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Modern Cost–Benefit Analysis of Hydropower Conflicts

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1. Introduction

Per-Olov Johansson and Bengt Kriström*

This book sheds some light on how current tools of welfare economics can be used to assess the benefits and costs of resource conflicts involving hydropower.[1] We have attempted to garner a set of chapters that paint a fairly broad picture of the issues involved. Consequently, we have tried to solicit papers from authors on both sides of the Atlantic, which inter alia means that we have been able to tap into the significant body of experience that already exists in the US. There is also a body of relevant knowledge in the hydropower-intensive countries of Europe, which are dominated by the Nordic countries. As these countries were electrified, resource conflicts surfaced and had to be resolved in some fashion. Along with decision-makers' need for comprehensive background information, methods and approaches were developed to meet this demand. Perhaps the most extensive investigations in this regard were carried out in Norway in the 1970s, when large investigative bodies were commissioned to look at hydropower investments using tools from a wide array of scientific disciplines.

We also tried to attract papers that shed some light on key methodological issues in our context, ranging from the intersection between cost–benefit analysis (CBA) and behavioral economics to appropriate statistical methodology for willingness-to-pay interval data with a particular form of censoring. In addition, hydropower supplies several different services into the electricity grid, including balancing power. Hydropower can conveniently be turned on and off on a time-scale that is unparalleled among the set of currently available technologies. Hence, when countries with substantial hydropower in the energy portfolio plan to expand their wind power share, balancing services become more valuable. Because electricity grids become increasingly interconnected, such as in Europe, this issue is not without interest even in countries that lack hydropower. This fact will be further discussed and illustrated below.

No major hydropower investment has been carried out in the Nordic countries for 20 years or so. For this reason, the CBA of hydropower in the Nordic countries is, it might appear, a barely flickering torch. However, two broad processes will change this state of affairs in a dramatic way in the Nordic countries as well as in the European Union proper.

First, the European Water Framework Directive (WFD) formalizes the demand for improved ecological status of water bodies within the Union in terms of quantified minimum levels. The WFD is crystal-clear on the significant need for cost–benefit analysis in this endeavor. For example, exemptions can be granted in cases where it can be demonstrated that the social costs of reaching the stipulated quality levels significantly outweigh the social benefits. Nick Hanley's chapter in this book (Chapter 2) provides a detailed account of the links between CBA and water quality management, with particular focus on the implications for CBA of the WFD.

Second, the Union has unleashed its 'triple 20 by 2020' policy, which inter alia includes reducing carbon emissions by 20 percent and increasing the share of renewable energy by 20 percent in the member states. If the EU is to reach its own goals on climate and renewable energy, there is little doubt that hydropower will have a role to play. The so-called energy certificate is one of the stronger policy instruments that is already in place in some member countries. The certificate simply mandates that a certain percentage of electricity production must come from renewable sources. In countries where hydropower is a viable option, we can be reasonably confident that proposals to augment this power source will be forthcoming. In the case of Sweden, there are already a number of such examples and they clearly illustrate that the resource conflicts involved are no less difficult to resolve today than they were in the past.

Several factors suggest that resource conflicts involving moving water are likely to be even more difficult to resolve in the future. In particular, the demand for recreation is well known to be positively correlated with income. In general, while the income elasticity of demand for environmental improvements is not necessarily greater than one, the bulk of the empirical evidence suggests that it is positive. John Loomis's chapter in this book (Chapter 3) provides an up-to-date and very thorough survey of the total economic values of free-flowing and restored rivers.

Because the benefits of free-flowing and restored rivers fetch no market prices, it is more difficult to estimate such benefits in monetary terms, compared with, let us say, the costs of constructing a dam. Yet, there are a number of issues that have to be resolved in any given application. Chapter 4 by Per-Olov Johansson and Bengt Kriström has a framework for CBA of hydropower conflicts, in particular perturbations of existing water flow regulation. The framework includes pertinent issues such as how to handle carbon permits, electricity certificates, the existing tax system, foreign ownership and a host of other issues that the analyst must confront in any given application. In addition they address the question of how to estimate the monetary benefits of a changed water regulation. Bo Ranneby and Yun Ju (Chapter 5) develop one of the proposed estimation methods (interval valuation questions) in the Johansson and Kriström chapter, hence providing a state-of-the art summary of one statistical approach to benefit estimation.

To some extent these three chapters provide a compact summary of certain technical issues that must be resolved when developing a framework for CBA in the case of hydropower conflicts. This is also true for the chapter by Finn Førsund (Chapter 6) on the effects on equilibrium prices and quantities of opening up electricity trade between countries with different generation technologies. A recent example is provided by the plans of the national grids of Norway and the UK to connect the two countries through an underwater power cable. The reason is that the UK plans a huge expansion of its wind power capacity. Since generation of wind power is very volatile there is a need for balancing power. Norway can supply such back-up power since it has a lot of hydropower that can be adjusted – upwards or downwards – almost instantaneously. In return Norway would receive British electricity on windy days. Similarly, Germany is planning a rapid expansion of its wind power capacity. Improved transmission capacity between the Nordic market – known as Nord Pool – and the German-based European Energy Exchange (EEX) market can provide Germany with relatively cheap Swedish and Norwegian back-up hydropower. This scenario is considered in the Johansson and Kriström chapter (but drawing on the assumption that the price will be set by the larger German market while Førsund considers the more general case in which prices adjust in both countries/markets).

Chapter 7 by Finn Førsund and Lennart Hjalmarsson pursues the analysis of balancing power in the context of the Nordic market, in which substantial investment in wind power is planned. This introduces additional supply-side uncertainty in the system and subsequently a number of pertinent questions regarding the future value of hydropower. A fairly common opinion holds that the increasing demand for balancing services increases the value of hydropower capacity. Because the additional supply of wind power is not demand driven – it is imposed via the certificates – Førsund and Hjalmarsson argue that prices may well drop on the average. If so, the value of hydropower does not necessarily increase as more wind power is added to the system.

The book closes by considering more general issues. The first is related to the present role of CBA in decision-making and the second to what lessons CBA practitioners can learn from the literature on behavioral economics. Thus, John Duffield's chapter (Chapter 8) asks: what impact does a CBA related to hydropower have on decision-making? Duffield summarizes his experience from a number of cases in the Western US and explains why some dams get built, not others, why some dams are being removed and the role economics might play in such decisions. Kerry Smith and Eric Moore then ask in Chapter 9 if and how CBA should be modified in light of the insights offered by behavioral economists. In recent years, economists have published a number of anomalies that all challenge basic assumptions about consumer behavior. Accessible recent accounts include Thaler and Sunstein (2009), probing microeconomic ideas, and Akerlof and Shiller (2009), who explore ideas of behavioral economics from a macro-perspective. CBA as it is currently practiced is based on standard utility theory, in which it is posited that a consumer is rational (in a very specific sense of the word). Seemingly incoherent choices can, in Smith and Moore's view, be due to unobservable constraints within the standard utility paradigm. They go on to test their theory using experimental methods.

What lessons can then be learned from this analysis for CBA? A key issue is usefully summarized in a quote from Beshears et al. (2008, p. 12):

> 'like doctors, governments (and other influential social institutions, like employers) are in a good position to advise autonomous agents without dictating to those agents, . . . Governments could play a constructive advisory role if (1) their advice is only given in circumstances when the many different measures of normative preferences discussed above tend to coincide, and (2) their advice is offered without any obligation to obey (e.g. an opt-out default). By contrast, in cases with ambiguous or contradictory measures of normative preference, we side with Hayek and Friedman – government should withdraw.

If one accepts this view, the role for CBA will be significantly diminished, not the least in cases of resource conflicts involving moving water. There is no opt-out option; either the dam is torn down or constructed and there is no meaningful way in which an individual can 'opt-out'. Smith and Moore are explicit on the implications: 'We have argued that the most carefully reasoned analytical arguments within the behavioral economics literature have nothing to offer for practical benefit-cost analysis.' Smith and Moore's analysis suggests that we currently do not have any viable alternative to approaches that are based on exploring trade-offs in the manner illustrated by the papers in this book.

NOTES

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2. Environmental cost–benefit analysis and water quality management

Nick Hanley

1 INTRODUCTION

Some of the earliest applications of cost–benefit analysis (CBA) in an environmental policy context concerned the management of water resources (Banzhaf, 2009). The desire by policy-makers, politicians, regulators and guardians of the public purse to have some sense of the efficiency with which public money was being spent on water resource management in the US led to a desire to compare the economic benefits and costs of individual projects (such as new dams). This way of thinking about the economic rationality of public spending was later extended to the UK, where the use of CBA in project appraisal in the 1970s was later extended to aspects of environmental policy analysis (DoE, 1991; Turner, 2005). Environmental CBA thinking also found its way into management strategies for public forests, with the non-market benefits of forestry increasingly being incorporated into investment analysis (Willis et al., 2003).

In Europe, the main policy within which CBA is now applied to water issues is the Water Framework Directive. The Water Framework Directive is a unifying measure passed by the European Union to harmonize water resource management, and to achieve a default target of 'Good Ecological Status' (GES) for all surface waters in the European Union (EU). GES is defined with respect to biological, chemical and morphological criteria. River basins are the focus for management actions, and it is in the drafting of 'River Basin Management Plans' that CBA comes into play. National agencies must identify cost-effective programmes of measures for each water body, which achieve the target of GES by 2015. However, they must also consider whether the costs of achieving GES on a particular water body are 'disproportionately costly'. This means a comparison is necessary between the likely benefits and costs of improvements to GES – potentially for every water body in each country! Because of this burden of work, countries have been seeking pro forma for identifying which water bodies are likely candidates for designation as 'disproportionate cost' cases. If the Environment Agency finds that benefits are considerably lower than costs for a particular water body, then the government can ask that either (1) a longer time-scale be allowed for that water body to achieve GES or (2) that a lower target for improvement be set. However, even if benefits exceed costs, a derogation for improvements to GES can still be sought if disproportionate costs are imposed on one particular sector or operator. In other words, distributional criteria are seen as very important.

In the UK, CBA is also used under the 'Periodic Review Process'. Water supply and sewerage services in England and Wales are undertaken by private sector companies. This means that water companies are both sources of water pollution (through their ownership and operation of sewage treatment works), and the beneficiaries of improvements in water quality. As part of the process of reviewing emission consents issued to water companies, the Environment Agency is required to consider the benefits and costs of proposed tightening of pollution regulation. This process, known as 'Periodic Review', is overseen by Ofwat, the Water Services Regulation Authority.

The first programme-wide assessment of proposed improvements occurred under the 'PR99' plan in 1999. The Agency carried out 700 individual multi-criteria assessments to prioritize and rank proposed improvements in water quality (e.g., by upgrading sewage works on a particular river). However, this process was criticized by Ofwat, on the grounds that multi-criteria analysis did not show which schemes would generate benefits in excess of costs (Fisher, 2008). The Agency then developed a *Benefits Assessment Guidance* manual to allow it to measure benefits under the following Periodic Review in 2004. This Guidance recommended extensive use of simple benefits transfer techniques (see section 4 of this chapter). The assessment used these guidelines to generate cost–benefit ratios for 437 individual water quality improvement schemes. This took 19 person-years of work at the Agency. Costs were provided by the water companies (i.e., by the polluters): this led to an apparent over-estimate of around 40 per cent in costs, once these figures had been scrutinized by Ofwat and the Agency (Fisher, 2008). Based on the benefits estimates generated by the Agency, and the (moderated) cost figures provided by industry, schemes were then classified according to cost–benefit ratio (Environment Agency, 2003).

Thus, at both the European and UK levels, environmental CBA is increasingly being used to assess water quality improvements as part of the policy and regulatory process. We now look at problems surrounding the measurement of these benefits and costs.

2. ISSUES WITH MEASURING COSTS

Given that the costs of measures to improve water quality will typically be valued by markets, it might be thought that the measurement of costs would be relatively straightforward, compared with the often non-market nature of benefits. However, the proper assessment of costs is often problematic. Costs depend first on what the determinants of low water quality are for a particular water body, and on what measures are proposed to ameliorate these impacts. Consider a policy target to improve the quality of a river from a currently degraded status to what is defined in the WFD as GES. How would costs be estimated? The first step is to identify the sources of the water quality problem. This might include direct pollution inputs from factories or sewage works, non-point run-off of nutrients and sediment from farm fields, abstraction of water by various users leading to lower dilution of pollutants, and morphological changes that limit fish migration. Table 2.1 shows the distribution of problem sources for water bodies in the UK in terms of the need to achieve GES.

Next, the possible measures that could be taken to achieve water quality targets need to be identified. Even for a particular water body, these could include a very wide range of measures, according to which pressures are targeted. For example, a programme to achieve GES for Loch Leven in Scotland, which currently suffers from blue-green algal blooms due to high nutrient inputs, could include:

- changes in farming practices to reduce phosphate run-off;
- capital investments at sewage works to improve nutrient removal capacity, and:
- reduced abstraction or increased in-flow of water during the summer.

The costs of a particular package of measures will then depend on how cost-effective these measures are. The costs of achieving a water quality target will depend very much on the mix of policy instruments or practical measures taken to achieve it. Moreover, from several perspectives, the distribution of these costs across sectors will be important to judging the acceptability of a programme of measures.

Two further problems may be identified for cost estimation. These are (i) uncertainty and (2) transferability. Uncertainty can be due to uncertainty over the engineering costs of measures, over the response of private agents to incentives such as payments for land management that reduces nutrient run-off, and over the extent of measures needed to reach a target. This latter will be influenced by stochastic determinants of water quality, such as rainfall and summer temperatures. The second concern

Table 2.1 Sources of failure to meet GES for water bodies in the UK

Country	Water Body	Proportion of water bodies affected by pressures						
		Abstraction	Alien species	Diffuse pollution	Flow regulation	Morph. alteration	Point source pollution	Unknown
England	Coastal	1	3	2	1	4	1	–
England	Groundwater	2	1	4	2	1	1	–
England	Lakes	1	1	3	1	3	2	–
England	Rivers	1	2	5	1	3	2	–
England	Transitional	1	2	2	1	5	3	–
Scotland	Coastal	1	1	1	1	1	4	1
Scotland	Groundwater	1	1	3	1	1	2	1
Scotland	Lakes	1	1	1	2	2	1	1
Scotland	Rivers	1	1	2	1	2	1	1
Scotland	Transitional	1	1	2	1	2	3	1
Wales	Coastal	1	3	2	1	4	1	–
Wales	Groundwater	2	1	4	2	1	1	–
Wales	Lakes	1	1	3	1	3	2	–
Wales	Rivers	1	2	5	1	3	2	–
Wales	Transitional	1	2	2	1	5	3	–

Note: 1 = <20% of Water Bodies at Risk from Pressure, 5 = >80%.

Source: Jacobs (2006).

arises when a policy requires the assessment of a great number of potential improvement schemes, as under the WFD. In such cases, methods will be needed to transfer cost estimates across water bodies and across sources. Can we predict which packages of measures will be cost-effective in achieving GES across similar water bodies?

3. MEASURING BENEFITS

In this section, I consider what kinds of benefits we are talking about, what methods have been used to measure them and in what contexts. I conclude by discussing some problems with benefits measurement.

3.1 Types of Benefit

Water quality benefits can arise from the amelioration of many sources of impairment. Benefits arise in terms of improvements to river (loch, lake, stream etc.) appearance/aesthetics and flows; improvements in bankside vegetation; improvements to in-stream ecology, water birds, mammals (e.g., otters) and amphibians; and reductions in bad odours and health risks (e.g., from cyanobacteria: Tyler et al., 2009). These physical changes in water bodies will generate both increased use values – for example, to fishers or kayakers (Johnstone and Markandya, 2006; Hynes et al., 2007) – and increased non-use values, for example because of an improvement in the survival probabilities of rare species. Many of these benefits will not be directly reflected in market prices, so that non-market valuation methods must be used to estimate them.

3.2 Means of Estimating Benefits

Non-market valuation methods can be divided into direct and indirect approaches, according to whether they are used to estimate direct impacts on utility, or indirect impacts via changes in production possibilities. Within the field of direct methods, both stated and revealed preference methods can be used. Stated preference methods include choice experiments and contingent valuation. Revealed preference methods include travel cost models and hedonic pricing (Hanley and Barbier, 2009). All have been used to measure the value of water quality improvements, dating back to early work by Smith and Desvouges (1986), Bockstael et al. (1987) and Mitchell and Carson (1989). A recent overview of methods that can be applied within the context of the WFD is given in Birol et al. (2006).

Stated preference methods can be used to estimate changes in both use

Impact \ Policy Option	Do Nothing	A	B
Number of agricultural jobs lost or gained in the local area	No loss no creation	Loss of five jobs	Creation of two jobs
Visual impact: number of months of low flow conditions in the river	Five months	Two months	Three months
Ecological condition of river	Worsening	Slight improvement	Big improvement
Increase in water rates per year	£0	£2	£2
Please tick the option you prefer			

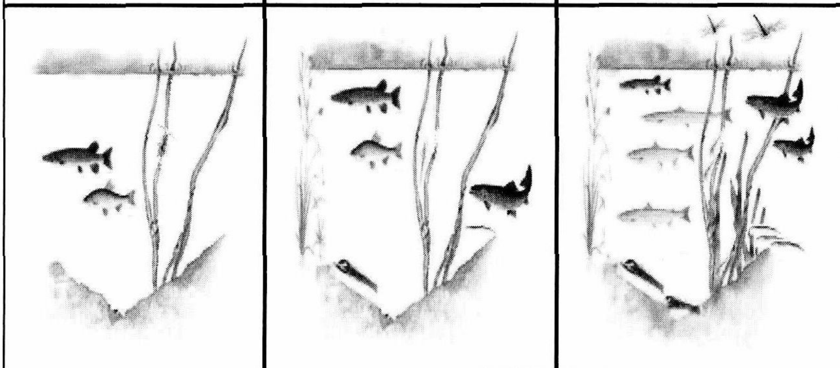
Figure 2.1 Example choice card from Hanley et al. (2006)

and non-use values. Contingent valuation has been very frequently used to value changes in water quality, starting with work in the 1980s (e.g., Mitchell and Carson, 1989). More recent examples include Holmes et al. (2004), who also study the benefits of ecosystem service restoration for the Little Tennessee River. In the UK, contingent valuation has been used to estimate the benefits of improvements to flow conditions in rivers subject to summer low flows (e.g., Hanley et al., 2003).

Choice Experiments have proved increasingly attractive to water quality researchers, since they enable separate values for different attributes of water quality to be assessed (e.g., in-stream ecology, aesthetics, bankside vegetation). Such attribute values are seen as being useful from a management or policy context, since they enable a more detailed picture of the relative benefits of water quality management options to be put together. Examples include Hanley et al. (2006), who looked at the value of improvements to two rivers in Eastern Scotland, and Smyth et al. (2009), who study the benefits of management actions for Lake Champlain on the Canada–US border. Attributes used in this latter study include water clarity, beach closures, the spread of an invasive plant and fish consumption advisories. An example of a choice experiment task, for the Hanley et al. (2006) study, is included as Figure 2.1, whilst Figure 2.2 shows in more detail how the varying levels of an attribute can be portrayed to respondents. Table 2.2 shows typical results in terms of marginal values for improvements in river attributes, again from the Hanley et al. (2006) study.

Revealed preference methods measure changes in use values. The 'travel cost' approach can be applied to water quality changes, both through an analysis of the effects of water quality on total recreational trips (count models) and on recreational site choice. Combining these approaches allows welfare measures to be calculated for specific changes in water

River life: fish, insects, plants



Poor	Moderate	Good
Low number and variety of fish, insects and plants: Coarse fish Tolerant species (water hog louse and weed) common	*Reduced* number and variety of fish, insects and plants: Coarse fish present, salmon and trout at risk Sensitive species (lamprey, crayfish, mayflies, native plants) occasionally present	*High* number and variety of fish, insects and plants: Salmon, trout as well as coarse Sensitive species (lamprey, crayfish, mayflies, native plants) present

Source: Author's own work, unpublished.

Figure 2.2 Explaining varying levels of an attribute

quality parameters. An example is the work reported in Johnstone and Markandya (2006), who relate anglers' choice of total fishing trips in a year and where those trips were made to water quality parameters such as the number of taxa within the river, organic pollution levels, habitat quality and the number of fish species. They found that both the number of angling trips and the distribution of these trips across sites were significantly related to most of the water quality measures used, although parameter signs are sometimes unexpected. They then use the combined participation and site choice model to measure welfare benefits (changes in consumers' surplus per trip) for a 10 per cent improvement in water quality measures. Table 2.3 shows some of their results. However, one problem with applying travel cost models to valuing water quality improvements is that the researcher often finds a high degree of multicollinearity between measures of water

Table 2.2 Example outputs from a choice experiment for two rivers in Scotland

	Motray River	Brothock River	Pooled Data
Independent preferences			
Local farm jobs	3.52	3.63	3.65
	(2.38; 4.66)	(2.41; 4.98)	(2.81; 4.48)
River flow conditions, per month improvement	3.87	2.70	3.00
	(2.52; 5.07)	(0.90; 4.21)	(1.74; 4.25)
Ecology slight improvement	8.97	10.53	9.45
	(5.41; 12.38)	(4.57; 17.19)	(6.25; 12.93)
Ecology big improvement	24.03	28.26	25.91
	(18.53; 31.08)	(19.65; 40.57)	(21.10; 31.74)
Correlated preferences			
Jobs	2.67	3.69	3.40
	(1.90; 3.42)	(2.64; 5.04)	(2.67; 4.13)
River flow	3.74	3.20	3.50
	(1.57; 5.55)	(1.28; 5.30)	(1.92 ; 4.72)
Ecology slight improvement	10.88	17.53	10.11
	(2.07; 19.29)	(1.88; 36.96)	(5.76; 14.39)
Ecology big improvement	23.67	36.13	25.65
	(14.99; 31.47)	(21.89; 55.71)	(21.04; 31.07)

Note: Units are £ sterling per household per year. Bid vehicle: local water taxes. Figures in parentheses are 95% confidence intervals.

Source: Hanley et al. (2006).

quality, making it difficult to identify individual effects. This was certainly a problem in the Johnstone and Markandya study noted above.

'Hedonic price' (HP) approaches relate variations in water quality to variations in house prices, as a way of measuring aspects of the benefits of improvements in quality. Leggett and Bockstael (2000) look at the effects of varying faecal coliform levels in coastal waters on property prices in Anne Arundel County, Maryland. The irregularity of this coastline means that water quality levels vary substantially within the housing market. The analysis was based on house sales data of waterfront properties from 1993 to 1997. The authors argue that faecal coliform counts are a good measure of water quality to use in HP studies, since it is something people are likely to care and know about, especially if they engage in water-based recreation such as swimming and boating, due to health risks. High levels

Table 2.3 Economic benefits to anglers in England from water quality improvements (change in consumers surplus per trip in £ sterling for a 10% improvement in statistically significant river attributes)

	Upland Rivers	Lowland Rivers	Chalk Rivers
Number of fish species		2.49	
BOD	–0.43		
Ammonia		–0.13	
DO	2.09		0.29
Flow	1.97	3.70	0.15

Note: Mean consumers surplus across all three rivers under current conditions was £25/trip. Blank cells imply that the water quality parameter was not significant in the choice model at 95%. BOD = biological oxygen demand; DO = dissolved oxygen.

Source: Extracted from Johnstone and Markandya (2006).

of faecal matter also make the water smell and look bad, and pose health risks to users. Leggett and Bockstael explore different functional forms for the hedonic price equation, including linear, double-log, semi-log and inverse semi-log. With the exception of the linear form, the measure of faecal coliform concentration is highly significant and negative as an explanatory variable. Welfare changes from reducing faecal coliform pollution are then estimated. Taking one particularly polluted stretch of coastline, where coliform levels currently range from 135 to 240 per 100 ml water, the authors find that property values would rise by $230 000 if levels were cut to 100 coliforms per 100 ml, a 2 per cent gain in value (based on the inverse semi-log hedonic price equation).

Stated and preference approaches can also be combined. For example, Hanley et al. (2003) use a contingent behaviour model, which amalgamates travel cost data on actual trips as a function of perceived water quality with stated changes in intended trips should pollution be reduced, to value the benefits of improving coastal water quality to beach users in Scotland. Travel cost models can also be combined with hedonic price approaches: see Phaneuf et al. (2008).

3.3 Contexts for Benefits Assessment

We can think about the context within which benefits assessments are carried out either in terms of the main determinant of poor water quality/