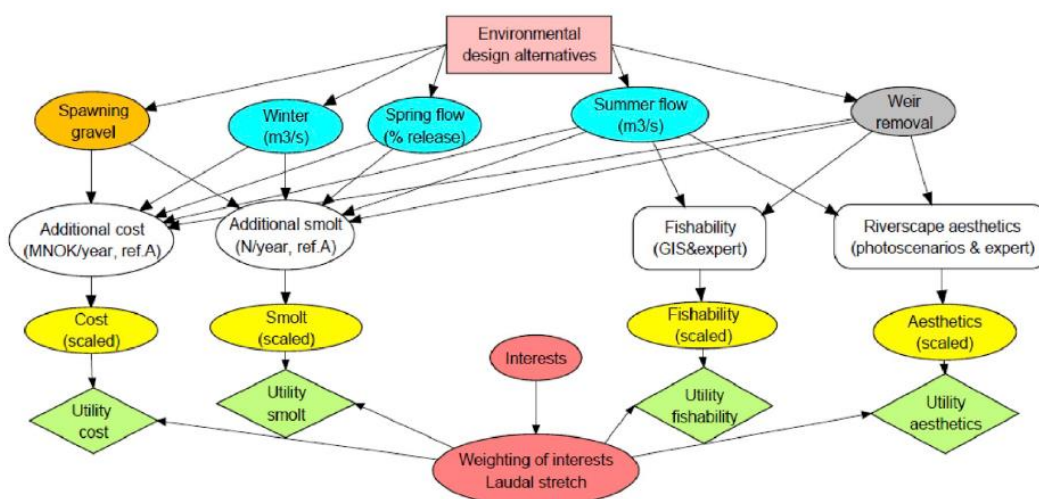


Figure 4 A schematic of the main components of the Bayesian network used in the MCA study by Barton et al. From Barton et al. (2020, Figure 3).

For each policy option, the impact was rescaled to a scale from –1 to +1. This is symbolised in Figure 4 by the transition between the third and fourth levels. The fact that the impact on all criteria is expressed in a uniform normalised scale is a typical feature of multi-attribute methods. This rescaling is an important step in the analysis, and we therefore look at it in more detail for each of the criteria:

1. Costs: Model simulations provided conditional probabilities for different cost outcomes given the policy options. Cost outcomes were expressed in SEK and rescaled in a broadly linear fashion to tenths intervals between –1 and 0. See Figure 5, which shows, for example, that the highest cost for any of the policy options was between NOK 6 and 6.5 million per year. This maximum cost was rescaled to the tenth interval between –1 and –0.9, and the simulations obviously indicated that the conditional probability is equal to 1 that the maximum cost falls in this tenth interval. For costs falling between 5 and 6 million SEK, the conditional probability is equal to 0.16 that the cost falls in the tenth interval between –1 and –0.9, equal to 0.64 that it falls between –0.9 and –0.8, and equal to 0.20 that it falls between –0.8 and –0.7, and so on.

Cost (scaled)							
Expression	dC / 6.5						
Additional c...	-6.5 - -6	-6 - -5	-5 - -4	-4 - -3	-3 - -2	-2 - 0	0 - 0
-1 - -1	0	0	0	0	0	0	0
-1 - -0.9	1	0.16	0	0	0	0	0
-0.9 - -0.8	0	0.64	0	0	0	0	0
-0.8 - -0.7	0	0.2	0.44	0	0	0	0
-0.7 - -0.6	0	0	0.56	0.08	0	0	0
-0.6 - -0.5	0	0	0	0.68	0	0	0
-0.5 - -0.4	0	0	0	0.24	0.4	0	0
-0.4 - -0.3	0	0	0	0	0.6	0.04	0
-0.3 - -0.2	0	0	0	0	0	0.32	0
-0.2 - -0.1	0	0	0	0	0	0.32	0
-0.1 - 0	0	0	0	0	0	0.32	0
0 - 0	0	0	0	0	0	0	1

Figure 5 Rescaling of costs in millions of Norwegian kroner per year to a scale in tenths from –1 to 0. From Barton et al (2020, Supplementary Material).

2. Smolt production: Again, modelling simulations were used to provide conditional probabilities for different production outcomes given the policy option. The outcomes were expressed in number of smolts per year and were rescaled roughly linearly to tenth intervals between 0 and 1 in a similar way as for costs.

For the impact on the criteria of costs and smolt production, it is clearly essential that recognised models are used to calculate the impact.

3. Fishing opportunities: Based on information about the impact on meso-habitats in the river, fishing opportunities were assessed by three different fishermen on a three-point scale, with each fisherman's judgement given equal weight in constructing an empirical probability distribution for the judgements on a scale of tenths from –1 to +1.
4. Aesthetics: Based on photomontages, the impact on aesthetics was assessed by the power company's environmental manager and an expert panel consisting of five scientists on a three-point scale, where the environmental manager and the expert panel were given equal weight in constructing an empirical probability distribution for the assessments on a scale of tenths from –1 to +1.

For the impact on the criteria of fishing opportunities and aesthetics, it is clear that the assessment depends on the people who acted as assessors and the weight given to each person's judgement. Barton et al. emphasise that the study was conducted for research purposes and therefore a convenience sample of assessors was used. If the aim had been to arrive at a rigorous decision-making basis, the selection of people making judgements would have needed to be done with great care. It can also be observed that since each individual judgement on fishing opportunities and aesthetics was an input to the analysis, the scaling is not necessarily linear. Non-linear scaling may well arise as a result of differences in the preferences of the impact assessors. There is a procedural difference here compared to SCORE, the application of which involves workshops with stakeholders to try to reach consensus in scoring. In principle, however, there is nothing to prevent the score assessment in SCORE being made as a weighing of individual input instead, but a potential disadvantage is that you then miss out on the interaction that takes place at a workshop and which can be consensus-building (see Söderqvist, 2019, for an analysis of the actors' experiences of SCORE).

The next type of input to the analysis is the weights for the criteria. These were set by different stakeholders based on how important each stakeholder considered each criterion to be. In the study, six different stakeholders representing different interests were consulted, of which the power company was one. Each stakeholder was individually asked to allocate 100 points between the different criteria based on their importance. The scores were scaled to conditional probabilities by dividing by 100. Not unexpectedly, there were large differences between the stakeholders in terms of the importance of the different criteria, see Figure 6.

Interests	Actor A ...	Actor B	Actor C	Actor D	Actor E	Actor F
Cost	0.53	0	0.1	0.05	0.2	0.2
Smolt	0.3	0.5	0.7	0.05	0.4	0.4
Fishability	0.1	0.3	0.1	0.05	0.2	0.3
Aesthetics	0.07	0.2	0.1	0.85	0.2	0.1

Figure 6 The six stakeholders' weighting of each criterion based on how important they considered the criterion to be in relation to each other. The weighting totals 1 for each stakeholder. The power company is actor A, the other actors are anonymous. From Barton et al (2020, Figure 4).

With all the above inputs, the analysis can be carried out using simulations. Figure 7 shows an example of the results of a run where each of the six actors has an equal weight in the analysis (each actor has probability 0.1667 in the Interests box). The equal weighting of the six actors in turn means that the final weight for each criterion is equal to an average of each actor's weighting in Figure 6. For example, the weight for cost in the form of the probability 0.18 in the Weighting of interests box is calculated from the Cost row in Figure 6 as $(0.53+0+0.1+0.05+0.2+0.2)/6$. The resulting expected benefits for each policy alternative are shown in the Environmental design alternatives box in Figure 7. The probabilities of 0.769 for each policy option represent that each policy option is a priori equally likely to happen. The expected benefits relative to the reference alternative are shown on the right of the box and indicate that the P2-Wp+G option is the best one (yielding a benefit of 0.51), where P2 refers to a particular water flow option, Wp to dam removal and G to the placement of spawning gravel. These expected benefits should then be interpreted given the structure of the MCA model as a whole and the inputs to the model in terms

of (a) the assessment of the impact of the policy options on the criteria, (b) the weighting of the criteria by the six actors and (c) the weighting of the six actors. The strength of the Bayesian network is that it is possible to make runs with changed conditions, and Barton et al. exemplify how the outcomes in the form of expected utility for the action alternatives change if a certain actor is given a weight of 1 or if a certain criterion is given a final weight of 1.

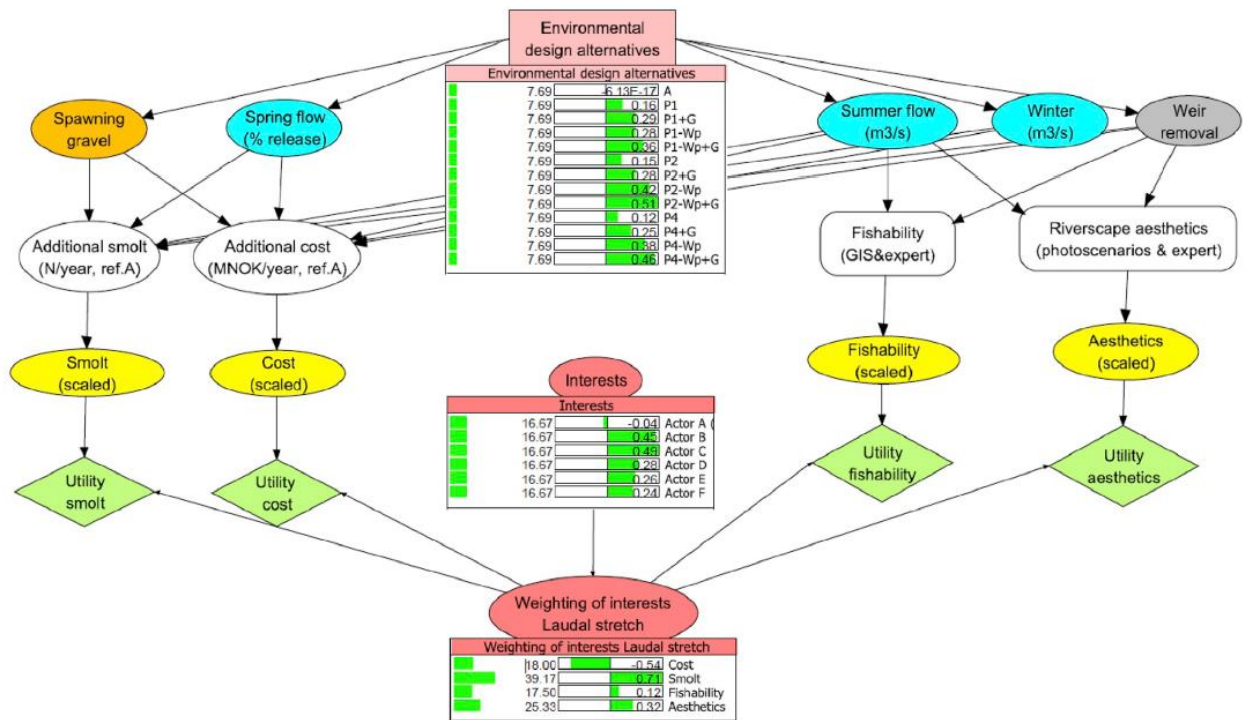


Figure 7 Example of results from the MCA analysis by Barton et al. From Barton et al. (2020, Figure 6).

In conclusion, the following observations can be made:

- Compared to SCORE, the study is based on more advanced modelling, which provides rich analytical opportunities, but it can be difficult for a layman to understand the mechanics of the model.
- Both the Barton et al. study and SCORE illustrate how dependent an MCA is on assessments of the impact of policy options on criteria, the way criteria are weighted, the selection of stakeholders to involve in assessments and weightings, and how this involvement is organised. As SCORE has been applied so far, scoring and weighting relies heavily on consensus building in workshops, including tools for such consensus building and documentation of workshop discussions, whereas Barton et al. rely on incorporating individual judgements and weightings into the model.
- In the study, the monetary component (costs) is converted into a kind of point scale. In SCORE, there is no such conversion. In both cases, a major question remains: whose preferences should be used to weigh monetised impacts against non-monetised impacts? There are many possible answers. From the point of view of CBA theory, the answer is that the preferences of the individuals concerned should be used, which explains why attempts to monetise environmental impacts through environmental valuation studies

make use of surveys and interviews aimed at the general public (stated preference methods) or of data on the market behaviour of individuals (revealed preference methods), cf. sections 3.1 and 3.2.

4.3.4 Example of an MCA using the AHP approach: Rosso et al.

The Analytical Hierarchy Process (AHP) and its relative, the Analytical Network Process (ANP), is a much debated MCA method due to disagreements about its properties (see, for example, Smith and von Winterfeldt, 2004). As mentioned above, one of the controversies is that rank reversal can occur even when logically it should not, for example, the introduction of a new policy option that is unrelated to previously analysed policy options can change the ranking of the previously analysed policy options. Nevertheless, AHP has become a common MCA method (cf. Cegan et al., 2017), probably because it is based on pairwise and relative comparisons. Such comparisons can be perceived as easy to understand.

Rosso et al. (2014) describe how AHP can be used to select the best hydropower project given a wide range of environmental, economic, technical and socio-political aspects. The background is that there is a potential for hydropower development in some parts of the Sesia Valley in northern Italy. The AHP is based on structuring the decision problem according to the following hierarchy:

1. Objective to be met: To make the best choice of course of action in any sense.
2. Main criteria for the assessment.
3. Sub-criteria of each main criterion.
4. Any sub-sub-criteria for each sub-criterion, and so on.
5. Action options to be assessed using the criteria.

Figure 8 shows how Rosso et al. defined this hierarchy in their application. The main criteria relate to sustainability dimensions, but compared to SCORE, Rosso et al. also include a technical dimension. The more precise definition of each sub-criterion is given in the article, but we do not go into these details here. As shown in the figure, they added an additional hierarchical level to the above list, namely a selection of stakeholders who made judgements on the criteria and whose judgements were weighted in the final assessment of the policy options. We will return to how this weighting was done at the end of the example.

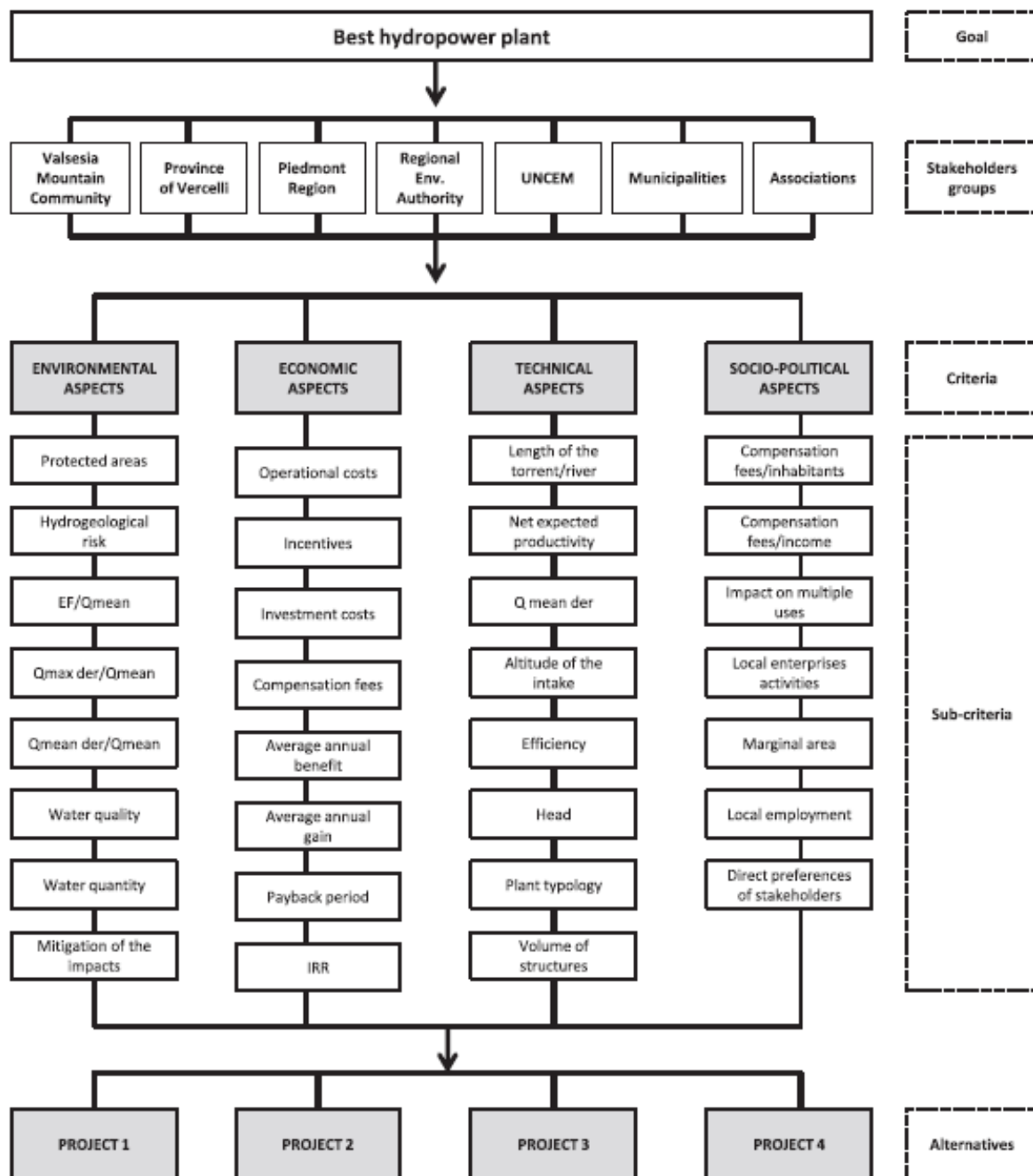


Figure 8 The hierarchical structure of the decision problem in Rosso et al. From Rosso et al. (2014, Figure 1).

There should be an equal number of sub-criteria under each main criterion and, for methodological reasons, the number of sub-criteria should not exceed nine. Each main criterion and all sub-criteria within each main criterion are assessed in pairs on a nine-point scale. Figure 9 shows the assessment made by a given stakeholder of the importance of each of the four main criteria relative to each of the other main criteria. These pairwise judgements were compiled in a 4×4 matrix with the main criteria as rows and columns and through mathematical calculations, which we do not go into here, the relative priority between the main criteria can be determined as weights for each main criterion between 0 and 1 that sum to 1.

Questionnaire for the evaluation of the criteria ("Stakeholder").

"With reference to the choice of the best performing project, which of the two aspects is more important? And to what extent?"																		
Environmental aspects	9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	Economic aspects
Environmental aspects	9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	Technical aspects
Environmental aspects	9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	Socio-political aspects
Economic aspects	9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	Technical aspects
Economic aspects	9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	Socio-political aspects
Technical aspects	9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	Socio-political aspects

Figure 9 Pairwise comparison of the importance of the main criteria. The value 9 on the left in the first row means that environmental aspects are extremely more important than economic aspects, and the value 9 on the right means that economic aspects are extremely more important than environmental aspects. A particular stakeholder's judgement is shown by the shaded fields, and the selected value 2 in the first row means that the stakeholder judged environmental aspects to be slightly more important than economic aspects. From Rosso et al. (2014, Table 5).

The same type of pairwise judgements was made for each set of sub-criteria within each main criterion. For example, there were eight sub-criteria within the main criterion *Environmental aspects*, see Figure 8. Consequently, $[8(8-1)]/2 = 28$ pairwise judgements were needed, which were then compiled into an 8×8 matrix, from which the weights for each sub-criterion could then be calculated.⁹

Finally, judgements were made on how each policy option performs in relation to the sub-criteria. In the study, these judgements were not made by stakeholders, but by experts. In this step, the policy options are compared in pairs for each of the sub-criteria, see Figure 10 for an example where the comparison of the policy options is done for the sub-criterion *Landscape quality and protected areas*. For each sub-criterion, this results in weights for each of the policy options.

Questionnaire for the evaluation of the alternative projects with reference to the landscape quality and protected area sub-criterion.

"With reference to the landscape quality and protected areas sub-criterion, which of the two alternatives is preferred? And to what extent?"																		
Alternative 1	9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	Alternative 2
Alternative 1	9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	Alternative 3
Alternative 1	9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	Alternative 4
Alternative 2	9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	Alternative 3
Alternative 2	9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	Alternative 4
Alternative 3	9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	Alternative 4

Figure 10 Pairwise comparisons of the four policy options based on their performance in relation to the sub-criterion Landscape quality and protected areas. From Rosso et al (2014, Table 10).

Using the calculated weights for the policy options, the sub-criteria and the main criteria, a final assessment of the policy options can be calculated for *each* actor, see Figure 11. Rosso et al. also calculated a final assessment of the policy options *across all actors* by assigning each actor a weight between 0 and 1, where the weights sum to 1. These weights were set using the type of stakeholder analysis described in the FRAM-KLIV report (Söderqvist et al., 2017, p. 36ff):

1. An identification of all relevant actors; Rosso et al. identified 42 actors.
2. An assessment of the level of interest and influence of each stakeholder; Rosso et al. used a score scale from 0 to 10, with 0 representing very low interest and influence and 10 representing very high interest and influence.

⁹ The number of pairwise comparisons is given by $[n(n-1)]/2$, where n is the number of items to be compared.

3. For each actor, Rosso et al. calculated the importance of each actor as the product of the two scores, after which a weight for each actor was calculated by normalising each actor's product to the sum of the product of all actors. These actor weights were then used to arrive at a final assessment of the policy options across all actors.

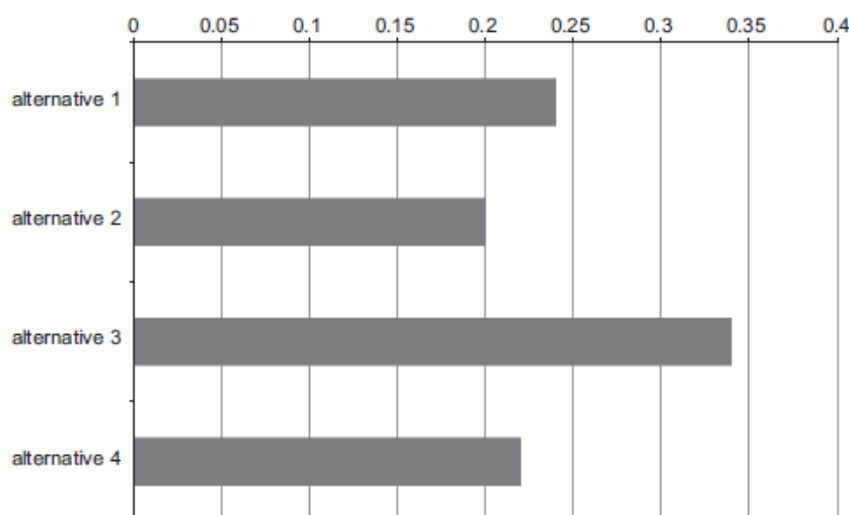


Figure 11 The ranking of the four policy options for a given stakeholder. Based on this stakeholder's assessment of the importance of the main criteria and sub-criteria, and the experts' assessment of how the policy options perform in relation to the sub-criteria, policy option 3 is ranked as the best. From Rosso et al (2014, Figure 4).

As with SCORE, other main criteria than sustainability dimensions can of course be used. Saaty (1994) exemplifies an AHP application where benefits and costs are two main criteria with a number of sub-criteria. The benefit side and the cost side are then examined separately with AHP in such a way that the action alternatives are ranked partly on the benefit side and partly on the cost side. Finally, a benefit-cost ratio is calculated to assess which policy option is the best relative to each other. It should be emphasised that this is a relative assessment and does not provide CBA-style information on whether the net benefits are positive or negative in absolute terms.

The pairwise comparisons and the use of specific mathematical calculations are two features that make the AHP method different from a linear additive method à la SCORE or a multi-attribute method à la Barton et al. (2020). But some fundamental challenges are common, for example:

- What criteria should be used?
- Who will make the judgements, both on how the policy options perform against the criteria and on how important the criteria are?
- How should the levels of semi-quantitative scales be interpreted? What basis should be used to assess which score or grade should be given?

4.3.5 Example of an MCA using an outranking method

In order to illustrate how outranking methods work, we use the illustration of the ELECTRE method found in DCLG (2009). Although generic, the illustration captures the essence of the method in a more fundamental way than in applications such as Maslov et al. (2014) and Saracoglu (2015). ELECTRE stands for *Elimination et choix traduisant la réalité* (Elimination and

choice expressing the reality) and was proposed in the 1960s as a method for multi-criteria analysis, see Roy (1968). The method has evolved over time into a family of ELECTRE methods (Figueira et al., 2013). Below, the method is described based on the basic ELECTRE I in order to provide an understanding of the pairwise comparison between policy alternatives on which the sorting is based.

Suppose that ten policy options have been identified to solve a problem, and the best policy option is to be selected based on how they perform against six different criteria. The performance matrix is shown in Table 3. The higher the value, the better the policy option performs. This is for each given criterion, i.e. the values cannot be compared column by column, only row by row. An interesting feature is thus that the performance of the policy options can be measured in different ways for different criteria. The criteria are given weights according to their importance, and these weights sum up to 1. In the example, the weights shown in the first column of Table 3 were adopted.

Table 3 Matrix of how policy options A-J perform with respect to criteria 1-6. The first column is the weights for each criterion according to their importance. From DCLG (2009), Table A6.1.

	Weight	A	B	C	D	E	F	G	H	I	J
1	0.25	6	2	16	10	11	5	16	17	10	5
2	0.10	300	450	350	500	380	250	390	400	410	250
3	0.15	27	21	27	20	23	31	24	22	16	18
4	0.25	18	19	12	12	20	10	18	26	23	21
5	0.05	570	400	420	450	400	430	510	380	410	400
6	0.20	12	23	18	20	16	18	21	23	20	22

To obtain information on which options dominate over other options, two different indices are calculated for each pair of policy options from the information in Table 3: one for concordance and one for discordance. The concordance index for the policy option pair (i,j) is equal to the sum of the criteria weights for those criteria where policy option i is at least as good as policy option j . For example, the concordance index for the pair (C,D) is equal to $0.25+0.15=0.40$, because policy option C performs better than D for criteria 1 and 3, but not for any others. The concordance index for the pair (D,C) is correspondingly equal to $0.10+0.05+0.20=0.35$.

The discordance index for the pair (i,j) is equal to zero if policy option i is better than j for all criteria. However, if policy option j is better than i for at least one criterion, the discordance is calculated for each criterion where j is better than i as a ratio there:

- The numerator is equal to the difference in performance between j and i .
- The denominator is equal to the maximum difference in performance for all pairs of policy options.

The discordance index of (i,j) is equal to the highest value of this ratio and lies between 0 and 1. For the pair (C,D), D performs better than C for criteria 2, 5 and 6:

- Criterion 2: $(500-350)/(500-250) = 150/250 = 0.6$
- Criterion 5: $(450-420)/(570-380) = 30/190 = 0.158$
- Criterion 6: $(20-18)/(23-12) = 2/11 = 0.182$

The discordance index for (C,D) is thus equal to 0.6. The main point of the discordance index is to be able to signal if any policy option is performing really badly, even if this bad performance would only apply to one of the criteria.

Which policy alternatives dominate over other alternatives is then examined by using the threshold values of the respective indices. An alternative i is characterised by dominance over another alternative j if the concordance index for (i,j) exceeds the concordance threshold while the discordance index for (i,j) is below the discordance threshold. By making such comparisons for all policy options, it is possible to sort out those policy options that dominate over at least one other policy option while not being dominated themselves. This sorting out results in a number of policy options that can be considered as promising solutions to the problem and should therefore be further investigated, for example by tightening the thresholds. If it turns out that no policy option dominates, it is instead appropriate to relax the thresholds. One of the strengths of the method is that this gradual sorting out stimulates discussions between decision-makers and other stakeholders about the strengths and weaknesses of different policy options, but a weakness is the arbitrariness that may exist in the choice of thresholds and their fine-tuning.

To make a selection of the ten alternatives of the example, the concordance index and the discordance index are compiled for all pairs of alternatives in the matrices in Table 4 and Table 5. To be decisive, the concordance threshold should be relatively high and the discordance threshold relatively low, but to make a first selection of alternatives, the thresholds can initially be set to the mean value of the respective index. These mean values are equal to 0.48 for the concordance index and 0.61 for the discordance index.

Table 4 Concordance matrix for policy options A-J. From DCLG (2009), Table A6.2.

	A	B	C	D	E	F	G	H	I	J
A	-	0.45	0.30	0.45	0.20	0.65	0.20	0.20	0.20	0.55
B	0.55	-	0.55	0.60	0.30	0.55	0.55	0.15	0.45	0.45
C	0.55	0.45	-	0.40	0.65	0.60	0.15	0.20	0.45	0.55
D	0.55	0.40	0.35	-	0.35	0.85	0.10	0.15	0.30	0.55
E	0.80	0.65	0.35	0.65	-	0.60	0.25	0.20	0.40	0.50
F	0.35	0.45	0.20	0.15	0.40	-	0.15	0.20	0.20	0.20
G	0.55	0.45	0.60	0.90	0.75	0.85	-	0.20	0.65	0.55
H	0.80	0.65	0.80	0.85	0.80	0.80	0.80	-	0.85	0.95
I	0.80	0.55	0.55	0.25	0.60	0.80	0.35	0.15	-	0.65
J	0.45	0.50	0.45	0.45	0.45	0.45	0.45	0.05	0.35	-

Table 5 Discordance matrix for policy options A-J. From DCLG (2009), Table A6.3.

	A	B	C	D	E	F	G	H	I	J
A	-	1.00	0.67	0.80	0.36	0.55	0.82	1.00	0.73	0.91
B	0.89	-	0.93	0.53	0.60	0.67	0.93	1.00	0.53	0.20
C	0.79	0.45	-	0.60	0.50	0.27	0.47	0.88	0.69	0.56
D	0.63	0.44	0.47	-	0.50	0.73	0.40	0.88	0.69	0.56
E	0.89	0.64	0.33	0.48	-	0.53	0.58	0.64	0.36	0.55
F	0.74	0.80	0.73	1.00	0.63	-	0.95	1.00	0.81	0.69
G	0.32	0.24	0.20	0.44	0.13	0.47	-	0.50	0.31	0.19
H	1.00	0.20	0.33	0.40	0.10	0.60	0.68	-	0.16	0.11
I	0.84	0.33	0.73	0.76	0.47	1.00	0.53	0.47	-	0.18
J	0.89	0.80	0.73	1.00	0.52	0.87	0.73	0.80	0.64	-

We can now begin by examining how policy option A relates to all other policy options. It turns out that A has a concordance index that exceeds 0.48 with respect to F and J and a discordance index that is less than 0.61 with respect to E and F. Thus, both the concordance condition and the discordance condition are only fulfilled with respect to F, i.e. A dominates over F. The following picture emerges from the corresponding investigation for all policy alternatives:

- A dominates over F
- B dominates over D
- C dominates over E, F and J
- D dominates over J
- E dominates over D, F and J
- G dominates over A, C, D, E, F, I and J
- H dominates over B, C, D, E, F, I and J
- I dominates over B, E and J

Overall, G and H look the most promising, as they dominate over a range of policy options while no policy option dominates over them. To get a clearer picture, the concordance and discordance thresholds can be gradually increased and decreased respectively to become more decisive. Different techniques are available to find the single best option and to obtain a ranking between the policy options.

In summary, the methodology is based on sorting out policy options by comparing the options against each other *for each criterion*. Thus, performance against the criteria need not be measured in the same way; for example, the measurement for one criterion may be in monetary terms while the measurement for another criterion may be square metres of a particular habitat, etc. This is an interesting feature, and the method's process of sorting out dominant policy options can be both a strength (stimulating discussion between stakeholders?) and a weakness (arbitrariness in how thresholds are chosen?), but the method itself does not solve some fundamental challenges for multi-criteria analysis:

- What criteria should be used?
- Who will make the judgements, both on how the policy options perform against the criteria and on how important the criteria are?

5 The Leipzig model: starting from reference costs

In Germany, where water management is the responsibility of the Länder, an approach to assessing disproportionate costs has been developed that represents a methodological approach that is partly different from CBA and MCA. One explanation for this approach is that the German water authorities generally do not support the routine use of CBA to assess disproportionate costs (Klauer et al., 2017). One reason for this is that there is a general scepticism in German public administration about using monetary environmental valuation (Martin-Ortega et al., 2014). Instead, the model often referred to as the Leipzig model has come into use. The name is explained by the fact that researchers at the University of Leipzig were pioneers in proposing the model (Ammermüller et al., 2011). The starting point of the model is to utilise information on past investments in environmental measures and, based on such information, calculate a reference cost that provides a threshold for what constitutes disproportionate costs. The threshold further takes into account the benefits of environmental measures through non-monetary expert judgements.

The application of the model takes place in four steps (Klauer et al., 2017):

1. Assessment of remedial costs of environmental measures needed to achieve good status/GEP in a water body.
2. Assessment of the benefits arising from the achievement of good status/GEP in the water body based on an assessment of how far the water body is from good status/GEP, known as the distance factor, which is assessed on a scale of 0-3. The value is set to 0 if the water body is already in good status for all assessed quality elements and to 3 if the water body is in poor status for all quality elements. The distance factor is assessed on the basis of available environmental monitoring data and existing status classification. In addition, the improvement of five additional benefits is assessed on a scale of 0-3, where 0 means that measures do not lead to any improvement in the relevant benefit.
3. Determination of a reference cost normalised to the river basin, and calculation of a disproportionate cost threshold for the water body.
4. Assessment of disproportionate cost by examining whether the costs of measures to achieve good status/GEP exceed the threshold.

The calculation of the threshold includes adjustments to the reference cost that take into account that a higher threshold is more reasonable if the current status is relatively far from good status/GEP and also that a higher threshold is more reasonable if the achievement of good status/GEP entails relatively large positive effects on the five benefits assessed in Step 2.

Klauer et al. (2017) describe two variants of the model, which differ in how the reference cost is calculated in step 3: the benchmark approach and the average cost approach. To arrive at a reference cost in Step 3, the latter approach uses past actual costs of action to achieve good status/GEP in water bodies, where the average cost of achieving good status/GEP is assumed to be a 'reasonable effort'. Instead, the benchmark approach is based on the actual national total public expenditure on water protection. In this way, the benchmark approach is less data intensive, which has been a request from practitioners.

The five co-benefits included in the assessment are:

- Improved habitats or improved connectivity.
- Flood defences.
- Improved quality or quantity of drinking water.
- Reduced erosion, reduced loss of soil fertility.
- Increased attractiveness for tourism and recreation, cultural heritage.

The calculated average cost of measures per square kilometre of water at 'reasonable effort' is multiplied by the summed average (i.e. a value between 0 and 3) of the additional benefits and by the distance factor calculated. The result shows the limit of how much additional measures can cost for the specific water body without being considered unreasonably costly. (Klauer et al., 2017). Macháč et al. (2016) applied the method to the Stanovice reservoir in the Czech Republic. They used the average annual investment of public funds from 1994 to 2009 to derive the average cost of measures per square kilometre of water as a starting point. The researchers experienced problems with not having data on various quality factors to assess how far from good status the water body was. They pointed out that if the problem is generalised across several water bodies, it may be necessary to review which indicators should be used. They also felt that the valuation of the co-benefits was very subjective and that this could affect the outcome of the analysis. In a Spanish application of the methodology, the five co-benefits used in Klauer et al. (2017) were excluded because they were considered difficult to assess objectively (Bolinches et al., 2020).

In Sweden, a variant of the average cost approach to the Leipzig model has been developed, the so-called BOKS model (Ivarsson et al., 2015a, 2015b). The BOKS model was developed as a screening tool to identify areas where the potential disproportionateness of intervention costs should be studied more closely, but there is nothing to prevent the BOKS model from being developed into a more rigorous tool that also determines thresholds as described above.

As shown in Klauer et al. (2017), the Leipzig model includes several different assessment elements in the form of scoring and weighting, and the results of the model may be sensitive to changes in these elements.

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Assessment methods for disproportionate costs according to the Swedish Water Management Ordinance (2004:660)

An overview

Assessing the disproportionate costs of environmental measures involves weighing up elements that describe competing interests in decision-making. To make such trade-offs, a range of different assessment methods are available. This report introduces the two methods most commonly used in the literature to weigh the costs and benefits associated with environmental measures and socially beneficial activities such as hydropower: *cost-benefit analysis* (CBA) and *multi-criteria analysis* (MCA). The report also describes the so-called Leipzig model.

SwAM, The Swedish Agency for Marine and Water Management is the responsible government agency tasked to protect, restore and ensure sustainable use of freshwater resources and seas including fisheries management.