Cost–Benefit Analysis and Water Resources Management

Edited By **Roy Brouwer and David Pearce**

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We are grateful to the 32 contributing authors to this book for sharing their experiences with us in the field of water and economics in different countries across Europe and North America. This book would not have seen the light without their kind collaboration and valuable inputs. Over the years we have conducted many cost–benefit appraisals ourselves and have commented on many others. We have learned a great deal from this process and we thank our colleagues in University College London, the University of East Anglia, the Institute for Environmental Studies (IVM), Vrije Universiteit Amsterdam, RIZA and various consulting companies for the insights they have provided. We hope this rich experience shows up in the chapters in this volume so that others can benefit in the way we have done.

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List of abbreviations

AAD	annual average damage
AFC	annual financial cost
AOC	Area of Concern
APRONA	Association for the Protection of the Alsatian Aquifer
ARIENA	Association for Environment and Nature in Alsace
B/C	benefit–cost
BAT	best available technology
BATEA	best available technology economically achievable
BAU	business as usual
BCA	benefit–cost analysis
BCT	best conventional technology
BMP	best management practice
BOD	biological oxygen demand
BPJ	best professional judgement
BPT	best practicable technology
BRGM	Bureau de Recherches Géologiques et Minières (French Geological Survey)
BWQ	bathing water quality
CADF	Federal District Water Commission
CAFO	concentrated animal feeding operation
CAP	Common Agricultural Policy
CBA	cost–benefit analysis/benefit–cost analysis
CCA	cost compliance assessment
CEA	cost-effectiveness analysis
Cemagref	Institut de Recherche pour l'Ingénierie de l'Agriculture et de l'Environnement
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
CFR	Code of Federal Regulations
CIIM	Cyprus International Institute of Management
CAN	National Water Commission
COI	cost of illness
CPB	Centraal Planbureau (Central Planning Agency)
CSERGE	Centre for Social and Economic Research on the Global Environment

CSIC	Spanish Council for Scientific Research
CSO	Storm Combined Overflow
CV	contingent valuation
CWA	Clean Water Act
DALYs	Disability Adjusted Life Years
DECRG	Development Economic Research Group
DEFRA	Department for the Environment, Food and Rural Affairs
DGCOH	Directorate General for Hydraulic Construction and Operation
EAR	Environmental Assessment and Review
EBE	Syndicat des Eaux de Ensisheim-Bollwiller et Environs
EC	European Commission
ECOWET	Ecological-Economic Analysis of Wetlands: Functions, Values and Dynamics
EEC	European Economic Community
EIA	environmental impact assessment
ELG	Effluent Limitations Guidelines
ENVECO	Environmental Economics Committee of DG XII
EO	executive order
EPA	Environment Protection Agency
ERDF	European Regional Development Fund
EU	European Union
EWW	English-Wabigoon-Winnipeg
EYATH	Water and Sewerage Corporation of Thessaloniki
FES	Family Expenditure Survey
FMP	fertilizers, manure and pesticides
FRN	Swedish Council for Planning and Coordination of Research
GDP	gross domestic product
GIS	Geographical Information System
GRAVAMEX	Regional Office for the Valley of Mexico
GW·MATE	Groundwater Management Advisory Team
ICR	Information Collection Request
IED	income elasticity of demand
IEG	Institute of Economics and Geography
IJC	International Joint Commission
INFORUM	macroeconomic forecasting group at the University of Maryland
IUCN	World Conservation Union
IVM	Institute for Environmental Studies
LA	local authority

LRD	Lower River Delta
LRMC	long-run marginal cost
LRMSC	long-run marginal social cost
MAFF	Ministry of Agriculture, Food and Fisheries
MAMC	Metropolitan Area of Mexico City
MAP	Mediterranean Action Plan
MARE	Marine Research on Eutrophication
MB	marginal benefit
MDPA	Potash Mining Company of Alsace
MISA	Municipal and Industrial Strategy for Abatement
MISTRA	Foundation for Strategic Environmental Research
MS	member states
MV	marginal value
MVP	marginal value product
NAO	National Audit Office
NGO	non-governmental organization
NNI	net national income
NOAA	National Oceanic and Atmospheric Administration
NPC	net present cost
NPDES	National Pollutant Discharge Elimination System
NPV	net present value
NRCS	National Resources Conservation Service
NSF	National Sanitation Foundation
NWPCAM	National Water Pollution Control Assessment Model
O&M	operation and maintenance
OECD	Organisation for Economic Co-operation and Development
OGWDW	Office of Ground Water and Drinking Water
OMB	Office of Management and Budget
OST	Office of Science and Technology
OW	Office of Water
OWM	Office of Wastewater Management
OWOW	Office of Wetlands, Oceans and Watersheds
PAH	polycyclic aromatic hydrocarbons
POTW	publicly owned treatment work
PPI	Potential Pareto Improvement
PT	public trust
PVTC	present value of the total cost
QUAIDS	Quadratic Almost Ideal Demand System
RAP	Remedial Action Plan
RBA	River Basin Authorities
RDD	random digital dialling

RIZA	Institute for Integrated Inland Water Management and Waste Water Treatment
SASE	Standard Annual Shortage Event
SCP	Southern Conveyor Project
SDAGE	Water Management Master Plan
SDWA	Safe Drinking Water Act
SOS	standards of service
STORET	Storage and Retrieval System
SCZM	Sustainable Coastal Zone Management
TAEC	total annual economic cost
TMDL	total maximum daily load
TSS	total suspended solids
UNEP	United Nations Environment Programme
UNESCO	United Nations Educational, Scientific, and Cultural Organization
USGS	United States Geological Survey
UV	ultraviolet
VAT	value added tax
VOC	volatile organic contaminants
VSL	value of statistical life
WFD	Water Framework Directive
WG	working group
WRM	Water Resource Management
WTAC	willingness to accept compensation
WTO	World Trade Organization
WTP	willingness to pay
WWTP	waste water treatment plant

1. Introduction

R. Brouwer and D.W. Pearce

1. BASIC ECONOMIC PRINCIPLES

Cost–benefit analysis (CBA) is around 70 years old if we date its first practical application to water resource developments in the USA in the 1930s. The theory of CBA is much older and its origins can be more precisely set in the 1840s with the writings of the French engineer and economist Jules Dupuit (Pearce, 2002). Dupuit was concerned with the issue of how to make public choices about investments that had no necessary commercial returns, such as roads and bridges. He established the notion of what today we call consumer's surplus, the consumer's net benefit from consuming something and measured by the excess of willingness to pay over the cost of acquiring the good. Along with any producers' surplus – the return received by the producer over and above the minimum he/she would accept to supply the good – it is the change in consumers' surplus that measures the benefit of providing more of a good. Moreover, these measures of surplus are general and apply regardless of whether the good in question is supplied through a market or if it is a public good, generally supplied by governments. That CBA was applied early on to water resources, albeit in very primitive fashion by today's standards, is no accident. Water has competing uses and for some of those uses it acts very much like a private good: A's consumption is at the expense of B's consumption. In other respects it is a quasi-public good: A's enjoyment of a water-based amenity is not affected by B's enjoyment of the same amenity (technically, a 'club good' since those enjoyments will tend to be diminished as more people seek out the amenity). It follows that, from an economic efficiency standpoint, water should be allocated to those users with the highest willingness to pay for it.

Figure 1.1 shows the supply (LRMC = long-run marginal cost) of, and demand (D = marginal benefit, MB, or value, MV) for water. For completeness, a backstop price (for example, desalination) is shown. It can then be seen that as demand grows over time (D shifts to the right) so the backstop technology could come into play. Because of its critical role in agriculture and its essential role in human consumption, water tends to be subsidized in many countries. Indeed, subsidies to water are often seen as some kind of 'right', with charging for water being seen as unethical. Yet water costs resources to supply, so water is an economic good: the resources used to supply it could have been used to supply something else. Figure 1.1 shows the effect of a subsidy, in this case not a subsidy that covers the entire cost of supply but part of it. The effect of the subsidy is to lower the ruling market price from P_M to P_S (the subsidy shifts the LRMC curve downwards to the right). (This is not shown simply to avoid too many lines in the diagram.) It can be seen that consumers gain because water is now cheaper. Their actual gain (of consumers' surplus) is given by area A + B + C. Producers also gain (producers' surplus) equal to the area above the supply curve between the effective price to them (P_S + the subsidy) and the original price, that is, D + E. Thus, together, consumers plus producers gain A + B + C + D + E. But subsidies are not 'free', they are paid for by taxpayers. The total subsidy cost is given by the new level of supply (Q_S) multiplied by the unit subsidy, that is, A + B + C + D + E + F. The difference between this cost and the consumers' plus producers' surplus gains define the net true cost of the subsidy. It is equal to area F. This is the 'deadweight' cost of a subsidy. Hence, regardless of one's views about water as an 'essential' good, subsidizing it for that reason (or any other) produces a net loss of social well-being for society as a whole.

Figure 1.1 illustrates a basic principle for the economics of water. Water should be priced at its market-clearing price, that is, where price equals the long-run marginal cost of supply.

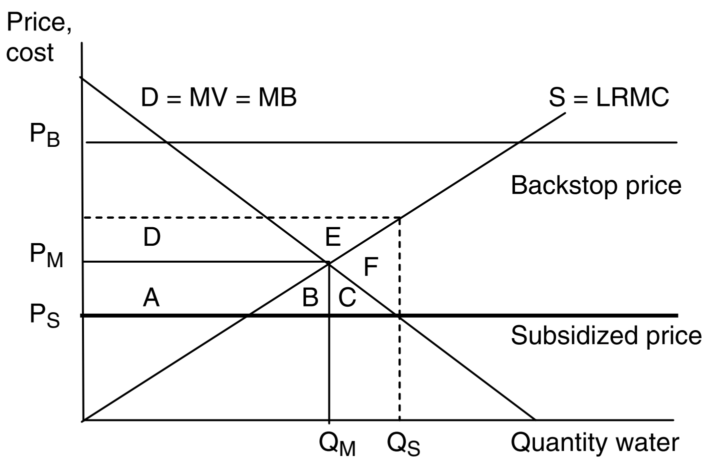

Figure 1.1 Basic case of supply and demand for water

Table 1.1 Subsidies to water ($\$10^9$ 1990s, per annum)

	Irrigation	Public supply
Africa	5.1	1.7
Latin America	3.1	5.2
SE Asia	—	8.6
W Pacific	—	10.9
Asia	11.4	—
E Mediterranean	—	2.2
Total: non-OECD	19.9	28.5
Total: OECD	15.0	

Source: Xie (1996); van Beers and de Moor (2001).

As noted above, in practice, water is priced well below the marginal cost of supply in a great many countries. Table 1.1 shows that water subsidies are widespread, especially, but far from exclusively, in developing countries. Note that the combined cost of the subsidies is over \$60 billion per annum, more than the amount given by rich countries to poor countries in official foreign aid. Yet many of the discussions about the world's 'water crisis' focus on the need to invest in new supplies to meet ever-increasing demand, without having regard to the fact that *existing* supplies are inefficiently allocated through improper pricing.

Figure 1.1 shows a demand curve for water. In practice, the demand for water is the sum of several different demands, the largest in many countries being the demand for irrigation water. For an efficient allocation of water, the marginal values of water should be the same and equal to the marginal cost of supply. The reasoning is simple, if marginal values are not the same, it will be possible to reallocate a given water supply at the margin away from those whose valuation is low to those whose valuation is higher, thus increasing overall social value. Suppose that the willingness to pay by A for an extra cubic metre of water is \$1 and that of B is \$0.3. The social value of water is increased if the water is allocated to A rather than B. Since the marginal valuations of A and B will vary with the amounts they consume, the social value of water is maximized when the two *marginal* valuations are equal. This is the *principle of equi-marginal valuation* and it is of vital importance to the efficient allocation of water resources. One immediate implication is that the different values for water should be investigated and measured. If irrigation water is valued at the margin less than, say, industrial water, water for domestic consumption, or even water in an environmental use (a wetland, say) then the equi-marginal valuation principle requires that we reallocate water away from irrigation and towards these other uses. How should such allocations be brought about? In principle the most effective way of ensuring at least an approximate compliance with the equi-marginal valuation principle is to establish a system of water trading or water rights trading. Then those users with the highest willingness to pay can bid more than the willingness to pay of those with the initial rights to the water.

These principles of efficient water pricing and allocation are explored in more detail in the chapters by Robert Young (Chapter 2) and John Briscoe (Chapter 3). They are of the utmost importance for anyone engaged in water use planning, regulation and investment. Detailed case studies of water supply issues in water-scarce countries are provided by Ben Groom and his colleagues for Cyprus (Chapter 14), by Josefina Maestu and colleagues for Spain (where major transfers of water from one region to another take place) (Chapter 15), and by Gloria Soto Montes de Oca and Ian Bateman for Mexico City (Chapter 16). Needless to say, even where economic principles are brought to bear on water issues, the outcome is unlikely to bear close resemblance to the textbook 'ideal' outlined above. These authors show how political factors intervene to determine what might be called a 'politico-economic' equilibrium in which competing interests for water influence the political process and vice versa and hence the actual outcome. That the final outcome may not look like the textbook solution should not be regarded as a failure of theory: it remains important that economists and others continue to prescribe on the basis of the received theory.

2. INVESTING IN WATER QUALITY

Pricing and efficient allocation of water to different users deals with the issue of water quantity. But the quest for ever-improving water quality is also of vital concern. Investments in improvements in drinking water define the very first environmental policies, although they were then seen as basic public health measures. The same investments are needed today in developing countries and in many middle-income countries. The World Bank (Lvovsky, 2000) estimates that just under 5 per cent of all the Disability Adjusted Life Years (DALYs: a measure of lives lost and quality of life lost) in developed countries arise from environmental factors, but the fraction is 18 per cent in developing countries. In turn, over a third of environmental-cause DALYs in the developing world are caused by poor water quality and perhaps a fifth in the rich world. Simply put, water quality matters crucially for human health in the poor world, and still matters in the richer world.

In the richer countries quality improvements have gone well beyond drinking water standards and relate to wider goals of ecosystem services – recreation, fisheries, biodiversity and general amenity. Cost–benefit analysis is just as applicable to water quality as it is to water supply. But now the principle of pricing at long-run marginal cost (LRMC) gives way to a more comprehensive rule: pricing at long-run marginal *social* cost (LRMSC). The difference between private and social cost is the externalities associated with water use. Excessive use of irrigation water can result in waterlogging of soils and compaction when they dry out. Excessive extraction of water can produce low flow situations, which lower water quality levels and then have serious ecological consequences. Diffuse pollution, mainly, but not exclusively, from agriculture, produces surface and groundwater contamination, and so on. Whereas LRMC involves calculations that rely largely upon fairly readily available engineering data, measuring LRMSC involves valuing the externalities in money terms. In keeping with the basic principles outlined above, it is individuals' willingness to pay for quality improvements that measure the benefits. Enormous efforts have gone into valuing individuals' preferences for changes in water quality in recent years. Those efforts are illustrated in this volume by the majority of the chapters: Denmark and Sweden (Chapters 6 and 7), Greece (Chapter 8), Canada and the USA (Chapters 9 and 10), The Netherlands (Chapter 11), the UK (Chapter 12) and France (Chapter 13). The type of water resource the quality change of which is being valued varies from groundwater (France), coastal and other bathing waters (the UK and The Netherlands), major lake systems (Canada) and rivers systems in general (Greece, the USA, Denmark and Sweden).

What these chapters tell us is that economic valuation techniques have advanced considerably, particularly with the use of 'stated preference' techniques. These techniques involve questionnaires, which either elicit an individual's willingness to pay directly (contingent valuation) or indirectly by presenting the respondent with choices between goods with the same characteristics but with the level of those characteristics varied (choice modelling, although the terminology varies). In choice modelling one of the characteristics is a price, and this enables the analyst to infer willingness to pay without asking directly what it is. Other valuation techniques remain relevant – hedonic property pricing measures water quality benefits by looking at the influence of water quality on property prices; travel cost approaches measure recreational benefits by inferring willingness to pay for an improved quality site from travel expenditures to the water site. No one would pretend that all the benefits of water quality improvements are currently being captured by these techniques. Notable difficulties, still being tackled in innovative studies, arise with the valuation of water quality effects on biodiversity, for example. The chapter by Charles Griffiths and Will Wheeler (Chapter 10) also shows how regulators add to the goals that have to be served by policy appraisals – notably, but not exclusively, distributional concerns. Ultimately, CBA ends up being part of a wider process rather than the sole means of making decisions. Nonetheless, the chapters in this volume show what advances have been made and how better decisions can be made with valuation techniques.

3. FLOOD CONTROL

Two other chapters in this volume, by David Pearce and Robin Smale for the UK (Chapter 4) and by Roy Brouwer and Jarl Kind for The Netherlands (Chapter 5) deal with flood control issues. Flood control was actually the focus of the very earliest cost–benefit studies of water resources. Today, fairly sophisticated CBA procedures are used on a routine basis in both countries. The focus is very much on probabilistic analysis of floods, the costs of their control and the damages avoided through that control. In turn, damages range across fairly easy to estimate effects such as property damage, through to impacts on wetlands, health risks and so on. In both countries only limited attention has been paid to the 'fear of floods', that is, the welfare losses arising from anxiety about future floods, but this can be expected to change as stated preference techniques are applied with more rigour in the future. Apart from describing how CBA has become central to flood control, two significant messages emerge from these chapters. First, Pearce and Smale point to an initially surprising result that benefit–cost ratios rise through time, contrary to initial expectations that the 'best' schemes will be implemented first. The substantial rise in the real values of property in the UK perhaps explain this. Second, Brouwer and Kind argue that, however imperfect, CBA in The Netherlands context has both stimulated the science of flood prediction and has provided an organizing framework for stakeholders to discuss the various aspects of costs and benefits. Cost–benefit analysts have long argued that one major advantage of CBA is 'cost–benefit thinking' – the organization and structuring of the arguments that supports social decision-making processes, not replaces them.

4. COST–BENEFIT ANALYSIS AND PUBLIC TRUST

We conclude this introductory chapter by drawing attention to an issue not addressed in the remaining chapters in the volume. As noted above, CBA is rarely if ever the sole procedure used for making public investment and policy decisions. Views differ on how desirable this state of affairs is, but political reality dictates that many other interests will be embedded in actual decisions. In recent years, however, the legal notion of 'public trust' has entered into decision-making, initially in the USA but now in European Union policy as well, although the terminology is rarely used.

The public trust doctrine arose in the context of environmental damage liability, and implies that any damage to natural resources and the environment must be negated, that is, some 'pre-damage' situation must be reinstated. Liability occurs in the context of some act that is not legally sanctioned. An oil spill or a toxic waste incident would be examples. Two versions of public trust appear to exist, both relating to liability for the recovery of costs relating to environmental damage to a specified resource. The first requires that the specific natural environment must be restored to its 'pre-damage' situation. The second requires that, if the specific asset cannot be restored, another 'like' asset must be created so as to compensate for the loss of the first asset. The doctrine requires that those who act as trustees, that is, management agencies, can use any money recovered from actions against liable parties only for enhancing or creating natural resources. Monetary compensation to damaged individuals, actual or hypothetical, would then have no role to play because, of itself, compensation does not restore the 'status quo'. As Jones states: 'public trustees do not have the authority to make individuals whole by providing such recoveries [money recovered from liable parties] directly to individuals; rather, trustees are allowed to spend their recoveries only on enhancing or creating natural resources' (Jones, 1996, p. 6).

Note how this contrasts with the economic efficiency view: to the economist, the status quo relates to the well-being of the individual. If, in a post-damage situation, an individual is compensated so as to be as well off ('made whole' in Jones's language) – in his or her own judgement – as they were in the pre-damage situation, then compensation is efficient and just. So long as the individual regards the compensation as a substitute for the damage done to the environment, it is not necessary for the damage itself to be 'undone'. Nor is there any need for the lost asset to be restored, either in itself, or through a like asset. For example, monetary compensation would be sufficient in the economic approach so long as the compensation conferred a level of utility or well-being equal to that, which existed in the pre-damage situation. Also, in CBA, this compensation need not be actual compensation, a point we return to. The public trust doctrine proceeds quite differently. It does not require that the status quo be measured in terms of individuals' well-being, but in terms of the state of the natural environment and regardless of ascertained public preferences. The implication for the economic valuation of damage is usually that any damages are measured by the costs of restoration, not by any attempt to elicit the willingness to pay of individuals for restoration of the pre-damage environment, or for the compensating asset.

According to Anderson (1993), the origins of the public trust doctrine in the USA are rooted in early nineteenth century state law and in common law. Under public trust, a nation's natural resources are held in trust for all citizens, now and in the future. In the USA, the courts steadily expanded the use of this doctrine, making its scope apply to wider and wider definitions of natural resources. Combined with *parens patriae* – the role of the state as guardian of persons under legal disability – public trust gives the state a right to protect the environment on behalf of its citizens. This right exists independently of ownership of the resource and derives from the state's duty to protect its citizens. As Kopp and Smith put it:

> Damage awards for injuries to natural resources are intended to maintain a portfolio of natural assets that have been identified as being held in public trust . . . Because this compensation is to the public as a whole, the payment is made to a designated trustee and the compensation takes the form of in-kind services . . . (Kopp and Smith, 1993, p. 2).

The combined doctrines were used to sue polluters in the late 1960s in the USA and the language of public trust began to enter environmental suits in the 1970s.

Public trust (PT) assigns a right to citizens to some predetermined state of the environment. In the liability context, this is the pre-damage state. But it is not difficult to extend the notion to non-liability contexts. All that needs to be done is to assign citizens a right to some other state of the environment which could be the current state, some state that existed at some time in the past, or some future state that reflects some chosen standard of quality. By assigning a right, public trust effectively downplays the importance of cost as a factor in determining the quality benchmark. It may not downplay it totally, but the US experience shows that occasionally it does make cost totally irrelevant, at least as far as the law is concerned. While public trust assigns rights to people, with the state acting as trustee, it does not seek to elicit people's preferences for restoration of the pre-damage environment. It assumes *on their behalf* that compensation is fulfilled by the restoration of the environment. Finally, if this assumption holds, then the losers have *actually* been compensated. In the CBA approach, no such actual compensation needs to take place. It is only necessary that we are satisfied that *if* they were compensated they would be 'made whole' in terms of their levels of well-being. So the central points of comparison between public trust and CBA are:

- public trust assigns a right to some defined state of the environment which, at the very least, is no worse than a pre-damage state;
- since rights need not be informed by preference elicitation – it is implicitly *assumed* that preferences coincide with the ascription of rights – compensation in the public trust approach is both full and actual;
- CBA makes no assumption about rights and preferences coinciding, and proceeds instead by seeking the hypothetical compensation that losers would require through the process of preference revelation;
- as far as compensation is concerned, the public trust and CBA approaches coincide *as far as benefits are concerned* if and only if 'making the environment whole' is the same as 'making people whole';
- CBA will in any event diverge from the public trust approach because it will compare the (hypothetical) compensation with the costs of restoring the pre-damage situation. It does this because it makes no prior assumption about the assignment of exclusive property rights. Since public trust operates with some notion of 'rights' to the benefits, there appears to be no case for comparing benefits with costs.

Public trust thinking has affected several regulatory developments in the USA. The US Supreme Court has, for example, ruled that the US Environmental Protection Agency is not obliged to consider the costs of regulations because Congress failed to mention cost when promulgating the Act. The curious logic is that cost should be important so that, by not mentioning it, Congress must have regarded it as unimportant. The Supreme Court's judgement was described as 'a plunge into the irrational' (Ross, 2001, p. 13) and as highlighting 'the intellectual bankruptcy of current US environmental policy' (Lutter, 2001, p. 1). While not explicitly traceable to the public trust doctrine, the similarities between public trust and the 'no cost' philosophy is evident.

Several procedures can be used to detect the growing influence of public trust in European policy. First, since EU Treaties require that some comparison of costs and benefits be made for new regulations (Article 130r of the Treaty on European Union 1992), the failure to attach a cost–benefit analysis to Directive proposals would provide fairly strong prima facie evidence that economic efficiency is being ignored. Second, if cost does not receive explicit mention as a balancing factor within the Directive itself, then, again we have evidence that the public trust philosophy is securing the upper hand. A weaker form of this test will be whether or not the Directive contains some reference to a notion of 'excessive' cost, that is, member states can seek derogations from the Directive if the costs of meeting the Directive's goals are, in some sense, excessive or disproportionate when compared to the goals. If an excessive cost clause exists, then only a weak form of public trust is being implemented. Mention of excessive cost will not, however, be sufficient to prove that economic efficiency is being adopted. For that, the legislation would have to be clear that 'excessive' means that costs exceed benefits and none of the Directives has such explicit language.

Table 1.2 summarizes the findings of Pearce (2004) on the use of CBA to evaluate Directives, and likely costs and benefits of the Directives for the UK (where regulatory impact assessment in some form or other is required for new legislation), supplemented with findings for The Netherlands. The suggestion is that, the closer one looks at these Directives, the less regard there has been to economic efficiency, with notions of 'excessive cost' or 'disproportionate cost' only appearing under pressure from member states and in turn reflecting concerns about competitiveness rather than welfare gains and losses directly. The question remains as to how, collectively, member states are agreeing to Directives that impose net social costs on them. Some of this disregard for cost has to be founded in the broader belief that, in the context of the environment at least, some form of the public trust doctrine has taken hold in Europe. It has not taken hold as formally as in the USA, although some of the court rulings on the Habitats Directive, for example, certainly make it look as if the courts are backing a 'no cost' doctrine. However, the concession in several of the Directives to notions of 'excessive cost' is a mitigation of the public trust doctrine.

In the public trust doctrine there is an implicit ascription of property rights for the citizens of the European Union. This is a right to an improved state of the environment or, at the very least, to the status quo, for example as embodied in the 'no deterioration' rule of the Water Framework Directive. At one level, the idea that environmental legislation should do anything other than improve the environment is obviously absurd. That is not the argument here. The issue is one of conflicting rights. Cost–benefit analysis ascribes rights to those who have to pay the costs of environmental improvement as well as to those who benefit. Environmental legislation that places greater weight on a unit of benefit than on a unit of cost has done two things: (a) it has assigned asymmetric property rights, and (b) it has overlooked the fact that net social cost involves the sacrifice of something else, maybe roads and railways, maybe health services and education. If nothing else, cost–benefit analysis is there to remind us of this important fact.

Table 1.2 The role of costs and benefits in EU environmental Directives in the UK and The Netherlands

Directive	CBA produced by the EC?	Do other CBAs exist?	Is a CBA test passed?	NSB to UK (UK£)	NSB to NL (€)
Habitats	No	No	Unknown	Unknown	Unknown
Packaging Waste:					
(a) 1994	No	No	No		Unknown
(b) Revisions	Yes	No	No	−212 million	Unknown
Landfill	No	No	No	−19/ton waste	Unknown
Nitrate	No	Partial	Unknown	Unknown	Unknown
Urban Waste Water	No	Partial	Unknown	Unknown	Unknown
Bathing Water					
1976	No	Yes	Yes	3.5 billion	Unknown
1994	No	Yes	Yes	6.5 billion	Unknown
2000	No	Yes	No	−2.6 to −4.0 billion	Unknown
2002	No	No	Yes	9.3–10.6 billion	2.3 billion
Water Framework	No	Partial	No	−807 million to −2.1 billion	Unknown

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2. Economic criteria for water allocation and valuation

R.A. Young

1. INTRODUCTION

The water resource presents an unusually wide variety of public management issues of interest to economists. In its varied forms, water supplies important benefits to humankind, both commodity benefits (to households, industries and farms) and public environmental values, including recreation, fish and wildlife habitat and a medium for carrying material residuals (pollution) from human production and consumption activities. Moreover, as a resource whose supply is determined by natural forces, too much or too little water creates other public management problems (Young, 1996a). With growth in population and income, serious conflicts over allocations of water are found throughout the world, and in many areas are rapidly becoming worse (Gleick, 1998). Economic evaluation can play a role in public assessments of proposals for addressing water management problems.

Resources have economic value or yield benefits whenever users would willingly pay a price for them rather than do without, that is, whenever resources are scarce.[1] Under certain conditions, market operation results in a set of values (prices) that serve to allocate resources and commodities in a manner consistent with the objectives of producers and consumers. In many parts of the world, the services provided by water have been plentiful enough that the resource could be regarded as a practically free good and, until recently, institutional arrangements for managing water scarcity in such locations have not been of serious concern.

When markets are absent or do not operate effectively (as are typical conditions in the case of water), prices as a basis for allocating resources are biased or non-existent. In such cases, economic evaluations of resource allocation decisions must be based on some non-market methods of estimating resource value. Resource value is measured in the context of a specific objective or set of objectives. The value of the resource reflects its contribution to the objective(s). In the field of water resources, governments have identified several objectives that may be relevant: enhancing economic efficiency (called national economic development in the federal planning literature in the US); enhancing regional economic development; enhancing environmental quality; and enhancing social well-being (US Water Resources Council, 1983; OECD, 1985). This chapter focuses on measuring resource values in the context of the economic efficiency objective.

Estimates of the economic benefits relating to water management are useful for several specific types of allocation issues. Perhaps the most familiar is the contribution to appraising investments on structural approaches to water management. Nations continue to make investments in water resources one of the most important components of public infrastructure budgets. Water-related investments – in irrigation, hydropower, urban and rural water supply, flood control and sanitation – have been designed to contribute to economic development and public welfare. Although most such investments may have been subjected to some sort of economic evaluation to assure that they would represent an economical use of scarce water and capital, many earlier water resource investments have yielded less return than anticipated and have proven to have been based upon overoptimistic pre-project economic evaluations. Among the projects yielding disappointing results, many, it is clear, were evaluated with less than rigorous procedures. Economic evaluation is important because it aids in determining if people want proposed projects and estimating the degree to which they are willing to pay for benefits. In the prevalent situation of constrained public budgets, conceptually correct and empirically valid estimates of the economic contribution of water-related investments are essential for making economically sound public expenditure decisions.

Another class of decisions in which economic values of water are useful is that evaluating non-structural or policy options. For example, as demands for fresh water grow against the finite world supply, estimates of the economic value of water are useful in the context of optimal allocation of water between and among water-using purposes and sectors. Water users will not be able to obtain all of the water they might possibly use. Sharing of the limited supply is a central issue of water management. In the context of water management, decision-makers in many nations face many other questions that invite economic evaluation, such as: how much water should be allocated to the agricultural sector for irrigated food production versus how much to cities with their household and industrial needs? How are needs to develop added food supplies to be balanced with the wish to preserve watercourses or wetlands for fish and wildlife habitat? How are wants for hydroelectric power generation and other in-stream uses to be balanced against demands for water for cities and farms? Each of the above cases are examples of the issue of optimal intersectoral allocation.

Several other non-structural water policy problems for which water valuations are useful come to mind. These include: how much groundwater should be pumped now and how much should be saved for future needs? How much groundwater versus how much surface water should be withdrawn to meet current water demands? And, how much treatment to apply to wastes discharged into watercourses? Considering another dimension – that related to finance and cost recovery – how much can beneficiaries afford to pay for water supplies? For each of these issues, estimates of the net economic contribution of the water resource are important for water policy decisions.

A common theme runs through the above survey of water allocation issues. Each of these are water management problems which involve choices as to how water should be combined with other resources so as to obtain the most public return from scarce resources. Included among the issues are the classic microeconomic resource allocation issues (Varian, 1993): how much of each input to use in production; how to proportion inputs in a production process; which products and how much of each to produce with scarce inputs; which technology to employ; and how to allocate use of resources and consumption of goods and services between the present and future uses. Therefore, these issues can be usefully cast as resource allocation problems and can be best understood within an economic framework.

A truism of applied policy analysis is that 'decisions imply valuation'. Rational decision-making presupposes the forecasting of consequences, and assignment of weights (values) to these consequences. Because of the limited role played by market forces in the allocation of water, market prices upon which to base water-related resource allocation decisions are seldom available. In the jargon of the economist, *shadow prices* reflecting the value of water must be developed in their place.

Economists have in recent decades developed a number of techniques for measuring the economic values or benefits associated with non-market allocation in the subject matter areas relating to the environment and natural resources. These techniques call for a wedding of economic theory and applied economic practice. The theoretical foundations of non-market economic valuation of environmental resources have come to be well developed (see, for example, Freeman, 2003). Applied methods for estimating economic benefits in actual cases relating to environment have been greatly advanced (see, for example, Garrod and Willis, 1999). Valuation techniques for producers' uses of water such as crop irrigation, hydroelectric power and industries, appear to have received relatively less attention (see Gibbons, 1986; and Young, 1996b).

This chapter summarizes the conceptual framework for economic valuation of non-market goods and services as applied to water resources.

It begins by reviewing some of the distinctive attributes that characterize supply and demand for water-related goods and services. Most effort is given to developing the basic concepts and definitions used in measuring economic value or benefits of public water projects or policies. The chapter concludes with a discussion of some issues in valuation relatively important or unique to appraising public decisions regarding the water resource.

2. THE DISTINCTIVE NATURE OF WATER SUPPLY AND DEMAND

A number of special characteristics distinguish water from most other resources or commodities, and pose significant challenges for the design and selection of water allocation and management institutions. On the physical side, water is usually a liquid. This trait makes it mobile: water tends to flow, evaporate and seep as it moves through the hydrologic cycle. Mobility presents problems in identifying and measuring specific units of the resource. Water supplies tend, due to natural climatic fluctuations, to be variable, so that the risks of shortage and of excess are among the major problems of water management.

Water, due to its physical nature, and for other reasons, is what economists call a 'high-exclusion cost' resource, implying that the exclusive property rights which are the basis of a market or exchange economy are relatively difficult and expensive to establish and enforce. Frequently, then, property rights in water are incomplete or, more likely, absent.

Turning to the demand side, humankind obtains many types of values and benefits from water. Because each of the different benefit types usually call for specialized evaluation and management approaches, it will be useful to group the types of water-related economic values into several classes. These are (a) commodity benefits, (b) public and private aesthetic and recreational values, (c) waste assimilation benefits; and (d) disbenefits or damages. Each of these categories clearly involves economic considerations, because they are characterized by increasing scarcity and the associated problems of allocating resources among competing uses to maximize economic value. Whether certain other values associated with water, such as intrinsic values associated with endangered species preservation, ecosystem preservation and certain sociocultural issues of rights to water, can be measured within the economic framework remains a matter of debate. Resolution of that issue is not attempted here.

To consider water demand more closely, note that the economic characteristics of water demand vary across the continuum from *rival* to *non-rival* goods (Randall, 1987). A good or service is said to be *rival* in consumption, if one person's uses in some sense preclude or prevent uses by other individuals or businesses. Goods that are rival in consumption are the types that are amenable to supply and allocation by market or quasi-market processes, and are often called *private* goods. Goods that are *non-rival* in consumption, meaning that one person's use does not preclude enjoyment by others, occupy the opposite end of the continuum. Goods that are non-rival are often called *public* or *collective goods*. Because non-payers cannot be easily excluded, private firms will not find it profitable to supply public goods. Water for agricultural, residential or industrial uses tends toward the rival end, while the aesthetic value of a beautiful lake or stream is non-rival.

The significance of non-rivalry can be better understood by noting its association with high exclusion costs. *Exclusion cost* refers to the resources required to keep those not entitled from using the good or service. Water is frequently a high-exclusion-cost good because of its physical nature noted above: when the service exists for one user, it is difficult to exclude others. In such cases, it is hard to limit the use of the good to those who have helped pay for its costs of production. (The unwillingness of some beneficiaries to pay their share of the provision of a public good from whose benefits they cannot be excluded is called the *free rider* problem. To circumvent the problem, public goods must normally be financed by general taxes rather than by specific user charges.)

The commodity benefits – the first type of benefit mentioned above – are those derived from personal drinking, cooking and sanitation, and those contributing to productive activities on farms and in businesses and industries. What are here called commodity values are distinguished by the fact of being mostly *rival* in use, meaning that one person's use of a unit of water necessarily precludes use by others of that unit. Commodity uses tend to be private goods or services.

Continuing with the discussion of commodity-type uses, some additional distinctions will be helpful. Those types of human uses of water, which normally take place away from the natural hydrologic system, may also be called *withdrawal* (or *off-stream*) uses. Since withdrawal uses typically involve at least partial depletion or consumption (for example, from evaporation and/or transpiration), they may further be distinguished as *consumptive* uses. Other types of economic commodity values associated with water may not require it to leave the natural hydrologic system. This group may be labelled *in-stream* water uses: hydroelectric power generation and waterways transportation being important examples. Since in-stream uses often involve little or no physical loss, they are also sometimes called *non-consumptive* uses. (Although in-stream uses do not 'consume' much water, in the sense of evaporating it into the atmosphere, they do often require a change in the time and/or place of availability – as are the cases with water stored for future use for irrigation or hydropower generation – and therefore exhibit some aspects of the rivalry of a private good.)

The economic benefits from water for recreation, aesthetics, and fish and wildlife habitat are a second group or type of value of water. Benefits in this class are also closer to the non-rival end of the spectrum. Although aesthetics and recreation were sometimes viewed as non-essential goods inappropriate for public concern, as incomes and leisure time grow, these types of benefits are increasingly important. In developed countries, the populace increasingly chooses to utilize water bodies for outdoor recreational activities. Even in developing nations, water-based recreational activities are becoming more important for their own citizens, and also often provide a basis for attracting the tourist trade. As with waste assimilation, recreational and aesthetic values are also nearer the public good end of the spectrum. Enjoyment of an attractive water body does not necessarily deny similar enjoyment to others. (However, congestion at uniquely attractive sites, such as waterfalls or mountain lakes, may adversely affect total enjoyment of the resource.) Significant in-stream values also are found as habitat for wildlife and fish forms a basis for sporting activities.

The economic benefit from waste disposal is a third general class of economic benefits of water use. Bodies of water are considered as a sink for carrying away a wide range of residuals from processes of human production and consumption. Water resources are used for disposal of wastes, diluting them and, for some substances, aid in processing wastes into a less undesirable form. They are therefore significant for what is called their 'assimilative capacity'. The assimilative capacity of water is closer to being a public or collective (rather than private) value, because of the difficulty in excluding dischargers from utilizing these services.

Dis-values (also called damages or negative benefits) of water represent an important related classification. Examples are found in connection with evaluations of floodplain and water quality management. Flood waters or excesses of pollutants reduce welfare. Conversely, reduction of disbenefits increases human welfare. In such cases, mitigation policies may be assessed by valuing the projected reductions in damages.

Non-use values are also an important consideration in water allocation, and for the economic valuation of water. It is observed, in addition to valuing the commodity benefits of water use, that people are willing to pay for environmental services they might neither use nor experience. Non-use values are benefits received from knowing that a good exists, even though the individual may not ever directly experience the good. Voluntary contributions toward preserving an endangered fish species represent an example. Most resource economists have concluded that non-use values should be added to use values so as to more accurately measure total environmental values (Freeman, 2003).

Because of differing conceptual frameworks, an additional useful distinction is between intermediate goods and final consumption goods. *Intermediate* goods (also called *producers'* goods) are employed to make *final* products (to be eventually used by consumers). Intermediate goods represent the largest class of off-stream uses of water by humankind. For example, water for crop irrigation, the largest single consumptive user of water in the world, is an intermediate good; cotton or maize grown under irrigated conditions are destined eventually to be further processed to become clothing or food. Industrial processing and hydroelectric power generation are other intermediate uses of water. *Consumption* goods are those providing direct human satisfactions. Residential water is an example of a final consumption good from the private (rival) good classification, while recreation and amenity services provide non-rival final consumption values. The importance of this distinction between intermediate and consumer goods is that the economic theory of a profit-maximizing producer provides the conceptual framework for the valuation of intermediate goods, while the theory of the individual consumer is the basis for valuing consumer goods.

Yet another useful distinction is between *real* and *pecuniary* economic effects. Real effects are actual changes in quantities of goods and services available, or changes in the amount of resources used. Real effects are positive or negative changes in welfare. Real effects are further subdivided into direct and indirect effects. *Direct* economic effects of water projects or policies are those which accrue to the intended beneficiaries; those that can be captured, priced or sold by the project entity, or – in the case of costs – which must be paid for. *Indirect* or *external* effects are those uncompensated side effects affecting third parties. Economists classify external effects as either technological or pecuniary. Technological externalities are real changes in production or consumption opportunities available to third parties, and generally involve some physical or technical linkage among the parties (such as with degraded water quality). This type of externality represents a change in welfare, and should be reflected in evaluation of the economic efficiency effects of policies or projects. *Pecuniary* impacts (often referred to as secondary economic impacts in the water planning literature) are those reflected in changes in incomes or prices (such as effected by increased purchases of goods and services in a regional economy). Secondary economic impacts typically represent income distribution impacts. From the larger perspective of nation or state, secondary impacts registered on a specific locality are likely to be offset by similar, but more difficult to isolate, effects on income of opposite sign elsewhere.

Economic convention therefore suggests that secondary impacts not be taken into account in economic evaluations, or only in special cases (see, for example, Boardman, et al., 2001, p. 114).

3. ECONOMIC VALUE VERSUS OTHER CONCEPTS OF VALUE

The economic approach is not the only way to assign values to natural and environmental resources. Broadly speaking, values can be termed *extrinsic* or *intrinsic*, both of which are relevant for water and environmental policy. The distinction rests on whether the basis for valuation derives from consequences for human welfare. Extrinsic (sometimes also called *instrumental*) values are those that arise because things or acts are instruments for humankind for attaining other things of intrinsic value. As an example, water resources may be valued (instrumentally) for their contribution to human health, welfare or satisfactions. Intrinsic values, in contrast, are assigned to things, actions or outcomes for their own sake, independent of means of providing or attaining other items or situations of value for humans (Anderson, 1993, pp. 204–6). For example, people often value environmental resources in ways other than from their use or consumption by humans; the public wishes to preserve endangered species or protect delicate ecosystems, without consideration of whether these offer immediate human utility.

It is important to recognize that both approaches to valuation are legitimately applied to environmental and resource policy (Pearce, 1993, pp. 13–15). However, the prevailing – although not unanimous view of philosophers – is that neither extrinsic nor intrinsic values are necessarily absolute. When values conflict, as they often do, a dilemma arises. In such cases, the only apparent solution is to make a practical judgement of how to compromise the competing goals (Maclean, 1993). Morgan and Henrion (1990, p. 27) describe a widely used method, called the *approved process* approach, which, roughly speaking, requires all relevant parties to observe a specified set of procedures or observe a concept of due process to estimate a policy's impacts on relevant measures of value. Any decision reached after an appropriate authority balances the competing values under the specified procedures is deemed acceptable. Standard water planning manuals (both the US Water Resources Council's *Principles and Guidelines*, 1983, and the Organisation for Economic Co-operation and Development's (OECD's) *Management of Water Projects*, 1985 – although neither acknowledge the underlying philosophical premises – appear to reflect an approved process approach. Both manuals call for a determination of environmental impacts (intrinsic values) to be balanced against human (economic and social) welfare (extrinsic value) considerations. Both manuals emphasize the display of impacts; the ultimate resolution or balancing of conflicting values is assumed to take place at the political, rather than the technocratic level.

The economic values discussed in this chapter are extrinsic (instrumental), in that they reflect people's assessment of a policy proposal's contributions or decrements to human welfare. These economic benefits will be appropriate to either a stand-alone economic analysis or as part of a more general multi-objective or approved process approach.

4. ECONOMIC CRITERIA FOR RESOURCE ALLOCATION AND VALUATION

Although the objectives of improving the distribution of income, enhancing environmental quality and attaining other non-market goals are important, the analysis here pertains exclusively to the objective of economic efficiency in the development and allocation of the water resource. There are two major reasons for this: first, under conditions of increasing scarcity and growing competition among water users, economic efficiency remains an important social objective and efficiency values have viable meaning in resolving conflicts; second, efficiency values provide a valuable means of assessing the opportunity costs of pursuing alternative objectives.

4.1 The Pareto Principle and Economic Efficiency

Economic efficiency may be defined as an organization of production and consumption such that all unambiguous possibilities for increasing economic well-being have been exhausted. Stated somewhat differently, economic efficiency is an allocation of resources such that no further reallocation is possible that would provide gains in production or consumer satisfaction to some firms or individuals without simultaneously imposing losses on others. This definition of economic efficiency (termed Pareto optimality) is satisfied in a perfectly functioning competitive economy. Abstracting from the mathematical elegance found in textbook expositions (for example, Just et al., 1982) and abstracting further from the time consideration in outputs and inputs of economic activities, Pareto optimality can be expressed quite simply in terms of the attainment of: (1) economic efficiency in production of goods and services; (2) economic efficiency in distribution of goods and services; and (3) resource allocation in a manner consistent with consumer preferences. Pareto efficiency is said to occur when the marginal benefits of using a good or service are equal to the marginal cost of supplying the good.

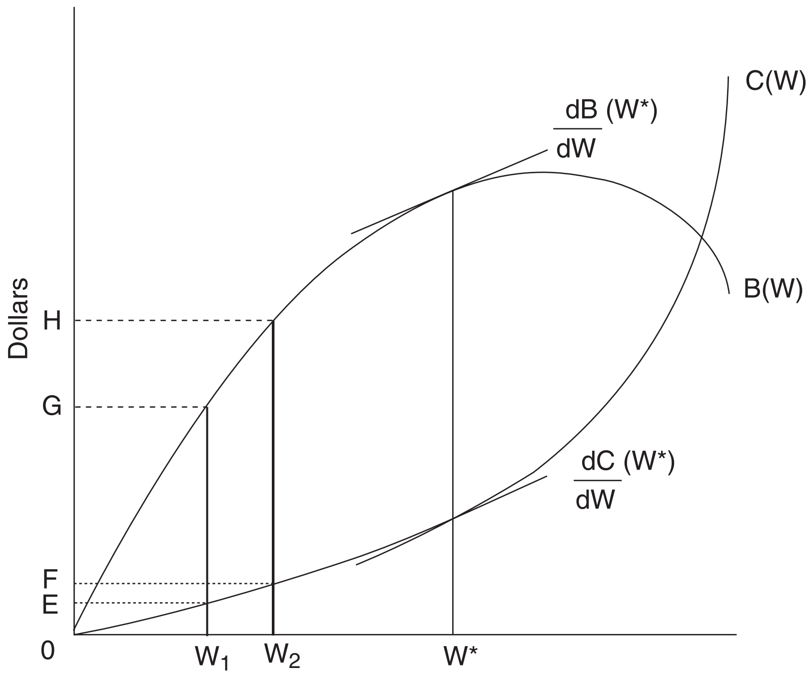
Pareto optimality rests on several central value judgements (Mäler, 1985). The first of these is the judgement that individual preferences count; the economic welfare of society is based on the economic welfare of its individual citizens. Second, the individual is the best judge of his/her own well-being. The third, highly restrictive, value judgement is that a change that makes at least one person better off while no one else becomes worse off constitutes a positive change in total welfare.

4.2 From Theory to Practice

Translating from the welfare economics theory to benefit–cost practice requires further steps. Because in a complex modern society, few policy changes that improve welfare for many would avoid lowering welfare of some individuals, few proposed changes would meet the strict Paretian standard of making no one worse off. However, welfare theorists circumvented this problem with the *compensation test*: if gainers could compensate losers and still be better off, the change would be judged an improvement. In practice, compensation is often impracticable; identifying and compensating all adversely affected parties is expensive and time-consuming. Hence, the compensation test becomes a test for a *Potential* Pareto Improvement (PPI). If gainers could *in principle* compensate losers, the change is deemed acceptable, whether or not the compensation actually takes place. Also, rather than evaluating all possible allocations in a continuous function framework, benefit–cost analysis typically examines fairly large discrete increments of change to assess whether the move is in the direction of Pareto efficiency. An action that generates incremental benefits in excess of incremental costs is termed Pareto-*superior*, because it leads to a condition superior to the *status quo ante*.

Following Smith (1986), Figure 2.1 illustrates the comparison of Pareto-efficiency and benefit–cost criteria. The curve denoted B(W) is a representation of aggregate benefits (that is, consumer or producer surplus) of alternative levels of water services (W), while C(W) represents the associated aggregate costs. These curves measure social welfare or aggregate utility and cost. Their general forms reflect the conventional assumption that benefits increase at a decreasing rate with increased output and costs increase at an increasing rate. The Pareto-efficient solution is at W* – the maximum vertical distance between B(W) and C(W). At W* the marginal benefits equal the marginal costs.

However, rather than seeking a full optimum solution, benefit–cost analysis (CBA) in practice typically considers whether a change from given



Source: After Smith (1986).

Figure 2.1 Comparison of Pareto-efficiency and benefit–cost criteria

conditions would represent a desirable shift. In Figure 2.1, such a change is represented by moving from W_1 to W_2. The conventional CBA test compares the aggregate increment in benefits (GH in Figure 2.1) with aggregate incremental costs (EF). If incremental benefits exceed incremental costs, as they are drawn to do in Figure 2.1, then the change is termed a Pareto *improvement*. Any act or policy judged a Pareto improvement would be recommended as preferable to the existing situation.

5. ECONOMIC VALUATION IN THE ABSENCE OF MARKET PRICES

Water management policies can have widespread effects on the quantity of water available, its quality, and the timing and location of supplies for both in- and off-stream uses. In general, these impacts have an economic dimension, either positive or negative, which must be taken into account in policy formation. Specifically, the decision process (resolution of conflicts) requires the identification and comparison of the benefits and costs of water resource development and allocation among alternative and competing uses.

Beneficial and adverse impacts to people are abstract and often ambiguous concepts. As noted earlier, mainstream economists treat values as extrinsic, and propose to measure impacts in terms of satisfaction of human preferences. To transform the concept of welfare into a single metric, the suggested measuring rod is that of money (Rhoads, 1985). A person's welfare change from some proposed improvement is measured as the maximum amount of money a person would be willing to forgo to obtain the improvement. Conversely, for a change that reduces welfare, the measure is the amount of compensation required to accept the change.

The economic evaluation of projects or proposals is based on balancing the predicted beneficial against the adverse effects generated by the proposal. *Benefits* are the 'good' or 'desired' effects contributed by the proposal, while *costs* are the 'bad' or 'undesired' impacts. This balancing of costs against benefits is called *cost–benefit analysis* (CBA). (For detailed treatment of the overall approach to CBA – particularly as applied to environmental and natural resource problems – the reader is referred to the extensive literature in that field, for example, Boardman et al., 2001; Dinwiddy and Teal, 1996; Johansson, 1993; Pearce, 1987; Zerbe and Dively, 1994.)

In applied CBA, the terms benefit and cost are assigned a narrow technical economic interpretation. The prices used in CBA are interpreted as expressions of *willingness to pay* (WTP) for a particular good or service by individual consumers, producers or units of government. Direct benefits are willingness of beneficiaries to pay for project services or policy impacts. Direct costs are willingness to pay for the forgone alternatives, or to avoid any adverse effects. In what follows, changes in producer surplus and consumer surplus, respectively, are accepted as the pertinent measures of willingness to pay or to accept compensation.

5.1 The Need for Shadow Prices

Howe (1971) has classified policy impacts into four categories that are paraphrased below:

1. Impacts for which market prices exist and market prices reflect scarcity values.
2. Impacts for which market prices may be observed, but such prices fail to accurately reflect true social values, but they can be adjusted to more accurately do so.

3. Impacts for which market prices do not exist, but it is possible to identify surrogate market prices.
4. Impacts for which market prices or surrogate prices are not meaningful.

The second and third cases are most typical in benefit–cost analysis for water resource planning; in these instances the prices employed (adjusted or estimated prices) are called *shadow prices* (or sometimes *accounting prices*).

Benefits and costs must be expressed in monetary terms by applying the appropriate prices to each physical unit of input and product. Three types of estimates are employed. Primary sources of the prices used for CBA are the result of observing the market activities. However, in the second type (often the case in water planning) it is necessary to make adjustments to observed market prices (for example, when agricultural commodity prices are controlled by government regulation or when minimum wage rates are set above market clearing prices). Finally, in many cases, it will be necessary to estimate prices that do not exist at all in any market (such as the value of water used for wetland preservation).

5.2 Defining Shadow Prices: The Willingness to Pay Principle

Whatever the source, the prices used in CBA are interpreted as expressions of *willingness to pay* or *willingness to accept compensation* for a particular good or service by individual consumers, producers or units of government. This presumption is obvious for market prices, since the equilibrium market price represents the willingness to pay at the margin of potential buyers of the good or service. For non-marketed goods, WTP also is the theoretical basis on which shadow prices are calculated. The assertion that willingness to pay should be the measure of value or cost follows from the principle that public policy should be based on the aggregation of individual preferences. Willingness to pay represents the total value of an increment of project output, that is, the demand for that output.[2] Willingness to accept compensation (WAC) is an important welfare measure in some contexts. Willingness to accept compensation is the payment that would make an individual indifferent between having an improvement and forgoing the improvement while receiving the extra money. Alternatively, it is the minimum sum that an individual would require to forgo a change that otherwise would be experienced. (Applied measurements of WTP and WTAC under the same conditions often find that estimates are not equal, in apparent conflict with economic theory. Various plausible explanations have been offered, both by economists and psychologists, but the issue seems to be unresolved. See Freeman, 2003, for further discussion.)

Therefore, *benefits* are defined as any positive effect, material or otherwise, for which identifiable impacted parties are willing to pay. *Costs* are the value of the opportunities forgone because of the commitment of resources to a project, or the willingness to pay to avoid detrimental effects. (Critics of certain applications of CBA from within the ranks of economists, observing that WTP is dependent on the existing distribution of income, properly caution against any unquestioning application of the technique for public investment decisions. However, few water policy initiatives would change the distribution of income enough to cause significant shifts in willingness to pay for benefits.)

5.3 Economic Surplus and Measures of Benefit

Economists base the concept of economic value on a decision framework within which rational individuals make the best use of resources and opportunities. The framework assumes that the individual members of the economy react systematically to perceived changes in their situation. Such changes can include – in addition to the quantity and quality of the water resource of primary interest here – prices, costs, institutional constraints and incentives, income and wealth.

Figure 2.2 illustrates the concepts of economic (producer's or consumer's) surplus under marketed commodity conditions. The curve denoted MBw in Figure 2.2 is a familiar demand curve, reflecting the maximum amount of the commodity W that consumers would be willing to take at alternative price levels. The demand curve slopes downward to the right, reflecting the desire for consumers to take more of the commodity W only as the price declines. The inverse demand curve (in which quantity is the independent variable and value is the dependent variable) can also be interpreted as the marginal willingness to pay for alternative quantities, so it is conventionally labelled in cost–benefit analysis, as in Figure 2.2, a *marginal benefit* (MB) function. *Consumer's surplus* is defined as the area above the price: it represents the difference between the maximum users would be willing to pay and what they would actually pay under a constant price per unit. The supply curves S_1 and S_2 represent a non-marginal shift in supply functions, such as from a project that increases the supply of some productive factor, such as water for crop irrigation.

Consumers enjoy two forms of gain: a decrease in unit price from P_1 to P_2 and an increase in available output (from W_1 to W_2). Producers also see a gain, from expanded output, but their price goes down. The area in Figure 2.2 circumscribed by the points P_1ABP_2 represents the gain in surplus enjoyed by consumers. With the change from W_1 to W_2, producer

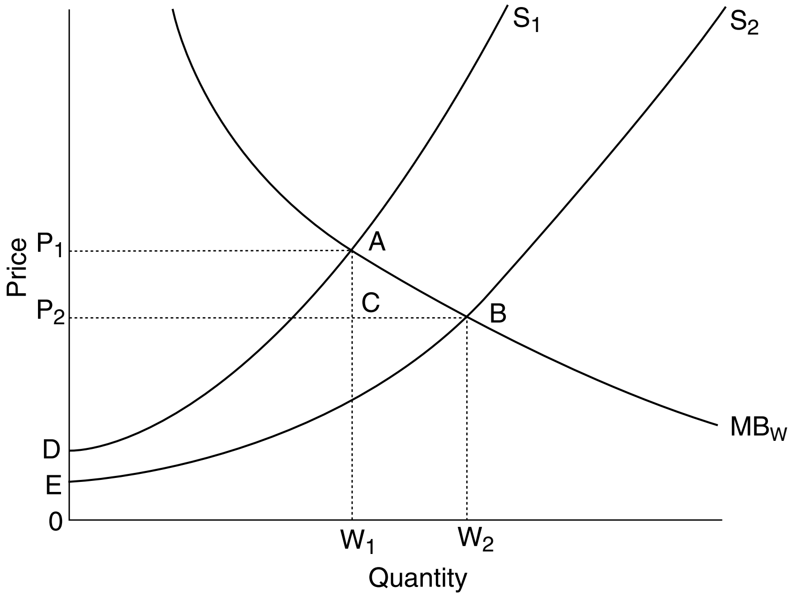


Figure 2.2 Price and quantity effects and change in economic surplus from non-marginal shift in supply of water

surplus changes from P_1AD to P_2BE. The net increase in economic surplus, the sum of gains and losses to producers and consumers, is DABE.

The economist reading the above paragraphs will note that the measures shown are for the ordinary *Marshallian* concept of demand and consumer surplus. More precise welfare measures, called Hicksian measures, are often reported in the applied welfare economics literature (Just et al., 1982). The Hicksian *compensating* version refers to the amount of compensation (received or paid) that would return the individual to his/her initial welfare position. The *equivalent* version refers to the amount of money that must be paid to the consumer to make him/her as well off as they could have been after the change. Whether to aim for the Hicksian formulation depends on the individual case. Marshallian demand functions are sometimes easier to estimate. Moreover, when purchases of the good or service in question accounts for only a small part of the household budget, it has been shown that the Marshallian measure is often a quite close approximation to the Hicksian measure. (See Freeman, 2003, for a more complete analysis.) For the case of water resources, which for the most part makes up a small fraction of consumers' budgets, the differences among the measures are probably smaller than the errors that occur in econometric estimation of the functions, so the Marshallian approximation will usually be acceptable in practical applications.

Figure 2.3 portrays a case frequently applicable to non-market valuation applied to water resources (Randall, 1987, ch. 13). It represents an increase in the availability of a non-priced water use from W_1 to W_2. Perfectly inelastic supply curves S_1 and S_2 shift from W_1 to W_2. The curve MB_w, as before, shows the downward sloping marginal benefit function. The area under MB_w between W_1 ('without change') and W_2 ('with change') represents the economic surplus attributable to the changed water supply. This area is that bounded by the points W_1ABW_2. It is this area that the economic analyst is attempting to measure in non-market valuation of changes in water and environmental amenities.

Note that the curve MB_w can, in addition to representing consumer demand, also portray the demand from producers. In the latter interpretation, MB_w is the producers' marginal value product (MVP) function, the marginal net return to increasing level of input. (See Johannson, 1993, s. 5.1, for a formal derivation of these properties of producers' welfare.) This

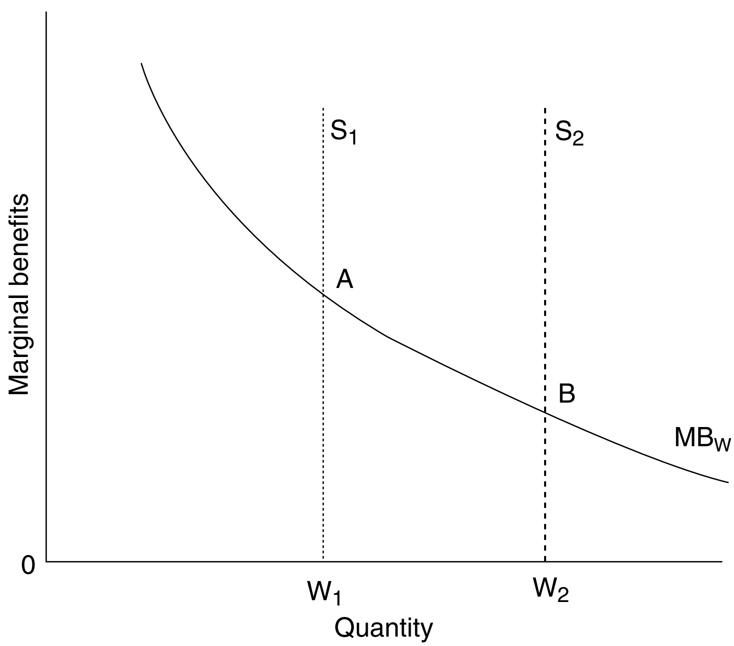


Figure 2.3 Change in economic surplus from non-marginal change in water supply

interpretation is, in fact, more applicable to valuation purposes than the producer's surplus depicted in Figure 2.2 (that is, the area above the supply curve S and below the price line). Also, in parallel with the Hicksian adjustment for income effects to consumer surplus measures, a corresponding adjustment for cost-minimizing allocation of other inputs or technology is appropriate for producer surplus measures (Johannson, 1993).

To recapitulate, the economic value of a non-marketed resource is measured by the summation of many users' willingness to pay for the good or service in question. Willingness to pay is a monetary measure of the intensity of individual *preferences*. Therefore, we can say that *economic valuation is the process of expressing preferences for beneficial effects or preferences against adverse effects of policy initiatives in a money metric*.

5.4 Opportunity Costs: Measuring Forgone Benefits of Reduced Water Use

Increasingly of interest are measures of opportunity costs of water resources. Opportunity costs are the benefits forgone when a scarce resource is used for one purpose instead of the next best alternative. When evaluating trade-offs of proposed reallocations, one needs a measure of the benefits of the proposed new use as well as the reduction of benefits associated with reduced water use in the sector currently benefiting. Hence, opportunity costs are conceptualized as the reverse of incremental benefits. Returning to Figure 2.3, a measure of opportunity costs would be the area under MB_w from, this time, W_2 to W_1. This is the same area as described before; in Figure 2.3 that bounded by the points W_1ABW_2. (Randall, 1987, Figure 13.5 conceptualizes this point more elegantly in a framework jointly accounting for increments or decrements of natural resource use.)

6. OTHER CONCEPTS USEFUL FOR APPLIED ECONOMIC EVALUATIONS RELATING TO WATER

A number of additional concepts are important for applied economic evaluation in water resource management. *The general point is that there is no single economic value of water. What is being measured is the welfare change associated with some policy-induced change in the attributes of the commodity*. It is important to keep clear what are the specific attributes of the situation and decision in question. A number of these issues are discussed in this section.

6.1 The With–Without Principle

The *with–without* principle holds that policy appraisal should contrast the 'state of the world' as it would be with the policy to the 'state of the world' as it would be without the policy. An important implication of the principle is that project evaluation is not adequately accomplished by comparing conditions before the project with conditions after its implementation. Many changes in the world from 'before' to 'after' would have occurred without the project, so such effects should not be credited or charged to the project.

The with–without principle directs the analyst to measure the impacts according to the status of the economy with the public intervention as compared to without the intervention. The intent is to identify only the impacts that are clearly associated with the project or programme, and not include as impacts any changes in the economy that would have occurred even without the project. Therefore, regional growth that would be due to private sector investment, or to other public projects should not be included in project impact measures. Project evaluations that measure impacts by comparing *before* with *after* the intervention are likely to overstate project impacts.

6.2 Accounting Stance

The *accounting stance* is defined here as *how* benefits, costs or other impacts are priced or counted in a cost–benefit analysis. The primary distinction is between *private* and *social* accounting perspectives, which differ as to how benefits and costs are measured. The private accounting stance measures impacts in terms of the prices faced by the economic actors being studied. In contrast, the social accounting stance draws on social prices (adjusted or shadow-priced so as to account for taxes, subsidies or other public interventions). The distinction between private and social accounting stances has seen most application in the case of agricultural water use, since many nations intervene in both commodity or input markets relating to agriculture. However, the analyst performing a social analysis may wish to use shadow prices for inputs such as labour, energy or capital in other contexts.

Although the terms *financial analysis* and *economic analysis* are used for the same distinction in some CBA manuals, particularly those from the World Bank (for example, Gittinger 1982), that terminology is avoided here. This is because these terms seem to me to be ambiguous and quite confusing to non-specialists, economists and non-economists alike. The methods termed financial analysis and economic analysis both employ the same basic economic techniques. The main difference is that they use different prices. Hence, the terminological distinction between private and social prices seems to capture the essential point.

6.3 Scope

Although this terminology is not in general use, the *scope* of a cost–benefit analysis is defined here as the geographical area or political entity or subdivision within which benefits, costs or other impacts are counted. A project or policy may have impacts that are confined to a local area, or they may extend to the nation or even internationally. For example, the economic benefits of an irrigation project may be confined to a local area. Some of the conventional direct costs of construction and operation might be met by water users (or taxpayers) in the project area, but part of the cost might be provided by the national government, so impacts would spread nationwide. Other costs, particularly indirect or external costs, such as forgone electric power generation or lower water quality imposed on downstream water users, will accrue well beyond the borders of the area benefited, but need to be accounted for in a full economic evaluation. Indirect benefits outside the project region can also occur. For example, interception of flood waters by irrigation or power reservoirs may yield benefits far downstream. Thus, both benefits and costs could extend well beyond the geographic area where direct benefits occur.

Ideally, the scope should be as encompassing as possible; real impacts should be accounted for no matter how far away or in what political jurisdiction they may occur. For example, indirect costs of water projects in upstream regions adversely affecting downstream neighbours (such as the forgone costs of depleting water) or indirect benefits (such as by intercepting flood waters) should be assessed. However, in practice, the choice of scope must be made on pragmatic grounds, balancing the gains in accuracy against the increased costs of spreading a wider net. Most national planning agencies suggest a national scope wherever possible, but in practice, few analyses give consideration to interests in downstream states or nations.

6.4 Long-run versus Short-run Values

Because policy decisions relating to water entail a range of cases, from major long-lived capital investments to one-off allocations in the face of immediate events such as droughts, it is often important to distinguish carefully between long-run and short-run values. The distinction relates to the degree of fixity of certain inputs, and is particularly important for cases in which water is a producers' or intermediate good, such as in irrigation, industry and hydropower.

A rational producer's willingness to pay for water will be based on net rents or returns to the input. In the short run, where some inputs are fixed, the estimate of the net increase in the value of output can ignore as sunk the cost of the fixed inputs. In the long run, where input costs must all be covered, these costs cannot be ignored. Therefore, we would expect that for the same site and production processes, values estimated for short-run contexts would be larger than values for the long run. Similarly, domestic water users exhibit different responses in the short versus the long run. Price elasticity of demand is less (in absolute value) in the short run when decisions are constrained, than in the longer-run decision context, when adjustments to shortages are possible. Accordingly, willingness to pay in the short-run planning situation is typically higher than in the long-run case.

However, most public water policy decisions involve situations where the long-run context is appropriate. Failure to observe this distinction has caused many non-specialists to erroneously use short-run measures for long-run decision contexts, thereby attributing too high a value to water uses. However, important cases occur – such as drought planning – where short-run values are appropriate.

6.5 Appropriate Measure of Water 'Use'

To assign an economic value to water, one must express it as a monetary value per unit water volume or quantity used. To the frequent confusion of non-specialists, several measures of water use are commonly found in the technical water literature. Moreover, at least one of the hydrologic terms for water use is the same word, but with a narrower meaning as that often adopted by economists. The need for different measures arises because, first, some water is typically lost between the water source and the water user, and second, because some additional amount of the water taken by the user is returned to the hydrologic system, where it sometimes is available to produce further human benefit.

Three measures of water use are possible candidates in an economic valuation. These are: *withdrawal*, *delivery* and *depletion*. Withdrawal refers to an amount of water diverted from a surface source or removed from a groundwater source for human use. Delivery is the amount of water received at the point of use (home, farm or factory). Withdrawal differs from delivery by the amount of *conveyance losses* to the point of use. That is, withdrawal minus conveyance loss equals delivery. (Conveyance losses are typically significant in water delivery systems. Losses of up to 30 per cent are not unusual in agricultural delivery systems, many of which often are of simple earthen construction and not sealed with impervious materials. Although urban water delivery systems are usually more efficient, losses of this magnitude or even higher may also be found in domestic water supply systems in developing countries. (See Nickum and Easter, 1994.) Depletion (often termed *consumptive use* or simply *consumption* by hydrologists) is a measure of water use referring to that portion of water withdrawn from a source that is made unusable for further use in the same basin. Depletion or consumptive use mainly occurs via evaporation and transpiration, but also may be due to contamination or drainage to a saline sink. Those who are not specialists in hydrology tend to use 'consumption' broadly as a synonym for 'use', so the term 'depletion' is suggested for the technical concept.

Next, almost all off-stream human uses of water *release* some water back to the hydrologic system. In urban settings, this may take the form of sewage discharges. In agriculture, a considerable amount of water is typically lost as drainage water, either through seepage into a groundwater system or overland flow via drainage ditches. *Return flow* in the technical water literature is a measure of that portion of water withdrawals which returns to the hydrologic system still usable for human purposes. Return flows comprise both conveyance losses and releases back to the hydrologic system. Thus, withdrawal minus return flow equals depletion.

The choice of withdrawal, delivery or depletion as the measure of water use will depend on the purposes at hand. For valuing off-stream uses, the quantity variable most often used is the amount *delivered* to the user. Alternatively, the measure may be the amount depleted. Economic values per volume of water will likely differ greatly, depending upon which measure is chosen. For the economist interested in predicting user behaviour in response to changing prices or entitlements, the delivery measure is often more appropriate, because that is the measure upon which water users base their allocation decisions. Hence, willingness to pay is usually conceptualized as of the point the firm or household receives the water. However, for river basin planning exercises, the net amount of water depleted in a particular use is the relevant measure, since that is the amount not available for further use downstream. Where necessary to consider quantities depleted, valuation can be made in terms of deliveries and adjustments to express benefits per unit depleted can be subsequently made.

Turning to non-depleting or in-stream uses, none of the above variables are precisely relevant. One must take any change in form, timing or location as a measure of water use. In the case of evaluating in-stream versus off-stream uses, incorporating a hydrologic model that can adjust for all these interdependent factors becomes an important aid.

6.6 Commensurability of Place, Form and Time: At-site versus At-source Values

Marketed economic commodities are priced according to spatial, quality and temporal attributes, and shadow pricing of water should follow similar rules. For example, another economically important liquid, petroleum, is always priced in terms of grade, location and date of delivery. A look at a daily newspaper's business pages reveals that prices for crude oil at the point of production are less than the cost per unit volume of refined gasoline (petrol) in bulk at some specified distribution point, which is in turn much lower than the price of gasoline at the local retail station. These considerations lead to a need for analysts engaged in comparative water valuation exercises to be careful to assure that the chosen measure of water value is *commensurable* in terms of a common denominator of place, form and time.

Water falls among the commodities which economists call 'bulky.' This means that the economic value per unit weight or volume of water tends to be relatively low. (For example, retail prices for water delivered to households are typically in the range of US0.0005–0.0008 per litre or about US$1.0 to $1.5 per ton, much less than other liquids important in contemporary life, such as petrol (gasoline), milk, soft drinks or beer. In crop irrigation, much of the water applied may yield direct economic values – profit net of costs of other inputs – of less than US$0.04 per ton.) Bulky commodities tend to exhibit high costs of transportation per unit volume, so that costs of transporting them become an increasingly important part of the total cost of supply. In the case of water, this point implies that water values are often highly site-specific.

Consider now the aspects of location and form. Because of its low value at the margin, capital and energy costs for transportation, lifting and storage tend often to be high relative to economic value at the point of use. Therefore, water at different locations may have widely differing values, and moving the commodity from one place to another frequently may become uneconomical due to conveyance costs. Thus it may be important to distinguish between *at-site* and *at-source* values, a consideration inadequately recognized in the water valuation literature. As the terms indicate, at-site values represent willingness to pay at the point of use or delivery, while at-source values measure willingness to pay at some point in the hydrologic system where water is withdrawn. At-source values are derived values that are sometimes called values for 'raw' or 'untreated' water. At-site values differ from (exceed) at-source values by whatever costs are required to transport, store, treat, and deliver the water from source to site. By convention in water supply project evaluations (but not by necessity) water supplies for off-stream uses are usually valued in at-site terms, and the storage and delivery costs are included in the total costs of providing water to users. In contrast, evaluations of intersectoral water allocations should use at-source values for each sector, so that the comparisons among sectors – be they producers' or consumers' uses and off-stream or in-stream – are in commensurate terms. Water in its raw (untreated) form in a river – or even in a reservoir or canal – is a distinctly different commodity than water delivered (perhaps after treatment and under pressure) to a farm, business or residence, and comparisons of value in alternative uses must recognize that point. Comparing values among uses is best performed with the comparisons made in terms of raw water supplies at some specified point of diversion. (Booker and Young, 1994, represents an early example of a class of combined hydrologic-economic models in which demands are initially expressed in at-site terms and which account for return flows and delivery costs so that in the final analysis both economic and hydrologic variables are expressed in at-source terms.)

Also, because of the variations in demand over the seasons of the year, the value will – other factors being equal – change with time. In many places, water has little value for irrigation in winter, but it may be quite useful at that time for power generation or industry.

7. SPECIFIC CASES OF ECONOMIC EVALUATION OF WATER RESOURCES ISSUES

In a river basin management context, the principal opportunities for economic welfare enhancement, and hence the need for measures of water value, are, first, investments in capturing, storing, delivering and treating new water supplies and, second, reallocation among water-using sectors. Other examples where marginal values of water might be useful include: optimal groundwater basin policy (for example, Provencher and Burt, 1994; Young, 1992) and pricing and cost recovery for investments in water supply systems. Of most interest are the cases of investment and reallocation decisions, discussed below in more detail.

7.1 Evaluating Investments in Additional Water Supplies

Consider now a simple framework (Equation 2.1) that shows the conditions for economic feasibility of a potential investment in water supply from the point of view of the private investor. All benefit and cost elements in the models presented below are assumed to be expressed in annual equivalent terms, employing a consistent interest rate and planning period and reflecting the same general price level.

$$DB_p > DC_p \tag{2.1}$$

where the symbols represent the following concepts: the subscript p denotes the private perspective; DB_p is direct private user benefit (willingness to pay for the initiative) and DC_p is the direct private cost. Direct benefit reflects the economic value of the physical increment in production due the increment in water supply. Direct costs are the costs of bringing the water supply to the user. Equation 2.1 asserts simply that the contemplated investment is economically feasible if, from the private investor's perspective, direct benefits exceed direct costs. The private investor is assumed to ignore any uncompensated indirect benefits or costs received by or imposed on third parties.

Turning next to evaluation of the impacts of an investment from the public or social accounting stance and national scope, three types of adjustments and additions should be made to Equation 2.1. First, benefits and costs are adjusted for subsidies or other government-induced market distortions. For example, crops produced with the aid of government support programmes – such as cotton or rice in the southwestern United States – would be valued at lower price levels, derived from estimated free market prices (which task is a challenge itself). Costs would similarly be adjusted for public subsidies (such as low-cost credit or energy) or penalties (for example, minimum wage regulations). On balance, these adjustments usually make the social net benefit of added water less than the private net benefit.

The terms new in Equation 2.2 are IB, representing indirect (real external) benefits, SB denoting secondary (pecuniary external) benefits, IC standing for real external costs and SC denoting secondary external costs. The other adjustments needed for a shift to the public accounting stance are to incorporate monetary estimates of any external effects, both real and pecuniary. These steps are represented in Equation 2.2, in which direct benefits and costs are expressed in social prices (adjusted for market price distortions, denoted by introducing a subscript s) and external impacts (both real and pecuniary) are incorporated in the formula. The Potential Pareto Improvement (PPI) hypothesis to be tested is:

$$\text{Is } (DB_s + IB + SB) > (DC_s + IC + SC)? \tag{2.2}$$

In words, is the sum of the present values of direct, indirect and secondary benefits greater than the sum of present values of direct, indirect and secondary costs?

Secondary benefits, the multiplier effects arising from increased purchases of production inputs and consumption goods when a project comes into operation, are typically concentrated in the project region. Secondary benefits are normally measured with specialized economic techniques (such as regional interindustry models). Regional models of this type simulate the effects of an increment of resources on the regional economy. Secondary costs (SC) are the pecuniary benefits forgone when a public investment draws funds (via taxes) from the economy at large. Secondary costs typically spread throughout the national economy and are very difficult to measure. As remarked in section 2 above, the conventional economic wisdom (embedded in public planning manuals and texts in CBA – for example, Boardman et al., 2001) is that from the national accounting stance, secondary or pecuniary costs are at least as large or larger than secondary benefits. Hence, the two effects offset each other and, except in special cases, secondary economic impacts can be ignored for national investment planning purposes.[3] Indirect costs and benefits, the other class of external effects, are real impacts and should be incorporated into evaluations adopting a public accounting stance. Indirect benefits are not often economically important in the context of water investments, but indirect costs are typically very significant. Examples of indirect costs of water withdrawals include reduced downstream water supplies or adverse effects on water quality downstream for off-stream (irrigators, industries, households) and in-stream (hydroelectric power plants, recreational water users and fish and wildlife habitat) water users.

In implementing this PPI test, economic valuation or shadow pricing will be required for the terms B_{it} and D_{jt}. (Of course, the PPI test can be also expressed in the alternative, but largely equivalent forms of benefit–cost ratios or internal rates of return. See for example, Gittinger, 1982, for discussion.)

7.2 Evaluating Proposals to Reallocate Water among Sectors

Another likely welfare improvement opportunity is for reallocating water among use sectors. The hypothesis (for a Potential Pareto Improvement) to be tested is: can a reallocation from sector i to sector j yield incremental gains to sector j in excess of the forgone benefits in the ith sector?

In applied cases, the hypothesis of sub-optimal allocation is tested for specific proposals for reallocation. Consider a proposal to reallocate water from agriculture to municipal uses. Indirect impacts are expected on the hydropower sector. The PPI test can be expressed by developing measurements for two conditions (Young, 1986).

The first condition is that the benefits (both direct and indirect) to the municipal sector exceed the sum of: (forgone direct benefits to the selling sector plus forgone indirect benefits to the selling sector plus forgone indirect benefits to the hydropower sector). Condition 1 can be written (assuming all benefit and cost expressions are in present value terms, employing a consistent planning period and price level):

$$DB + IB > FDB + FIB + FIB + TC + CC \quad (2.3)$$

where:
DB: direct economic benefit (value) to receiving sector
IB: economic benefit to indirectly affected sector(s)
FDB: forgone direct benefit (value forgone) in source sector
FIB: forgone benefit in indirectly affected sector(s)
TC: transactions costs (for information, contracting and enforcement)
CC: conveyance and storage costs
A further condition is that the direct forgone benefits in irrigated agriculture be the least-cost source of water for the purchasing sector:

$$FDB + FIB + TC + CC < AC \quad (2.4)$$

That is, condition 2 asserts that the sum of direct and indirect foregone economic benefits and the transactions and conveyance costs should be less than the cost of the next best alternative water source.

Economic analysis of both issues – as well as the other resource allocation and cost recovery problems mentioned in the introduction – require the estimation of incremental or marginal benefits of changes in water supply or use. The overall challenge is critically to examine methods for estimating the various manifestations of incremental benefits.

This discussion has focused on measuring benefits of increments or decrements of water supply. To this point, the analysis has abstracted from two other important dimensions of water supply – water quality and supply reliability. These are taken up in the next two subsections.

8. THE BENEFITS OF IMPROVED WATER QUALITY

The quality of water, of course, also influences its economic value. Water in natural environments is never perfectly pure. Humankind uses water bodies as sinks for disposal of numerous wastes from production and consumption activities. The extent to which micro-organisms, and dissolved or suspended constituents are present varies greatly, and in sufficiently high concentrations can affect health, and reduce aesthetic values and productivity. Therefore, the content of pollutants or, conversely, the degree to which the water is treated for various uses is important in determining its economic value.

Estimating benefits of improved water quality raises some complex and challenging issues. For the important cases of degradable effluents – those that are transformed after discharge into receiving waters – the detrimental effects depend on the nature of downstream water uses, the distance downstream, temperature, rates of flow and the quality of receiving waters. Willingness to pay for water quality improvement is usually assumed to reflect damages to subsequent water users. The damage function is a measure of the effect of the concentration of pollutants on the utility or costs of receiving entities. Benefits are the damages avoided from a given project or regulative policy.

The framework for conceptualizing the benefits of water quality improvement can be readily derived by extending the model developed earlier for increments of water supply. All other factors (prices, incomes, technologies, and so on) held constant, an improvement in quality of water for either producers or consumers will shift the demand or marginal benefit curves to the right. The increment in producers' or consumers' surplus accruing to the change will be the appropriate measure of benefits of an improvement in water quality. (See Spulber and Sabbaghi, 1998, ch. 2, for a rigorous exposition. A more advanced formulation, with application primarily to groundwater contamination is found in Bergstrom et al. 2001.)

A related example responds to the need for measuring economic damages from releases of harmful materials into public water bodies. This issue has increasingly come into prominence in the USA in response to the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) of 1980 (see, for example, Kopp and Smith, 1993).

9. THE BENEFITS OF IMPROVED WATER SUPPLY RELIABILITY

The degree of certainty with which supplies are available, in addition to its quantity and quality, is another important factor influencing the willingness to pay for water. Domestic, industrial and agricultural water demanders all place a higher value on reliable water supplies than on supplies with high risk of availability. At least two cases can be envisioned for which the potential for changed reliability might have value. The major source of water supply unreliability comes from normal hydrologic risk; reflecting the inevitable swings in precipitation and runoff. (For individual users, hydrologic variation may be exacerbated by the institutional arrangements for sharing shortages. Where the rule for allocating shortages is a priority – first in time/first in right – system, high priority users may be little affected while low priority classes experience more than proportional fluctuation.)

Another problem is the short-term lack of reliability of water supply systems, due to either inadequate capacity or to breakdown. Some Third World cities, for example, lack sufficient capacity to be able regularly to deliver water to all customers on demand. A policy of rotating supplies among different geographic sectors of the city's system may serve as a rationing method. In such cases, even customers with piped residential connections are unable to obtain water on demand throughout the full 24-hour day, or even are unable to obtain some water every day (Nickum and Easter, 1994).

Increasing reliability comes at increasing costs, so trade-offs are necessary between cost and risk. Conventional technical risk analysis as applied to water supply reliability selects a risk level roughly reflecting the potential severity of adverse effects, and designs projects to satisfy the selected degree of risk (Renn, 1992). Reliability standards typically vary among use classes: for reasons of health, sanitation and, implicitly, willingness to pay, water supply reliability is usually set higher for domestic supplies than for irrigation.

The technical approach treats all affected areas and parties equitably, but it ignores economic efficiency considerations. Under technical reliability standards, investments to improve reliability may not be subjected to systematic comparison of costs of improved reliability with the expected losses averted. Therefore, large expenditures may sometimes be made which have little prospect for a corresponding reduction of damages. In contrast, the economic approach goes beyond the identification of the probability of some adverse event to the measurement of the disutility of such events to humans.

Howe and Smith (1994) developed a model for assessing reliability and apply it to the case of municipal water supply. Of interest here is how they formulate a function reflecting the economic benefits of reliability to compare with costs of achieving reliability. They defined the 'Standard Annual Shortage Event' (SASE) as a drought of sufficient severity and duration that certain specified restrictions on water use would be put in place. (Howe and Smith's case study was for cities in the semi-arid western United States, so the hypothesized drought-induced restrictions were on summer, outdoor water usage for lawns and gardens.) Here, the discussion abstracts from the optimization model formulated by Howe and Smith to focus on the marginal economic benefit of improved reliability. System reliability, R, is defined in terms of probability (P) of occurrence of the SASE:

$$\mathrm{R(SASE)} \equiv 1 - \mathrm{P(SASE)} \tag{2.5}$$

Next, a *loss function* L(SASE), is introduced, which represents the reduction in economic value accruing if the SASE were to occur. The desired economic measure, the *marginal benefit of improved reliability*, is given by the incremental reduction in expected losses (denoted E(L)):

$$\delta E(L)/\delta R \tag{2.6}$$

Howe and Smith implement their model empirically with a contingent valuation survey. Griffin and Mjelde, 2000 represent a more recent endeavour at valuing water reliability, one that illustrates the problems of empirical measurement of willingness to pay for uncertain outcomes. Valuing reliability has received relatively little attention, but more effort on this topic is clearly warranted.

10. UNCERTAINTY AND SENSITIVITY ANALYSIS

Estimating benefits for long-run water investment or allocation decisions by its nature requires forecasting the behaviour of a number of economic, hydrologic technological and social variables for a many-year planning period. Because of the limited predictabilities in these factors influencing water management decisions, no analyst can expect to be fully accurate in such a situation. It is desirable that some recognition of uncertainty be incorporated into benefit analysis. Basing a plan simply on best-guess projections may bring about an unwarranted degree of confidence in the results.

A number of formal treatments of uncertainty are applicable to evaluation of water investment and allocation decisions (for example, Morgan and Henrion, 1990). The techniques recommended in these sources – usually based on estimating objective or subjective probabilities of occurrence of key variables – are typically used in evaluating flood risk reduction measures and may be used by academic researchers. However, adoption of such formal techniques will often require too much in the way of analytic expertise and study resources to be useful under many actual planning conditions.

A more practical alternative for acknowledging uncertainty is to use 'sensitivity analysis'. The effect of (sensitivity to) important variables on the estimated value of water is determined by varying one element at a time to determine the sensitivity to erroneous forecasts (Gittinger, 1982). For example, a study of the economic benefits of irrigation should test for sensitivity to assumptions about future crop yields, crop prices or production costs. The cost of capital, represented by the interest or discount rate, is an important variable of uncertain value, and sensitivity to its potential values often should be tested. A sensitivity analysis cannot reduce the risk of a given plan. Sensitivity analysis does not change the facts, but shows the impacts of incorrect assumptions regarding key parameters.

A variation on sensitivity analysis is the 'switching value' test. The switching value test investigates how far a key element in the analysis would need to change in an unfavourable direction before benefits fell below zero.

11. CONCLUDING REMARKS

The economic valuation of goods and services whose prices are in some way distorted or for which markets do not even exist is an important aspect of environmental and resource economics. Economists recognize that people value things – including many important services of the earth's water supply – that they do not purchase through a market or that they may value for reasons independent of their own purchase and use. Further, not everything that reduces utility – such as pollution – is adequately costed in markets. Although economists are sometimes equated with Oscar Wilde's cynic (who 'knows the price of everything and the value of nothing'), environmental economists in fact spend much of their professional efforts attempting to estimate the public's value (often called a shadow price) for non-marketed goods and services.

This chapter has reviewed the conceptual framework for estimating economic efficiency benefits of decisions relating to water supply, allocation and quality. The modern economic paradigm assumes that values of goods and services rest on the underlying demand and supply relationships that are usually, but not always, reflected in market prices. Economics is not just the study of markets but, more generally, it involves the study of preferences and human behaviour. The prices used in cost–benefit analyses are interpreted as expressions of willingness to pay for a particular good or service by individual consumers, producers or units of government. Direct benefits are willingness of beneficiaries to pay for project services or policy impacts. Direct costs are willingness to pay for the forgone alternatives, or to avoid any adverse effects. The numerous techniques developed for applied non-market valuation of water are based on these principles.

Much of the applied non-market valuation literature has dealt with water resources in one or another of its many ramifications, but there is not yet any single publication that brings all these disparate methodologies together for all types of water uses. Moreover, although many of the resource valuation techniques, particularly on the topic of environmental quality, have been subject to critical scrutiny and testing, some important areas of water valuation have received less attention. Particularly for the intermediate or producers' goods derived from water – such as crop irrigation, hydroelectric power and industrial and commercial water use – procedures for empirical applications of valuation techniques appear to be less developed and seem to have received less application and critical confrontation. An important next step will be to extend the applied paradigm to meet that challenge.

NOTES

1. The term 'value' takes on a narrow meaning in economics, referring to money measures of changes in economic welfare (Freeman, 2003, p. 7). 'Economic benefit' and 'economic value' will be used interchangeably here to refer to positive welfare changes resulting from investment projects or policy initiatives.
2. Some authors, unfortunately, in addition to this broad meaning, use 'willingness to pay' to refer to a specific type of non-market valuation study which directly questions people on their valuations for environmental changes. To avoid ambiguity, these specific techniques would best be identified by the name of the relevant elicitation process – that is, 'contingent valuation'.
3. Regional models have occasionally been used, incorrectly in my view, to measure direct economic benefits according to a 'value added' concept. See Young and Gray (1985) for a critique.

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3. Water as an economic good

J. Briscoe

1. THE THEORY OF WATER AS AN ECONOMIC GOOD

There is an emerging consensus that effective water resources management includes the management of water as an economic resource. The Dublin Statement of the International Conference on Water and the Environment, for example, states that 'water has an economic value in all its competing uses and should be recognized as an economic good'. But there is little agreement on what this actually means, either in theory or in practice. This chapter provides a simple framework for unbundling the different components of water as an economic resource, provides some data on critical variables and discusses the policy implications.

The idea of 'water as an economic good' is simple. Like any other good, water has a value to users, who are willing to pay for it. Like any other good, consumers will use water so long as the benefits from use of an additional cubic meter exceed the costs so incurred. This is illustrated graphically in Figure 3.1(a), which shows that the optimal consumption is X^*. Figure 3.1(b) shows that if a consumer is charged a price P^1 which is different from the marginal cost of supply, then the consumer will not consume X^*, but X^1. The increase in costs (the area under the cost curve) exceeds the increase in benefits (the area under the benefit curve) and there is a corresponding loss of net benefits called the 'deadweight loss'.

But what about groups of users, how is welfare maximized for the group and society as a whole? The simple logic of Figure 3.1 applies in the aggregate – for society as a whole, welfare is maximized when:

- water is priced at its marginal cost; and
- water is used until the marginal cost is equal to the marginal benefit.

So far so good, but what actually do we mean by 'benefits' and 'costs', how are these dealt with in different water-using sectors and what are the implications? These issues are explored in the next section of this chapter.

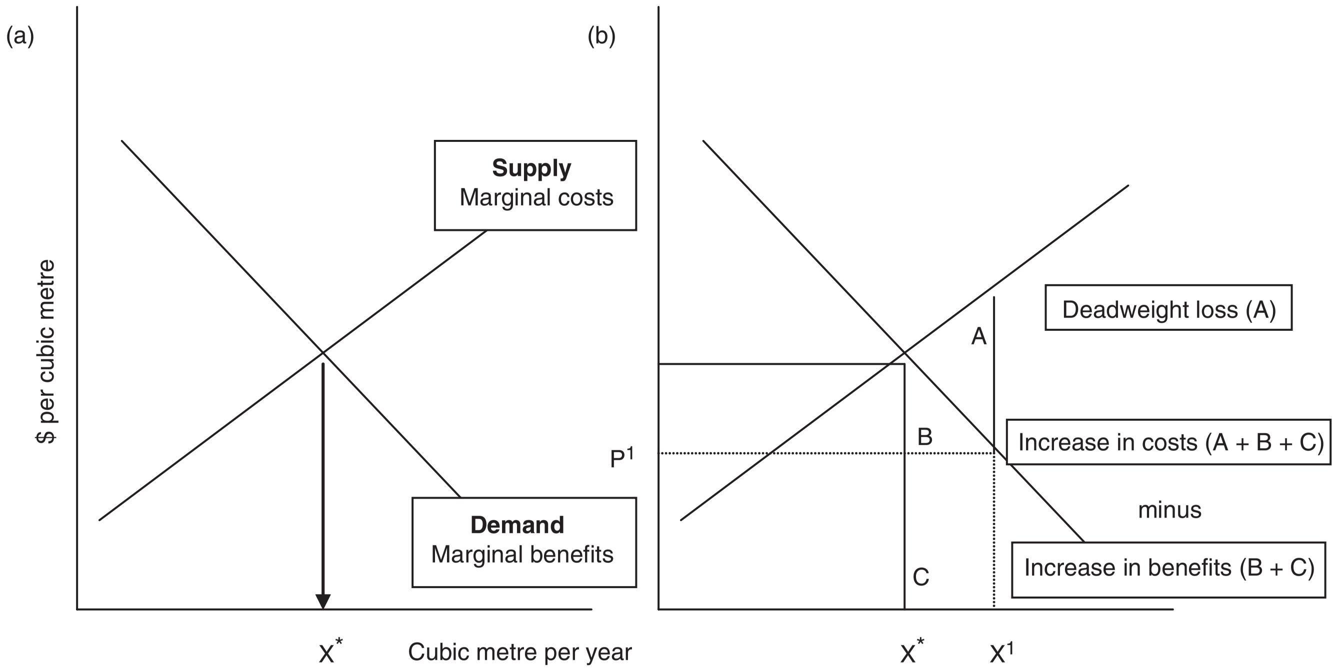


Figure 3.1 Optimal consumption and deadweight losses if water is underpriced

2. THE VALUE OF WATER

The value of water to a user is the maximum amount the user would be willing to pay for the use of the resource. For normal economic goods which are exchanged between buyers and sellers under a specified set of conditions, this value can be measured by estimating the area under the demand curve. Since markets for water either typically do not exist or are highly imperfect, it is not simple to determine what this value is for different users of water. A hodgepodge of methods are used to estimate the value of water in different end uses (Gibbons, 1986). These methods include:

- estimating demand curves and integrating areas under them;
- examining market-like transactions;
- estimating production functions and simulating the loss of output which would result from the use of one unit less of water;
- estimating the costs of providing water if an existing source were not to be available;
- asking (with carefully structured 'contingent valuation' questions – Arrow et al., 1993; Griffin et al., 1995) how much users value the resource.

What is the point of estimating these values, given the crude and inexact nature of the estimates, and given that the value of water varies widely depending on factors such as the use to which it is put, the income and other characteristics of the user, the location at which it is available, season and time, and quality and reliability of the supply? Most certainly these 'ball-park estimates' can never, and should never, be used to make technocratic decisions on allocations and prices (as has sometimes been proposed). But examination of the values which emerge from these estimates do show some striking and remarkably consistent themes which have major implications for policy. To illustrate these themes, it is useful to work with some actual values. Figure 3.2 summarizes some data (presented by Moore and Willey, 1991) from the western United States, where most valuation work has been done. Other compilations (for example, in Gibbons, 1986) show similar patterns in terms of the relative value of water in different uses.

Conclusions which emerge from Figure 3.2 (note the log scale on the Y axis) and consistently in similar studies and in meta-studies which draw together large amounts of available data include the value of water for:

- irrigated agriculture;
- hydropower;
- household purposes;

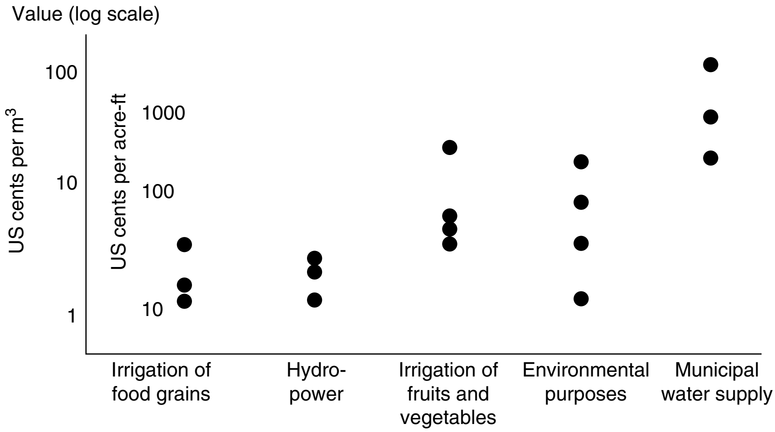


Figure 3.2 Typical market and non-market values for water in the western United States

- industrial purposes; and
- environmental purposes.

2.1 Value of Water in Irrigated Agriculture in Industrialized Countries

It is, first, important to note that irrigated agriculture accounts for a large proportion of water use, especially in many water-scarce areas. The value of water for many low-value crops (such as food grains and fodder) is universally very low. Where reliable supplies are used on high-value crops, the value of water can be high, sometimes of a similar order of magnitude to the value of water in municipal and industrial end uses.

2.2 Value of Irrigation Water in Developing Countries

The picture in developing countries is similar. Consider the case of India. In western India (Shah, 1993) groundwater is exploited by private farmers and is provided in a timely and responsive fashion to users (the farmers themselves and others to whom they sell the water). The water is used on high-value crops (including fruits, vegetables and flowers). The value of water, as reflected in active and sophisticated water markets, is high (typically around US 5 cents per cubic metre). In public (mostly surface) irrigation systems in the same country, the quality of the irrigation supply is poor, food grains are the major crop produced, and the value of water is typically only about 0.5 cents per cubic metre (World Bank, 1994a), orders of magnitude lower than in the private groundwater schemes. Similar very large and persistent differences are found in publicly run irrigation schemes throughout the developing world.[1]

2.3 Value of Water for Hydropower

The short-run values for water in hydropower in industrialized countries are typically quite low, often no higher than the value in irrigated agriculture (Gibbons, 1986). Long-run values are even lower. Whether hydropower is an economic proposition depends greatly on particulars – of the economy, of the power sector and of the water sector. Where water is abundant and there are few competing uses, hydropower is likely to be economically viable; where water is scarce (and therefore competition high), the case for hydropower is less clear-cut.

In developing countries the demand for power is growing very rapidly. Although energy conservation is important here (as it is in industrialized countries), large capacity expansion is inevitable and essential. It has been argued (Goodland, 1996) that the high environmental costs of alternatives (especially fossil-fuel based generation) means that hydropower is a particularly attractive alternative in many developing countries. Interestingly, data suggest that the environmental costs – as measured by flooded area per kw and number of oustees per kw – are substantially smaller for big dams than smaller dams (less than 100 megawatts of installed capacity).

It is frequently argued that hydropower is a non-consumptive use and therefore does not impose costs on others. It is this notion which has, for instance, been behind the creation of two separate categories of water rights – 'non-consumptive' and 'consumptive' – in Chile (Gazmuri and Rosegrant, 1996). What is evident – in Chile and elsewhere – is that the situation is not so simple. By modifying flow regimes and the timing of water to downstream users, hydropower installations can impose major costs on other users (Briscoe, 1996b). The key issue is not consumptive or non-consumptive use, but the costs imposed on others by a particular use of a resource.

2.4 Value of Water for Household Purposes

This value is usually much higher than the value for most irrigated crops. Not surprisingly, the value for 'basic human needs' and for household uses is much higher than the value for discretionary uses (such as garden watering). An important finding (similar to that emerging from the irrigation data) is that people, even poor people in developing countries, value a reliable supply much more than they value the intermittent, unpredictable supplies which are the norm in most developing countries (World Bank Water Demand Research Team, 1993).

2.5 Value of Water for Industrial Purposes

This value is typically of a similar order of magnitude to that of supplies for household purposes.

2.6 Value of Water for Environmental Purposes

The value of water for environmental purposes such as maintenance of wetlands, wildlife refuges and river flows also vary widely, but typically fall between the agricultural and municipal values, as shown for the western United States in Figure 3.2. In developing countries, most similar work has been done on the value of mangrove swamps (in El Salvador, Malaysia, Indonesia and Fiji), which are critically dependent on inflows of fresh water. These data, too, show quite high values (primarily due to the off-site impacts on fisheries) (Lai, 1990).

Before discussing the policy implications of these remarkably consistent findings, it is relevant to summarize a related area of work on the economic value of water, which also has major impacts for policy. There is a substantial literature assessing how users react to changes in the price of water. The concept used is that of 'elasticity', with the measure being defined as the percentage change in use of water for each percentage increase in the price of water. Once again, there is a striking consistency to the findings (and to their import for resource management, as discussed later). Figure 3.3 presents some values (again from Gibbons, 1986) which do not purport to be universal, but which illustrate consistent findings in the literature.

In assessing data on elasticity, it is necessary to clear up a confusion generated by a piece of economic jargon. When the price elasticity of demand is less than -1.0 (that is, when the percentage change in consumption is less than the percentage change in price) then economists say 'demand is inelastic with respect to price'. The common-sense (but erroneous) interpretation is that demand is not reduced as prices change. In fact, as long as price elasticity is negative, demand is reduced when prices increase.

An obvious omission from Figure 3.3 – the lack of estimates of the price elasticity of demand in irrigated agriculture – needs to be explained. This is best done with reference to the place where it has been most studied – the western United States. In the western USA the price elasticity of demand for irrigation water is low. The reason for this low elasticity is not that farmers do not respond to prices (as is often inferred), but rather because users' reactions to price changes depend on the original price and

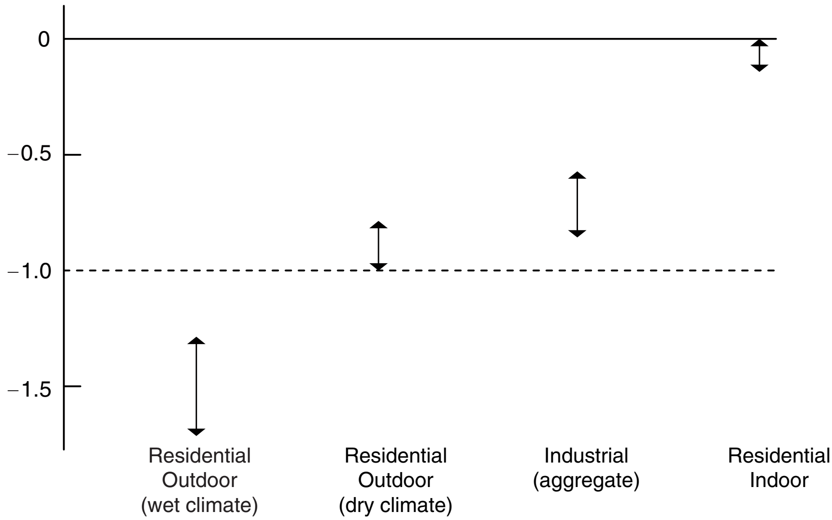


Figure 3.3 Range of price elasticities of demand for water in the United States

because irrigation water costs are held artificially low (Gibbons, 1986). In California, for example, where water is priced at $3 per thousand cubic metres, a 10 per cent price increase causes a 5 per cent decline in water use, whereas where water is priced at $14 per thousand cubic metres, a 10 per cent price increase results in a 20 per cent drop in use (Rogers, 1986).

The major point that emerges from the (quite large) literature on the price elasticity of water demand is that, in developing and developed countries alike, the price elasticity is significantly negative, meaning that users react to price increases by reducing demand. A second important point is that the price elasticity is, as common sense would suggest, related to the price level – the higher the price, the greater the elasticity. Obvious and commonsensical as these findings may be, they contradict a large body of folklore about 'non-responsiveness to prices' in the water profession.

Before concluding this discussion of 'value', it is relevant to focus on the issue of the 'value' of waste water treatment, or the 'value' of environmental quality. The usual approach to this has been to assume that it is impossible to assess this value and, instead, to promulgate standards (by type of treatment required, quality of effluent stream, or quality of the receiving stream). This is often perceived as a way of 'getting round' the issue of value. As was shown in a seminal work by Harold Thomas (1963), setting of a standard is equivalent to imputing a value for the resource. As will be discussed later, there are institutional arrangements for setting standards which violate (at great cost) this understanding, but there are also institutional arrangements which provide practical and proven methods for taking these values into account implicitly in setting standards.

3. THE COST OF WATER

So much for the value side of the equation – what of the cost side? In thinking about 'the cost of water' it is first necessary to acknowledge that there are two different types of costs incurred in providing water to, say, a household or a field. The first (obvious) cost is that of the constructing and operating the infrastructure necessary for storing, treating and distributing the water. In this chapter this is referred to as the 'use cost'. The second, less obvious, cost is the 'opportunity cost' incurred when one user uses water and, therefore, affects the use of the resource by another user. For example, greater abstraction of water by a city might affect the quantity and quality of water available to downstream irrigators, thus imposing costs on these users.[2]

3.1 Use Cost

In discussing 'use costs', it is first necessary to define three concepts. First is the concept of 'historical costs'. Consider the example where a water board constructs a reservoir from which it supplies water to its customers. What should the board charge its customers for the service provided by the reservoir? Frequently, the charging system mimics the mortgage payers of a homeowner – the board charges its users that which is necessary to pay for the remaining portion of the debt incurred in financing the dam. This is known as 'historical cost' pricing. The second, less intuitively obvious concept is that of 'replacement cost pricing'. Accountants will argue that the value of the asset (the dam in this case) is not correctly measured by its historic costs (which are often heavily distorted by government intervention), but rather the cost that would be incurred in replacing the asset. The analogy here is that of the housing rental market. If a homeowner has paid off his or her mortgage, he or she does not charge a tenant nothing – rather, he or she charges a rental fee that reflects the replacement cost of the asset. The third concept is that of marginal cost. Economists argue that when someone is thinking about using a bucket of water, they should not be told (through prices) what it costs to produce that water but, rather, be told the cost that will have to be incurred if capacity needs to be expanded to produce another cubic meter of water (Turvey and Warford, 1974). Where cost curves are relatively flat, the distinction between the former (average costs) and the latter (marginal costs) is unimportant. When costs are falling (as happens where there are economies of scale, for instance in treatment plants), marginal costs are less than average costs. For raw water, however, the situation is just the opposite, because the closest, cheapest sources are those which are used first. The cost curve for raw water, then, is almost always rising, and marginal costs are greater than average costs.

3.2 Opportunity Cost

It is obvious that measuring the opportunity cost of water is a difficult task. It needs a systems approach and a number of more or less heroic assumptions about real impacts and responses to these. What can be said with certainty is that:

- Opportunity costs are related to value in a non-transitive way. That is, if a city and an irrigation district lie on opposite banks of a stream, the opportunity costs imposed by abstraction by the high-valued user (the city) will be much lower than the opportunity costs imposed by abstraction by the low-value user (the irrigation district).
- Opportunity costs increase substantially as the water in a basin becomes more 'densely used' (both in quantity and quality terms) and are, therefore, substantially higher, all other things being equal, in arid, heavily used basins.
- The existence and imposition of opportunity costs can give rise to conflicts amongst users, unless there are institutional mechanisms for recognizing these costs and for ensuring that these are taken into account by users (on which more later in this chapter). Such conflicts are, of course, not a new phenomenon – the etymology of the word 'rivals', originally meant 'one living on the opposite bank of a stream from another' (Oxford English Dictionary, 1971).

4. THE BALANCING OF VALUE AND COSTS

The overall 'economic cost of water', therefore, comprises two separate components – the use cost and the opportunity cost. It is useful to maintain and deepen this disaggregation in thinking about how the idea of 'the cost of water' is understood, and how this understanding frames the public, political and theoretical discussions of water management. In doing this, it is instructive to recognize that there are a variety of ways in which the use cost and opportunity cost are perceived, and how different institutional arrangements mean that users are faced with different vectors of 'use' and 'opportunity cost'.

In exploring these relationships it is useful to first define the 'golden standard', namely, that combination of use and opportunity costs which ensure that users take the full economic costs of using water into account. As illustrated in Figure 3.4, a user faces the full economic cost when he or she (a) has to pay a 'use cost' which corresponds to the marginal financial cost of supplying the water to him or her and (b) incurs an opportunity cost which reflects the value of water in its best practical alternative use. This combination of 'use cost' and 'opportunity cost' is shown in the upper right-hand corner of Figure 3.4.

So much for theory, what about practice? This varies by sector and by country. A few examples will illustrate the general situation.

4.1 Urban Water Supply in Industrialized Countries

Practice in urban water supply in industrialized countries deviates from 'the economic optimum' in two ways, which are significant in theory, but of little importance in practice. Regarding 'use charges', water utilities in industrialized countries are generally operated on commercial or quasi-commercial principles (World Bank, 1994b), and recover the full average financial costs (level III in Figure 3.4) from users. There are two reasons why few utilities operate at level IV (the economic optimum).

First, although there are negative economies of scale for raw water, there are positive economies of scale for the major civil works, which account for much of urban water supply costs. Accordingly, marginal costs may not be different from (and may actually be less than) average costs. Second, setting tariffs to cover average costs is a simple, transparent process, which mimics that of commonplace financial transactions. A corollary is that the (small) economic benefits of moving to marginal cost pricing have to be weighed against the (large) administrative and governance costs of dealing with a system which 'defies common sense' for most customers.

Urban water tariff setting also deviates from the economic optimum in that the opportunity costs of water are often not visible to the utilities (except in well-functioning water resource management systems, two of which are described later in this chapter). In any case, these opportunity costs are, from the point of view of urban water supplies, usually very small relative to the financial costs of abstracting, transporting, treating and distributing water. For the urban water sector Figure 3.4 would usually look like a 'tall L', as shown in Figure 3.5.

The 'tall-L' shape for urban water arises both because the value of raw water for municipal uses is typically (as shown in Figure 3.2) an order of magnitude higher than the value of the next best use, and because the costs of raw water constitute only a minor part (typically less than 20 per cent)

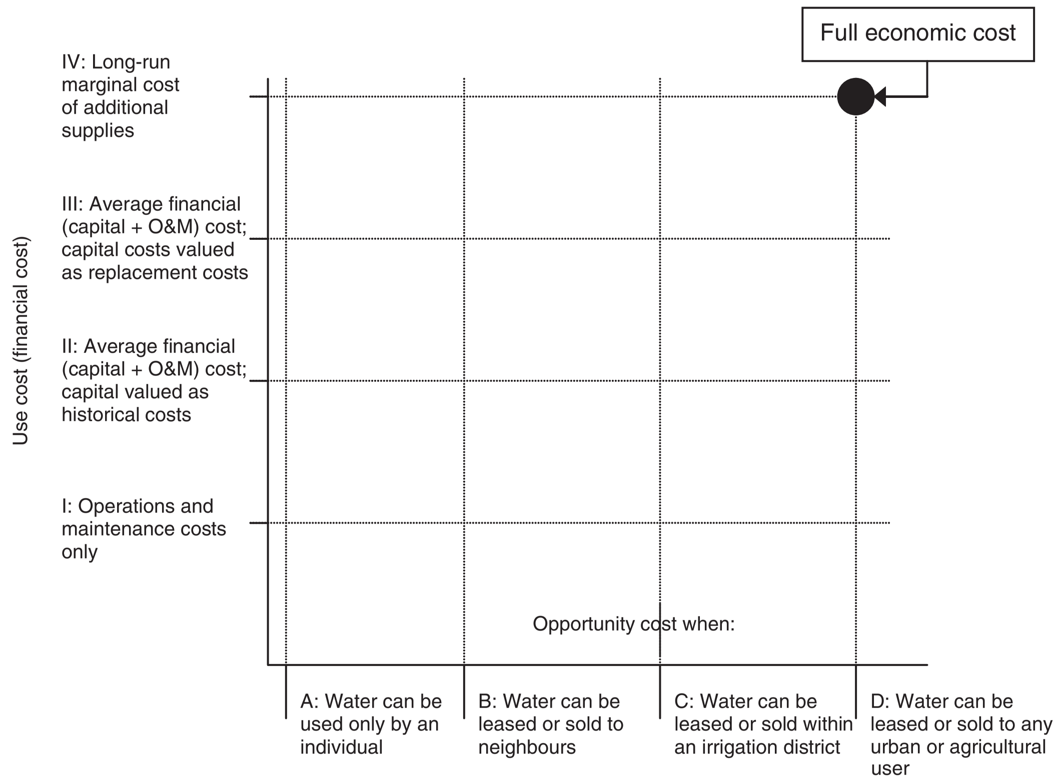


Figure 3.4 Schematic representation of the definitions of use cost and opportunity cost

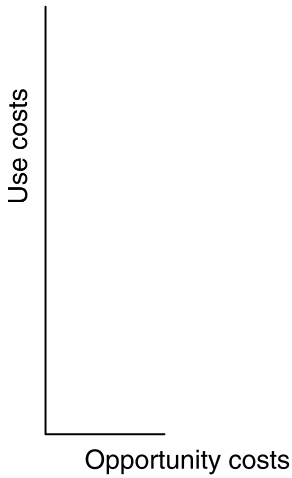


Figure 3.5 The relative magnitudes of use costs and opportunity costs for urban water supply

of the cost of water as delivered to the customer. The bottom line then is that, although opportunity costs are often not taken into account, the 'tall-L' shape of Figure 3.5 means that, in practice, urban water supply pricing in industrialized countries deviates little from the economic optimum.

4.2 Urban Water Supply in Developing Countries

In developing countries the situation is quite varied and generally quite different from that in industrialized countries. The first difference comes on the cost side. Many cities in developing countries are growing rapidly. In many cities incomes are also increasing and industrial demand is growing. The net result is that the demand for municipal water is often growing very fast and new sources have constantly to be found. A consequence is that the costs of urban supplies from new sources are growing rapidly – in current World Bank financed projects the cost of a cubic metre of raw water for a city is typically two to three times greater (in real terms) than was the case in the last project (World Bank, 1992). In terms of Figure 3.4, this means that the difference between marginal (level IV) costs and average (level III) costs are typically substantially greater for developing countries than for industrialized countries. Unfortunately the story does not stop there. Urban water supplies in most developing countries have been financed but of general revenues. In many cases these costs are fully subsidized, with the utility responsible only for operation and maintenance costs (level I).

In other cases the costs are computed in historical terms, which typically greatly undervalue the assets of the utility.

With regard to opportunity costs, the situation is similar to that in industrialized countries – they are not taken into account, but are also usually small relative to real financial costs. In a typical case in India, for instance, average financial costs ('use costs') are about US 50 cents per cubic metre, whereas the opportunity cost of water (for irrigation of food grains) is about 0.5 cents per cubic metre, a difference of two orders of magnitude.

The important challenge for urban water utilities in developing countries, is, therefore to:

- reduce costs by more efficient operation, which increasingly means substantial involvement of the private sector (Serageldin, 1995; World Bank, 1994b); and
- raise tariffs from their very low levels, which typically cover less than one-third of costs (World Bank, 1992). Worrying about opportunity costs they impose – the short leg on the L in Figure 3.5 – is not a priority problem for urban water utilities in developing countries.

4.3 Privately Financed Irrigation

The great distinction here is not between industrialized and developing countries, but rather between publicly and privately financed irrigation schemes. In most countries private irrigators bear the full financial costs of the schemes they construct and thus implicitly face financial costs at level III in Figure 3.4. In a number of countries this is not the case, with subsidies substantially reducing the financial costs incurred by private irrigators.[3]

Private irrigators seldom face any opportunity costs for the water they use. Where groundwater is used, this has led to the unsustainable pumping of aquifers, sometimes on a huge scale, such as the Ogallala aquifer in the United States (Rogers, 1986). Where surface water is used, this is often in the context of a 'prior appropriation' water doctrine, which implicitly encourages the ignoring of opportunity costs.

4.4 Publicly Financed Irrigation

Public irrigation systems throughout the world share several striking characteristics. First, as has been documented in countries as different as the United States (Bradley, 1996; Worster, 1992; Reissner, 1986), and India (Wade, 1986), they have been enormous sources of political patronage. Typically these investments have been subsidized almost completely by the state. In most developing countries charges have been much lower than those required even to pay for operations and maintenance costs (World Bank, 1995). In Bihar in India, for example, water charges are not sufficient even to cover the costs of collection (Rogers, 1992).

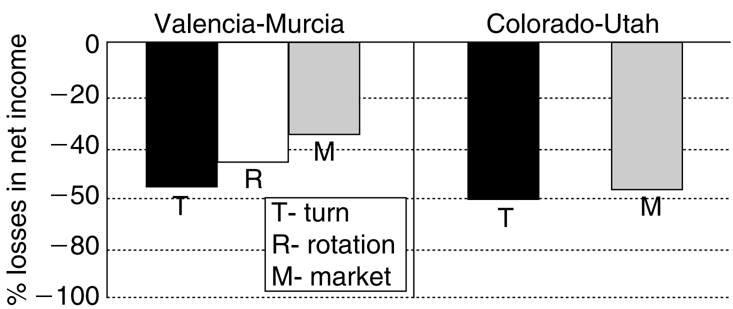
The issue of 'recovering the costs of operations and maintenance' has been the focus of much debate in the irrigation community. This is an important debate, first, because the associated issue of ensuring that systems are maintained and provide a good-quality service to users such as farmers is obviously appropriate and central to improving irrigation performance. This issue thus deservedly occupies centre stage in reviews, such as a recent one by the Operations Evaluation Department of the World Bank (1995). An important finding from such reviews is that the supply side of this question is at least as important as the demand side. It has been shown repeatedly that cost recovery in irrigation systems makes little positive difference unless the revenues so collected are applied to improving the quality of service received by the farmers. Where these revenues go to a central treasury (as is frequently the case), there is little improvement in irrigation performance if 'costs are recovered'.

The 'opportunity cost' axis is an important and subtle one in canal irrigation systems (the dominant technology in public irrigation districts). A typical situation is one in which users are charged a small amount (often zero) for the 'use cost', but where they do take account of one restricted measure of the opportunity cost of the resource. The best-known example of this is the rotational rationing system of north India (the so-called 'waribandi system'). As students of the system have pointed out, in this setting water is often the limiting production resource. Each farmer, therefore, faces an 'opportunity cost' which influences the way in which he uses that resource. While this is true (and is often neglected in criticisms of such systems) it should be observed that the opportunity cost varies considerably depending on 'alternative uses' which come into play. In the waribandi system, the 'opportunity cost' is essentially that of the opportunities which the individual farmer forgoes on another (non-irrigated) field, assuming he has one. The 'opportunity cost' would evidently be greater if all farmers in a particular distributory were included, since it is the value placed by the highest alternative use which defines the opportunity cost.[4]

Similarly, if it were possible (as is increasingly the case) to transfer the water among a wider universe of potential users of that water (which will usually include other farmers, and may include neighbouring towns and industries), then the 'opportunity cost' would be greater still. While 'the best alternative use' needs to take into account location and the hydraulic connections possible between users, it is certain that the restrictive 'opportunity cost' implicit in rationing systems (like waribandi) will often represent large underestimates of the true opportunity costs and will therefore mean that farmers are facing both use and resource costs which represent substantial underestimates of the true costs. Under such circumstances, as explained earlier, deadweight losses are likely to be substantial.

The magnitude of these losses has been estimated in a seminal assessment of different irrigation systems in Spain and the United States. Maass and Anderson (1978) did simulation analyses of the effects of different water allocation procedures on the economic impact of water shortages. In the 'turn' system, farms are served in order of location along the canal. When water reaches a farmer, he takes all he needs during the period, before the next farmer is served (a procedure followed in Valencia). In the 'rotation' system each farm has a reserved time in which to irrigate in each period, but the water delivered in this time varies on each rotation depending on the flow in the ditch (a procedure followed at the time of the study in Fresno, Utah and Murcia.) In the 'market' system, all water users bid each period for the water used to irrigate their crops and the water is allocated to the highest bidders (a procedure followed in Alicante). As shown in Figure 3.6:

- the market system is far superior in terms of overall productive efficiency; and
- the differences between the market system (which incorporates the opportunity costs within the command area) and the turn and rotation systems (which do not incorporate these opportunity costs) is large.



Source: After Maass and Anderson (1978).

Figure 3.6 Relative efficiency of different American and Spanish water management procedures when water to an irrigation district is reduced by 10 per cent

A relevant aside is to note the effects of different water management regimes on the distribution of losses amongst farmers when there are shortfalls in water availability. The standard measure for inequality is that of the Gini coefficient – as shown in Figure 3.7. The Gini coefficient is:

- zero when losses are equally distributed equally across the land; and
- unity when all losses are concentrated in a single farmer.

As shown in Figure 3.8, in both Spain and the United States, the market system was markedly superior to the turn and rotation systems in terms of

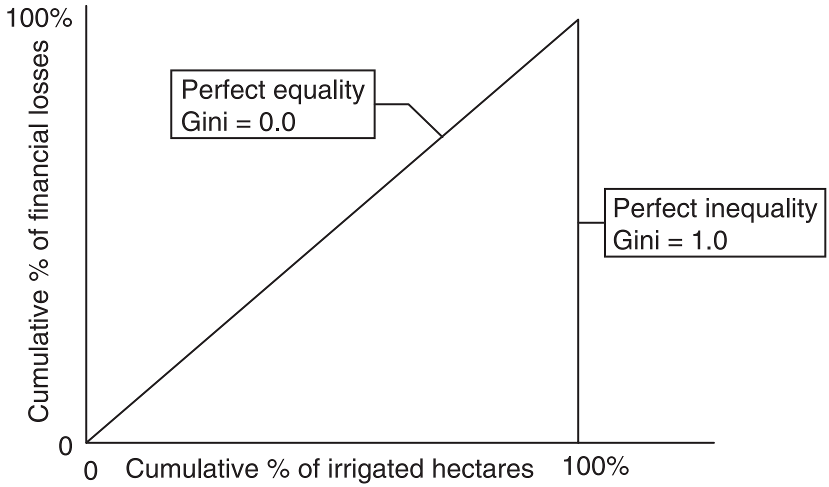


Figure 3.7 Measures of equality – the Gini coefficient

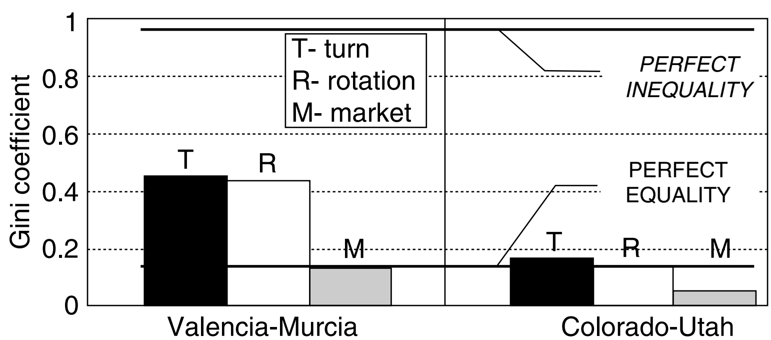


Figure 3.8 The equity of different water allocation systems

the equity of distribution of the losses resulting from a water shortage. As pointed out by the authors,

> although it is a doctrine of many welfare economists that procedures that rank high in efficiency will do poorly in distributing income equally among beneficiaries, while procedures that do well in distributive equality will be inefficient . . . this conventional wisdom does not apply to a wide variety of conditions in irrigated agriculture. (Maass and Anderson, 1978, p. 391)

4.5 The Implications for Irrigation vis-à-vis Urban Uses

In summary, when considering the relative magnitudes of the use cost and opportunity cost of irrigation, the situation is almost exactly the opposite of that pertaining for urban water supply. Financial costs of irrigation systems are usually much lower (per unit of water) than they are for urban water, and opportunity costs are much higher, both absolutely and relatively, as shown in Figure 3.9.

Ignoring opportunity costs is thus a matter of minor practical importance when it comes to the economic management of urban water supplies, but a matter of huge practical significance when it comes to irrigation. As illustrated schematically in Figure 3.10, the shape for irrigation is a 'flat L' in contrast to the 'tall L' in Figure 3.5 for urban water supply.

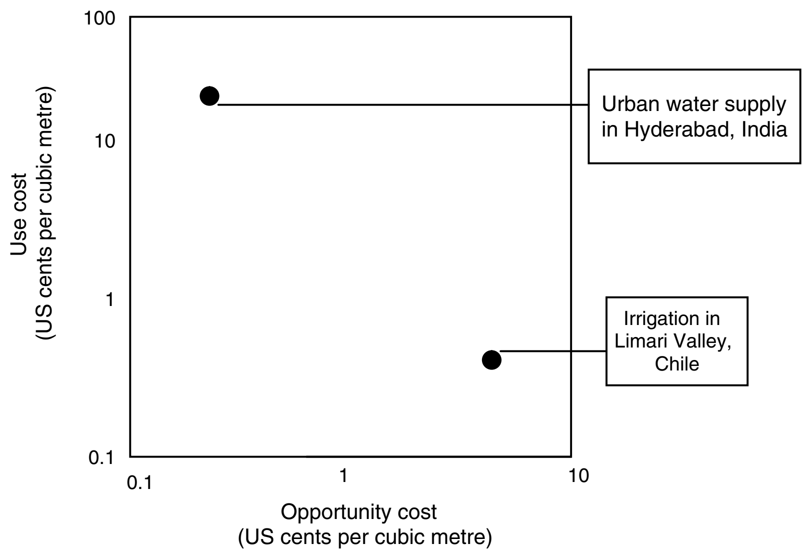


Figure 3.9 Illustrative values of use and opportunity costs for urban supply and irrigation opportunity costs

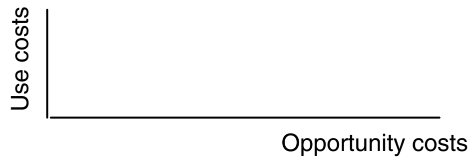


Figure 3.10 The relative magnitudes of use costs and opportunity costs for irrigation

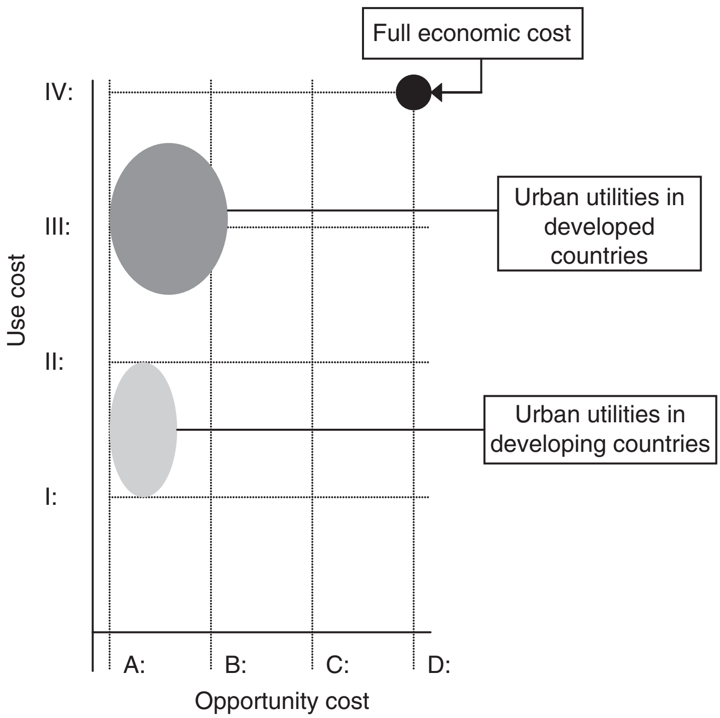


Figure 3.11 Schematic representations of deviation from economic pricing for urban water supply

Finally, it is instructive to return to the graphical format developed in Figure 3.4 to summarize the issues on use and opportunity costs as they pertain to different water using sectors. Figures 3.11 and 3.12 provide a schematic representation of how the management of different water using sectors deviate from the economic optimum.

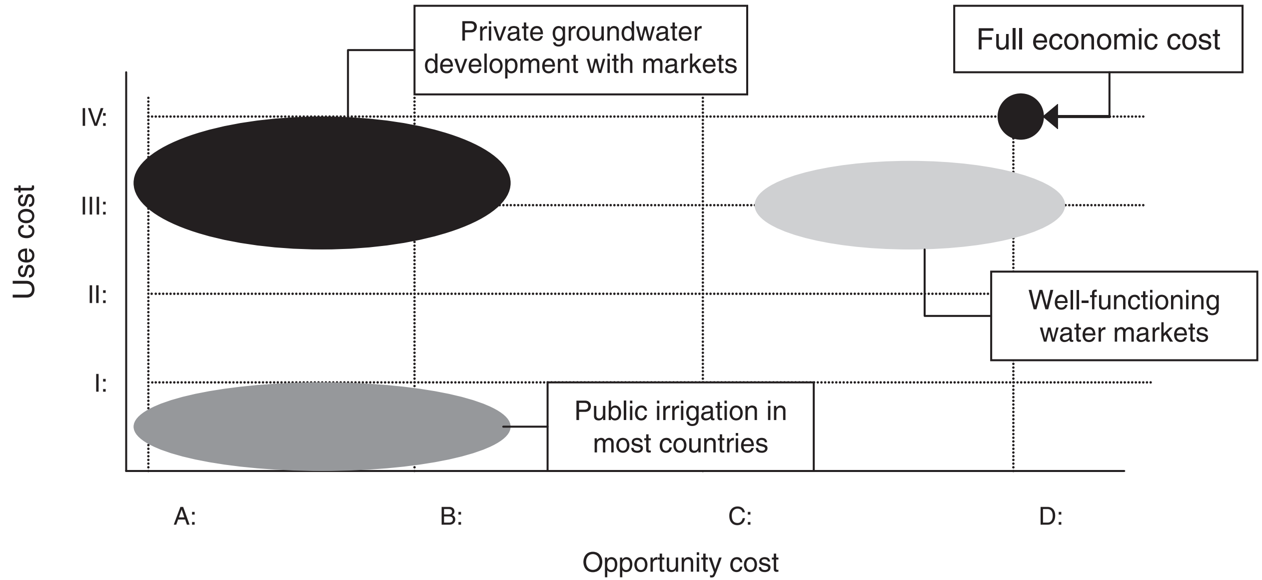


Figure 3.12 Schematic representations of deviation from economic pricing for irrigation

5. EXAMPLES OF GOOD PRACTICE

5.1 Where Water Quality Management is the Principal Challenge – the Ruhr/French Model

Probably the most widely admired water resource management model is that which was developed in the Ruhr Basin in Germany in the early part of the twentieth century, and subsequently adapted on a national scale by France in 1964. The evolution and details of the Ruhr and French experiences have been described elsewhere (Cheret, 1994; Ruhrverband, 1992; Serageldin, 1994). The core elements of this system are:

- management of the basin by a policy-making 'water parliament', comprising all important stakeholders in the basin, supported by a high-quality technical agency; and
- the extensive use of negotiated abstraction fees and pollution charges.

How does the economic value of water come into play in the Ruhr/French type of system? With regard to use costs the answer is simple: the users pay the full financial cost of the infrastructure required to deliver water to them. The way in which the model deals with opportunity costs is more important and less obvious. Abstraction fees are set through a negotiation process. If there is a shortage of water and a potential user without access wants water (or an existing user wants more water), then that user's voice will be heard in the parliament in pushing for higher abstraction prices so as to bring supply and demand into balance. In economic terms this 'next best use' is precisely what is meant by 'opportunity cost'. On the quality dimension (of dominant importance in industrialized countries), the operation of the basin agency is similar: the costs imposed on others in the basin are revealed in both the work of the technical agency and in the course of negotiations, and pollution fees accordingly set in part to take account of these 'externalities'.

On the one hand, then, opportunity costs do come into play in decisions on prices. On the other hand, this expression is indirect and muted by a complex administrative process. As a result, the signals on opportunity cost in such a system do not have the desired specificity and flexibility. While administratively set prices in these systems are affected by opportunity costs, they cannot mimic a market, which, as described in the next section, automatically differentiates by location, quality, season and other complex and changing variables.

5.2 Where Water Scarcity is the Principal Challenge – Experience with Water Markets

In arid areas of the world the foremost water resources management problem has long been that of allocating scarce water among competing uses and users. A wide variety of approaches have been taken, and are taken, to this problem.

In the twentieth century, the most common approach has been a combination of 'first come-first served' (known as the 'prior appropriation doctrine' in the western United States (Worster, 1992)), and the augmentation of supplies through massive investments and allocation of the additional water on political grounds. The problems with such an approach has become manifest throughout the world – the financial costs are enormous, precious water is wasted on low-value activities, while high-value uses cannot secure adequate supplies, and environmental destruction and degradation are the norm (Postel, 1992; Reissner, 1986; Worster, 1992). Recently there has been a surge of interest in the use of water markets as a means of performing this allocation function in an efficient and consensual fashion.

Water markets have a long history both informal, as documented by Shah (1993) for groundwater in Western India, and formal, most notably in Spain (Maass and Anderson, 1978). There have been major developments in Australia (Dudley, 1994), and innovative proposals on the use of markets to solve international water disputes in the Middle East (Fisher, 1994). Most of the attention, however, has been focused on the western United States, where, a wide range of water markets have developed (Saliba and Bush, 1987), with some sophisticated developments (such as the recent development of electronic water markets for the huge Westlands Water District in the Central Valley of California (Zachary, 1996).

In the context of the present discussion of the economic management of water, it is instructive to concentrate on a single, much discussed case, that of the water markets in Chile. The key policy decision in Chile was the separation of land and water rights in 1981 and the simultaneous encouragement of trading of water without restriction. The water market is a brilliant conceptual solution to the enduring problem of reconciling practical and economic management of water. On the one hand, 'common-sense pricing' suggests that the water management unit charges users for the use costs – the investment and operating costs incurred in storing and delivering the water to the user (it is this which is done by users' associations who operate water systems at various levels in Chile).

The problem arises because these financial costs are much lower (often an order of magnitude) than the opportunity cost.[5] The existence of a water market means, however, that behaviour is not driven by the financial cost of the water, but rather by the opportunity cost. If the user values the water less than it is valued by the market, then the user will be induced to sell the water. This is the genius of the water market approach: it ensures that the user will in fact face the appropriate economic incentives, but de-links these incentives from the tariff (which is set on 'common-sense' grounds).

In well-regulated river basins in arid areas of Chile, the water markets function as one would wish: within a particular area water is traded from lower-value uses to higher-value uses. Prices are responsive to both temporary (seasonal) scarcity as well as longer-term scarcity and trading is quite active. Two comments are appropriate here. First, it is evident that no administrative mechanism, even the very good Ruhr and French systems, can mimic water markets in transmitting information on opportunity costs in such a flexible and specific way. Second, it is important to note that water markets are not a simple panacea. The major challenge facing water resources managers in Chile is more effective basin-level management, which will both complement and enhance the workings of the water markets (see Briscoe, 1996).

From the perspective of the economic management of water, a critical issue is the 'breadth' of the water markets, with the dictum being 'the less restrictions there are on water trades, the more the true opportunity cost will come into play'. In Chile, where water can (and is) traded from agriculture to towns, a farmer who owns water rights faces the full opportunity cost of the resource. In many instances (such as the water market of Alicante, and the large market in the Northeast Colorado Water Conservation District) there are specific, and sometimes absolute, prohibitions on the sale of water to non-agricultural users. In such situations, the opportunity costs are obviously truncated, with important resulting distortions in the economic signals.

6. CONCLUSIONS

In this chapter, an attempt was made to develop a framework for thinking about management of water as an economic resource and to assess the policy implications in light of available empirical evidence.

Three principal conclusions emerge from the discussion. First, economic development and environmental sustainability in many countries depend on considering water as a scarce resource, and using economic principles for its management. Second, the challenge is particularly great with respect to irrigated agriculture, which is, simultaneously, the largest user of water in many countries and the sector which is managed (in most places) least like an economic resource. Third, while it is clear that the distance between the 'bad' bottom left-hand corner of Figure 3.4 and the 'good' top right-hand corner is great (particularly for irrigation), there are also examples of good practice which show that change is possible and how it can be effected. Finally, it is important to acknowledge that the idea of 'water as an economic good' is but one of a triad of related ideas which will increasingly shape the way in which societies are organized (and water managed) in the twenty-first century. These ideas are:

- broad based participation by civil society in decisions (including those on water management) which were previously often treated as the province of technocrats alone;
- the hegemony of the market model of development, and the corresponding move to using market-like and market-friendly instruments for managing all elements of the economy (including water); and
- the emergence of the environment as a major focus of concern.

NOTES

1. A comprehensive review of World Bank-financed irrigation schemes (World Bank, 1995) showed that food grains were the predominant crop in 90 per cent of such schemes.
2. Technically speaking, the 'opportunity cost' is defined as the value of the water in its highest value alternative use.
3. Subsidized energy prices for water pumping is widely practiced, from the United States to India. While it has been, or is being, phased out in many countries, in some – India is a prime example – farmers benefit from large subsidies for irrigation pumping.
4. This is confirmed by the fact that, although not formally sanctioned, limited water markets – often involving only neighbours – exist in waribandi-like systems.
5. In the Limari Basin, in Chile, for example, the use cost is about 0.5 cents per cubic metre, and the opportunity cost about US 5 cents per cubic metre.

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4. Appraising flood control investments in the UK

D.W. Pearce and R. Smale

1. INTRODUCTION

The UK government has generally assumed the role of financing flood defence and coastal protection (hereafter just 'flood protection'), but just how much should government spend? For any given budget constraint, appraisal procedures used by the government ministry responsible, the Department for the Environment, Food and Rural Affairs (DEFRA) make use of cost-benefit analysis (CBA) as part of an overall 'scoring and weighting' procedure to assign priority to different schemes. But the size of the budget constraint should itself be determined by a comparison of the social returns to flood protection and the social returns from alternative uses of that money. This chapter focuses primarily on the second question, that is, what is the appropriate size of the flood protection 'budget'? Economic analysis would suggest that if there are higher social returns from expanding the existing budget than the returns on other uses of the money, then flood protection should be expanded. This may amount to changing the 'return period', that is, the probability of a flood in any given time period, so that risks are lowered relative to current design standards and effective current risks.

We argue that:

- on the basis of the appraisal procedures currently used by DEFRA, there are extremely high net benefits from increased flood protection;
- benefit–cost ratios from added expenditure appear to be rising, rather than falling as might be expected;
- existing appraisal procedures understate benefits because of the general omission of categories of benefit not covered by property damage, and because of several conceptual factors.

2. WHY IS THE GOVERNMENT INVOLVED IN FINANCING FLOOD PROTECTION?

To fix some definitions, several forms of expenditure are associated with floods. First, there is *ex ante* expenditure to avoid floods – for example, sea defences, gullies, river dredging and so on. Nearly all of this expenditure is undertaken by central and local government, but there will be a private component, the scale of which is probably unknown.

Second, there is expenditure to mitigate the effects of floods, and this takes the form of *ex ante* insurance premia, and *ex post* compensatory payments. Insurance premiums are paid by those at risk. Compensatory payments are made by the insurance companies and by government. Allowing for the normal profits of insurance companies, premiums should just equal insurance pay-outs, so the two cannot be added together.

Third, there are any clean-up costs that are not included in compensatory payments. These costs will be financed by private individuals, corporations, local and central government.

Historically and currently, most flood protection expenditures are financed by central government in England and Wales. The operating authorities responsible for implementing this expenditure are the Environment Agency, inland and maritime local authorities, and Internal Drainage Boards. Households and firms obviously also undertake some private protection measures as well. Two questions immediately arise:

1. How large should the total of public plus private flood protection expenditure be?
2. How large should the public fraction of total expenditure be?

The answer to the first question is that flood protection expenditure should be incurred up to the point where the expenditure of one extra £1 is just equal to the social benefit secured by spending that £1, that is, marginal cost should equal marginal benefit. Expenditure beyond this point will incur costs greater than the extra benefits received, signalling a waste of resources. Expenditure below this point suggests that benefits greater than costs could be secured by increasing expenditure. While simply stated, there are formidable problems of implementing this rule.

First, the benefits are probabilistic (and stochastic) in nature: they relate to avoided flood damage arising from an event, or set of events, the timing and magnitude of which are not known. There are various ways in which this probabilistic feature of flood damage can be taken into account. A simple rule would be to take the *expected value* of the damage, that is, damage multiplied by the probability that damage will occur. It is well known that the expected value does not account for *risk aversion*, so that a better rule is based on *expected utility*. This is discussed later.

Second, identifying the optimum requires knowledge of benefit and cost functions, that is, the relationship between expenditure and damage has to be identified across a range of expenditures. Invariably, only very limited knowledge exists about damage and cost functions. Indeed, as a general point, even 'point estimates' of damages are hard to come by since little research has been done on the various components of total damage associated with floods.

Even when some approximate answer to the first question can be provided, the second question remains. There will be a desirable balance between the public and private financing of the optimal level of flood protection. The nature of flood protection favours its central provision by government. The reason for this is that, even if the damage from a flood is to private goods (housing and so on), flood protection is a public good. A public good is one which, when supplied to one person, is automatically supplied to others, and where it is difficult, if not impossible, to exclude the others from the benefits of the good ('joint consumption and non-exclusion'). An example might be clean air. In practice, many goods have joint consumption features (for example, a beach) up to some point where the good becomes congested and overall well-being is impaired by adding new beneficiaries, and have varying degrees of excludability. No one private individual will finance flood protection because of the inability to appropriate the benefits that all other individuals would secure. Also, with public goods individual beneficiaries are likely to understate their willingness to pay for the public good, thus resulting in under-supply. This is the traditional argument for the public provision of public goods. It is not an inviolable argument, but tends to hold in the flood defence case. If it is accepted that flood defence has a very large public goods content, then control expenditures would be very largely public expenditures. However, an issue remains of the balance between preventive expenditure and mitigation expenditure. This may not be obvious. Essentially, however, governments should not spend extra money on flood defence if that expenditure exceeds the (expected) damage of the flood events avoided by more flood control. In that context, it would be better to 'allow' some damage to occur and for mitigation costs to be met via insurance premiums and tort liability. The amount of allowable damage is the optimal residual damage. The public–private split in protection expenditures should be influenced by the efficiency with which the relevant sectors can manage risks. From a social standpoint, the aim should be to minimize the costs of protection, regardless of who finances those costs. It has been suggested that, in the USA, insurance is cheaper than tort liability because the latter has very high transaction costs (Freeman and Kunreuther, 1997). As such, a larger fraction of collected funds goes to compensation and restoration under insurance than under civil liability actions.

There are other factors that are relevant to the share of protection costs that should be borne by government. First, some floods are 'natural events' and arise from the stochastic nature of the weather and natural events. The 'cause' is therefore an 'act of God'. In many cases, however, the cause may be some human act to which some degree of negligence attaches. Such acts may actually cause the event, for example, by making a 'normal' weather event into one associated with 'abnormal' damage, or by exacerbating an extreme event. Broadly, there will be natural and human-induced events. Now suppose the source of the flood risk is housing development in a flood plain. Here there may be a good case for negligence-based liability across several parties:

- the local government for giving planning permission when it should have known the relevant risks;
- the developer for not taking those risks on board when selling properties; and
- the house owner or his/her agent (for example, professional surveyors) for failing to appraise the risk.

Where negligence is involved, tort law applies and insurance companies would not be expected to meet claims (or would be expected to contest such claims) and government also should not pay. The principle that property developers in the UK should contribute to the costs of flood defence is already embodied in Planning Guidance Note 25 (DETR, 2001).

Second, if an individual or agency can be identified as the cause of flood damage, then liability is ascribed to that individual or agent. In principle, we would expect someone who causes flood damage to be responsible for restoration of the pre-damage situation. Another way of stating this situation is that liability implies property rights. In this case, the property rights to a 'risk free' situation reside with the person suffering the consequences of the flood. Liability may be *strict liability*, in which case there is a presumption that the party causing the risk is liable, or liability may be *negligence based*, in which case the suffering party will have to demonstrate both cause and negligent behaviour, that is, that the causal agent failed to adopt an appropriate duty of care. It can happen the government, acting on behalf of society, is a causal agent. For example, water run-off from roads may legitimately be regarded as a flood-source that is the responsibility of central government (trunk roads) or local government (other roads). Watercourse modification also appears as a likely cause of increased flooding. Hence those government sources should pay for the restoration of the pre-damage situation, including compensation for disruption and distress. In contrast, flood risks brought about by agricultural drainage would be the liability of the farmer and not of any public agency. However, as recent events have shown, difficult ascription of liability may arise if the agency responsible for drainage beyond the farmer's fields has failed to maintain drains 'properly'. Climate change represents a human-induced cause of increased flooding but since the causal agents are worldwide and public, government would also have the responsibility of dealing with floods due to this factor.

These examples are sufficient to show that determining the 'correct' amount of public flood protection expenditure is far from straightforward. The presumption is that the public good nature of protection means that flood protection expenditure should be public. A cost–benefit rule can determine how large the expenditure should be, with residual damages being offset by insurance premiums. However, the ascription of liability affects what it is legitimate for governments to spend. If government is liable it should pay, but if individuals are liable then they should pay. It is possible to say that, at a minimum, the public sector proportion will comprise:

- all expenditures where the private and public benefits to non-liable parties exceed the costs; and
- all expenditures arising from government liability for floods.

3. THE ARGUMENT FOR MORE RELIANCE ON INSURANCE

Rather than relying on government to finance more flood protection, it could be argued that those at risk should bear more financial responsibility for mitigating flood damage. If so, the need for CBA of public expenditure is minimised. Civil action involves those who cause damage paying for it after the risk event has occurred. Such actions do not therefore directly finance *preventive* activities, but do so indirectly by giving an incentive to potentially liable parties to avoid *ex post* liability through preventive measures. Hence liability 'finances' some prevention of risk through private expenditures. Civil action is therefore a sub-component of private expenditure.

Individuals may spend money to prevent risks in the sense of mitigating a risk should it occur. Individuals typically cannot prevent flood risks on their own – it requires the actions of a community to do that. But they can engage in expenditures to mitigate risks because of the damage those risks will do to themselves. A major form of expenditure by private individuals is insurance. Again, insurance will not directly reduce the risks of floods, but may do so indirectly so long as insurers can influence the behaviour of the insured, for example, by varying premiums across risk classes. A requirement for a property developer to take out insurance against flood-plain risks might, for example, deter the developer from building in flood plains. Hence insurance can be an indirect means of financing some flood protection in the sense of damage avoiding activities.

It has been argued that *more* private insurance, and correspondingly *less* public expenditure, is advisable in the case of certain risks (Freeman and Kunreuther, 1997). For this argument to be valid in the current context, flood risks must be 'insurable'. In turn, insurability requires certain conditions to be met. These are discussed briefly.

3.1 Risk Pooling

The loss must be capable of being *pooled*, that is, the risk must be shared out across a significant number of people. The bigger the group of people over which the risk is pooled, the better. For risk pooling to work it is also essential that the risks faced by any one individual are uncorrelated with risks faced by other individuals, otherwise a significant part of the insured group could be making simultaneous claims which could not be met by the revenues from premiums. This is a significant issue. If the good being protected (conserved) has public good properties, then its loss for any one person will be a loss for other people too. The basic requirement for insurance breaks down. In the flood risk context this suggests that while private property damage risk will be insurable, virtually all environmental risk will be uninsurable.

3.2 Clear and Definable Loss

Any loss must be reasonably definable, measurable and must occur within a clearly defined period of time. If risks are 'fuzzy', the insurance company cannot assess the likelihood that it will have to pay out, and/or the amount it has to pay out, and hence cannot know its own profit situation. The same goes for unquantified risks and risks that may stretch over some undefined period of time. Losses must also be verifiable – there must be an accepted standard of evidence such that the insured can prove loss and which the insurer can verify. Floods may damage, for instance, biological diversity, but is biodiversity insurable? It would be difficult to prove what the consequences of biodiversity loss would be. In practice, the issue is not one of insurance or no insurance, but one of the scale of the premium when there is lack of clarity about the risk. Insurers tend to be both risk-averse and averse to 'ambiguity' of risk, that is, its lack of definition. Various mechanisms exist to share the risk between the insurer and the insured when risk is not precisely quantifiable, and there may be some sharing with governments as well.

3.3 Loss Frequency

Nearly all insurance works on the basis that there is prior information that acts as a source of data to calculate premiums. Theft is an everyday occurrence and hence it is relatively straightforward to determine the premium for theft insurance, and relatively easy to modify the premiums as experience changes. Weather is a further insurable event, but climate, for example, may not be. Climate change can be thought of as the long-run trend of weather events. While this may make it seem that climate should be insurable because there is a lengthy history of 'weather', climate change may combine features of temperature change and levels of temperature for which there is no historical precedent, or, at least, no precedent for which information is available. The long-run incidence of floods is likely to be very much affected by climate change, but if climate change is uninsurable, then the related floods may themselves be uninsurable as well.

3.4 No Moral Hazard

The insurer needs to be able to predict the behaviour of the insured and this underlines the relevance of past data on frequency and type of event. But if the insured change their behaviour once they are insured, and if this behaviour cannot be predicted, then premiums could be set too high or too low. If the behaviour of the insured changes so that they actually increase the frequency of adverse events, the premiums will be too low and the insurer will lose money. If this phenomenon is significant then the insurer may withdraw from the market. Significant moral hazard would therefore reduce the chance that insurance will succeed. One way of overcoming moral hazard is to devise mechanisms for monitoring the behaviour of the insured. Moral hazard is a problem of *asymmetric information*, the insurer has one lot of information, but the insured has information to which the insurer is not party. Changing asymmetric to symmetric information, whereby the insurer knows what the insured is doing, is thus the means for overcoming moral hazard. Moral hazard therefore provides the insurer with an incentive to secure the missing information. Once the information is secured, it can be used to adjust the premium upwards to allow for any anticipated increase in risky behaviour. Other ways of dealing with moral hazard are well known. The insurance contract may involve a *deductible*, a sum that the insured pays towards the cost of the compensation if the adverse event occurs. In this way, compensation costs are shared, giving the insured an incentive not to relax the degree of care and caution about the adverse event. Deductibles tend to take the form of absolute sums, for example, the first £500 of loss is payable by the insured. *Co-insurance* is similar in nature and involves the insured agreeing to pay a declared percentage of the costs of the loss. *Upper limits* act in the same way – that is, the insurer agrees to pay sums up to an upper limit, anything beyond being payable by the insured. For those in private property affected by flood risks, it is arguable that they behave in a less risk-averse way, because they have the protection of insurance. It was noted above that individuals have little opportunity to control flood risks, although they can control, to some extent, the scale of the damage associated with the risk.

3.5 No Adverse Selection

Adverse selection is another phenomenon arising in contexts of asymmetric information. In the example of insurance premiums set out earlier, the implicit assumption was that each individual was equally likely to suffer the adverse event. But some people may be more likely than others to suffer the event and they may well be people (or firms) who will suffer large losses. This would not matter, in terms of setting the premiums, if the insurer knows who this group is, but it will matter if they do not have this information. Since insurers are risk averse, they will tend to set premiums very high in the absence of the information, making insurance unattractive to all the insured (since the insurer does not have the information to discriminate). The market may then fail. Again, the solution is to acquire the information and adjust the premiums according to risk groups. This is a common practice in insurance. An important procedure for acquiring the relevant information is to *audit* the insured, either directly or by requiring that the insured submit to some independent form of audit. By and large, flood risks can (with the advent of geographical information systems) be audited and insurers can adjust premiums according to those risks. There appears to be therefore no case here for insurers not to be operating in this market.

3.6 Enforceability of Contract

Finally, insurance will not work unless the insured pays the premium and the insurer honours the obligation to pay out for damages in the event they occur and the losses are genuinely suffered. Some *legal force* relating to an insurance contract must therefore exist and there is also likely to be a legal basis for damage *liability* – the right of an injured party to claim damages from the individual or agent causing the damage. This may seem rather obvious, but it affects the applicability of insurance to some environmental risks.

The argument that increased insurance, rather than increased public expenditure, should be relied upon is seen to be weak since:

- insurance generally does not affect flood prevention;
- insurance cannot apply to public goods damage;
- insurance may actually increase damage through moral hazard; and
- unless regulated to do so, insurance cover will tend to be withdrawn when events become uninsurable, for example, because of the unknown consequences of climate change.

4. THE MEASUREMENT OF FLOOD DAMAGE

The damage done by a flood comprises a number of elements:

- temporary or permanent damage to household, commercial and industrial buildings, plus contents;
- temporary or permanent damage to infrastructure (for example, roads, telecommunications and so on)
- damage to environmental assets;
- damage to cultural assets (heritage);
- human damage: morbidity, trauma, distress;
- human damage: relocation and disruption costs;
- output-related damage: losses in productivity due to work days lost;
- clean-up costs: costs of emergency services, restoration costs; and
- any loss of well-being by those who are not directly affected.

Determining the 'right' way to measure actual or potential flood damage is complex. There are three broad notions of damage:

- The first might be termed the '*public trust*' doctrine. It measures damage by whatever it costs to restore the flood-damaged asset, or set of assets, to their pre-damage situation. This notion of 'damage' has no clear foundation in economics, but is present in some legal doctrine (see below).
- The second measure rests on the economic concept of damage as whatever individuals collectively would be willing to pay to avoid the flood damage. This measure of damage is prima facie the correct one

when those at risks from floods do *not* hold the property rights to an absence of risk. Hence it would be the correct concept in a context where the risk was voluntarily assumed or where the risk is 'natural'.

- The third measure is also economic and is the amount of money that would be required in compensation by the individual at risk so that he is as well off after the risk as he was before. This notion of compensation most clearly applies when those at risk do hold the property rights to risk protection.

There are some crucial differences between these three measures of 'damage'. To anticipate the evidence, as far as the economic concepts are concerned, the 'compensating variation' (willingness to pay) measure could be several times as large as the 'equivalent variation' (willingness to accept compensation) measure, and neither need have much relationship to the legal notion of cost of restoration. In determining the amount that should be spent on flood control, therefore, the choice of damage concept could matter a great deal.

The public trust doctrine has emerged in the USA in the context of damages to natural assets. The doctrine basically states that citizens have a 'right' to the state of the environment that existed before some event changed that state. The context is one of human-induced damage now or in the past (for example, land contamination). Natural resources are regarded as being held in trust by the state and federal governments of the USA for existing and future generations. The doctrine implies that damage to natural resources must be negated, that is, the natural environment must be restored to its pre-damage situation. The public trust doctrine also requires that those who act as trustees can use any money recovered from actions against liable parties only for enhancing or creating natural resources (Jones, 1996). Monetary compensation, actual or hypothetical, would then have no role to play because, of itself, compensation does not restore the 'status quo'. As Jones states: 'public trustees do not have the authority to make individuals whole by providing such recoveries [money recovered from liable parties] directly to individuals; rather, trustees are allowed to spend their recoveries only on enhancing or creating natural resources' (Jones, 1996, p. 6).

The doctrine contrasts with the standard economic view embodied in the second and third notion of damage. To the economist, the status quo relates to the *well-being* of the individual. If, in a post-damage situation, an individual is compensated so as to be as well off ('made whole' in Jones's language) – in his or her own judgement – as they were in the pre-damage situation, then compensation is efficient and just. So long as the individual regards the compensation as a substitute for the damage done to the environment, it is not necessary for the damage itself to be 'undone'. The public trust doctrine proceeds quite differently. It does not require that the status quo be measured in terms of the individual's well-being, but in terms of the state of the natural environment. Hence any damages are measured by the costs of restoration and those costs can legitimately be recovered from the parties responsible for the damage.

Thus there appear to be three 'contenders' for the conceptual basis of measuring damages from floods. The first is legal and the other two are economic.

The first, the *public trust doctrine*, says that damage is measured by whatever it costs to restore the pre-flood situation in terms of the state of the environment. Extended to non-environmental assets, it would be the equivalent of the insurance principle of restoring assets that are, say, stolen from insured premises. The insurance settlement does not contain any element of compensation for the distress of being burgled.

The second rests on the notion that the agent at risk does not have the property rights to a risk-free situation. It is then the *willingness to pay* (WTP) to avoid the risk. The basis for the measure is the well-being of the individual so that all losses of well-being are relevant. While it is tempting to think this WTP measure will be greater than the expenditure needed to restore assets, there need to be no particular link. Willingness to pay will be influenced by the assets at risk and by the unwanted experience of floods. On the other hand, theoretically WTP for the assets could be less or more than what it costs to restore them.

The third measure rests on the notion that the agent at risk does have the property rights. In this case, it is the minimum willingness to accept (WTA) compensation by the agent at risk that matters. As will be seen, there is evidence to suggest that WTA is greater than WTP in some contexts. The link between WTA and restoration costs is again indeterminate. WTA will comprise both a sum relating to the lost assets and a sum for lost well-being unrelated to assets, but WTA for the assets lost could be greater or less than the cost of restoration.

There is only limited and debatable support for the public trust notion in economics. Both notions of WTP and WTA are economically integral to CBA.

4.1 Why Property Benefits Understate True Social Benefits

Benefit–cost (B/C) ratios based on property damage alone may understate 'true' benefits for a number of reasons. This section investigates the following sources of understatement:

- infrastructure damage needs to be included;
- loss of environmental assets need to be included;

- distress, trauma and morbidity is known to be a real cost of floods (Tapsell et al., 1999);
- non-use values need to be included; and
- damages will rise with climate change.

These damages should be added to property damage to secure a better reflection of the true damage. The effect would be to raise B/C ratios. Other possible sources of understatement arise from:

- the use of expected values rather than expected utility values for damage; or
- the possible role of willingness to accept compensation rather than willingness to pay or restoration costs.

It is also possible for floods to have benefits. However, the only widely countenanced one appears to be saltwater intrusion creating saltwater marshes, which have wildlife benefits. The DEFRA guidance on CBA (MAFF, 1999; 2000a; 2000b) acknowledges all of these additional benefits. To date, there appear to be no studies, which measure distress/morbidity in monetary terms.

Damages in flood control cost–benefit studies are estimated using expected values, that is, the estimated damage (or actual damage done) multiplied by the probability of the flood event. Underlying this assumption is risk neutrality. This means that individuals (or government as the representative of society) are indifferent between two probability distributions of flood damage each with the same expected value. If we now suppose that the two distributions have the same expected value, but different ranges of outcomes (measured by dispersion), then the expected value approach will again be indifferent between the two distributions. But the distribution with the larger dispersion may encompass floods with extremely large damages. It seems unlikely that individuals, or government, would be indifferent between two distributions of this kind. The likelihood is that a larger 'weight' would be attached to the severe damage possibility – a loss of say £100 million would be valued at more than a gain of £100 million. *Expected utility* theory is an attempt to account for such variable weights. Without deriving the result, it can be shown that the effect of reformulating the investment decision in expected utility terms produces a revised cost–benefit formula. Instead of

$$NSB = \sum_t (B_t - C_t)/(1 + s)^t \tag{4.1}$$

where NSB = net social benefits, B = benefits, C = costs, t = time, and s is the social discount rate, we would have:

$$NSB = \sum_t (B_t - C_t - k_t)/(1 + s)^t \tag{4.2}$$

where k is the 'cost of risk bearing'. In this formulation, the expected value of net benefits (B − C) is adjusted downwards if the project involves the probability of 'large' losses. The opposite would be the case if the project protects against large events and a risk adjustment involves in this case an addition to (B − C). The expected utility argument is relevant to defence design standards. Intuitively, if extra defence protects against a low probability but high damage event, allowance needs to be made for the size of the high damage. What expected utility theory suggests is that the high damage event will have a larger influence on the decision than it would if expected value theory alone was applied. Arguments against introducing this risk adjustment tend to focus on the fact that large risks can often be distributed across large populations, so that the 'per capita' risk is small or negligible. This would be relevant to flood defence in so far as government is the source of finance, that is, large risks are distributed across many millions of taxpayers. Nonetheless, the intuition behind the expected utility argument can be appreciated.

While theory predicts that WTA measures of damage should diverge only marginally from WTP measures, in practice divergence can be substantial. The evidence comes mainly from contingent valuation studies in which WTP and WTA questions are asked. Horowitz and McConnell (2002) review 45 studies in which WTA and WTP estimates are derived. The average ratio of WTA to WTP is 7, that is, WTA is on average 7 times WTP for the same good. This ratio relates to all kinds of goods. For public non-market goods, the ratio is even higher (10), which is also the ratio for health and safety. They reject the view that such differences are statistical artefacts. They also find that the ratio varies directly with the 'distance' that a good is from being an ordinary 'private' good. In other words, the further away the good is from being a private good, the higher the disparity between WTA and WTP. Even for private goods, however, the WTA/WTP ratio is a little under 3.

The implications of the disparity for public policy are potentially formidable, but it is less clear if they affect the CBA of flood protection. Recall that WTA is relevant when those at risk have property rights, in this case a right to protection from floods. Assume this 'right' is to total protection. Hence the WTP of those at risk to secure increased flood protection is not relevant. It is their WTA compensation to forego improved flood protection that matters. Using property prices to measure avoided damage and residual damage is then incorrect as the basis for CBA. The reason for this is that property prices are market prices and market prices are based on WTP.

As far as understatement of damages is concerned, then, the damage from floods, and hence the benefits from flood protection, are understated if the focus is solely on property damage. First, there are other sources of loss of well-being attached to floods and these need to be valued using available economic valuation techniques. Second, it is possible that risk aversion is not being fully accounted for through the use of expected values for property loss. Third, there is some basis for arguing that WTA rather than WTP is relevant to at least some flood impacts, although we found that the prevailing legal presumption is against this.

4.2 Caveats

There are several arguments that might be raised against increasing flood protection expenditure. First, as far as risks to private property, agricultural land and commercial/industrial property are concerned, it is arguable that the risks to at least some property and land are already internalized. Essentially, the price of property reflects the (discounted) sum of the individual values attached to the positive and negative characteristics of that property. Thus, taking a house as an example, the price of the house reflects the 'price' of the size of the property, nearness to features such as transport facilities, shops, schools and so on, and associated land, amenity, disamenity such as noise and air pollution. In principle, there is no reason to exempt flooding risks from these characteristics. Other things being equal, a house in a flood risk area should command a lower price than one in a lower or zero risk area. The house price differential (the 'hedonic' price) therefore compensates the owner for the incremental risks. Put another way, the risk is 'internalized' in the house price.

Economic analysis tells us that such internalized risks are not relevant to the computation of social cost and should not therefore be regarded as a cost that should be offset by social expenditure. In the various government documents there appears to be little or no appreciation of this possibility. Direct evidence to support the view that property prices already internalize risks appears not to be available. Somewhat surprisingly, hedonic property models seem to have accounted for all kinds of other effects, but not flood risk. Thus the evidence that risks might be internalized to some extent is indirect. Other hazards do appear to affect house prices – for example, noise, air pollution, radon risks.

Is there an inconsistency? The issue is complex. Most policy contexts for hedonic models concern improvements to environmental quality. It is then correct to argue that increases in house prices measure the welfare gain from those policies. In the flood protection context, the benefits could similarly be measured by the increase in house prices brought on by better protection. We suspect that this may be the underlying rationale for the property value approach in the Ministry of Agriculture, Food and Fisheries (MAFF)/DEFRA appraisal guidance. But what the guidance is concerned with is the estimate of physical damage done by a flood and this need not be the same as the welfare gain from protection. There are at least two reasons for this. First, damage to property does not capture all the welfare losses, and second, the value of incremental gain need not be the same as the value of equal incremental loss.[1]

But there might be an additional argument for ignoring the 'internalization' argument. For risks to be internalized, householders must perceive the risks correctly. If they understate the risks then, at best, only part of the risk will be capitalized into house prices. If they overstate the risk, there will be too much adjustment of house prices. Evidence on flood-risk perception suggests that people typically understate risks. The literature on risk perception dates back some 40 years when Kates (1962) interviewed US householders in areas of flood risk. Various reasons were given for not taking account of flood risks: some did not believe it would happen, some thought they were protected (when they were not), most thought that if a flood had happened it would not happen again for a long while, and many were 'in denial', arguing that it would not happen to them, it was an Act of God anyway, that there never had been any real floods anyway, and that past floods were 'freak' accidents. More generally, people overstate tiny risks and understate bigger risks, and few people understand the nature of a random event (see the essays in Slovic, 2000 and Viscusi, 1998). Tunstall and Tapsell (1997) report a risk perception survey for 13 case studies in the UK. They found that:

- most people in the study areas were aware of past flooding and most knew of a risk of flooding when they moved to the area;
- but those who knew of a risk tended to underestimate its scale, both in terms of the likelihood of the event recurring and the severity of the flood;
- those who knew of the risk also had a degree of over-confidence in river managers to control floods;
- many did not know of the risk, for example, surveyors' searches failing to detect such risks; and
- among those who knew of risks were a sizeable proportion who chose to accept the risks of being near a river because of its amenity value.

The evidence therefore lends support both to those who believe property prices will not internalize risks (lack of perception of risk, lack of information about a risk) and those who might argue for some internalization (amenity benefits outweigh flood risks).

Second, the MAFF/DEFRA guidance rightly acknowledges that some permanent or persistent seasonal flooding may be beneficial to wildlife. Floodplain flooding could, for example, mean the abandonment of arable areas and conversion to wet grasslands. Other floods are harmful to wildlife: recent floods in the Ouse washes have seriously affected breeding seasons for resident birds. Economic valuation studies for wetland areas are fairly numerous but, obviously, any use of such values relates to site-specific damages. It would be difficult to justify incorporating a nation-wide figure into estimates of annual average damage.

5. EXAMPLE

On the basis of CBA, more flood protection would be justified if two conditions are met:

- benefits exceed costs; and
- the resulting benefit–cost ratio exceeds that which could be obtained if the funds required to increase flood protection were spent elsewhere.

Halcrow et al. (2000) set out the measurement of assets at risk from flooding and coastal erosion. The procedure is to focus on property as buildings, their associated land, and agricultural land. Potential damage to property is estimated on the basis of flood characteristics and maps which detail the incidence of property in flood-prone areas. Table 4.1 gives the market value of the total *stock* of assets at risk in England and Wales.

Table 4.1 Total assets at risk in England and Wales (£ billion)

Assets at risk	Market value
Buildings/land	215.6
Agricultural land	7.1
Total	222.7

Note: Assumed to be at 2001 prices, but no indication of the year prices is given in Halcrow et al. (2000).

Information in Halcrow et al. (2000) can be used to derive benefit–cost ratios for actual and increasing flood defence expenditures. The data reported are for average annual damages associated with different standards of service (SOS), and costs of flood protection. These data enable average and marginal benefit–cost ratios to be calculated. As far as the debate over whether or not increased expenditure is justified, it is the marginal ratios that matter, that is, the incremental benefits that can be obtained for an increased expenditure on flood defence. Nonetheless, the average B/C ratios are also of interest. The Halcrow et al. (2000) data relate to damages in England and Wales, but the cost data relate to England only. Accordingly, we have extracted the damage data for England only. The damages in question are for property, agricultural land and traffic delays. Property damages dominate. The damages are reported as annual average damage (AAD) rather than net present values. As long as damages and costs can be thought of as annualized flows, then a ratio of annual benefits to annual costs gives the same result as the ratio of the present value of benefits and costs, hence nothing is gained by discounting (assuming that the distribution of costs and benefits through time is valued in the same way). Table 4.2 sets out the basic data and reports the calculated B/C ratios.

Table 4.2 casts light on the expected costs and benefits of raising indicative standards to the 1/100 return period. Moving to the 1/100 standard of service would secure substantial incremental benefits for very modest increases in costs. Moreover, since incremental B/C ratios are rising, it suggests that, as far as *benefits* are concerned, there is no reason to suppose that incremental benefits from moving to an even stricter standard of say a return period of once every 200 years would be any less than those of meeting current target standards.

The results presented in Table 4.2 are surprising. Benefit–cost ratios of around 7 are extremely high in terms of the experience of public policy cost–benefit appraisal. An incremental B/C ratio of around 17 suggests that it is even more beneficial to invest in higher flood defence standards. Typically, one would expect marginal B/C ratios to be declining as more investment occurs. If past expenditures had been ranked according to their benefit–cost priority, then the 'best' schemes would be done first and the least attractive would be considered last. Hence the marginal B/C ratio should decline and increasing ratios of 17 might be treated with some suspicion. However, there are various reasons why we consider these ratios to be realistic.

First, we note that the high ratios are entirely consistent with the B/C ratios reported in a National Audit Office report (NAO, 2001) since (a) the ratios for individual schemes are very high, and (b) the ratios rise over time. The NAO (2001) examined 108 schemes requesting Environment Agency

Table 4.2 Benefit–cost ratios for flood protection in England

Scenario	Source	Annual costs (expenditures)	Annual damage costs avoided	Average B/C	Incremental B/C
Do nothing	Halcrow, Table 3.2	0	0	0	—
Business as usual: maintain current protection expenditures	Halcrow, para. 3.5.2	0.24	1.80	7.5	—
Maintain current design standards (return period 1/40) requiring additional investments	Halcrow, Table 3.3a Halcrow, Table 3.4a	0.36	2.55	7.1	6.2*
New standards: return period 1/100	Halcrow, Table 3.4a Halcrow, Table 3.4b	0.39	3.05	7.8	17.3**

Notes:
See Halcrow, para. 3.5.2. However, this paragraph refers to paragraph 3.3.3. as the source of the quoted benefit–cost ratio of 7. Paragraph 3.3.3 refers to the expenditure to maintain current standards (£0.361 billion p.a.), not actual expenditure (£0.24 billion p.a.), and the ratio is 7.5 not 7. The text is confusing, but we have taken it to mean that a benefit–cost ratio of 7.5 would be achieved 'instantly', but this would not be sustained, because of the need to raise expenditures to £0.361 billion p.a.
* (2.55–1.80)/(0.36–0.24).
** (3.05–2.5)/(0.39–0.36).

Source: Based on Halcrow et al. (2000).

grant aid. The results were that all schemes had B/C ratios in excess of unity (1) and average ratios were:

Year	*B/C ratio*
• 1997–98	7
• 1998–99	19
• 1999–2000	20

Second, the nature of new flood defence schemes is such that the additional costs of those schemes are low. Existing defences are being improved rather than constructed from scratch. This is consistent with the cost data reported in Halcrow et al. (2000). Hence marginal B/C ratios are likely to rise. Moreover, as incomes per capita rise over time, the level of assets at risk is likely to rise both in 'physical' terms and in terms of relative prices. In some cases, for example, certain house contents, we would expect real prices to decline, but even here there is likely to be an offsetting effect due to ownership of more durables. House prices do show a rising relative price. Finally, there could be a 'learning effect' whereby the later the CBA, the more likely it is to be based on improved information about risks and assets at risk.

Halcrow et al. (2000) suggest that maintaining current standards secures B/C ratios of 10 for fluvial flood defence schemes, 9 for tidal/sea flood defences, and under unity (1) for coastal protection. Some other evidence reporting low B/C ratios comes from Dunderdale and Morris (n.d), who evaluate the benefits of river maintenance for agricultural output, measured by crop productivity, grazing season length and animal stocking rates. The results shown below are for ratios in economic prices.[2]

- four have B/C ratios > 1.0;
- two have B/C ratios = 1.0; and
- nine have B/C ratios < 1.0.

The B/C ratios for measures to protect agricultural land are low and, on balance, such schemes would fail a benefit–cost test. What is not known, however, is the extent to which the schemes studied would have brought benefits other than the protection of crop and grazing land. Overall, this evidence does not invalidate the presumption of high B/C ratios for defences that protect significant amounts of property.

While the B/C ratio for incremental flood protection is extremely high, it could be the case that it is no higher than with alternative investments. If the capital budget is fixed for some reason, then the relevant B/C ratio that flood protection schemes must beat is given by the marginal ranked project that is just included within the budget. Put another way, projects should be ranked by their B/C ratios and projects should be accepted in order of ranking until the budget is just exhausted. Whatever the B/C ratio is on the 'last project', this is the cut-off B/C ratio. Note that this ratio cannot be predetermined: it depends on the budget and the array of projects to be appraised. The idea that there is some predetermined cut-off rate is not therefore correct, even in a context where there is a budget constraint. The issue at stake here is whether flood protection is under or over-funded. Taking some predetermined B/C rate of, say, 2 or 3 or 4 and saying that all projects above this rate will be funded and all those below it will not, implies that alternative uses of government funds will secure even higher B/C ratios.

The UK has tended not to publish the B/C ratios for public investments, even though they may have been calculated within the required project appraisals. Clearly, the political and economic constraints on taxation and public borrowing mean that funding has to be prioritized to projects with the most favourable B/C ratios. The public administration sector with the strongest record of publication is transport, and it is useful to know that this sector is also concerned with infrastructure. Here, major projects with B/C ratios in excess of unity, and even up to 2, have failed to attract funding. The 'Crossrail' project in London is an example.[3] However, our review of government studies found no instances in other sectors of B/C ratios as high as those we have noted for flood defence. This suggests that the case for additional flood defence expenditure is likely to be high relative to comparable project investments in other sectors.

6. CONCLUSIONS

In this chapter, we have argued that flood control has significant public good features, which make it the proper province of government policy rather than private insurance. These public good features imply that the size of flood control expenditures should be determined by a proper comparison of social costs and benefits. However, current appraisal in the UK focuses virtually exclusively on damage to property, ignoring the many other social costs associated with floods, including the anxiety and stress associated with them. Unfortunately, data on these non-property costs are scarce. Focusing solely on property damage reveals an initially surprising result when actual schemes are investigated. Not only are benefits substantially in excess of costs, but the benefit–cost ratio appears to be increasing over time and with increasing levels of protection. We have offered several reasons for this outcome, including decreasing costs of incremental schemes, and rising property values. We do not rule out the possibility of a faulty underlying methodology, but the balance of evidence suggests the rising incremental benefit-cost ratios reflect real underlying phenomena.

NOTES

1. This latter point is essentially the same as the issue of the disparity between WTP and WTA.
2. Economic prices being the right prices for national appraisals. Economic prices exclude transfer payments like subsidies, which do not reflect real economic losses. Financial prices would include subsidies.
3. Crossrail, as its name implies, refers to a proposed plan to provide further rail links across London.

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5. Cost–benefit analysis and flood control policy in The Netherlands

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1. FLOOD CONTROL AND SAFETY POLICY IN THE NETHERLANDS

For centuries the Dutch have reclaimed and drained land and raised dikes to protect themselves against flooding. Protection against flooding has always been the government's primary water policy objective in a country of which approximately two-thirds is situated below sea level. Dikes have always been the most important means to achieve this. Over the years, 53 different dike enclosures have been constructed for those areas located below sea level. Each of these enclosures has a different safety level, expressed in an acceptable probability at which dikes and other water retaining structures along the coast, the rivers Rhine and Meuse and the IJsselmeer district have to hold, that is, not breach and prevent flooding. These safety levels have legal status and range from once every 1250 years to once every 10 000 years (Figure 5.1).

As can be seen in Figure 5.1, the safety levels are highest in the western part of The Netherlands and become gradually lower when moving from west to east. The safety levels are based upon the probability of flooding and its consequences for both people and the material damage caused by flooding to buildings and economic losses in the area vulnerable to flooding. The western parts of The Netherlands are the most densely populated areas in the country, with large cities like Rotterdam, The Hague and Amsterdam, and, furthermore, where most of the country's economic production value is generated (Figure 5.2). Hence the relatively higher safety levels in this part of the country.[2]

The Water Act (Wet op de Waterkering), which was adopted in 1996 and defines the safety levels, obliges the responsible water managing authorities to report every five years about the extent to which the legal safety levels are met.

More recently, especially after the high waters in 1993 and 1995, government policy is also focusing on alternative ways to maintain existing flood

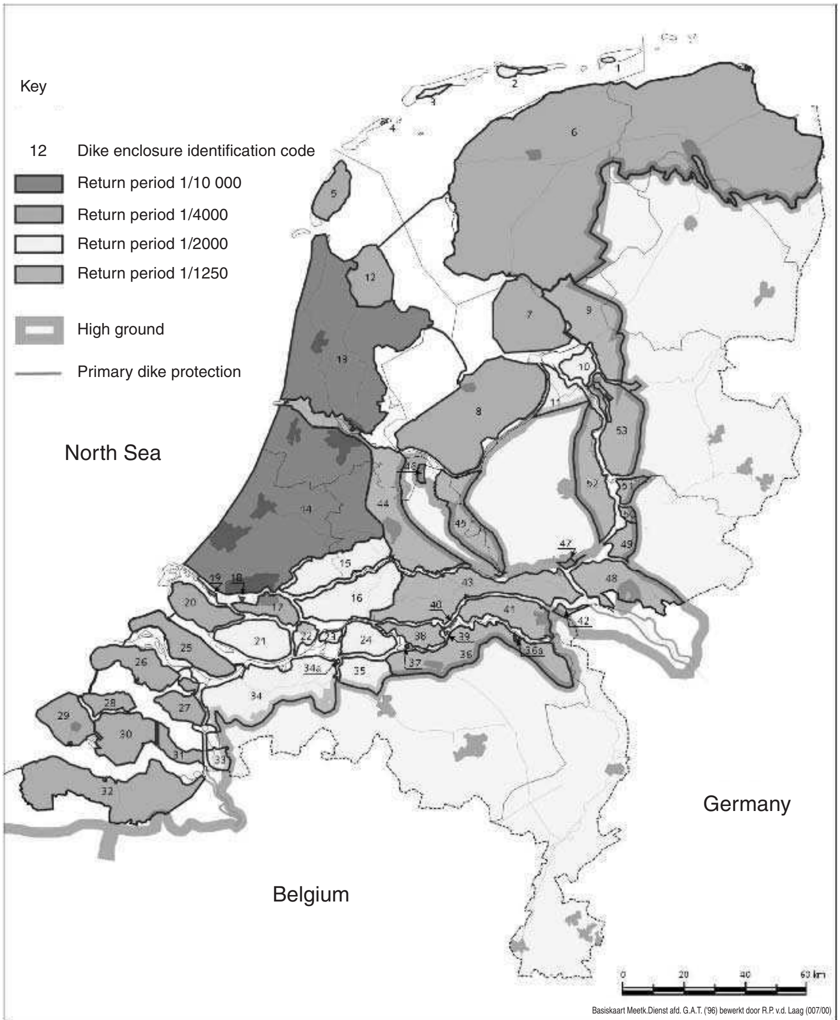


Figure 5.1 Dike enclosures and safety levels in The Netherlands

protection and safety levels in the future, such as managed realignment. Countries facing climate change, sea-level rise and land subsidence such as The Netherlands and the UK are questioning the long-term sustainability of the traditional technical 'engineering' approaches to flood control, where dike strengthening and heightening are the most common measures. A new approach is promoted, first put forward by science and now also endorsed by policy, to use the dynamics and resilience of water systems as

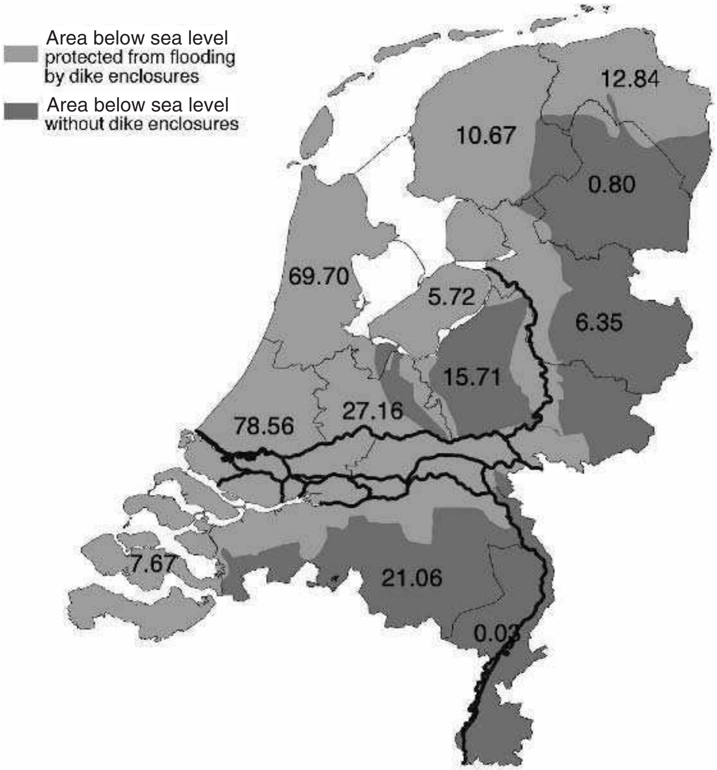


Figure 5.2 Gross domestic product (GDP) in millions of euros generated within areas situated below sea level and protected from flooding by dike enclosures in 2000

an effective means to reduce the risks and damages associated with flooding in the long term. The natural dynamics and flexibility of water systems have been severely undermined in the past through normalization of rivers, drainage of land and increases in the built-up area in traditional wetlands and flood plains. Managed realignment schemes include the realignment of rivers, estuaries and coastal defences, retreat to higher grounds, set back of dikes, widening and restoring flood plains and changing current land use patterns.

Besides long-term sustainability, managed realignment is believed to create additional socio-economic benefits compared to traditional policies of 'holding the line' such as dike strengthening. These benefits include the creation of new wildlife habitats, nutrient and contaminant assimilation and recycling, recreational and amenity values. Learning to live with floods through managed realignment is also believed to increase public awareness and appreciation of water system dynamics and resilience, and result in a reduction of future damage by changing the nature of economic activities in places at high risk of flooding.

This new Dutch policy was laid down first in the *Fourth National Water Policy Document* published in December 1998. Objectives for integrated water management policy stated in this policy document include (Ministry of Transport, Public Works and Water Management, 1998):

- Increase water system resilience, which will simultaneously also result in nature conservation.
- Increase, correspondingly, the coherence between water policy, nature conservation policy and spatial planning policy.
- Involve different stakeholders and the public at an early stage of the new policy.

These objectives were also endorsed by the Advisory Committee Water Management Policy in the 21st Century, who published their report in August 2000 (Commissie Waterbeheer 21e Eeuw, 2000). The committee concluded that without additional efforts climate change and land subsidence will cause safety levels to fall and water related problems to occur more frequently. At the same time, the number of inhabitants requiring protection and the economic value of properties to be protected are increasing. The committee classified current and future management of the water system as inadequate. The government acknowledged the potential future problems associated with climate change, sea-level rise and land subsidence and supported the recommendations by the committee in its official position *A Different Approach to Water Management; Water Management Policy in the 21st Century*, published in December 2000. In fact, the government went a step further and introduced the following three principles in order to ensure safety and reduce water related problems in the future:

1. Water management policy should be based on the principle of anticipating potential problems instead of reacting to them.
2. Water management problems and administrative responsibilities should not be passed on throughout river basins but should be based on a three-step approach: retain water in sub-basins, store water in sub-basins and drain water in sub-basins.
3. Besides traditional 'hold the line' measures, managed realignment options, creating more space for water, should be considered as a structural solution to future water related problems.

Finally, given the fact that the risks of flooding can never be reduced to zero, a special Commission for Flood Disaster Areas (Commissie Luteijn) also investigated the residual risks of flooding over and above existing safety levels and advised the Dutch government in 2002 about potential emergency measures, based on so-called designated flood disaster areas, which can be deployed in the case of emergency.

In view of the spatial differentiation of risks and the increasing economic interests protected in dike enclosures, flood control policy has to be more and more underpinned by sound economic analysis and evaluation. The use of cost–benefit analysis (CBA) has increased enormously in the past years. The main objective of this chapter is to present Dutch experiences with the use of CBA in this specific domain. Section 2 gives an overview of the use of CBA in Dutch water policy in general, while section 3 presents case study examples of CBA for large flood control projects in The Netherlands. Section 4 discusses the experiences with the use and usefulness of CBA in actual policy and decision-making so far and looks forward.

2. THE USE OF CBA IN DUTCH WATER POLICY

One of the first CBAs carried out in The Netherlands for a large-scale water management project was in the 1960s for the so-called Delta Commission (Tinbergen, 1960), which was established after the flood disaster in 1953 in The Netherlands when more than 1000 people died. The Delta Plan involved the closing off of the gateways to the south-western estuaries in the South-West of The Netherlands, except the Rotterdam Waterway and the Western Scheldt. The net costs of implementation of this plan compared to the improvement of the existing dike system in the area were estimated at about 1.1 billion Dutch guilders (price level 1955). This was a large amount of money in those days (4 per cent of the net national income – NNI).[3] Although the total direct costs of the Delta Plan were only 200 million Dutch guilders higher than the strengthening of the dikes, the Delta Plan was expected to result in considerably more indirect gains, including non-priced benefits such as public safety, drought damage reduction and recreation. The study concluded that compared with the material damage of the 1953 floods, the incremental costs of 1.1 billion are lower and the Delta Plan can therefore be justified. As a result of insufficient knowledge at that time, the study did not include a probabilistic analysis of future flooding events and their impact on the future economic damage avoided, but did acknowledge the importance of such an analysis for the outcome of the CBA, as well as the fact that the economic value of the probability of future damage avoided diminishes at positive discount rates.

Although knowledge and information about risks of flooding were very limited in the 1950s and 1960s and the study by Tinbergen was unable to quantify these risks, the study was nevertheless amazingly comprehensive in those early years in its coverage of the relevant issues. The concept of risk is central to any CBA looking at alternative flood control options and scenarios. Risk is defined here as the product of both the probability of flooding, which is a function of a variety of factors, including water levels, wind, geomorphology, strength of dikes and other water-retaining structures, and the financial and economic consequences of flooding, which is a function of the economic values of building structures, activities and so on in the area prone to flooding. A reduction in flood risks is the main economic benefit in any CBA in the context of flood control. However, it was not until the end of the 1990s that these benefits could actually be quantified in CBAs of large-scale flood control projects based upon advanced hydraulic and flood probability models (for example, Delft Hydraulics, 1998) and a newly developed national information system which enables assessment of the damage costs of flooding with the help of damage functions (DWW, 2000).

Also the estimation of other socio-economic benefits of non-traditional flood control alternatives such as managed realignment has received increasingly more attention, especially after the publication of the *Fourth National Water Policy Document* published in 1998. In this policy document the government addresses for the first time not only the issue of the costs of water management measures, but also the financial and economic consequences of these measures for society as a whole. The benefits of water management policy were even more explicitly addressed by the Advisory Committee Water Management Policy in the 21st Century in their report published in 2000. The committee concluded that policy-maker and public awareness of the benefits of water management measures are low and benefits should be addressed more explicitly in the future. Regarding the new policy oriented towards managed realignment, the Advisory Committee concluded furthermore, based upon a rough assessment of costs and benefits, that the additional costs of alternative flood protection measures now are justified based upon the social and economic benefits in the future.

In CBAs of flood control projects, the inclusion of the economic value of benefits other than damage avoided is believed to play an important role as this economic value is expected to be decisive in favour of managed realignment compared to traditional 'hold the line' solutions. However, their estimation and valuation is not without problems. The ecological functions of floodplains, fluvial wetlands or inter-tidal salt marshes, such as the provision of wildlife habitat and nutrient assimilation, and spatial quality and diversity (landscapes) are especially difficult to translate in economic terms. They are so-called non-priced public goods.[4] Although various economic methods and techniques have been developed over the past decades to value non-priced public goods in money terms, very few studies exist in The Netherlands or abroad, which have estimated the additional non-priced public benefits of managed realignment compared with, for example, traditional dike-strengthening. Economic valuation of these benefits can be a costly and time-consuming undertaking. In practice, benefits transfer is often used as a cost-effective alternative to value these benefits. Benefits transfer implies that previous valuation results, usually found under different geographical, socio-economic, institutional and political circumstances, are used to estimate the benefits of environmental changes in a new context (Brouwer, 2000).

Non-priced benefits such as spatial and ecological quality and the public perception and valuation of life threatening risks were considered in a pre-feasibility study carried out by the Central Planning Agency (CPB) on behalf of the ministry in 2000 for six different coastal and fluvial managed realignment projects, but could not be monetized (CPB, 2000). Nevertheless, it was this pre-feasibility which really paved the way for the new policy based on other (managed realignment) than conventional flood control measures (dike-strengthening). The agency concluded for most of these projects that the socio-economic benefits of managed realignment exceed their costs and are hence likely to be beneficial to society as a whole. The study is heavily quoted by the government in their official policy document 'A different approach to water', which was published only one month later (Ministerie van Verkeer en Waterstaat, 2000). The study is considered to provide sufficient evidence to justify continuation of the newly introduced policy towards flood control. The use of CBA in water management policy was given a major impetus in this policy document, as the government stated that 'concrete (managed realignment) measures will have to be tested based on their costs and benefits. In this test also non-priced costs and benefits will be taken into account, such as costs and benefits related to nature and spatial quality' (ibid., p. 31).

At the same time, the ministry also issued, together with the Ministry for Spatial Planning, their new policy line *Space for Water* (*Ruimte voor de Rivier*), which outlined the next steps from policy formulation to policy implementation and project planning (Ministry of Transport, Public Works and Water Management, 2000). Also in this official document to Parliament the government refers explicitly to the study carried out by the Central Planning Agency, showing that alternative, non-traditional flood control measures are likely to be beneficial to society as a whole. It was furthermore emphasized that, also in the policy implementation and project planning phase, flood control alternatives will be evaluated and compared on the basis of their costs and benefits, following the guidelines set out by the Central Planning Agency (see Box 5.1).

BOX 5.1 FORMAL PROCEDURAL STEPS FOR SETTING UP A CBA FOR LARGE INFRASTRUCTURE PROJECTS

Step 1: Problem definition
Step 2: Project definition and definition of the baseline scenario
Step 3: Identification of exogenous developments
Step 4: Estimation of investment and running costs
Step 5: Identification of project impacts
Step 6: Estimation and valuation of project impacts
Step 7: Set-up of a cost–benefit sheet
Step 8: Sensitivity and uncertainty analysis

Source: Eijgenraam et al. (2000).

In order to enhance comparability across studies, the CBAs currently carried out for flood control projects all follow the general CBA guidelines produced by the Central Planning Agency under commission of the Ministry of Economic Affairs and the Ministry of Transport, Public Works and Water Management (Eijgenraam et al., 2000). These guidelines were developed especially for large infrastructure projects related to transport, not for water infrastructure. Furthermore, they do not provide guidelines regarding non-priced public costs and benefits. The guidelines nevertheless provide a structured way of carrying out a CBA (see Box 5.1). How these CBAs are set up specifically for flood control is detailed in the next section.

3. CASE STUDY EXAMPLES

In this section, three different case study examples are presented, each of which addresses a different issue relevant to CBA in the specific area of flood control. The case studies cover three different areas: the non-tidal part of the Meuse river basin in the south of The Netherlands, the tidal parts of the Rhine and Meuse river basin in the western part of The Netherlands (also referred to as the Lower River Delta) and the non-tidal part of the Rhine river basin in the eastern part of The Netherlands.

Section 3.1 presents the outcome of a CBA carried out for different long-term flood control strategies, including traditional dike-strengthening and four managed realignment alternatives, for the Meuse river basin (Brouwer, 2003). This sub-section will specifically address and discuss the effect of flood probability modelling in combination with scenarios of future change and discounting on the outcome of CBA.

Section 3.2 presents one of the first full CBAs carried out for alternative flood control measures in the Lower River Delta located in the south-west of The Netherlands, where the rivers Rhine and Meuse enter the North Sea. In this study, benefits transfer was applied to estimate the non-priced benefits of public safety, long-term ecological quality and landscape amenities of managed realignment measures compared to a 'do nothing' scenario (Brouwer and van Ek, 2004).

Finally, section 3.3 discusses the results of a CBA carried out in 2002–03 in which the economic effects of controlled flooding in the case of emergencies, by allocating and designing flood disaster areas in The Netherlands, are evaluated (HKV and Delft Hydraulics, 2003; Kind, 2003). In this case study, it is the residual risk that is of primary importance in the CBA.

3.1 CBA of Long-Term Flood Control Strategies for the Meuse River Basin

From 2001 until 2003, a government-led working group investigated the possibilities of anticipating a 20 per cent water discharge increase of the river Meuse from 3800 m^3/s to 4600 m^3/s over the next 50 years as a result of climate change and land subsidence.[5] Besides traditional dike-strengthening, four different long-term managed realignment strategies were developed, including a most cost-effective managed realignment strategy, in order to safeguard existing safety levels. These strategies were subsequently evaluated in terms of their costs and benefits. The river basin can be divided in two parts for which different safety levels apply. The southern part of the river basin is not protected by dikes, due, amongst other reasons, to the geo-morphological and topographic characteristics of the river basin, and is currently protected in such a way that flooding is allowed to occur once every 250 years. Downstream, dikes protect the area prone to flooding and a higher legal safety level applies of once every 1250 years (Figure 5.3). The expected financial damage to buildings and economic activities in case of flooding is much higher in the area protected by dikes than in the non-protected area (Figure 5.4).

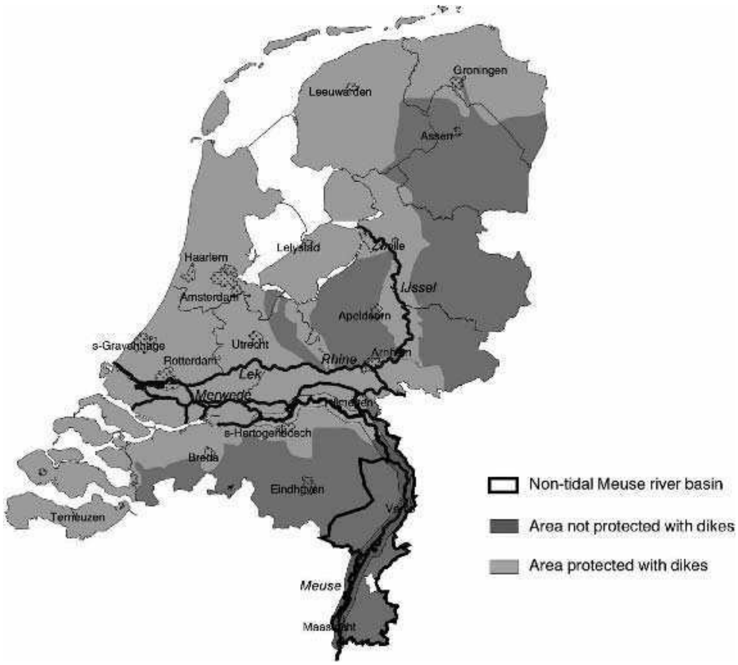
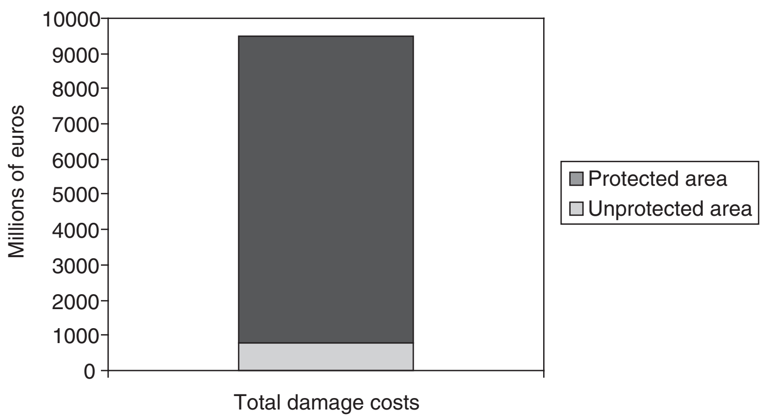
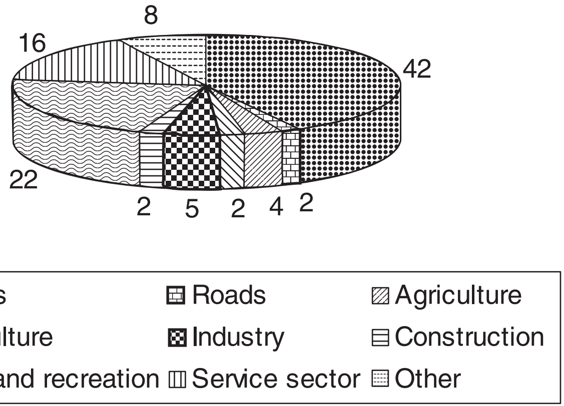


Figure 5.3 The Meuse river basin divided into an unprotected and protected part



Source: Brouwer (2003).

Figure 5.4 The financial value of total damage at a river discharge of 4600 m^3/s (price level 2002)



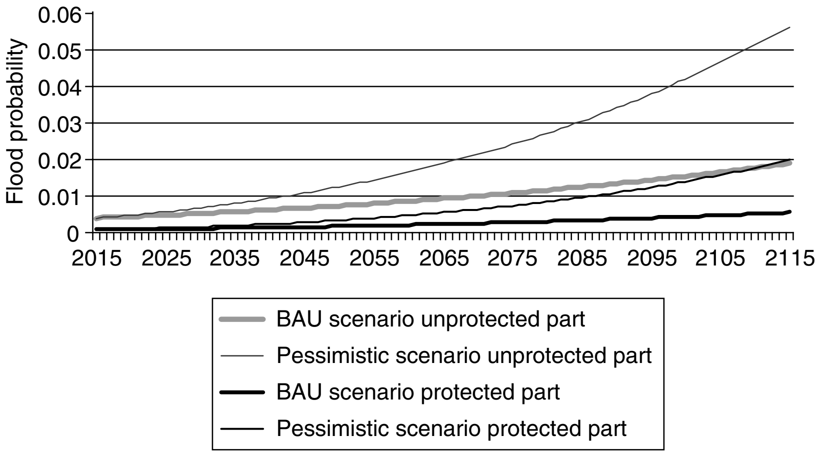
Source: Brouwer (2003).

Figure 5.5 Distribution of the total current damage costs in the protected and unprotected part across various damage categories (in percentages)

Based on current protection levels ('do nothing scenario'), a river discharge of 4600 m^3/s is expected to result in a total damage of approximately €9.5 billion. Ninety-two per cent of this damage (€8.7 billion) occurs in the area protected by dikes, the remainder (€800 million) in the area not protected by dikes. This damage is estimated with the help of damage functions developed especially for the Meuse river basin.

As can be seen from Figure 5.5, most of the current damage consists of damage to houses (42 per cent), trade and recreation (22 per cent) and service sector (15 per cent). In the case of trade, recreation, the service sector, agriculture and industry, also damage is taken into account as a result of clean-up, repairs to buildings and business interruption. No significant differences exist between the protected and non-protected parts of the river basin with respect to the various damage categories.

The current situation is not static and various developments can be identified that are expected to change in the future. Besides climate change and land subsidence, these include economic developments. Figure 5.6 shows the increase in flood probabilities as a result of two different climate change scenarios for the area protected by dikes and the area not protected by dikes.[6] The first scenario assumes, among others, an increase in precipitation of 20 per cent over the next 100 years, while the second scenario assumes an increase of 40 per cent. Under the most pessimistic climate change scenario, the probability of flooding increases along the unprotected part of the river Meuse from once every 250 years now to once every



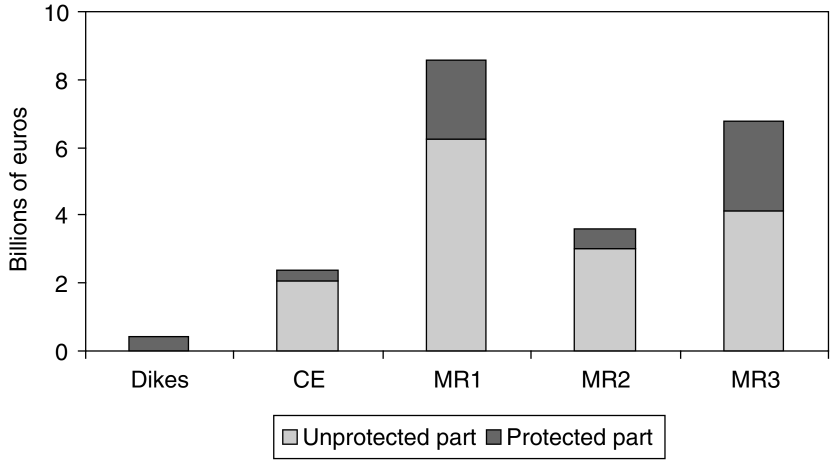
Note: Translation of the Advisory Committee's scenarios specifically to the Meuse by hydraulic experts. BAU is 'business as usual'.

Figure 5.6 Expected increase in flood probabilities as a result of climate change for the area protected and the area not protected by dikes

80 years in the year 2050 and once every 18 years in the year 2100. For the protected area, these return rates increase from once every 1250 years now to once every 300 years in 2050 and once every 50 years in 2100. The original CBA was based on this most pessimistic scenario.

In order to be able to account for autonomous economic change, three different national scenarios developed by the Central Planning Agency were used: Divided Europe, European Coordination and Global Competition (CPB, 1996). These national scenarios were translated to river basin scale through interviews with five different Chambers of Commerce in the area. The original scenarios only cover the time period 1995–2020, but were extrapolated to 2050 and 2100. The overall economic growth rates in the three different scenarios are 1.5 per cent for the scenario Divided Europe, 2.75 per cent for European Coordination and 3.25 per cent for Global Competition. After consultation of the regional Chambers of Commerce, it was decided to carry out the original CBA based on an economic growth rate of 2 per cent per annum.

The total costs for traditional dike strengthening and the four managed realignment strategies, all maintaining present safety levels over the next 100 years, are presented in Figure 5.7. A distinction is made between costs of measures taken in the protected and non-protected part of the river basin. Although most of the damage avoided is found in the area protected by dikes



Note: CE is the most cost-effective managed realignment (MR) alternative.

Figure 5.7 Distribution of the total costs across the area protected and the area not protected by dikes in billions of euros (price level 2002)

(see Figure 5.4), Figure 5.7 shows that most of the costs are found in the non-protected area, in some cases even up to 80 per cent. The distribution of costs and benefits across the river basin is hence significantly skewed. Most of the benefits are found in the already protected area, while most of the costs are borne in the unprotected area.

The total costs of the four managed realignment strategies are considerably higher than the total costs of strengthening the dikes along a stretch of 200 kilometres. The total costs of dike-strengthening amount to €428 million. The total costs of the most cost-effective managed realignment strategy is almost six times higher, while the most expensive managed realignment strategy is even 20 times higher. Using the 'do nothing' scenario as the baseline scenario, the net benefits of dike-strengthening and the most cost-effective managed realignment strategy are positive, while the net benefits of the three other managed realignment strategies are negative (Table 5.1). Hence, do nothing is not a feasible option (current safety standards have to be maintained anyway according to the law unless the law is changed).

Different assumptions about the cost calculations do not result in significant changes in the benefit–cost ratios of the various managed realignment strategies. The costs of these strategies have to decrease substantially in order to break even. In most cases the net benefits remain negative.

Table 5.1 *Costs and (net) benefits of various flood control alternatives in millions of euros (price level 2002) and benefit–cost ratios calculated under different assumptions*

	Dikes	CE	MR1	MR2	MR3
Original calculations					
Investment costs and O&M	428	2.387	8.590	3.620	6.780
Risk reduction	2.927	2.927	2.927	2.927	2.927
Net benefits	2.499	540	−5.663	−693	−3.853
B/C ratio	6.8	1.2	0.3	0.8	0.4
Sensitivity analysis			B/C ratios		
1) Damage costs					
Lower (50%)	3.4	0.6	0.2	0.4	0.2
Higher (50%)	10.3	1.8	0.5	1.2	0.6
2) Flood probabilities					
BAU scenario	3.1	0.6	0.2	0.4	0.2
3) Economic growth					
Lower (1% p.a.)	3.5	0.6	0.2	0.4	0.2
Higher (3% p.a.)	14.2	2.5	0.7	1.7	0.9
4) Discount rate					
Lower (2%)	24.2	4.3	1.2	2.9	1.5
Higher (6%)	2.6	0.5	0.1	0.3	0.2

Note: CE is the most cost-effective managed realignment (MR) alternative.

The calculated benefits in terms of the risks avoided depend on implicit or explicit assumptions related to the:

1. area flooded in case of an increase in river discharge and the associated total damage based on the estimated damage functions;
2. probability of flooding and hence the estimated expected damage;
3. future economic growth in the area which may result in higher or lower economic damage; and
4. valuation of future flood damage through the use of discount rates.

Using different assumptions results in different benefit estimates and hence benefit–cost ratios. Table 5.1 also presents the outcome of the benefit–cost ratios under varying assumptions. If the damage avoided calculated on the basis of the current situation and corresponding damage functions in the flood-prone areas is higher than expected, for instance because the area which floods is larger than expected, the inundation depth is higher than expected or the indirect economic effects are more substantial than expected, then the benefit–cost ratios of the alternative managed realignment strategies increase. However, only one of the three alternatives becomes larger than 1. If the reverse is the case and the damage costs are substantially lower than expected, only traditional dike-strengthening remains economically efficient.

In the original CBA, the most pessimistic climate change scenario and hence the highest flood probabilities were used to calculate the risks involved. Using a more moderate climate change scenario and correspondingly lower probabilities of flooding over the next 100 years and thus lower risks when doing nothing, the original benefit–cost ratios are, on average, reduced by about half.

Lower economic growth has a similar effect on the benefit–cost ratios. Assuming higher economic growth, and hence higher future damage costs, the second managed realignment strategy becomes economically efficient. The benefit–cost ratio of traditional dike strengthening becomes a high as 14. In the analysis, the possible effect of managed realignment on economic growth in flood-prone areas was not taken into account. However, it has been argued that the strategies themselves also influence socio-economic developments in flood-prone areas. Traditional dike-strengthening is expected to result in higher damage costs as economic developments in risk areas are effectively not discouraged, whereas managed realignment is assumed to result in increased awareness of the risks of building, living and working in flood-prone areas, ultimately resulting in reduced damage costs in the future.[7] As said, these potential benefits were not accounted for in the CBA.[8]

Finally, the way future flows of benefits (reduced risks) are valued through the use of positive discount rates appears to have the biggest effect on the calculated benefit–cost ratios and the net benefits of all the project alternatives. In the original CBA, all costs and benefits are discounted at the 4 per cent discount rate prescribed by the Dutch Treasury for government investment projects in risk-free environments. Obviously, the use of higher discount rates means that risk reductions further in the future are valued less and the benefit–cost ratios of all the managed realignment alternatives approach zero. Dike-strengthening remains economically efficient, but has an internal rate of return, that is, where costs and benefits are the same, of 9 per cent (based on a pessimistic climate change scenario and corresponding flood probabilities and an annual economic growth of 2 per cent). On the other hand, using lower discount rates than 4 per cent results almost immediately in positive net benefits for all flood control alternatives. Using a discount rate of 1 per cent,[9] the benefit–cost ratios become as high as 50 and 10 for dike-strengthening and the most cost-effective managed realignment strategy respectively. The ratio for the most costly managed realignment strategy (the first one) is 2.5 in that case. It is important to point out that the impact on the benefit–cost ratio of the time period over which costs and benefits are considered in the CBA (for example, 50 or 100 years) increases as the discount rate becomes lower as lower discount rates obviously imply that costs and benefits further away in the future are valued higher.

3.2 CBA of Managed Realignment in the Lower River Delta

From 1998 until 2000 an administrative working group (WG) led by the regional government responsible for the Lower River Delta (LRD) looked in a pre-feasibility study at various managed realignment options (land use changes and floodplain restoration) along the rivers Lek, Merwede, Meuse and Waal in The Netherlands (Figure 5.8). The main reasons for this work were the critical situations during the floods of 1993 and 1995 when polders were threatened and hundreds of thousands of people had to be evacuated. Following these floods in 1993 and 1995, existing dike structures were strengthened. However, in order to maintain present safety levels and anticipate expected river water level rises between 20 centimetres and 1 metre and 15 centimetres over the next 50 years (based on different climate change and sea-level rise scenarios), alternative managed realignment measures were identified to be implemented stepwise between 2000 and 2050. Based on the legally defined safety norms in the area, these measures were part of a planning strategy designed to prevent, where possible, new rounds of dike-strengthening and encourage multi-functional use of land and the development of biological diversity (de Jong et al., 2000).

As part of the WG's objectives, the aggregate effects of these managed realignment options were examined and assessed in a pre-feasibility CBA. The expected impacts of the proposed managed realignment measures compared to a 'do nothing' baseline scenario are shown in Table 5.2.[10] In Table 5.2, a distinction is made between priced and non-priced effects, and direct and indirect effects. The most important non-priced positive effects in the case of the proposed managed realignment measures (compared to the do nothing scenario) are changes in the water system's discharge capacity, public (perception of) safety and biodiversity restoration. The investment costs needed to implement the managed realignment measures and consequently the damage costs avoided are examples of direct priced effects. The investment costs are borne by the principal who carries out the

Figure 5.8 Location of the Lower River Delta in The Netherlands

project (the government). Important user groups in the region are people who live and own houses in the area, farmers and industry. Their properties and current and future economic interests will be protected by the proposed measures (at the expense of the relocation of a smaller number of houses and businesses). Third parties which benefit from the proposed managed realignment measures are the sand and grit exploitation companies in the area and, consequently, the construction industry, and possibly dredging companies as a result of increased sedimentation.

In view of the positive effects on nature and landscape, the area is expected to become more attractive for recreational activities. Based upon autonomous developments in the recreation and tourism industry, it is

Table 5.2 Expected impacts of managed realignment compared to a 'do nothing' baseline scenario

		Priced		Non-priced	
		Efficiency	Redistribution	Efficiency	Redistribution
Direct	Principal	Investment costs		Discharge capacity	
	Users	Damage costs		Public perception safety	
	Third parties	Benefits from sand and grit extraction	Income losses in agriculture and industry	– Biodiversity conservation – Public perception dislocation	Employment in agriculture and industry
Indirect		– Recreational benefits – Commercial shipping benefits		Change in water infrastructure	

expected that a large part of new recreational activities will take place in the LRD. That is, the expected increase in recreation is not the result of substitution effects, where visitors are attracted who normally visit other sites in The Netherlands (which would result in a redistribution effect on a national scale), but a real increase. The attraction of extra visitors will hence create more income in the region and nationally. These recreational benefits are considered an important indirect effect.

The possible effects of the proposed alternative flood control measures on commercial shipping are also indirect effects. However, the net effect on commercial shipping can be positive or negative. On the one hand, the deepening of river beds and floodplains and the creation of additional watercourses are expected to increase commercial and recreational shipping possibilities, and the change in the water infrastructure may enhance the accessibility of the area at the same time. On the other hand, widening the rivers also lowers water levels throughout the river basin, in which case the shipping possibilities decrease.

Efficiency effects are included in the CBA, while redistribution effects are excluded. Redistribution effects refer to effects which may have important institutional and financial consequences, but which do not influence the economic output of a country, measured in terms of national income or value added. Examples in this case are the loss of income and employment

Table 5.3 Present value of the costs and benefits of the proposed managed realignment measures compared to the 'do nothing' baseline scenario in billions of euros (price level 2000)

Costs		Benefits	
Investment costs	2.4	Economic risks avoided	3.3
Production loss agricultural land	1.8	Net welfare loss	2.2
Operation and maintenance costs	1.3		
Total	5.5	Total	5.5

in agriculture and industry in one area or region as a result of the implementation of the proposed land use changes and floodplain restoration measures, which are expected to be offset by income generation elsewhere in the country as a result of the relocation of farms and businesses.

A preliminary assessment of the economic costs of the proposed managed realignment measures showed that this was a very costly option and much more expensive than for example traditional dike strengthening. The total costs of the managed realignment measures were estimated at approximately €5.5 billion (Table 5.3). The most important reason for this high preliminary cost estimation was the fact that the measures are proposed in one of the most densely populated and economically developed areas in The Netherlands with an enormous complex infrastructure, which was expected to be affected substantially by the proposed managed realignment measures. Estimation of the economic risks avoided resulted in a discounted economic value of €3.3 billion. The benefits of sand extraction and recreation were also estimated, but relatively low (less than €200 million). Hence, estimation of the priced costs and benefits showed a negative net present value of €2.2 billion.

Members of the WG expected that economic (monetary) estimation of the non-priced benefits of managed realignment might be decisive in concluding whether the proposed managed realignment measures are beneficial to society as a whole. Since there are no original economic valuation studies in The Netherlands investigating the non-priced benefits of managed realignment, the assessment of the economic value of the expected non-priced social and environmental benefits (public safety, biodiversity restoration and landscape amenities) was based on a meta-analysis carried out by Brouwer et al. (1999) for 30 international studies looking at the economic value of various wetland ecosystem functions. These different studies produced just over 100 willingness to pay (WTP) values. These values were examined in detail and related to four main hydrological, geochemical and biological ecosystem functions performed by wetlands: flood water retention, surface and groundwater recharge, nutrient retention and

Table 5.4 Economic values of wetland ecosystem characteristics

Wetland characteristic	Average WTP (€/household/year)[1]
Wetland type	
Salt water	70
Fresh water	75
Wetland function	
Flood water retention	120
Surface water and groundwater recharge	30
Nutrient retention and export	70
Wildlife habitat and landscape diversity	95
Wetland value	
Use value	85
Non-use value	45
Use and non-use value	80
Continent	
North America	90
Europe	40

Note: [1]Rounded indicative figures. Price and purchasing power level 2000.

export and nursery and habitat for plants, animals and micro-organisms and landscape structural diversity. The economic values associated with these four functions are presented in Table 5.4.

The economic values associated with the various wetland ecosystem characteristics are expressed in average WTP per household per year. The values presented in Table 5.4 show an average WTP ranging from €30 for the wetland function surface and groundwater recharge to €120 for flood water retention. The fact that the function flood water retention is valued highest conforms to expectations regarding the possible risks to life and livelihood as a result of flooding and the capacity of floodplain wetlands to reduce this risk. No significant difference exists between the average values for fresh and salt water ecosystems. Use values for wetland ecosystems are significantly higher than non-use values. Table 5.4 also shows that use and non-use values can not simply be added in order to get a total economic value, as predicted by theory (Hoehn and Randall, 1989). Finally, average WTP is more than twice as high in North America than in Europe, due, amongst other reasons, to higher income levels.

The total economic value of the non-priced benefits such as public perception and valuation of safety, biodiversity preservation and landscape change is calculated based on the economic value for flood water retention (€120/household/year) and wildlife habitat and landscape diversity

Table 5.5 Present value of the costs and benefits of the proposed managed realignment measures compared to the 'do nothing' baseline scenario, including the non-priced socio-economic benefits, in billions of euros (price level 2000)

Costs		Benefits	
Investment costs	2.4	Economic risks avoided	3.3
Production loss agricultural land	1.8	Economic value public safety, biodiversity and landscape amenities	3.1
Operation and maintenance costs	1.3		
Net welfare gain	0.9		
Total	6.4	Total	6.4

(€95/household/year). These values are adjusted for the income differences found between countries (see Brouwer et al., 1999, for the regression results) and the fact that use and non-use values can not simply be added. These corrections result in an average WTP for both flood water retention, wildlife and landscape amenities of approximately €80/household/year.[11]

Next, the market size is determined in terms of number of households, which are expected to benefit from the proposed managed realignment measures. Together with the WG, it was agreed that more or less the whole population of the province South-Holland will benefit. In South-Holland approximately 1.5 million households are found. Multiplying this with an average value of €80/household/year results in a total economic value of €120 million per year.[12] Discounted at the prescribed 4 per cent discount rate by the Dutch Treasury over the next 100 years results in a present value of the total economic value of €3.1 billion. Including this economic value in the CBA results in a net welfare gain of almost 1 billion (Table 5.5). Even if a lower average value is used for the biodiversity and landscape amenities based on a previous study carried out in the area looking specifically at biodiversity conservation and landscape amenities on agricultural land (see note 11), a net welfare gain results of 0.5 billion euros.

3.3 CBA of Designated Flood Disaster Areas in The Netherlands

In May 2002, the Commission Luteijn presented its report 'Controlled Flooding' to the Dutch Government (Commissie Noodoverloopgebieden, 2002). This Commission was established after the evacuation of 250 000 people in flood-prone areas in 1995 along the Rhine and Meuse. The Commission's aim was to advise on 'controlled flooding' as means of limiting the consequences of floods in extreme situations along the rivers Rhine and Meuse.

The Commission advised to designate three flood disaster areas – for the Rhine basin Rijnstrangen and Ooipolder and for the Meuse basin Beersche Overlaat. These areas, measuring some 6000 hectares along the Rhine and 7000 hectares along the Meuse, are relatively sparsely populated areas with mainly extensive agricultural activities. Rijnstrangen has approximately 500 inhabitants, Ooipolder 13 000 and Beersche Overlaat about 26 000. The flood disaster areas can be used in extreme situations, that is, when river discharges exceed the maximum levels that are used for designing the dike structures. Such extreme discharges present acute dangers of flooding, the locations of which are usually unknown in advance. Through the principle of *controlled flooding* in designated flood disaster areas, the risk of uncontrolled downstream flooding of densely populated areas with important economic values is reduced. The frequency of using one of the flood disaster areas is estimated by the Commission at once every 1250 years.

The costs involved in establishing the flood disaster areas mainly consist of the costs of constructing new dikes or heightening existing dikes around the flood disaster areas, building inlet and outlet structures and dikes in the areas to prevent larger villages located in these areas from flooding when the flood disaster areas are actually used. The total costs of constructing the flood disaster areas were estimated at 1.25 billion euros. These costs were expected to be justified by the economic damages avoided downstream.

However, the Commission's report evoked a lot of criticism, mainly questioning the effectiveness and efficiency of the proposed areas, the estimated costs and benefits and the fact that alternatives for the flood disaster areas had not been sufficiently studied. Following the Commission's report, the Dutch government commissioned further pre-feasibility studies of the costs and benefits of strategies dealing with residual risks in the Rhine and Meuse river basins. Residual risks are defined here as the risks, which remain after the legal safety standards for flood protection are met. Hereafter we will discuss some of the results of the study carried out for the Rhine basin.

In the follow-up pre-feasibility studies, a distinction was made between structural strategies and emergency strategies. The distinction between the two is that for structural strategies no human intervention is required, whereas with emergency strategies human intervention is required to apply the emergency measure. Among the structural strategies investigated were measures like dike-strengthening (Figure 5.9), managed realignment and stretches of overflow-resistant dikes.[13] Hence, these strategies include measures over and above the measures required to meet the legal safety standards as explained in the earlier sections – for this specific study area once every 1250 years.

The emergency strategies included, amongst others, three different options for flood disaster areas: (1) flood disaster areas as proposed by the

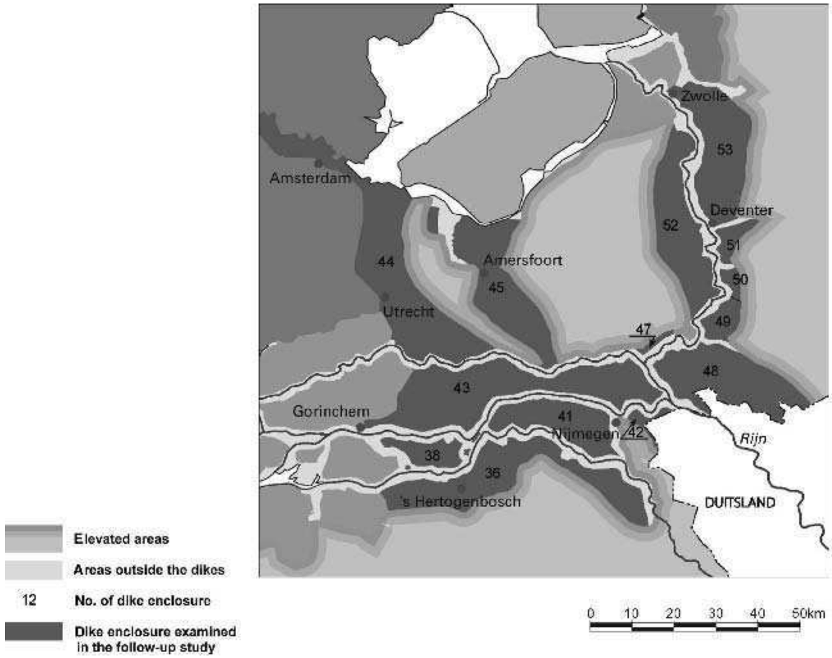


Figure 5.9 Dike enclosures examined in the follow-up study

Commission (Figure 5.10), (2) designated flood disaster areas in the upper part of the river basins without the construction of any further dikes or other protective structures, and (3) designated flood disaster areas in the upper part of the river basin with only an inlet structure.

The costs of these different strategies vary enormously, as illustrated in Table 5.6. Benefits were calculated in terms of the decrease in risk (expected material damage as well as expected damage due to the temporal interruption of businesses) due to flooding with and without the strategies. Without the strategies, the present value of the residual risk is estimated at approximately €1.1 billion.[14]

The structural strategies, dike strengthening and managed realignment, both show an important impact on the probability of flooding. They reduce the probability of flooding by a factor of five (that is, from 1/1250 to 1/6250). If no strategies are implemented, the impact of the managed realignment strategy on flooding is negligible, whereas the consequences of flooding in the case of dike strengthening are more severe due to the expected increase in inundation depth. The effects of flood disaster areas with no further protective structures are rather dramatic. According to the hydraulic

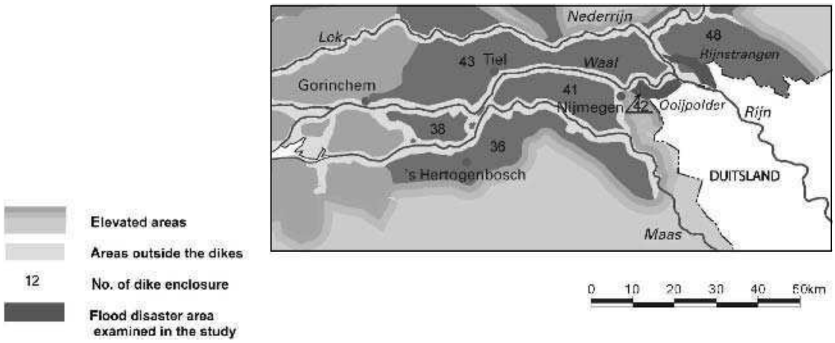


Figure 5.10 Location of the flood disaster areas for the Rhine proposed by the Commission Luteijn

Table 5.6 Costs of different measures to cope with residual risk in millions of euros (price level 2000)

Measure	Total costs
Dike-strengthening	600
Managed realignment	3000–9000*
Overflow-resistant dikes	90
Flood disaster areas according to the Commission	440
Flood disaster areas without any structures	Nil
Flood disaster areas with only an inlet structure	5

Note: *A range is shown because some strategies had additional ecological and socio-economic objectives, which cost extra.

analyses carried out, several dike enclosures along the river IJssel (nos 49, 50, 51, 52 and 53) are expected to fail, one after the other, as the limited storage capacities of the dike enclosures are reached (Figure 5.11).

Overflow-resistant dikes, flood disaster areas as proposed by the Commission and the flood disaster area with only an inlet, limit the damage in the area where the strategy is carried out and reduce the probability of flooding, but not the consequences outside those areas. Hence, also in the areas where the measures are taken, the risks can be reduced. Table 5.7 shows the estimated costs and benefits of the selected strategies. In the table, an increase in risk due to flooding in certain dike enclosures or flood disaster areas is included as 'costs', whereas a decrease in risk in other areas is included as 'benefits'.

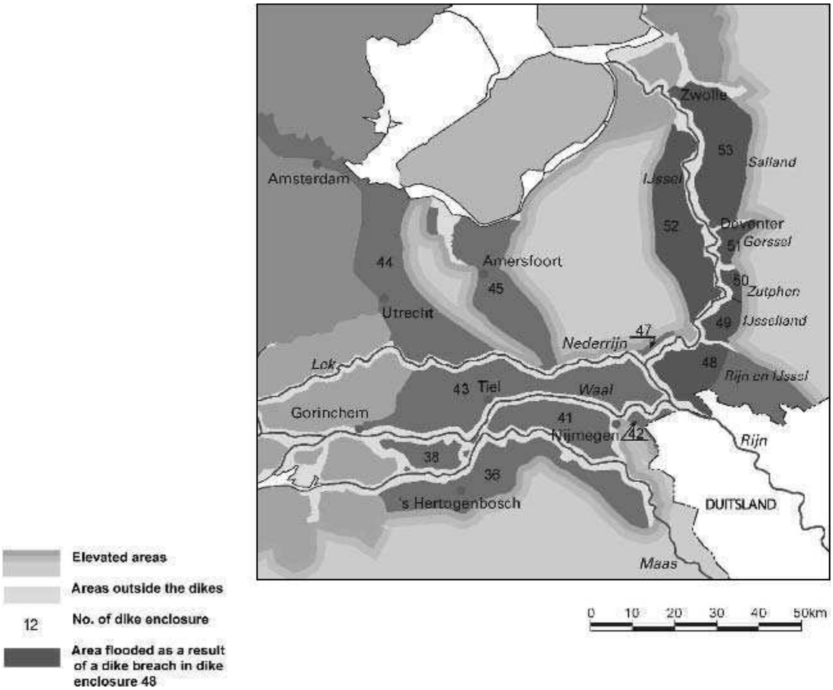


Figure 5.11 Dike enclosures which are expected to flood as a result of a dike breach

Table 5.7 shows that most risk reduction over and above the present legal safety levels results from the structural strategies such as dike strengthening and managed realignment. Based on these strategies, the residual risk of 1.1 billion euros can be reduced by 70 to 80 per cent. The benefit–cost ratio of the strategy managed realignment falls below 1. However, other socio-economic and environmental benefits are not included in the total benefits assessment. New untested strategies like overflow-resistant dikes and flood disaster areas with only an inlet deserve further attention in view of their high benefit–cost ratios. They reduce the residual risk of 1.1 billion euros by 10 to 30 per cent. Although the extent to which the risks can be further reduced with the latter two strategies has not been assessed (yet), it is expected to be less than for dike-strengthening and managed realignment, as controlled flooding is an integrated element of these strategies.

The results of the pre-feasibility study support the general conclusion by the Commission that controlled flooding is to be preferred instead of uncontrolled flooding. However, the investments proposed by the Commission for the Rhine basin to reduce the damage when the flood

Table 5.7 Costs and benefits of structural and emergency strategies in millions of euros (price level 2000)

	Structural strategies			Emergency strategies		
	Dikes	MR	ORD	FDA1	FDA2	FDA3
Costs						
Investment, O&M	600	3000	90	440	—	5
Increased risk (A)	—	—	10	5	620	35
Total costs	600	3000	100	445	620	40
Benefits						
Decreased risk (B)	750	870	360	170	535	165
Total benefits	750	870	360	170	535	165
Net benefits	150	−2130	260	−275	−85	125
B/K ratio	1.3	0.3	2.6	0.4	0.9	4.1
Residual risk without strategy (C)	1100	1100	1100	1100	1100	1100
Net reduction in residual risk (B − A)	750	870	350	165	−85	130
Residual risk remaining (C −(B − A))	350	230	750	935	1185	970
Percentage of residual risk reduced ((B − A)/C)	70%	80%	30%	15%	−10%	10%

Note: Dikes refers to dike-strengthening, MR to the cheapest managed realignment option and ORD to overflow-resistant dikes. FDA refers to the three different Flood Disaster Area strategies, where 1 is the strategy as proposed by the Commission Luteijn, 2 the strategy with no further protective constructions and 3 the strategy with only an inlet.

disaster areas are deployed, are not justified in view of the benefits of the associated risk reduction. The results of the analysis furthermore raise the questions whether (a) the present safety standards (1/1250 in the study area) are set at efficient levels, and (b) whether probability of flooding is the right measure for flood control policies, or whether a combination of probability and consequences (risk) should be the central focus in these policies. These questions, along with a fully integrated CBA of the strategies, including socio-economic and ecological costs and benefits, require further attention.

4. THE USE AND USEFULNESS OF CBA

Central to CBA in the context of flood control is the concept of risk and uncertainty. Our understanding of flood risks has improved significantly since the first CBA evaluating flood control alternatives after the flood disaster in 1953. Although policy or project appraisal will always be surrounded by a variety of uncertainties, they too are investigated in a more systematic way nowadays, for instance through the use of scenarios.

In recent years, alternative flood control management like managed realignment and flood plain restoration instead of traditional engineering approaches such as dike-strengthening has furthermore stressed the potential for long-run efficiency increases by focusing on other non-priced benefits as well. These benefits include raising public awareness and environmental improvements. Awareness that risks can never be reduced to zero and that people have to learn to live with floods is expected to reduce damage-sensitive (capital-intensive) activities in flood-prone areas in the long run as liabilities and risk premiums will inevitably change in the face of climate change. Also environmental improvements such as ecological and landscape amenities are expected to be valued higher as spatial quality becomes increasingly scarce in a densely populated country like The Netherlands. However, these assumptions will have to be substantiated through empirical evidence from practical case studies and in CBA evaluations through a more thorough valuation of public safety and ecological and landscape amenities than currently is the case.

In this chapter, we examined the specific *use* of CBA in the context of flood control in The Netherlands. In this final section, we will also briefly discuss the *usefulness* of CBA in informing and advising decision-making about flood control alternatives.

First, it is important to point out that CBA has proven to be much more than simply a technical tool or instrument. Setting up a CBA is an interactive process between experts, decision-makers and other stakeholders. In principle, all relevant stakeholders, including the decision-making authority, have a place in the CBA as a CBA describes and reflects current and future socio-economic interests, and the expected impact of policy alternatives on these interests. Hence, setting up a CBA in an interactive or participatory – bottom-up – way, has proven to be a powerful communication tool and facilitator of decision-making processes. This communication and facilitating role is especially apparent when going through the steps: (a) definition of the specific problem and baseline scenario, (b) identification of feasible and efficient policy alternatives and (c) identification of the effects of different policy alternatives.

Interaction between the expert(s) and the stakeholders involved in and during these steps is absolutely necessary in order to ensure sufficient support for the CBA and its results afterwards. Different stakeholders may have different ideas about the specific problem, its severity and its causes. Discussing the exact nature of the problem and the way the problem can be solved will also take place without a CBA, but is nevertheless an explicit first step in any CBA requiring careful thought. Related to this is the issue of how the problem will evolve in the 'without project' situation (baseline scenario). The fact that a CBA asks decision-makers to think about the relevant baseline scenario is remarkable. In The Netherlands, the baseline scenario of an environmental impact assessment (EIA) is, for instance, a standard 'do nothing' scenario. In CBA, this is not the case, especially so for flood control where safety standards are legally defined and 'do nothing' is therefore currently legally impossible. However, defining the baseline scenario requires careful thought about both exogenous driving forces such as climate change, economic developments, urbanization and possible continuation of current policy into the future. Questioning this policy may result in a different baseline scenario, allowing, for example, the search for more efficient risk levels and safety measures in the context of flood control.

Involving stakeholders at an early stage in setting up a CBA can also substantially increase the efficiency of the CBA exercise itself as the CBA expert does not have to look for the variety of different impacts of flood control on society and the economy all by himself. It also diminishes the risk of leaving out important 'hidden' stakeholders and impacts.

Second, the use of CBA has stimulated scientific thinking about flood probabilities and their consequences. Estimation of risks implies that data and information has to be collected about both these two elements. In practice, both probabilities of flooding, or changes in these probabilities as a result of human intervention, and the socio-economic impacts of flooding are often unknown as flooding is a stochastic instead of a deterministic process dependent upon a variety of factors. In The Netherlands, it is fair to say that economics has been one of the driving forces behind the more systematic modelling and collection of data and information of the risks of flooding.

Furthermore, although still in its infancy, the role of uncertainty and the way uncertainty can be managed in scientific risk assessment is increasing. On the other hand, the importance of the role of social risk perception as opposed to scientific risk assessment is increasingly acknowledged and more and more researched in this specific field.

NOTES

1. The views expressed in this chapter are those of the authors and do not necessarily represent the views of the Dutch Ministry of Transport, Public Works and Water Management.

2. Another reason for the higher safety standards for dike enclosures in the western part of The Netherlands is the shorter warning period for flooding dangers from sea (hours) compared to flooding dangers from rivers (days).
3. The total direct costs of the Delta Plan were estimated at 1.8 billion Dutch guilders (price level 1955), which amounts to almost 7 per cent of the country's NNI in 1955.
4. Basically, all the major water management policy issues raised in the *Fourth National Water Policy Document* – public safety from flooding, emission of pollutants to water and contaminated sediments and water scarcity as a result of drought – relate to the provision of non-priced public goods.
5. The strategies follow an extensive programme of measures which are currently implemented to ensure current safety levels are maintained until and including 2015. Hence, the strategies presented here would be implemented after 2015.
6. These scenarios were developed by the Advisory Committee Water Management Policy in the 21st Century.
7. Besides differences in the consequences of flooding, there may also exist differences in flood probabilities of traditional dike-strengthening compared with managed realignment. Currently, hydraulic models are unable to account for these subtle differences.
8. Also not accounted for here in money terms are the additional non-priced ecological and landscape amenities as a result of managed realignment. The managed realignment alternatives are much more expensive than dike-strengthening, because they were designed based on specific spatial quality principles. The extra costs for these alternatives are therefore basically the price paid for these additional benefits. The results of a separate environmental and landscape impact assessment were used together with the economic results in a multi-criteria analysis to rank the various flood control alternatives.
9. Assuming that policy requires that investments now should benefit current and future generations in the same way and the long-term opportunity costs of capital approach zero.
10. The structure of Table 6.2 is based on the CBA guidelines published in 2000 by the Dutch Ministry of Transport, Public Works and Water Management and the Ministry of Economic Affairs (Eijgenraam et al., 2000).
11. First, the average values are multiplied by 0.61 (based on the estimated regression coefficient) to correct for income differences. Second, the income adjusted average values are added and multiplied by 0.62 ([use and non-use]/[use] + [non-use] = 80/130 = 0.62) to account for the fact that use and non-use values cannot simply be added. Comparing the income adjusted value for biodiversity and landscape amenities (€58/household/year) with the results found in a study carried out in 1995 looking specifically at the economic value of biodiversity preservation and landscape amenities on agricultural land in the Province of South Holland (Brouwer and Slangen, 1998), the value used here is about 40 per cent higher than for the inflation and purchasing power updated value found in the latter study (€40/household/year).
12. Distance-decay effects were assumed not to be present.
13. Normally, Dutch dikes do not resist a considerable overflow of water; they fail, resulting in high damages.
14. Calculated over an infinite period of time at the prescribed 4 per cent discount rate and an annual real growth rate of GDP of 2 per cent.

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6. Cost–benefit analysis of river restoration in Denmark

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1. INTRODUCTION

Denmark is a densely populated country where few natural obstacles have prevented the appropriation of land for arable farming and urban uses. In the beginning of the nineteenth century up to 60 per cent of the total land area was occupied by heath, meadows, dry pastures, and bogs. Today less than 9 per cent of the country is covered by these types of extensively managed natural habitats. This has resulted in a decline in biodiversity as well as the loss of aesthetic and cultural values. During the past couple of decades the trend has turned and much emphasis is now placed on nature restoration. Restoration of wet meadows in river valleys has a prominent position in the nature restoration programme (Minister of the Environment, 2002). This is due to the fact that much of the biodiversity lost in Denmark is connected with wetlands and riparian areas. In addition, restored wetlands will often be able to reduce nitrogen and phosphorus pollution and provide new recreational opportunities.

The magnitude and significance of these benefits will vary depending on the type of restoration, the size of the area, and the geographical location. Likewise, the costs of restoration depend on the level of ambitions regarding the magnitude and multitude of benefits and the alternative use value of the land. In other words, when selecting areas for nature restoration decision-makers are confronted with the questions: how can generically different benefits be measured in comparable terms and how should different levels of restoration costs be weighed against benefits? Here, economic valuation methods and cost–benefit analysis (CBA) offer an opportunity to guide policy-making.

Till recently economic valuation methods and CBA had received little attention in Danish environmental policy-making. This is now changing and it is the explicit objective of the present government to get 'value for money' spent on environmental protection. The CBA of the Skjern river restoration project is an *ex post* analysis carried out after the project had been initiated and implemented. It was commissioned in 2001 by the Wilhjelm Committee in charge of drafting a national action programme for biodiversity and nature protection. The primary purpose of the study was to test the relevance of welfare economic analysis of Danish nature restoration projects, and together with other CBAs it should indicate whether nature restoration could be justified by economic efficiency considerations. Most Danish nature restoration initiatives have focused on low cost projects, often on marginalized land. As nature restoration efforts continue, low-cost alternatives will be in shorter supply. Choosing the Skjern river project meant that the efficiency hypothesis was put to a tough test since the restoration costs here are relatively high. This is due to the fact that major construction works had to be undertaken to re-meander the river and a significant share of the land affected was of relatively high agricultural value.

The main objective of this chapter is to illustrate the application of CBA in a specific area of river restoration in Denmark and to investigate if a relatively expensive nature restoration project – such as that of the Skjern river – can be justified by the economic efficiency test of a CBA.

2. DESCRIPTION OF THE SKJERN RIVER PROJECT

The primary purpose of the Skjern river project is to re-establish a large coherent nature conservation area providing good conditions for wetland fauna and flora. The location of the area is presented in Figure 6.1.

In terms of volume, the Skjern river is the largest river in Denmark with an average discharge of 35 m^3/s. The catchment area covers 2500 km^2 and the river has a length of 95 km. The river discharges into the Rinkøbing Fjord – a shallow 300 km^2 costal lagoon, which is connected with the North Sea by a floodgate. The Skjern river delta and Ringkøbing Fjord have been designated as an international bird protection area for wading birds and as an EU habitat area. The river system is home to a number of red-listed species in Denmark.

Before the 1960s the Skjern river floodplain was managed as extensively grazed meadows and hayfields. During the 1960s the lower 20 km of the river were straightened and diked. Pumping stations were established and 4000 ha of meadows were drained and converted to arable land. In 1987 the Danish Parliament decided to study restoration possibilities. Detailed surveying and designing started in 1995 and re-meandering work began in 1999. The river restoration works were completed by mid-2003.

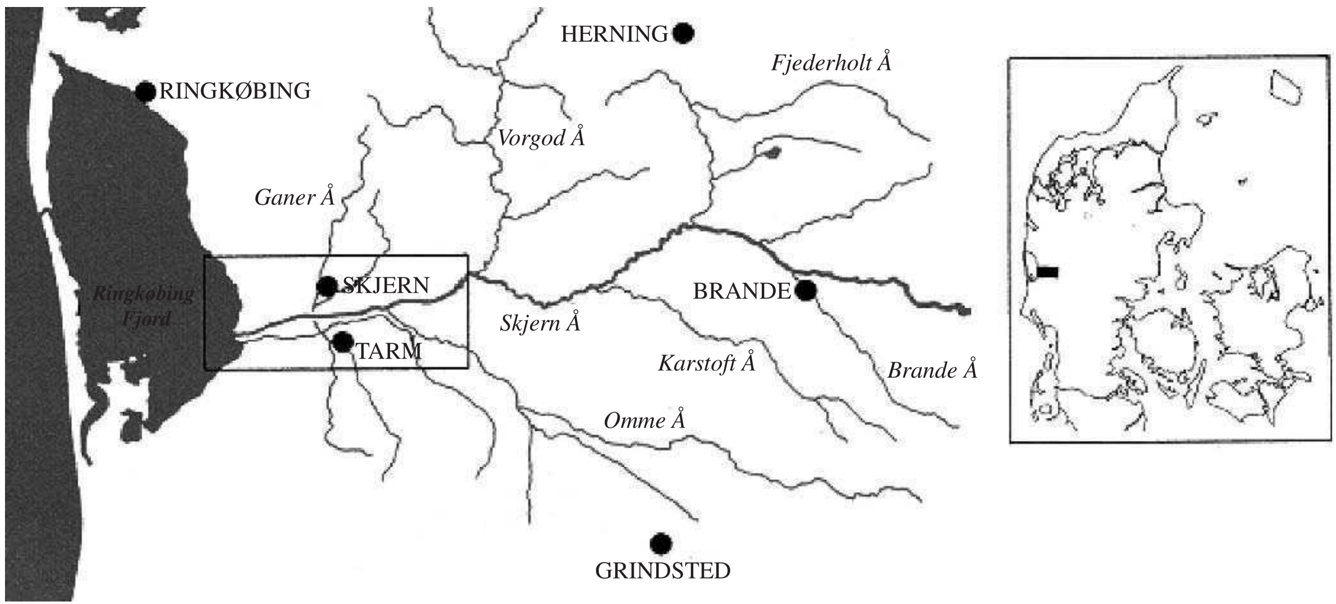

Figure 6.1 Location of the Skjern river project area

Of the 4000 ha reclaimed in the 1960s 2200 ha were included in the project. The entire project comprises the following initiatives (Danish Forest and Nature Agency, 1998):

- The lower 19 km of channelled river was turned into a 26 km meandering course.
- The river has been laid out with several outflows to the Fjord, which, in time, will create a delta of approximately 220 ha.
- Creation of a lake of approximately 160 ha.
- Re-establishment of the contact between the river and riparian areas by permitting periodical floods on land within the project area.
- Transfer of 1550 ha of arable land to extensive grazing.

The project will improve the water quality of the Skjern river system, living conditions for the wild flora and fauna, together with the recreational value of the area.[1] The flora of riparian areas and the river will become more diversified and is expected to include rare species like Elisma natans in flowing water and calamus, water soldier and cowbane in still water. The area will become increasingly attractive to breeding birds, especially species specifically found in wetlands, reed, and meadows. A significant factor is the establishment of a large coherent area with improved possibilities for nesting and feeding. Key areas free from hunting and disturbance will be established to ensure resting, foraging, and breeding possibilities for birds and mammals. Bird species like kingfisher, bittern, water rail, crake, reed bunting, reed warbler, bearded tit, ducks and geese are expected to breed in the area. Large amounts of ducks and geese have already been registered and the populations of migrating and resting birds are expected to increase. In addition to a varied bird life, an increase in the population of endangered amphibian and reptile species is expected, and populations of otter in central Jutland will be able to migrate as a result of the removal of human-made barriers in the landscape. Improved water quality, environmentally friendly maintenance practices, and the re-establishment of spawning grounds will have a positive effect on the salmon and trout populations in the river system.

The river discharges into the Rinkøbing Fjord, which is a shallow costal lagoon considerably affected by excessive loads of nutrients. A major programme is underway aiming at reducing nutrient emissions to the Fjord. The Skjern river project will contribute to this programme due to the retention of nutrients and other particles in the wetlands of the river valley. The reduction is partly obtained by the transfer of arable land to more extensive land uses, but more significantly by the re-creation of the natural ecology and hydrology of wetlands, which will filter and absorb nutrients and other particles in the river water during flooding.

Finally, the nature restoration project will increase the possibilities for recreation in the area. The size of the project area facilitates activities such as hiking and biking, boating, camping, studies of flora and fauna, angling and hunting. Accessibility has been improved by the establishment of new trails, access to grazing areas and the construction of outdoor recreation facilities.

3. THE COST–BENEFIT ANALYSIS

The present CBA was preceded by an economic feasibility study of the Skjern river project (see COWI, 1998). However, this study did not include potentially important non-market benefits like outdoor recreation and the non-use value of biodiversity. The economic value of these services can be approached by measuring people's (hypothetical) willingness to pay (WTP) for the benefits in question. Conducting economic valuation by state-of-the-art criteria is time-consuming and costly. In the present study time and budget limitations precluded the use of an original valuation survey. Instead value estimates were obtained through benefit transfer from Danish and other international economic valuation studies. The following value categories provided by the ecosystem are included in the present CBA:

- Value as a factor of production (farm land, reed production, and so on);
- Value of the ecosystem services provided (retention of nutrients, flood risk reduction, and so on);
- Extractive outdoor recreation values (hunting, angling);
- Non-extractive outdoor recreation values (hiking, boating, wildlife observation, and so on); and
- Non-use values which individuals place on the mere existence of biological diversity.

The project is considered socially advantageous if the sum of discounted consequences (benefits and costs) is positive.

3.1 Transfer Payments and Price Conversions

The analysis of the Skjern river project is based on the principles of welfare economics. In the following sub-sections we explain some of the main criteria used with regard to the treatment of transfer payments and the adjustment of prices in the analysis.

3.1.1 Delineation of society and transfer payments

A methodologically important issue in a CBA is the definition or delineation of society. In most cost–benefit analysis it is (implicitly) assumed that society is the nation-state. It is a well-known principle of CBA that transfer payments (subsidies and taxes) should be disregarded in evaluating public projects and policies. This is due to the fact that transfers *between members* of a society net out in the aggregate. However, as Mishan (1972, p. 63) points out: 'transfers are not to be disregarded if they take place across national territories. A tariff on foreign imports may be held to benefit the tariff-imposing nation even though it inflicts a loss on foreigners that exceeds the gains of the tariff-imposing nation'.

In the present case it makes a considerable difference whether the nation-state Denmark or the EU is defined as the community. If the nation-state delineation of society is applied, subsidies from the EU are to be regarded as an income flow comparable to export revenues – and as such they should be incorporated in the cost–benefit analysis. On the other hand, if the EU is considered the relevant delineation of the community, EU subsidies should be disregarded. From an idealistic perspective the latter alternative might seem the preferable approach. Yet, we have difficulties imagining that the EU is in fact perceived as the relevant community by national decision-makers. Accordingly, in the present CBA society is defined as the state of Denmark. Transfer payments within the national boundaries are therefore disregarded while transfers from the EU are included in the analysis.[2]

The national delineation of society also means that project benefits experienced by foreigners, for example tourists, should be excluded from the CBA. The same holds for global benefits in the form of reduced greenhouse gas emissions, unless such emissions are eligible for inclusion in Denmark's reduction obligations under the Kyoto agreement. If that is the case these benefits can be evaluated using the replacement cost method.

3.1.2 Adjustment of prices

Ideally, prices used in a CBA should reflect the social opportunity costs of resources. A Pareto efficient allocation requires that each individual's marginal rate of substitution between any pair of goods must equal each firm's marginal rate of transformation between that pair of goods. Achieving this in a market economy requires (among other things) that producers and consumers are faced with the same set of relative prices. However, taxes drive a wedge between producer prices and the market prices faced by consumers. The need to address this problem is accentuated by the fact that economic valuation of environmental benefits typically renders value estimates at the consumer price level, whereas project costs are typically in producer or factor prices.

If we assume that all resources in the economy are fully employed, the use of labour and produced inputs in a project will crowd out other production activities. Ultimately, this will result in the crowding out of consumption possibilities, that is, goods evaluated at consumer prices. Based on these assumptions, Johansson (1993, p. 82) establishes the following price adjustment criteria for a CBA:

- Produced inputs should be valued at consumer prices, that is, inclusive of VAT and other commodity-specific taxes.
- Labour should be valued at market wages, plus social fees, plus value added tax (VAT) and other commodity-specific taxes.

In the present CBA costs and benefits measured at factor prices are converted to the consumer price level using a *standard conversion factor*. The standard conversion factor can be calculated as the ratio between the gross domestic product and gross factor income (see Møller et al., 2000). For the period in question the standard conversion factor is equal to 1.17. For internationally traded goods the conversion factor should reflect the difference between consumer prices and prices in international trade. In the present CBA the conversion factor for internationally traded goods equals 1.25.

Individuals' WTP for environmental benefits can be interpreted as trade-off ratios between the non-market resources in question and market goods (see Freeman, 1993). Consumers' optimization of their consumption bundles is based on market prices including commodity taxes. Thus, when demand curve valuation approaches are applied – such as the travel cost methods, hedonic pricing, and contingent valuation – the values obtained are estimated at consumer prices and no conversion is required. On the other hand, if an environmental resource is priced via market goods at factor prices the resulting value must be adjusted by the standard conversion factor. One such approach is pricing via the *costs of alternatives* or *replacement costs*. An example is the costs of sewage treatment as an alternative to the retention of nitrogen and phosphorus on a restored floodplain.

3.2 Estimation of Costs

Most of the project costs are associated with construction activities and the loss of economic rents from agricultural land. Total construction costs are calculated as the expenses already incurred and the expenditures budgeted for the remaining project period. As for the loss of land rent, the calculations take into account an estimate of marginalization due to soil settling in the absence of the Skjern river restoration project.

Table 6.1 Present value of investment costs in constant (2000) prices

	Million DKK
Surveying and designing	21.3
Nature monitoring programme	9.5
Information and nature education facilities	17.4
Construction	119.7
Other	8.2
EU subsidies	−32.3
Net total	143.7

3.2.1 Direct project costs

The distribution of the direct project costs is presented in Table 6.1. In order to make the cost figures comparable through time, expenses incurred before the base year 2000 are inflated using the national wholesale price index. Opportunity costs of time are added to budget expenditures before 2000 (at 3 per cent per annum). Budgeted costs after 2000 are discounted to a present value at 3 per cent per annum. All budget costs have been multiplied by the standard conversion factor (= equal to 1.17).

The restoration of the Skjern river involved considerable preparation expenses in the form of surveying and designing. A nature surveillance programme monitors the development in hydraulics, nutrient flows, vegetation and wildlife before and after project completion. Approximately 17 million Danish Crowns (DKK) have been invested in a wide-ranging information programme and the construction of nature education facilities.[3] The construction costs of nearly 120 million DKK comprise the re-meandering of the river and the re-establishment of infrastructure in the form of roads and bridges across the river.

The European Union has contributed about 32 million DKK to the implementation of the project. As explained above, trans-boundary transfer payments are treated as benefits. In Table 6.1 the EU subsidies have been entered as a negative cost. After this correction, net direct project costs sum up to approximately 144 million DKK.

3.2.2 Costs of land reallocations

In addition to the direct construction and operating costs of the Skjern river restoration project, the central government has also purchased land in the area. The land required was purchased during the period 1991–2000. Total *budget expenditures* on land purchases were 84.4 million DKK, corresponding to an average price of about 40 000 DKK per hectare. This average covers a considerable variation in prices paid per hectare – mainly due to the fact that land prices in general rose by more than 70 per cent during the period when the land was acquired.

Budget expenditures for land purchases do not represent a use of resources. From a social point of view they can be seen as transfer payments from the state to the former owners. However, the change in land use represents a resource cost in the form of land rent losses. It is in terms of the capitalized value of rent forgone that the economic value of land enters the present CBA. European Union subsidies to arable farming are regarded as social benefits and included in the land rent – as explained in section 3.1.1. Land rent calculations are detailed below.

Draining and arable farming in the project area have resulted in soil settling, that is, land shrinkage caused by oxidation of organic soil and mechanical compression. This process has to be taken into account because it affects the economic value and hence the opportunity costs of land use in the longer term. Drained land is considered marginalized when soil conditions no longer permit arable cultivation, that is, when the distance between the groundwater level and the top soil is so small that the land becomes too wet to carry machines for sowing and harvesting. Soil settling rates were calculated on the basis of observations from similar areas (see Viborg County, 1996).

Approximately 1750 of the 2200 project hectares were arable land until the restoration of the river. Of the 1750 ha originally reclaimed for arable farming it was estimated that approximately 500 ha could be considered marginalized prior to the implementation of the project, whereas about 400 ha would have become marginalized during the following two decades. For approximately 100 ha the remaining cultivation time has been estimated to 40 years. The remaining 700 ha could have been cultivated for many years to come.

When land becomes marginalized, land rent is usually assumed to be zero. But this is not necessarily the case under the present acreage payment scheme of the EU, which involves an obligation to set land aside. The rational farmer is expected to meet his set-aside obligations by using more or less marginalized land in the project area for this purpose. As long as (partly) marginalized land qualifies for acreage payments, it is rent bearing. The implementation of the project has meant that (arable) land outside the project area alternatively has to be set aside. In the analysis presented here the rental value of set aside land in the project area is calculated as the opportunity cost of setting aside more valuable land outside the project area. Under the assumption that the quality of land surrounding the project area equals, on average, the land quality in South and West Jutland, the opportunity cost of using marginalized land for the Skjern river restoration project is estimated as the average land rent obtained in these two regions. However, it is unlikely that set-aside subsidies could have been obtained indefinitely for land, which is in fact marginalized. We tentatively assume that in the absence of the project set-aside payments for marginalized land will end 20 years from now.

The land rent from arable land is calculated for the following types of soil in the project area: loam, sand and sandy loam. The land rent estimates for the various soil types are based upon Schou et al. (2001), who used regional land rent data from the Danish Institute of Agricultural and Fisheries Economics' 2000 accountancy statistics for agricultural enterprises. Land rent from loam in the project area is assumed to equal the average land rent recorded for the Eastern Islands, where loam can be considered as the representative soil type. Rent from organic soil or humus is assumed to equal the rental value of loam (Wiborg, 2001). Land rent from the sandy soils is assumed to equal the average land rent in South and West Jutland, where sandy soils predominate. Sandy loam is supposed to provide a land rent equal to the average of the rent levels in the two above mentioned areas.

The calculated land rent for arable land and marginalized land is shown in Table 6.2. The rent figures were transformed to the consumer price level using the standard conversion factor. As can be seen, rents within the project area vary from 1450 DKK/ha/year for sandy soils or marginalized/set aside land to 2580 DKK/ha/year for loam and humus soils.

Implementation of the project implies furthermore that 1550 ha are expected to be used for extensive grazing (without fertilization or use of pesticides). The stock of dairy cattle in the region is expected to be sufficiently large to supply the necessary young stock (heifers) to graze these areas. The land rent from grazing is estimated based on similar land use in the nearby coastal grazing areas, Tipperne and Skallingen, where dairy farmers pay for grazing their heifers on pastures – with fencing and surveillance provided by the Forest and Nature Agency. The land rent from extensive grazing in the project area is calculated based on information about costs and revenues from the pasture activities in the above mentioned

Table 6.2 Arable land rents in the project area

Soil type	Rental value
Humus and loam	2580 DKK/ha
Sand	1450 DKK/ha
Sandy loam	2015 DKK/ha
Marginalized land	1450 DKK/ha

Source: Schou et al. (2001) and own calculations.

areas (Jensen, 2001). Land rent from grazing in the project area is estimated at approximately 170 DKK/ha/year. This means that grazing will recover – depending on the quality of land – between 5 and 10 per cent of the land rent forgone by giving up arable farming.

The total economic costs associated with land use changes are calculated as the net present value of land rent forgone by transforming arable land into pasture land for grazing and nature areas. The present value of land rent in the absence of the project consists of two main components: (1) the discounted flow of land rents from arable land over an infinite time horizon and (2) the discounted value of land rents from (subsidized) set-aside land over a 20-year time horizon. The net costs of land use change are calculated as the present value of the land rent forgone minus the present value of the land rent from extensive grazing after completion of the project.

Table 6.3 shows the calculated net costs due to the change in land use. As can be seen, land use changes account for a total cost of approximately 76 million DKK when the annual rent is discounted at a 3 per cent rate. Using a higher discount rate of 7 per cent reduces the net present value to 41 million DKK. This corresponds to an average reduction in land values of 43 000 DKK/ha at 3 per cent and 23 000 DKK/ha at 7 per cent. As noted previously, the average price paid for land purchased by the government was 40 000 DKK per ha.

Land rent forgone is calculated under the assumption that termination of arable farming in the project area will not significantly affect livestock production in the region. Two conditions must be fulfilled for livestock production to remain unaffected. First, it must be possible for livestock farms to meet the land–livestock balance requirements in view of the fact that legislation demands a sufficient amount of arable land being available for the disposal of animal manure. Second, the cattle farms must be able to maintain the necessary production of roughage. It is unlikely that the project will significantly affect roughage production, since fodder crops were not grown on large areas of arable land in the project area anyway.

Table 6.3 Present value of land rents forgone due to land use changes (price level 2000)

Discount rate	Million DKK
3%	75.8
5%	52.5
7%	41.3

According to the Ministry of Agriculture (1997) there is ample supply of land not used for manure in the area affected (Skjern and Egvad municipalities). Thus, the manure balance requirements are not expected to necessitate reductions in animal production.

3.3 Estimation of Benefits

A great deal of attention was given to the valuation of improved possibilities for outdoor recreation, hunting and angling, together with the substantial non-use value expected to be present as a result of an increase in the area's biodiversity. Value estimates of the more straightforward benefit components are based on the *costs of alternatives* or *replacement costs* approach. The latter benefit components have been transformed to the consumer price level using the standard conversion factor. A number of the *replacement cost* estimates have been drawn from the COWI (1998) study. The various benefits associated with the river restoration project will be dealt with in more detail below.

3.3.1 Savings on pumping costs

Conversion of the arable land in the project area means that pumping water out of the area is no longer required. The saved pumping costs are considered a project benefit. Savings amount to 356 000 DKK annually from 1999 onwards, when the project was actually implemented (COWI, 1998). The present value of this flow of saved costs over an infinite time horizon is approximately 12 million DKK at a 3 per cent discount rate.

3.3.2 Improved land allocation

The government has purchased approximately 400 ha farmland outside the project area in connection with the project. Where possible this land has been exchanged for land in the project area. At the same time, land has been reallocated between farms in the area. The farms who owned the land in the project area are mostly located outside the river valley, some of which even up to more than 10 km away from their land in the project area. The redistribution of land in connection with the project has shortened the overall distance from the farms to the fields. The land exchange and redistribution schemes affected in this way about 1000 ha. For the users of this land, the distance to the fields has been reduced by 3 km on average (COWI, 1998). The associated saved transportation costs amount to 225 DKK/km/year. COWI estimates that total savings are 860 000 DKK/year (price level 2000). The present value of this amount over an infinite time period and at a 3 per cent discount rate is 30 million DKK.

3.3.3 Increased reed production

According to the project proposal, 300–400 ha of reedbeds are expected to develop in the project area. Based on COWI (1998), it is estimated that it will be possible to harvest approximately 250 ha annually from the year 2005 onwards. The net return or land rent from reed production is about 1400 DKK/ha (price level 2000) or 350 000 DKK annually. Over an infinite time horizon and at a 3 per cent discount rate, the economic value of increased reed production is approximately 10 million DKK.

3.3.4 Flood risk reduction and closure of fish farm

There are a couple of small-scale project effects in the form of flood risk reduction and reduced pollution associated with the closure of a fish farm, which are addressed briefly below. The restoration of the Skjern river provides flood protection by allowing excess water levels to flood the restored floodplains. Approximately 30 houses are expected to benefit from this. Based on estimates of reduced flood risks and information from the National Floods Council on compensation expenses, the estimated annual benefit is about 30 000 DKK. This amounts to a present value of 1 million DKK at a 3 per cent discount rate over an infinite time horizon.

A fish farm in the project area was closed to stop emissions of organic material, which would otherwise have ended up in Ringkøbing Fjord. The capitalized net benefits (value of reduced pollution minus loss of resource rent) have been estimated at 3.9 million DKK at a 3 per cent discount rate over an infinite time horizon. The net benefits from the closure of the fish farm and reduced flooding risk are entered in Table 6.5 under the heading 'Miscellaneous benefits'.

3.3.5 Nutrient and metal reduction

The project will lead to a considerable reduction of the emission of nitrogen, phosphorus and ochre. This is due to reduced leaching from the converted arable land and more significantly the re-creation of the natural ecology and hydrology of the floodplain. This will restore the natural ability of the soil to filter nutrients and other particles. It is important to note that the benefits from reduced water pollution in the project area as such are incorporated in the value estimates of improved recreational opportunities, biodiversity, and so on. The additional value of nutrient reductions is a spill-over effect from the project in the sense that it alleviates the pollution pressure on the adjacent habitat in the Ringkøbing Fjord. During the last few decades the ecology of this habitat has been heavily degraded, and local authorities are currently planning a programme aiming at significantly reducing nutrient emissions into the Fjord. The economic value of the emission reductions due to the Skjern river project can be measured via the *costs of alternatives* – in this case the costs of extra sewage treatment or the establishment of wetlands elsewhere.

Nitrogen and phosphorous According to the Ministry of the Environment and Energy (2001a) the Skjern river project is expected to reduce nitrogen emissions by a total of 211 tons annually. The annual retention of phosphorus in the project area is expected to be 14.5 tons, or approximately 6 kg P/ha (Danish Forest and Nature Agency, 1998). The Danish Institute of Agricultural and Fisheries Economics (2000) investigated the social costs of alternative approaches to limiting emissions of nutrients into the marine environment. Under the assumption that land of relatively low agricultural value is available, the analysis shows that establishing wet meadows is one of the cheapest alternatives.[4] Conversion of agricultural land to wet meadows is assumed to cost 1500 DKK per ha in terms of land rent forgone (after adjustment by the standard conversion factor). The estimated costs of ground work are 10 000 DKK per ha – equivalent to annual capital costs of 300 DKK/ha at a 3 per cent interest rate.

In the analysis by the Institute of Agricultural and Fisheries Economics it is assumed that wet meadow land will reduce nitrogen losses by approximately 350 kg N per ha annually. This is probably a somewhat optimistic assumption. The Ministry of the Environment and Energy (2001a) estimates that the wetlands established in the Skjern river project area will at best reduce nitrogen emissions by 220 kg N per ha. Using the latter estimate, unit cost of nitrogen reduction can be calculated at approximately 8 DKK per kg N. Transferred to the Skjern river restoration project the total value of nitrogen reduction equals 1.7 million DKK annually (that is, in terms of the costs of the cheapest alternative). The corresponding present value is 57 million DKK at a discount rate of 3 per cent over an infinite time horizon.

Concerning phosphorus, it is assumed that the ratio between nitrogen and phosphorous reductions is the same in the Skjern river project as it would be in alternative wet meadow projects. Accordingly, the benefits from phosphorous reductions are covered by the costs of alternative wetlands estimated above.

It should be noted that if the alternative were removal of nitrogen and phosphorus at water treatment plants, the cost would be significantly higher (COWI, 1998). Hence, the present estimate of nutrient reduction benefits rests on the assumption that low-cost alternatives are available to the extent required to achieve the targeted reduction of nutrient emissions to the Ringkøbing Fjord. In this respect the calculations above can be considered a conservative approximation of the benefits from nutrient retention.

Ochre Drained pyritiferous soil strata are leaking ferrous substances, which are converted into ochre and in turn precipitates into streams and fjords. From the environmental impact assessment, it was estimated that the Skjern river project is responsible for an annual reduction of ochre emission of 635 tons. The alternative ochre treatment costs are estimated at 1.97 DKK per kg ochre (COWI, 1998). This amounts to 1.3 million DKK annually for the expected reduction – or a present value of approximately 41 million DKK at a 3 per cent discount rate over an infinite time horizon.

3.3.6 Effects on greenhouse gas emissions

The increased water level in the project area is expected to affect the emission of several greenhouse gases. It is estimated that the net result is an annual CO_2 reduction of approximately 15 000 tons (COWI, 1998). Reduced emission of greenhouse gases is a global benefit. A change in Denmark's greenhouse gas emissions has a negligible effect on the damages associated with global warming. Therefore, when measured at the national level, the CO_2 effect can only be considered a social benefit if it can be included in Denmark's CO_2 reduction obligations according to the Kyoto protocol. Under the present standards this is probably not possible. Consequently, the effect on CO_2 emissions is not included in the cost–benefit analysis.

If the estimated CO_2 reductions were allowed to enter the national CO_2 account, their value could be estimated at the price of CO_2 quotas traded internationally. A Danish government report expects a CO_2 quota price in the area of 90 DKK per ton (see Ministry of Finance, 2003). At this price level the value of CO_2 reductions from the Skjern river project would amount to 1.4 million DKK annually, which is equivalent to a capitalized value of 45 million DKK (at a discount rate of 3 per cent over an infinite time horizon). If the alternative were CO_2 reductions through an expansion of offshore windmill capacity, the social costs would amount to 270 DKK per ton CO_2 (ibid.). This corresponds to 135 million DKK in capitalized value of the project's CO_2 reductions. Thus, if it becomes possible to enter such reductions in a country's CO_2 account, this could contribute significantly to the economic profitability of wetland restoration.

3.3.7 Hunting benefits

Hunting will be permitted on about 1100 ha in the project area (Ministry of the Environment and Energy, 2001b). Larger populations of migrating and resting birds will increase the hunting value. In the CBA, it is the *increase* in the area's total hunting value as a result of the project which should be included on the benefits side. Theoretically, the increase equals the sum of higher *hunting rents* (producer surplus) and the increase in *consumer surplus* accruing to hunters. However, to the best of our knowledge no valuation studies exist, which would enable the estimation of a WTP function for access to hunting in this area or areas with similar characteristics. On the other hand, the increase in the rental value of hunting (that is, producer surplus) has been approximated by the Danish Forest and Nature Agency, who expects an increase in the hunting value from 200 to 600 DKK/ha/year – depending on the level of restrictions on hunting. When a nature preserve was established in the nearby located Nissum Fjord area, compensation for a hunting ban on meadowland amounted to 500 DKK/ha/year (KOFE, 1998). We estimate that, on average, the increase in hunting value will be approximately 250 DKK/ha/year.

The implementation of the Skjern river project reduces the available hunting area by 1045 ha. At first sight, the lost hunting value seems to constitute a project cost. However, in combination with the restrictions on hunting in the remaining state owned areas, the ban on hunting is expected to lead to an overall increase in game density. The value of this hunting improvement on adjacent land is tentatively estimated at 400 DKK/ha/year or an annual net hunting benefit of 200 DKK/ha for an area equal to the hunting free area. In total the hunting improvement represents an annual economic benefit of approximately 0.5 million DKK, which is equivalent to a present value of 15 million DKK at a 3 per cent discount rate over an infinite time horizon. In principle, the potential consumer surplus should be added to this value.

3.3.8 Angling benefits

The following project features are of particular relevance to potential angling benefits:

- Restoration of the lower 20 km of the Skjern river;
- Establishment of a 160 ha lake; and
- Creation of a 220 ha delta.

These changes are expected to improve angling opportunities considerably – not only along the restored part of the river, but also in the remaining parts of the river system. Of particular importance are the expected improvements in salmon and sea trout fishing. Furthermore, it is likely that the aesthetic values, created by the restoration, will have an additional value for many anglers.

No original valuation study has been carried out looking at anglers' WTP for the angling improvements created by the project. The valuation of improved angling opportunities is therefore based on a transfer of benefit-estimates. The only study looking at WTP for angling access in Denmark is part of a project conducted for the Nordic Council by Toivonen et al. (2000).

This investigation uses various formats of the contingent valuation method. It was estimated that Danish anglers' consumer surplus – associated with their present angling activities – equals on average 616 DKK/angler/year (measured as the hypothetical WTP for access to angling minus expenses).

The survey also asked about anglers' WTP for changes in the availability and quality of angling opportunities. For the Skjern river project, the following question stated in Toivonen et al. (2000, p. 61) is of importance:

> Imagine that there were a river near your home which for many years had been closed for recreational fishing . . . The river has a natural stock of salmon and sea trout, which allows for an above average chance of catching these fish species. Imagine that the river is opened to recreational fishing with rod and line . . . To get access you will have to pay a rent that would grant you 12-month right to fish in the river . . . What is the most you would be willing to pay . . . ?

In Denmark, the estimated WTP for the above-mentioned scenario lies in the interval 550–921 DKK/year per angler. The improvement of angling opportunities in the Skjern river, however, is not quite the same as the scenario in the above WTP question. The most important difference is that there was also a possibility of catching salmon and sea trout in the Skjern river before the project. So the improvement here is not as great as in the valuation scenario where it is assumed that a new river (with an above average chance of catching salmon) is added to the choice set.

Based on information from local anglers' unions, it is estimated that some 5000 anglers are currently using the area (COWI, 1998). It is likely that a larger number of anglers will use the area after completion of the project. COWI estimated that the restoration project will probably bring twice as many anglers to the area. However, this estimate is surrounded by large uncertainties. We assume here that the number of anglers will remain unchanged.

Assuming that these 5000 anglers will be willing to pay an extra 550–921 DKK per year, the value of angling will increase by 2.8–4.6 million DKK annually. At a discount rate of 3 per cent over an infinite time horizon the present value equals between 93 and 153 million DKK. In the cost–benefit assessment, the lower figure is used as a conservative estimate of the total economic benefits from angling.

Finally, besides the estimated consumer surplus, extra rent may also accrue to the owners of angling rights in the Skjern river system (producer surplus). The extent to which the potential consumer surplus will be converted to rent payments depends on the circumstances, in particular whether the state is willing to offer fishing rights to local anglers on favourable terms. This is a matter of distribution, however, which is not in itself relevant to the results of the CBA and will therefore not be considered here in more detail. From a welfare economics point of view it is the sum of the benefits that matter, not the distribution between consumers and producers respectively.

3.3.9 Non-extractive outdoor recreation benefits

Besides extractive recreational uses in the form of angling and hunting, outdoor recreation played a minor role prior to the project. However, the size and character of the newly created area allows for several types of outdoor recreation activities in the form of hiking, boating, bird watching and so on. Based on visitation patterns in similar nature areas, it is expected that the project will lead to a substantial increase in the number of visitors, largely due to the enhancement of the amenities and improved accessibility of the area. The relevant population here are recreational users who visit the area and who are not hunters or fishermen.

In Denmark there is free access to non-extractive outdoor recreation activities in nature areas. Thus, outdoor recreation will not provide any return to the owners of the land in the project area (producer surplus). The benefits consist of a consumer surplus obtained by the visitors in relation to their recreational activities. No original valuation study has been undertaken to assess the recreational benefits from nature restoration in the area. Hence, the consumer surplus is estimated again with the help of benefits transfer. As a point of departure, it is assumed that the Skjern river area will obtain a status similar to that of other nature areas of national significance, for example, the landscape Mols Bjerge. Willingness to pay for access to outdoor recreation in Mols Bjerge was investigated in a previous study (see Dubgaard, 1996). The results from this study are used as the basis for assessing the public's (hypothetical) WTP for access to outdoor recreation in the Skjern river area once the project has been completed.

The conservation area Mols Bjerge is situated in eastern Jutland approximately 40 km north of Århus. With an area of 2500 ha it is somewhat larger than the project area of 2200 ha. The topography and flora and fauna of the two areas are considered to be of minor importance for the users' WTP for access. What is essential is the fact that the Skjern river valley as well as the Mols Bjerge are unique natural areas in a Danish context. Mols Bjerge, however, is situated closer to large urban areas than the Skjern river area. Thus, the average travelling distance to the Skjern river area will be longer than to Mols Bjerge.

In Dubgaard (1996) a total of 3300 visitors were interviewed on-site about their use of the Mols Bjerge area. Willingness to pay for access to the area was elicited using the contingent valuation method. Angling and hunting are not permitted in the Mols Bjerge area, which means that the valuation estimates obtained do not overlap with angling and hunting benefits. Average WTP for access to Mols Bjerge ranged between 30 and 50 DKK per visit – depending on the question format (ibid.). The interviews and therefore the price level date from the period of 1991–92. Adjusted for the year 2000, WTP amounts to 40–60 DKK per visit. In the total benefits estimation procedure for the Skjern river again the lower figure will be used as a conservative estimate.

To calculate the potential outdoor recreation benefits from the Skjern river project an estimate of the expected number of visits has to be obtained. The registered number of visits to Mols Bjerge can be used as one of the indicators, but due to differences in the distance to adjacent densely populated areas, this figure is probably higher than the visitation potential of the Skjern river valley. Therefore, visitation counts from similar areas in western Jutland were also taken into consideration. The Tipper peninsula in the southern part of the Ringkøbing Fjord area has characteristics similar to the Skjern river valley. It consists of the privately owned area Værnengene in the south and the state owned area Tipperne in the north. Tipperne is a nature preserve where public access is restricted to guided tours only during a limited number of hours each week. During the past few years, the number of visits has been in the range of 7000–10 000 annually (Christiansen, 2001). In Værnengene, there is public access on roads and trails. There is a nature exhibition in this area where a counter registers the number of visits. According to Gregersen (2001), the registered number of visits is between 30 000 and 40 000 annually (41 000 in the year 2000). However, not all visitors visit the nature exhibition. Gregersen estimates the total number of visits to Værnengene at about 60 000 annually.

The number of visits to the Tipper Peninsula would probably be higher if there would be free access to the nature preserve. The Skjern river valley will presumably possess natural amenities and bird life equal to the Tipper Peninsula, but in addition a greater variation in outdoor recreation opportunities and recreational facilities. Visitation to the Tipper Peninsula is therefore considered a lower bound approximation of the number of visits to the Skjern river area in the long term. The registered number of visits to Mols Bjerge was between 160 and 170 thousand annually (Dubgaard, 1996). Mols Bjerge is situated in a densely populated area. For this reason this number of visits is considered an upper bound approximation of the expected number of visits to the Skjern river area. An annual number of visits in the order of 90 000–100 000 therefore seems a reasonable (cautious) estimate for the Skjern river valley.

With an annual number of visits of around 90 000 and an average WTP to visit the area of 40 DKK per visit, the expected recreational value of the

Skjern river area is 3.6 million DKK per year. At a discount rate of 3 per cent over an infinite time horizon, the calculated present value equals 120 million DKK.

3.3.10 Increased non-use value of biodiversity

The Skjern river project will improve wildlife habitat for a number of rare and endangered species in Denmark. Krutilla (1967) pointed out that individuals' may place a value on the mere existence of biological variety and its widespread distribution. Several largely synonymous terms are now used for this value category: non-use value, passive use value, existence value, preservation value, bequest value, and others (see Carson et al., 2001). Hanemann (1995) defines use and non-use value within the framework of utility theory. Use value of an environmental good is defined in terms of weak complementarity. If the environmental good is access to bird watching in a nature preserve, the complementary good is visits to the preserve. If no visits are made, for example because transportation costs have increased, then a change in the variety of birdlife in that particular preserve is irrelevant from a use value perspective. Non-use value, on the other hand, is attributed to preferences, which are separate from the use of market goods. As a result, revealed preference methods cannot be used to estimate non-use values, whereas the contingent valuation method can.

Numerous empirical investigations have established that people are willing to pay for the preservation of species they do not expect to be able to observe or make use of otherwise (see, for example, Loomis and White, 1996). We assume that this also holds for most people in Denmark. Thus, in contrast to recreational use values, where only visitors are affected, non-use benefits of biodiversity conservation accrue to everybody in Denmark who cares about this. Unfortunately, no original studies of non-use values have been conducted in Denmark. In the present CBA the existence value of enhanced biodiversity is estimated with the help of benefit transfer.

The non-use benefit estimate applied is from a valuation study of nature protection and restoration in the Pevensey Levels in England, an area comparable to the Skjern river valley (see Willis et al., 1996). The Pevensey Levels study used a variant of the contingent valuation method. Willingness to pay was elicited from a representative sample of the entire population in Great Britain, that is, users as well as non-users. Although not exactly the same, the non-use value of biodiversity preservation and enhancement was approximated in this case by the non-users' WTP for the project.

In order to obtain realistic value assessments through the transfer of benefit estimates from other projects and studies, the following conditions must be fulfilled (see Desvousges et al., 1992):

- In the baseline situation (that is, prior to the implementation of the project) the characteristics of the areas must be similar, such as nature type and area size.
- The benefits derived from these natural assets must be comparable.
- The affected populations must be comparable with respect to socio-economic characteristics.

In the present case, the two areas are reasonably comparable with respect to size (with a Skjern river project area of 2200 ha and Pevensey Levels of 3500 ha). The difference in size is accounted for by transferring benefit estimates per hectare. Furthermore, both the projects aim at ensuring and re-establishing the biodiversity of wet meadowlands with nearly identical biological characteristics. The Skjern river project, however, is more comprehensive than the programme in Pevensey Levels, since the latter area had not experienced drainage as much as the Skjern river valley. Hence, the estimates of WTP for biodiversity improvements in Pevensey Levels can perhaps be considered a lower bound approximation relative to the Skjern river project, everything else being equal. Finally, the socio-economic and cultural differences between Denmark and Great Britain are considered modest and should not present an obstacle to the transfer of benefit estimates.

The greatest problem is difference of scale concerning the size of the populations of the two countries. To solve the scale problem the benefit estimate is converted to unit benefits per household. Thus, the existence value estimate from the research area is divided by the number of households in Great Britain. Further, to adjust for the difference in area size WTP per household is divided by the number of hectares in the research area. The value per household/ha is transferred to the Danish project area. Here total existence value is calculated by multiplying the transferred benefit estimate with the number of hectares in the project area and the number of households in Denmark.

The calculation procedure is illustrated in Table 6.4. According to these calculations, the non-use value associated with enhanced biodiversity in the Skjern river area amounts to 2.7 million DKK annually. At a 3 per cent discount rate over an infinite time horizon, the present value equals 86 million DKK.

We acknowledge the uncertainties related to the measurement of non-use values and the problems associated with benefit transfer (see Brouwer, 2000,

Table 6.4 Benefit estimation of the economic value of biodiversity

	Pevensey Levels (3500 ha)	Skjern river (2200 ha)
Number of households	21.7 million	2.4 million
Economic non-use value/ha/year	£858	1207 DKK
Economic non-use value/household/year	£0.14	1.11 DKK
Total economic value	£3 million	2.7 million DKK

for an assessment). Still, the alternative to accepting these uncertainties would be to omit this potentially important benefit from nature restoration. The question is whether this error would be grater than the possible error associated with estimation and benefit transfer. Admittedly, this is a question which is difficult to answer. It seems that American administrative agencies have adopted benefit transfer of non-use values on a routine basis, rather than omitting these values from liability assessments etc. (see Penn, 2002; US EPA, 2000). In the present CBA we take a similar approach assuming that the number we have arrived at *is* better than no number.

4. RESULTS OF THE COST–BENEFIT ANALYSIS

The calculated costs and benefits are assembled in Table 6.5, which shows three scenarios using discount rates of 3 per cent, 5 per cent and 7 per cent. The Skjern river project turns out to be clearly beneficial for society at a discount rate of 3 per cent – with the present value of net benefits amounting to 228 million DKK. The project is still beneficial at 5 per cent with a net present value of 67 million DKK. However, at a 7 per cent discount rate the project provides a net present value close to zero. Thus, the magnitude of the discount rate is essential to the outcome of the CBA. This is not surprising since a sizeable part of the costs are incurred in the initial stages, while the flow of benefits is expected to continue indefinitely. These are the usual characteristics of nature restoration projects.

Unfortunately, no agreement exists on which discount rate can be considered the most relevant in social CBAs. A group of economic analysts from the Danish Environmental Research Institute and agencies under the Ministry of the Environment recommend a social discount rate of 3 per cent in social CBAs (Møller et al., 2000, p. 140). This recommendation is based on an estimate of consumers' time preference rate – measured as the real rate of interest (after tax) in the capital market during the 1990s. However, the Ministry of Finance (1999) recommends a social discount

Table 6.5 Cost–benefit results of the Skjern river project

Discount rate	3%	5%	7%
Project costs	143.7	143.0	142.2
Operation and maintenance	17.0	14.9	14.7
Forgone land rent	75.8	52.5	41.3
Total costs	236.5	210.4	198.2
Saved pumping costs	12.1	7.4	5.4
Better land allocation	29.7	19.4	15.2
Reed production	10.1	5.0	3.0
Miscellaneous benefits	5.0	2.4	1.3
Reduction of nitrogen and phosphorus	56.7	34.0	24.3
Reduction of ochre	40.5	27.0	21.3
Improved hunting opportunities	15.3	9.0	6.3
Improved fishing opportunities	89.0	52.4	36.7
Outdoor recreation	120.1	70.7	49.6
Non-use value of biodiversity	85.9	50.6	35.5
Total benefits	464.2	277.6	198.6
Net benefits	228	67	−1

rate within the range of 6–7 per cent. According to the estimates by this Ministry, this level can also be inferred from consumers' rate of time preference as well as the opportunity costs of capital (Ministry of Finance, 1999, p. 72).

The USA has a similar discrepancy between the recommendations of the environmental and financial authorities. Concerning intra-generational discounting (that is, short and medium term), the recommendation of the US Environmental Protection Agency is to use a social discount rate of 2–3 per cent – based on consumers' rate of time preference, estimated from the market interest rate after taxes (see US EPA, 2000, p. 48). When dealing with inter-generational discounting (that is, the very long term, where the welfare of future generations is involved), the EPA recommends sensitivity analyses with discount rates at levels down to 0.5 per cent (ibid., p. 52). However, the American Office of Management and Budget (OMB) recommends a standard discount rate of 7 per cent for social CBAs – based on the opportunity cost of capital (see US OMB, 2000, p. 7). Still, the OMB agrees with the EPA that using the consumers' rate of time preference would result in a social discount rate of 3 per cent (ibid.), but this is not the rate recommendable for social project assessment. Norway employs social discount rates of 3.5–8 per cent, depending on the risk concerning the returns from the project (see Ministry of Finance, 1999). The recommended social discount rate in Great Britain is 6 per cent (ibid.).

5. CONCLUSIONS

In addition to the disagreements about the relevant social discount rate we have the uncertainties associated with valuation of non-market environmental benefits. Add to this the uncertainty concerning the long-term development of price relations. Extrapolating the development in price relations is connected with great uncertainty and this has not been attempted in the present analysis. Implicitly this presumes that price relations will remain unchanged over an infinite time horizon – a rather simplistic assumption of course. These uncertainties mean that the results of an environmental CBA should not be considered the final answer. Rather, one should regard economic valuation and CBA as experiments testing the robustness of a project to alternative assumptions regarding the magnitude of costs and benefits, and – not least – the various social demands with respect to the return on invested capital.

From this perspective, the present CBA indicates that the socio-economic efficiency of the Skjern river project is quite robust. The benefit estimates applied are generally drawn from the lower end of the value intervals found. This implies that the calculated net benefits of the project must be considered a conservative estimate. In the international literature, arguments can be found for high as well as low discount rates. When it comes to long-term environmental effects, however, there seems to be a tendency to prefer low discount rates, that is, approximately 3 per cent. Discount rates in the interval 5–7 per cent must be considered fairly high capital remuneration requirements for long-term investments. The results of the present CBA show that only when the discount rate exceeds 7 per cent the project fails the present value test. We must conclude, therefore, that the resources, which have been allocated to the Skjern river project, have been put to good use from a socio-economic point of view.

As noted in the introduction, the present analysis reflects an *ex post* study conducted to test the relevance of CBA in environmental project analysis. It follows that the Skjern river CBA could not have affected current decision-making, but alongside with other environmental CBAs it did support the conclusion that nature restoration is a social objective worth pursuing. Recently, the Danish Forest and Nature Agency has commissioned a number of policy assessment studies focusing on the protection and restoration of wetlands, heaths, and so on. Important aspects here are the estimation of biodiversity existence values and the development of decision support systems for the selection of sizeable areas for nature restoration. Thus, economic valuation and CBA are gradually becoming more integrated in Danish nature policy decision-making. The big challenge ahead though is to create a stronger basis for the evaluation of nature and biodiversity benefits.

NOTES

1. In the first years of the project, a number of extensive studies of the physical, chemical and biological aspects of the project were carried out, which have been reported, for example, in COWI (1997) and Danish Forest and Nature Agency (1998). The description of the expected environmental and ecological effects is based on these previous studies.
2. The member states contribute to the EU budget. Reduced EU subsidy payments will benefit the individual member states in the form of reduced contributions to the EU or otherwise. Such savings, however, will be distributed among all the member states in proportion to their share of total budget contributions. For a small country like Denmark the 'refund' will make up a negligible share of the revenue lost when a project *in Denmark* leads to a reduction in for example, agricultural subsidies to this country.
3. In 2000, one Danish Crown (DKK) equalled, on average, about €0.13 and US$0.12.
4. This corresponds with findings in Sweden (see Gren et al., 1997).

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7. Cost–benefit analysis and complex river basin management in the Stockholm archipelago in Sweden

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1. INTRODUCTION

The main objectives of this chapter are twofold. First, a general overview will be given of the use of cost–benefit analysis (CBA) in Sweden as part of project evaluations including environmental impacts. Questions which will be addressed are: who are the users of this instrument and what determines whether CBA is used or not? Some Swedish universities and research institutes have a tradition of conducting CBA as part of their environmental economics research, but who are the users of CBA outside academia, especially in water-related projects? Second, we will illustrate how state-of- the-art CBA is applied to a complex Swedish water management example. This case study demonstrates the need for integrating both natural and socio-economic science and policy design. The case study is considered representative of the kind of knowledge and expertise needed in cost–benefit analyses of similar issues in Sweden or elsewhere.

This chapter is based upon two studies carried out independently of each other. The overview of the use of CBA by Swedish governmental agencies and non-governmental organizations was carried out in 2002 by Frykblom and Helgesson (2002) on behalf of the Swedish Environmental Protection Agency. The main findings of their survey are presented in section 2. The second study is a CBA study of eutrophication reduction measures in the Stockholm Archipelago. This CBA study was carried out in close collaboration with a number of scholars, including Scharin (2003), Söderqvist and Scharin (2000), Soutukorva (2001) and Sandström et al. (2000).[1] The set-up and main findings of the CBA and a discussion of the results are found in section 3. Section 4 concludes.

2. THE USE OF CBA IN SWEDEN

2.1 Embedding of CBA in Current Legislation

Before we look at the actual use of CBA, it might be of interest to find out what is required by the current legislation. As Sweden is a member of the European Union (EU), that is where we start off.

The Amsterdam Treaty came into force 1 May 1999. It develops the basic principles for protection of the environment and emphasizes that environmental impacts shall be integrated in the development and execution of politics also within other sectors. Section XIX says that the EU environmental policy shall contribute to the maintenance, protection and improvement of the environment, to protect people's health, ensure a careful and rational use of natural resources, and to promote actions at the international level to solve regional or global environmental problems. Article 174:3 of the Amsterdam Treaty states that when the EU forms its environmental policy, it shall take into consideration potential benefits and costs that are associated with the undertaking, or lack of undertaking, of actions. It is, however, not clear whether the comparison of costs and benefits necessarily implies a monetary comparison of effects, that is, a CBA. The Commission is clearer about the use of environmental impact analysis, such analysis shall be undertaken whenever there is a risk of significant environmental impacts. There is clearly a growing interest within the European Commission to account for the potential social benefits and costs of their actions. In order to have consistency between sectors, the General Directory of Environment has developed guidelines for how a CBA can be carried out. While the guidelines are available to all member states, there are no requirements for individual members to apply them (Vainio, 2001).

On 1 January 1999, Sweden got new legislation related to environmental issues (*Miljöbalken*). All previous environmental laws and directions fall under this new piece of legislation. Although the legislation states that economics is one of the aspects to be considered when ensuring a long-term management of resources, an explicit requirement to perform CBA is nowhere found. In April 1999, the Swedish Congress unanimously accepted 15 general environmental quality objectives:

1. clean air
2. good-quality groundwater
3. flourishing lakes and streams
4. thriving wetlands
5. balanced marine environment, flourishing coastal areas and archipelagos

6. no eutrophication
7. natural acidification
8. sustainable forests
9. varied agricultural landscape
10. magnificent mountain landscape
11. good built environment
12. non-toxic environment
13. safe radiation environment
14. protective ozone layer
15. reduced climate impact.

Each of the 15 objectives has been delegated to different federal agencies. The five goals directly related to water quality (2 to 6), or use of it, are all under the responsibility of the Swedish EPA, with the exception of groundwater which is allocated to the Geological Survey of Sweden.

The objectives are meant to work as benchmarks for all environment-related development in Sweden, regardless of where it is implemented and by whom. The overriding objective is to solve all the major environmental problems within one generation. Even if there is no explicit mention of CBA, the government proposition specifies that methods to consider costs of environmental impacts must be incorporated into economic and public decision-making models. The responsibility to conduct a CBA is again a bit elusive, there are however more explicit documents.

Twenty-four federal agencies, listed in Table 7.1, have been assigned responsibility to integrate environmental and natural resource concerns, and to work towards ecological sustainability within their sector. The assignment is further specified in a governmental publication: 'The responsibility includes, among other things, to identify the role of the government agency, propose possible goals and actions, *describe the cost–benefit consequences* and work towards the implementation of actions' (Regeringskansliet, 2000, p. 62, own translation and emphasis added).

2.2 Actual and Future Use of CBA in Governmental and Non-governmental Organizations in Sweden: Results from an Institutional Survey

The first 24 federal agencies listed in Table 7.1, and another 14 ministries, governmental institutes and organizations (numbers 25 to 38 in Table 7.1) were included in our investigation of the use of CBA by Swedish governmental and non-governmental organizations. In total, 38 agencies were asked to participate.

All organizations listed in Table 7.1 were sent a questionnaire by mail.

Table 7.1 Government and non-government organizations and agencies included in the survey

1.	Swedish EPA	13.	Swedish Work Environment Authority
2.	Geological Survey of Sweden	14.	The National Heritage Board
3.	The Swedish Rescue Services Agency	15.	The National Agency for Education
4.	The National Board of Housing, Building and Planning	16.	SwedAid
5.	The Swedish National Rail Administration	17.	The National Social Insurance Board
6.	The Swedish National Road Administration	18.	The National Board of Forestry
7.	Swedish Maritime Administration	19.	The National Board of Trade
8.	Swedish Business Development Agency	20.	The Swedish Energy Agency
9.	The Swedish Consumer Agency	21.	Swedish Armed Forces
10.	The Swedish Board of Agriculture	22.	National Chemicals Inspectorate
11.	National Board of Fisheries	23.	The Swedish Civil Aviation Administration
12.	The Labor Market Administration	24.	The National Board of Health and Welfare
25.	Swedish National Road and Transport Research Institute	32.	Ministry of Justice
26.	Swedish Institute for Transport and Communications Analysis	33.	Ministry of Industry, Employment and Communications
27.	Ministry of Finance	34.	Statistics Sweden
28.	Ministry of Defence	35.	National Institute of Public Health
29.	Ministry of Agriculture	36.	National Institute of Economic Research
30.	Ministry of Environment	37.	Greenpeace
31.	Ministry of Health and Public Affairs	38.	Swedish Society for Nature Conservation

The questionnaire was divided in three sections to capture three different types of users. The first group consists of those that have used CBA within the last five years, the second of those that do not use CBA yet but plan to do so in the near future, while the third group consists of those that neither use the method at present nor foresee any future use of it.

Thirty questionnaires were completed and returned. Table 7.2 shows our categorization of the respondents into the three groups. Federal agencies with assigned responsibility to use CBA within their sector are written in

Table 7.2 Use of CBA in governmental and non-governmental organizations in Sweden during the period 1997–2001

Group 1 (Yes)	Group 2 (No, but soon)	Group 3 (No)
SWEDISH EPA	THE SWEDISH RESCUE SERVICES AGENCY	***GEOLOGICAL SURVEY OF SWEDEN***
THE NATIONAL BOARD OF HOUSING, BUILDING AND PLANNING	THE SWEDISH ENERGY AGENCY	*NATIONAL BOARD OF FISHERIES*
SWEDISH MARITIME ADMINISTRATION	Statistics Sweden	**THE NATIONAL BOARD OF FORESTRY**
THE SWEDISH BOARD OF AGRICULTURE		SWEDISH ARMED FORCES
NATIONAL CHEMICALS INSPECTORATE		THE NATIONAL BOARD OF TRADE
THE SWEDISH NATIONAL RAIL ADMINISTRATION		THE NATIONAL SOCIAL INSURANCE BOARDS
THE SWEDISH CIVIL AVIATION ADMINISTRATION		THE NATIONAL AGENCY FOR EDUCATION
THE SWEDISH NATIONAL ROAD ADMINISTRATION		SWEDISH BUSINESS DEVELOPMENT AGENCY
National Institute of Economic Research		THE NATIONAL HERITAGE BOARD
Swedish Institute for Transport and Communications Analysis		THE SWEDISH CONSUMER AGENCY
		SWEDISH WORK ENVIRONMENT AUTHORITY
		THE NATIONAL SOCIAL INSURANCE BOARD
		Ministry of Health and Public Affairs
		The Labour Market Administration
		Ministry of Industry, Employment and Communications
		Ministry of Justice
		Greenpeace

capitals, agencies with a responsibility for water-related issues are printed in italic, while agencies with an assigned responsibility for one of the 15 environmental goals are printed in bold.

2.2.1 Group 1

The agencies were first asked about their use of CBA during the period 1997–2001. Approximately one-third of the respondents have used CBA during this period. In a follow-up question, the agencies that had used CBA were presented with a 10-point scale to indicate the extent of use. One signified 'To a very small extent' and 10 'To a very large extent'. Only three agencies report a frequent use. These agencies are the Swedish National Rail Administration, the Swedish National Road Administration and the Swedish Institute for Transport and Communications Analysis. The first two have an assigned responsibility to use CBA. The remaining eight agencies in group 1 report a limited or very limited use of CBA. It is striking that no agency responsible for any of the national environmental objectives uses CBA regularly. One of the two agencies responsible for issues directly related to water does use CBA, namely the Swedish EPA. It was reported that CBA is mainly applied to analyse new investments/large projects. It is worthwhile pointing out that the National Institute of Economic Research undertakes CBA projects as part of their green accounting work.

We further asked the respondents why CBA is used; here it was possible to choose multiple pre-listed reasons. Five answered that CBA is used because it corresponds with the agency's policy or a governmental directive. Equally, five organizations reported the nature of the environmental effects or the project as an important reason. Two reported that the specific interest of the project leader is the main reason.

Cost–benefit analysis is usually conducted by employees within the agency, even though it is also common to hire consultants. There is a large variation in the use of valuation methods, where contingent valuation is mentioned most often, followed by benefit transfer. Other methods include dose-response, hedonic pricing, travel cost and human capital studies. The latter approach includes estimation of the direct and indirect costs to society as a whole from changes in the population's general health status. In most cases, the results are or were used as input together with other information and background material in the decision-making process. Other uses are publication of results, such as public reports or media coverage.

The organizations in group 1 foresee an increased future use of CBA. This is predicted to be more systematic than today, as not only the knowledge of CBA will increase, but also the persistent need to analyse economic consequences. When asked about the need for future research on CBA, two categories of responses dominate. The first is the possibility to include other values into the monetary analysis, such as environmental and health effects and effects on historical heritage. The second category consists of the wish to find easier and more transparent general techniques that are easily understood by a broader public.

2.2.2 Group 2

Agencies that did not use CBA during the 1997–2001 period were asked whether there are any plans existing for future use. While three out of the 19 organizations have such plans, only one (Statistics Sweden) is willing to also present a time plan for implementation. Ironically, this is the only agency of the three that does not have a responsibility to conduct CBA. None of the three agencies mentioned any particular area as important for future research. The Swedish Energy Agency, one of the agencies with an assigned responsibility, foresees that CBA is likely to be used in connection with the 15 national environmental goals.

2.2.3 Group 3

The respondents in group 3 did not use CBA from 1997 until 2001 and do also not foresee any future use within their agency. The group includes 12 of the agencies with an assigned responsibility to conduct CBA, of which two agencies work with issues directly related to water, and one of those two is also responsible for one of the 15 national environmental objectives.

The group was further asked about their perception of the advantages and disadvantages of using CBA. Reported advantages include a structured way to account for consequences of actions with a well-known and standardized method. The disadvantages can be divided into mainly two different types of answers. First, the method is perceived as either being insufficient for their type of problems or their sector does not have any significant environmental impacts. Second, CBA is outside the realm of what they normally do and the organization lacks the competence to conduct such an analysis. Reported needs for further research include the use of discount rates, more user-friendly CBA and the problem of (lack of) historical values.

Responses to the questionnaires and follow-up interviews made it obvious that there is a hesitance to apply CBA. This is found in many places, not only by individuals rejecting the idea that there exist trade-offs, but also among people with knowledge of and experience with economics and CBA. Objections are based on technical and methodological issues and ethical concerns.

A number of actions were identified which could increase the use of CBA. Several agencies expressed a need for having experts within their own organization. Even if consultants conduct the CBA for the governmental agency, internal competence is needed not only when defining the project, but also later when results are to be used. Co-ordination of CBA at government level is expected to increase the transparency, compatibility, use and acceptability of the method.

It is difficult to identify specific fields where there is a real need for research on CBA in Sweden. If agencies are required to undertake CBA, it is also of paramount importance that there are individuals with the necessary competence. Universities have a role to play here by offering, for instance, state-of-the-art training products. However, CBA is more than simply estimating costs and benefits, it is also important to consider carefully what it takes for the method to be applied further. It is not uncommon, in Sweden or internationally, that results from a CBA are ignored when a decision regarding the selection or implementation of a project is to be taken (see Gren et al., 2002). Implementation aspects may therefore be an important field of future research.

Given the limited use of CBA in Sweden, it may not come as a surprise that there are few Swedish examples of cost–benefit analyses of complex water resources management issues. It is also not surprising that in Sweden, CBA for issues on a larger scale than a local one are primarily found in the academic world. Some examples are: Gren (1995) on nutrient abatement by restoring wetlands, Gren et al. (1997) on an international abatement programme for reducing eutrophication effects in the Baltic Sea, and Hjalte (1977) on the restoration of lakes. Notwithstanding these interesting and useful analyses, we believe that the CBA to be presented in the next section is one of the most advanced present-day Swedish CBA applications to water resources management on a regional scale. It is therefore likely to be instructive in showing what lessons can be learnt from this study.

3. CASE STUDY: COSTS AND BENEFITS FROM REDUCED EUTROPHICATION IN THE STOCKHOLM ARCHIPELAGO

3.1 Study Area

Marine eutrophication, that is, an increased supply of nutrients stimulating the growth of algae, is one of the major threats to the environment of the Baltic Sea. It is likely to be a consequence of a substantial increase in the anthropogenic load of nutrients (phosphorus and nitrogen) during the twentieth century, though this has been subject to considerable debate (Elmgren, 2001). In any case, there are phenomena that are attributed to eutrophication, including decreased water transparency, increased frequency and severity of oxygen deficiency in bottom waters, and changes in water fauna (Cederwall and Elmgren, 1990; Kautsky and Kautsky, 2000; Wulff et al., 2001). Some of the eutrophication effects are easily visible to the general public and likely to decrease the quality of seaside recreation. Increased turbidity is one such effect, which in turn influences the algae flora along the shores. Different fine-threaded and unpleasantly slippery algae (for example, *Cladophora glomerata*) are not negatively affected by a reduced water transparency, while the opposite is true for bladder-wrack (*Fucus vesiculosus*), a traditionally abundant species (Cederwall and Elmgren, 1990; Kautsky and Kautsky, 2000).

One area where eutrophication affects recreational quality significantly is the Stockholm Archipelago (Figure 7.1). This has probably affected a substantial number of people, since the archipelago is one of the most important recreational areas along the Swedish Baltic Sea coast. It consists of a cluster of approximately 24 000 islands, all in different sizes and shapes. It is situated in Stockholm County, which has about 1.8 million inhabitants, that is, about 20 per cent of the total Swedish population. Primarily during the last 50 years, the archipelago has transformed from being a place of permanent residents combining small-scale agriculture and fishing to a place for recreation and tourism. The central and especially the outer part of the archipelago is a 'sleeping' community most of the year until the summer when it is invaded by tourists and cottage owners. During the summers of 1998 and 1999, on average about 650 000 of the inhabitants living in Stockholm County and its neighbouring Uppsala County made about 3.4 million trips to the archipelago (Sandström et al., 2000; Soutukorva, 2001). Visitors from other parts of Sweden and abroad are not included in these figures.

The importance of the Stockholm Archipelago as a recreational area suggests that reducing eutrophication will be beneficial to society. The obvious question is whether these benefits will outweigh the associated costs of measures to abate nutrient emissions. The results of a cost–benefit analysis aimed at answering this question are presented below. Figure 7.2 shows the underlying natural scientific relationships which have to be quantified first in order to be able to connect nutrient abatement measures and their associated costs with the benefits they cause by reducing eutrophication effects.

As indicated above, there are many different eutrophication effects, some of which are less visible to the public in general and more difficult to express quantitatively. In order to keep the CBA manageable, we choose to focus on one particular eutrophication effect – increased turbidity – and to specify one particular environmental improvement – a one-metre increase in the average summer Secchi depth in the archipelago.[2] In most of the

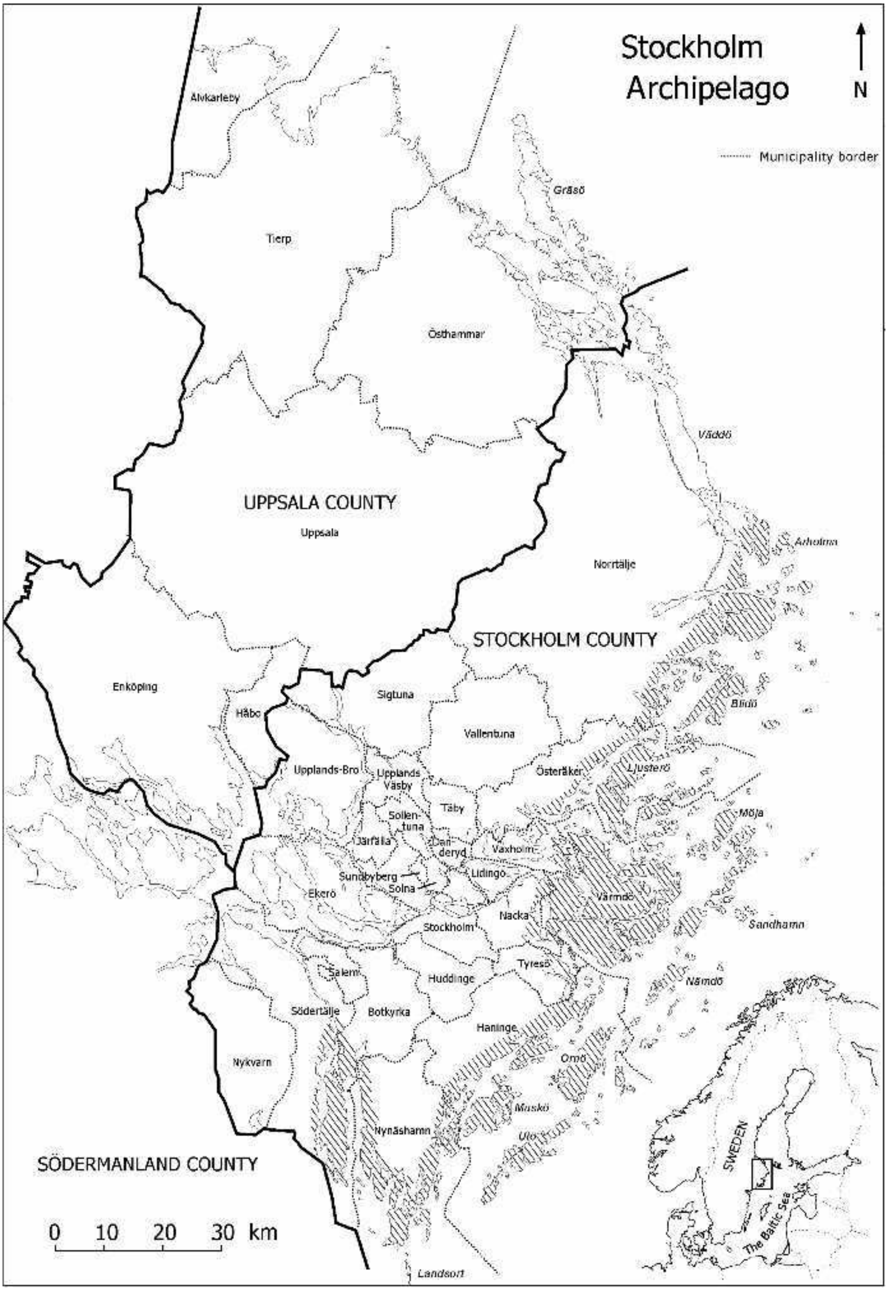
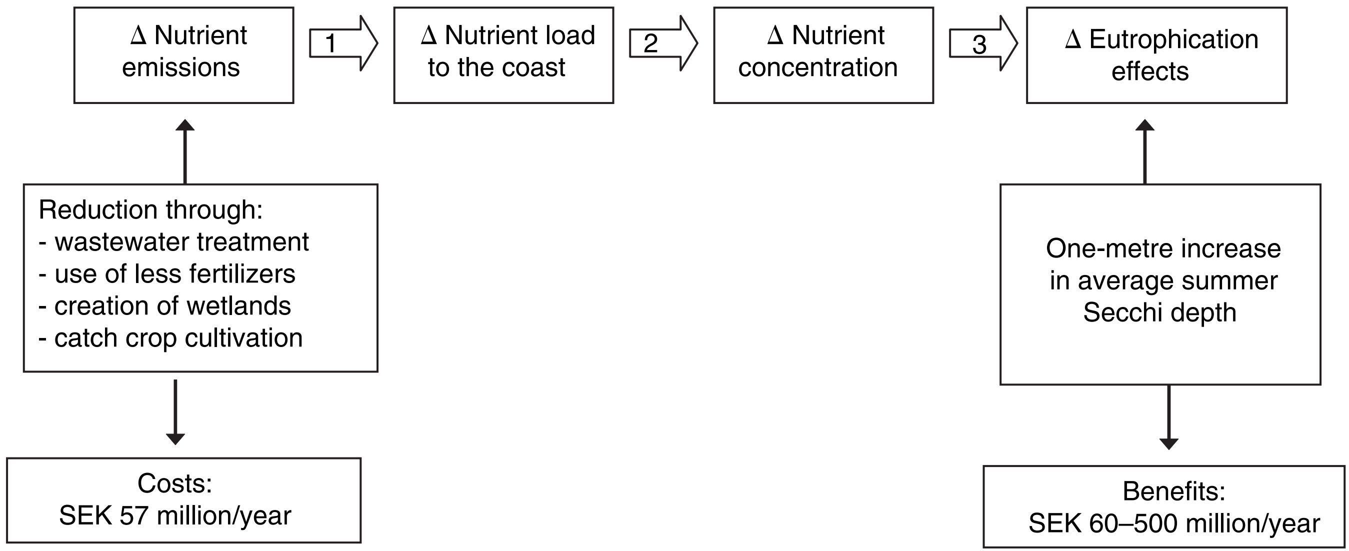


Figure 7.1 Map of the Stockholm Archipelago



Notes:
Δ = change in.
Arrow 1: The transportation and retention of nutrients from emission sources in the river basin of the archipelago to the recipient water body, that is, the coast.
Arrow 2: The relationship between the nutrient load at the coast and nutrient concentrations in the archipelago waters.
Arrow 3: The relationship between nutrient concentrations and eutrophication effects.

Figure 7.2 Physical relationships underlying the Stockholm Archipelago CBA study inner parts of the archipelago, such an improvement means that visitors will be able to see their feet on the bottom of the water when they bathe in the archipelago.

Following from Figure 7.2, the benefits of an increase in Secchi depth are related to the cost of achieving that improvement using different nitrogen abatement measures.[3] This relationship introduces an important spatial dimension to the case study, due to the fact that the impact nutrients have on a recipient does not only depend on the quantity and quality of emissions, but also on the buffering characteristics of their pathway from source to recipient. This buffering is caused by natural processes whereby a proportion of the emitted nutrients is subject to retention through sedimentation and reduction through denitrification. The marginal abatement costs to the archipelago will often differ significantly between measures in different sub-basins and their specific characteristics. This heterogeneity of sub-basins and corresponding abatement costs implies that any kind of uniform abatement policy will be inefficient as emission reduction targets can often be reached at lower costs.

3.2 Nitrogen Sources, Pathways and Loads in the Stockholm Archipelago

The major part of the nutrient load to the archipelago originates from depositions within its river basin. The Stockholm Archipelago river basin is divided into six catchments (labelled 1 to 6 in Figure 7.3). These six catchments consist of 33 sub-basins of which 11 are located adjacent to the archipelago in catchment 6. The relationship between nitrogen flows and Secchi depth for these 33 sub-basins and the archipelago will be described in this section. The nitrogen load from five of the six catchments initially enter one of the five lake basins of Lake Mälaren, while one (catchment 6) discharges its load directly into the archipelago and is therefore not influenced by the nutrient sink capacity of Lake Mälaren. The nitrogen is transported within Lake Mälaren eastwards towards the archipelago. Only the final nitrogen load from the river basin to the archipelago is considered, since it is mainly this load that determines the Secchi depth in the archipelago.

In order to be able to determine the costs of reaching the Secchi depth target, the first step is to assess the extent to which emissions from different sources have an impact on the Secchi depth in the archipelago. The sources and their emissions of nitrogen for each catchment are summarized in Table 7.3. Emissions from agriculture and forestry refer to leaching, while the other sources involve direct discharges into a water body. This explains their relative sizes, which incorrectly suggests that agriculture is the dominant source of nitrogen. The atmospheric deposition in water bodies

Figure 7.3 *The river basin for the Stockholm Archipelago divided into six catchments*

Table 7.3 *Nitrogen emissions from different sources within the six Stockholm Archipelago river basin catchments in tons per year*

Catchment	Agriculture (leaching)	Forestry (leaching)	Waste water	Industry	Atmospheric deposition
1	914	901	499	337	503
2	1882	490	1343	320	675
3	586	124	246	230	358
4	1354	323	568	0	113
5	186	32	38	0	94
6	764	318	2449	0	164
Total	5686	2188	5143	887	1907

is indirectly also included in the estimates of agriculture and forestry. The major emission source in this region appears to be waste water treatment plants (WWTPs). Emissions from WWTPs account for a relatively large share in catchments 2 and 6, while the share of agriculture is relatively large in catchments 1, 2, 3, 4 and 5.[4]

Except for the direct discharges in catchment 6, all other emissions are subject to either the buffering capacities of the previous catchments or to the nutrient sink capacity of Lake Mälaren. The actual impact on the recipient is therefore less than the emissions from all the sources reported in Table 7.3. The fraction of these nitrogen emissions ending up in the archipelago is determined by the buffering capabilities of the aquatic ecosystems along their pathway to the archipelago.

The fraction of nitrogen that reaches the archipelago increases when moving from lake basin 1 to 5. While only 32 per cent of the first lake basin's nitrogen load reaches the archipelago, this is 77 per cent in the case of lake basin 5. The effect of measures taken in region 1 is therefore much smaller than those taken in region 5 in view of the fact that a discharge into lake basin 1 is subject not only to its own nitrogen sink capacity, but also the capacity of lake basins 2, 3 and 5.

These loads and their distribution across the six catchments are presented in Figure 7.4. The retention capacity of Lake Mälaren manifests itself through the differences found between each catchment's nitrogen load into the nearest catchment and its final contribution to the archipelago. Due to the nutrient sink capacity of Lake Mälaren, only 58 per cent of the total annual N discharges into the lake of 11 823 tons reaches the archipelago. The final load of nitrogen into the coastal zone of the Stockholm Archipelago is 6814 tons per year. More than half of the total load to the archipelago comes from sources within its adjacent catchment area (that is,

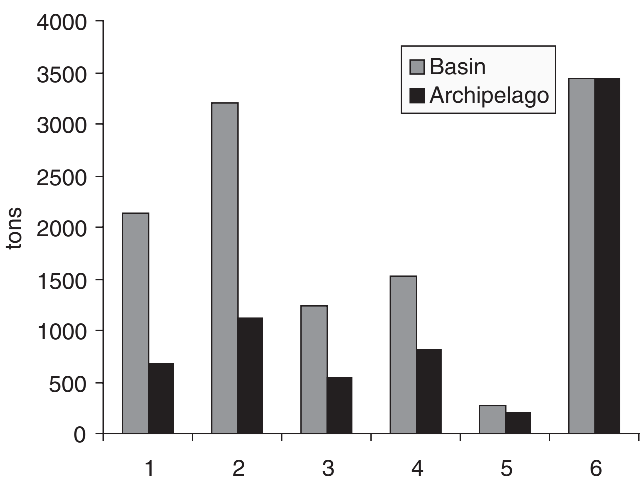


Figure 7.4 Contribution of each of the six catchments to their adjacent catchments and to the Stockholm Archipelago

catchment 6). This can be explained by the presence of a few large WWTPs in this catchment, which treats the sewage from the city of Stockholm and discharges nitrogen effluent directly into the archipelago.

The nitrogen contribution from WWTPs exceeds the contribution from agricultural sources in this region even though the emissions from the latter source are larger. Given the focus on the water quality in the archipelago, this illustrates the importance of finding out the relationship between nitrogen emissions and the resulting nitrogen load to the archipelago, see Figure 7.2. The nitrogen load to the archipelago affects the Secchi depth through the resulting nitrogen concentration in the water. Time series data on nutrient concentrations and Secchi depth in the Stockholm Archipelago have been analysed, and nitrogen concentration turned out to be a significant determinant of Secchi depth (Färlin, 2002). During an average summer, a Secchi depth of 1.5 m is not uncommon in the inner part of the archipelago. The relationship estimated by Färlin (2002) suggests that a Secchi depth of 1.5 metres corresponds to a nitrogen concentration of 592 mg per m^3. The relationship also suggests that a 1-metre improvement to 2.5 metres requires a reduction of the nitrogen concentration to 426 mg per m^3, that is, a reduction of approximately 30 per cent. It can therefore be concluded that in order to improve the Secchi depth by at least one metre in the whole archipelago, a reduction of the nitrogen concentration of at least 30 per cent is needed.

Linking levels of nitrogen concentrations to nitrogen loads to the Stockholm Archipelago as a whole requires detailed water exchange modelling for the archipelago. Research aimed at building such models is on-going. However, no results are available yet. In the CBA, it was therefore assumed that a load-concentration function estimated for the Baltic Sea can also be applied to the archipelago. This function suggests that a 40 per cent reduction of nitrogen load is required to reduce the nitrogen concentration by 30 per cent (Wulff, 2000). In absolute terms this means that a 40 per cent reduction of the nitrogen load to the coast corresponds to a reduction from 6814 to 4089 tons per year, that is, 2725 tons per year.

3.3 Nutrient Abatement Measures

In this study, four different abatement measures are considered in the Stockholm Archipelago river basin: (1) reducing discharges from WWTPs, (2) reducing the application of fertilizers, (3) cultivation of catch crops, and (4) construction of wetlands. These measures also play a role in present-day agri-environmental policies in Sweden. However, these policies do not take into consideration the spatial heterogeneity of marginal abatement costs. In the Stockholm Archipelago river basin, 50 per cent of the construction costs of a wetland are subsidized, including a yearly payment of SEK 3000 per ha to the landowner as a compensation for the opportunity cost of the land, while SEK 800 per ha per year is given to fund necessary wetland management activities (Swedish Board of Agriculture, 2003).[5] In Sweden, a tax on fertilizers exists, which was SEK 1.80/kg N in 1998 and is the same for the whole of Sweden (SCB, 2000). A subsidy of SEK 900 per ha for catch crop cultivation exists for certain parts of Sweden, however not for this region, as a result of the low effect they have on reducing leaching in this part of the country (Swedish Board of Agriculture, 2003). Only the nitrogen emissions of coastal WWTPs located south of Norrtälje (see Figure 7.1) are regulated by law. Annually, plants connected to over 100 000 individual households are not allowed to discharge, on average, more than 10 mg N per litre, while the limit for those connected to less than 100 000 households is 15 mg N per litre (Swedish Environmental Protection Agency, 2003). Wastewater treatment plants in the region that are not located along the coast, but discharge into a river or Lake Mälaren, are only obliged to use some kind of nitrogen abatement. Present policies do not, therefore, really create any incentives for plants to further reduce their nitrogen discharges, which might explain their reluctance to invest in improved abatement technology.

3.4 Abatement Costs

The calculation of the abatement costs as well as the nitrogen load to the coast are based on the final impact of each sub-basin on the archipelago. Abatement costs are thus estimated taking into account the impact on the archipelago, converting the marginal cost at the source to a marginal cost at the recipient water body (that is, the archipelago). Marginal abatement costs at the source for the first two measures (abatement of discharges from WWTPs and fertilizers) were calculated for the 33 sub-basins, while costs for the last two measures (catch crops and wetlands) were calculated for the whole region. The impact of nitrogen leaching was calculated for each sub-basin taking into account the retention as well as the nutrient sink capacity of Lake Mälaren. In the river basin, direct discharges – mainly by WWTPs, but also by industrial discharges and atmospheric deposition in water – account for the largest part of the load to the archipelago (68 per cent).

The question was what would be the least costs to accomplish a reduction of the nitrogen load to the coast by 2725 tons per year in order to increase the Secchi depth by 1 metre? An important part of the CBA was to search for the cost-effective combination of abatement measures, that is, an allocation of abatement measures across the basin in such a way that any

Table 7.4 Abatement costs at source and recipient water body for various abatement measures (price level 1998)

Measure	Marginal costs at source (SEK/reduced kg N/year)	Marginal costs at recipient water body (SA) (SEK/reduced kg N/year)
Fertilizer reduction	0–4.93	0–514
Catch crop cultivation	127	141–1984
Wetland construction	33.5	33.5–105
Waste water treatment	5–32	9–38

Note: SA = Stockholm Archipelago.

reallocation would result in an increase in the total cost of reaching the target.

Table 7.4 summarizes the abatement costs for the different measures at the source as well as at the recipient water body.[6] The range of cost estimates at the recipient water body (that is, the archipelago) are the result of differences regarding the estimated impact of discharge locations in the various sub-basins. As a rule, the marginal abatement cost of a certain measure is higher in upstream sub-basins. Assuming that each WWTP adopts the best available technology to reduce nitrogen emissions, this generates an estimate of the marginal abatement cost of N reduction at WWTPs. The marginal abatement cost for reducing the amount of fertilizers was estimated as the loss in farmer producer surplus. The cost of growing catch crops consists of seed and sowing costs and loss of profit due to smaller yields and is in this region estimated at SEK 380 per ha (Olsson, 2002). Dividing the fixed costs of constructing a wetland by the lifetime expectancy results in an annual fixed cost, which when added to the yearly opportunity costs (based on foregone profits from crop cultivation and management costs), yields a marginal abatement cost for wetlands.

The abatement costs at the source are fairly similar for three of the four measures (wetlands, fertilizers, and waste water treatment). Catch crop cultivation, however, is an extremely expensive measure in this region. Differences are even more accentuated when estimating the cost of reducing one unit of nitrogen at the recipient water body. In this case, the same measure differs in costs in different catchments. For fertilizer reduction and catch crop cultivation, this cost increases in the presence of downstream measures.

All those measures are selected whose marginal costs for reduction at the recipient are less or equal to the shadow price of the restriction (that is,

Table 7.5 Nutrient abatement and associated costs to achieve a 1-metre Secchi-depth improvement with the help of various measures (price level 1998)

		Costs (SEK million per year)
Total reduction to archipelago	2725 ton	57.0
Allocation of total load reduction:		
Waste water treatment	91.5%	54.0
Fertilizer reduction	8.5%	3.3
Catch crop cultivation	0%	0
Wetland construction	0%	0

minimizing the cost of a 1-metre Secchi-depth improvement). An optimal allocation of measures is defined by the fact that no further reduction can be made without increasing the costs. Table 7.5 summarizes the optimal allocation for accomplishing the target of a nitrogen load reduction of 2725 ton per year. The total costs amount to SEK 57 million per year. The shadow price of the target constraint is SEK 32 per kg of N, implying that any abatement measures with a marginal abatement cost at the recipient water body equal to or less than this number should be implemented in a cost-effective solution. Table 7.5 also shows the optimal allocation of measures in order to reach the target: waste water treatment and fertilizer reduction account for 91.5 and 8.5 per cent of the emission reduction respectively, while the measures for construction of wetlands and growing catch crops should not be implemented in view of their relatively high costs. As to the spatial distribution of the measures, a general result is that measures in sub-basins distant from the recipient are less attractive due to their lower impact on the archipelago. Sixty-six per cent of the load reduction can be achieved by taking abatement measures in catchment 6, while the rest is dealt with in upstream catchments.

3.5 Recreational and Other Benefits

The benefits of a 1-metre increase in average summer Secchi depth in the archipelago were estimated using two different approaches: (1) a travel cost approach, where people's demand for recreation in the archipelago is estimated with Secchi depth (our measure of water quality) as one of the explanatory variables, and (2) a contingent valuation approach, where people's willingness to pay for a nutrient abatement programme implying a

1-metre increase in Secchi depth is estimated. The travel cost approach gives information about the recreational benefits of improved water quality. However, it might very well be the case that also non-users of the archipelago care for its water quality and would be willing to pay something for an improvement. Contrary to travel cost studies, contingent valuation is able to estimate 'total' benefits in that also non-users' willingness to pay can be taken into account and also non-use values. On the other hand, compared to travel cost studies contingent valuation may introduce a hypothetical bias.

Data for both the travel cost study and the contingent valuation study were collected by two mail surveys on travel behaviour and willingness to pay, carried out directly after the summers of 1998 and 1999. The questionnaire was sent to a sample of households living in the counties of Stockholm and Uppsala. Four thousand individuals were sampled randomly in 1998 and 1500 in 1999. The response rates were 49.5 per cent and 62.5 per cent respectively. The difference in response rate is probably due to the fact that the first questionnaire contained more detailed questions on travel behavior than the second one, and hence took longer to complete, resulting in a lower willingness to complete and return the questionnaire.

In the travel cost study, travel data obtained through the surveys were combined with data from Secchi-depth measurements in the archipelago, and a conditional logit model estimating recreationist choices of sites in the archipelago. Three explanatory variables were included in the conditional logit model: travel costs, Secchi depth and site accessibility. The relationship between travel costs and the probability of selecting a specific site was found to be negative and significant, whereas Secchi depth and accessibility had a significantly positive impact on this probability. A similar positive relationship between Secchi depth and recreational demand was found by Sandström (1999) in another travel cost study on seaside recreation along the Swedish Baltic Sea coast. The estimated model was subsequently used for computing an aggregate willingness to pay for recreational benefits in the counties of Stockholm and Uppsala for a 1-metre increase in average summer Secchi depth. The result is SEK 60 million per year if total travel costs do not include the cost of travel time and SEK 110 million per year if the costs of travel time are included and estimated arbitrarily at 30 per cent of the individuals' wage rate.

In the contingent valuation study, the valuation scenario included in the questionnaire described a nutrient abatement programme and was followed by questions about respondent willingness to pay (WTP) (see Figure 7.5). The payment vehicle used in the questionnaire is an increase in expenses due to higher prices of tap water and agricultural products. An increase in an earmarked taxation was also tested as a payment vehicle in pre-tests of

The water in the Stockholm Archipelago can be improved if measures are taken against nutrient emissions from, for example, agriculture and household sewage.

Suppose that an abatement programme is proposed. According to this programme, farmers and sewage treatment plants in the counties of Stockholm, Södermanland and Uppsala have to finance measures against the nutrient emissions. This would in turn result in increased prices of agricultural products and tap water in these three counties. The following would also happen:

The measures would improve the water quality in the archipelago. For example, the water transparency in the inner and central parts of the archipelago would, on average, increase with about 1 metre in 10 years. This would mean that, for example, in the summer in the inner parts of the archipelago the water transparency would increase from the present average of about 1 metre to about 2 metres in the next 10 years.

As a rule, it would thus in 10 years be possible to see one's feet on the bottom wherever one bathes in the archipelago.

If no measures are taken, the water quality continues to get worse, and the water gradually becomes more turbid.

X. Would you agree or not agree to pay something in terms of increased expenses in order to make it possible to carry out this nutrient abatement programme?

I WOULD DEFINITELY AGREE → ***go to question Y***
I WOULD PROBABLY AGREE → ***go to question Y***
I WOULD NOT AGREE

Y. What is the maximum increase in expenses that you would be willing to pay for this purpose?

Please remember that your income can be used for other expenses too!

Answer: NOT MORE THAN SEK ______PER MONTH DURING THE NEXT 10 YEARS

Figure 7.5 *Valuation scenario and WTP questions used in the contingent valuation survey*

the questionnaire, but not used in the main survey because of the relatively high amount of protest answers, mainly as a result of scepticism about the possibility of earmarking tax payments for nutrient abatement measures in the Stockholm Archipelago.

Table 7.6 Computation of aggregate WTP based on the contingent valuation study

Mean WTP per resident (SEK/year)	Total regional WTP[a] (SEK million/year)	Present value regional WTP[b] (SEK million)
436	624	5057

Notes:
[a] The population in the counties of Stockholm and Uppsala consists of 1 431 700 residents between the ages of 18 and 75 years.
[b] Time horizon: 10 years (as specified in the valuation scenario). Discount rate: 4 per cent.

Table 7.6 reports the results for mean and aggregated WTP. The problem of non-respondents in the survey was approached in the following way. In a short follow-up questionnaire to a random sample of 500 non-respondents, a question was posed about the reason for not completing and returning the original questionnaire. About 25 per cent of the respondents to this follow-up questionnaire stated that they do not visit the archipelago or are not interested in the archipelago. When mean and aggregate WTP were computed, 25 per cent of the non-respondents to the original questionnaire were therefore assumed to have a genuine zero WTP, whereas other non-respondents were assumed to have a WTP that equals on average the WTP of the other respondents who did complete and return the questionnaire.

As shown in Table 7.6, the total economic value of the annual flow of recreational benefits due to clearer bathing water in the Stockholm archipelago amounts to approximately SEK 600 million. The present value of this annual flow is about SEK 5 billion, or about SEK 500 million per year.

3.6 Balancing Costs and Benefits

From the previous sections it follows that the estimated least total costs for accomplishing the 1-metre Secchi-depth increase (SEK 57 million per year) are equal to or less than the recreational benefits as estimated by the travel cost study (SEK 60–110 million per year). However, the cost estimates do not include transaction costs for implementing and enforcing abatement measures, which might increase the total costs considerably, maybe even rendering the net benefits negative. Net benefits are likely to remain positive if the costs are compared to the total benefits estimated by the contingent valuation study (SEK 500 million per year). Based on this finding there are therefore strong indications that it is beneficial for society to reduce the nitrogen load to generate a 1-metre increase in average summer Secchi depth over the next 10 years.

The estimates and results in this study rely heavily on existing natural scientific research. Integration of economics and natural sciences such as ecology and physical geography is vital in order to obtain meaningful economic results that can be used to advise policy and decision-making. The importance of a spatially differentiated (river basin) cost-effectiveness analysis in order to achieve a maximal net gain cannot be understated. Current management of environmental issues within given administrative units can generate the wrong incentives for solving the eutrophication problem of the Stockholm Archipelago. There will be little incentive for an upstream municipality to finance abatement measures improving the water quality for municipalities located further downstream, unless the spatial distribution of costs and benefits is made explicit.

The spatial asymmetry between emission sources' effect on the recipient has vital implications for the result and the choice of policy instruments. It is also important to understand that the choice of target and recipient affects the final solution with regard to total costs and the selection and allocation of abatement measures. Including other recipients with their respective target does not, however, necessarily imply a need for increased abatement. It might very well be that the reductions made in order to achieve a 1-metre Secchi-depth improvement in the archipelago also generates the necessary improvements in other recipient water bodies, such as Lake Mälaren and the Baltic Sea. It can also be questioned whether the 1-metre Secchi-depth improvement target is the optimal one for the Stockholm Archipelago. According to basic economic theory, net benefits are maximized when the marginal abatement cost at the source equals the marginal benefit of the reduction. Based on the questionable assumption that marginal benefits are constant, such an analysis was made by Scharin (2003) in which he came to the conclusion that net gains in that case would be SEK 516 million per year and imply a load reduction of approximately 51.5 per cent.

4. CONCLUSIONS

We believe that the CBA presented here illustrates the need for an advanced linking between economics and natural sciences for analysing complex water resources management issues. Given the general reluctance to use CBA among Swedish governmental and non-governmental organizations outside the academic world, it is hardly surprising that they tend not to take the lead in carrying out similar analyses. An additional reason is probably that similar analyses are likely to need results from the research frontier, which generally calls for involvement of academic institutions. In the case of the Stockholm Archipelago, water exchange models for an archipelago area can serve as an example of a component which is still subject to ongoing research.

This chapter also illustrates the need for an integrated approach to complex water resources management issues. This involves at least three important aspects. First and (almost) needless to say, water issues can clearly not be analysed separately from land issues – coastal water quality problems can generally not be solved without looking at the inflow of pollutants from land. A second aspect of integration is the need for combining knowledge from several scientific disciplines. This calls for joint, multidisciplinary research ventures. However, such co-operation is not always successful. In a discussion of the design of multidisciplinary research, Gren et al. (2002) argue that such joint ventures require a careful balancing between the research project's policy relevance and scientific relevance. There is typically a trade-off involved, which might result in the project getting caught in either of two possible traps. Too strong a focus on policy relevance might imply a risk of reduced scientific relevance through, for example, putting pieces of research results together in a way that is suitable for getting a rapid answer to a policy question but that is inattentive to the peculiarities and uncertainties associated with each piece.

On the other hand, too strong a focus on scientific relevance might imply that the research gets too detailed for being policy relevant. The result of this second trap might be a thorough understanding of individual systems, but not necessarily of how the systems are linked to each other. For increasing the chances of successful multidisciplinary co-operation, the existence of these traps should be taken into account in the design of the co-operative work.

Finally, the need for an integrated approach is also evident from the dependence between the results of a CBA and the design of policy instruments. The results reported for the case of the Stockholm Archipelago are conditional on the possibility to apply spatially differentiated policy instruments in the region. The socio-economic gains suggested by the results of the CBA can be realized with such instruments. However, if uniform instruments are instead applied in the region, total abatement costs would amount to SEK 104 million per year, compared to SEK 57 million per year under a spatially differentiated nutrient abatement policy. The former figure exceeds the maximum benefits derived from the travel cost study, but is still less than the contingent valuation estimate. So a socio-economic gain might still be realized, but it will be a smaller one than in the case of spatially differentiated policy instruments. This dependency between the choice of policy instruments for implementing a project and the results of a CBA of the project illustrates the importance of integrating the issue of policy design into cost–benefit analyses.

NOTES

1. The study was carried out as a research activity within the research programmes Marine Research on Eutrophication (MARE) and Sustainable Coastal Zone Management (SUCOZOMA), funded by the Foundation for Strategic Environmental Research (MISTRA), and the research project Ecological-Economic Analysis of Wetlands: Functions, Values and Dynamics (ECOWET), funded by the European Commission/DGXII Environment and Climate Programme (Contract No. ENV4-CT96-0273) and the Swedish Council for Planning and Coordination of Research (FRN).
2. Secchi depth is a measure of water transparency. It is measured using a Secchi disc; a white disc 20 cm in diameter. This is lowered until it is no longer visible, then brought up until it is visible. The average of these two measurements gives the Secchi depth. It is assumed that perceived water transparency can be measured as the Secchi depth.
3. Since nitrogen (N) is the limiting nutrient of coastal areas of this part of the Baltic (Granéli et al., 1990), phosphorus (P) is excluded from the analysis. The limiting nutrient is the one currently determining the level of primary production. No reductions of primary production will occur by reducing the non-limiting nutrient, that is, phosphorus.
4. Data concerning depositions, land use, retention and other vital characteristics were used in order to estimate the nitrogen discharges from each catchment. Most of these data are available on a sub-catchment level. Catchments 1 to 6 can be divided into 33 smaller sub-catchments in the model. The estimates and results are for the sake of simplicity presented at the level of the six catchments, even though they are available for all 33 sub-catchments. Nitrogen discharges to Lake Mälaren and the final load to the archipelago are estimated using available information about depositions and flows. The nitrogen sink capacity of Lake Mälaren (sedimentation and denitrification) is significant in determining the different impacts of the depositions from adjacent sub-basins. The exchange of nitrogen between the lake basins is in one direction, that is, there is only a flow from upstream to downstream basins and no flow in the opposite direction.
5. SEK = Swedish Crowns. By the end of 1998 (1998 being the price level base year in this chapter), one SEK equalled about 0.10 euro.
6. The potential effect of any kind of payments distorting the costs (such as subsidies or taxes) was ignored in the analysis. The same holds for potential transaction costs for implementing and enforcing abatement measures.

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8. The costs and benefits of implementing the European Urban Waste Water Directive in Greece

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1. INTRODUCTION

The Greek peninsula is endowed with over 13 700 km of coastline. As a result, 'The majority of Greek national territory is coastal land'.[1] A handful of indicators aptly demonstrates the importance of the coast and, by the same token, its vulnerability to human pressures: coastal areas represent 72 per cent of total Greek territory, 86 per cent of the Greek population lives in coastal areas, 88 per cent of employment in manufacture is found in coastal areas as well as 90 per cent of all tourist activities and 90 per cent of energy consumption.[2] Protecting coastal water quality deserves (and has in part acquired) a prominent position on the official Greek policy agenda as well as in environmental activists' plans. This interest is reflected in a number of national and international initiatives addressing the issue of land-based marine pollution into the Mediterranean basin.[3]

The record of policies and measures directed to the protection of coastal waters in Greece shows a mixed picture. On the one hand, within the EU 'Blue Flag' programme almost all Greek beaches (97.14 per cent)[4] meet the highest quality standards. On the other hand, pollution levels are high in the vicinity of large urban centres and industrial sites. As stated in the latest EU report on urban waste water treatment and the implementation of Council Directive 91/271/EEC: 'Greece has been very late in identifying its sensitive areas in August 1999, five and a half years after the deadline mentioned in the directive.' Since then, the Ministry of the Environment has declared that increased financial means will be devoted to the treatment of waste water through its Operational Environmental Programme.[5] On every occasion, and at the latest by 31 December 2005, Greece has to comply with article 4 of Directive EEC/271/91 requiring that urban waste water from agglomerations between 2000 and 10 000 person equivalents (p.e.) will be subject to secondary or an equivalent treatment.

Since 1980, when the first waste water treatment plant (WWTP) was constructed, 280 WWTPs are now in operation serving a total of 55–60 per cent of the population (Tsagarakis and Aggelakis, 2000). Table 8.1 presents the population connected to sewerage systems in the EU. A study of the 241 plants in operation in Greece today reveals that 209 of them are activated sludge systems, 24 are natural systems, five use bio-filters and only three use primary treatment methods. Of these plants, 52.7 per cent are operating, 14.5 per cent are completed but not operating, while the rest face a number of functional problems. Funding for the operation of WWTPs is based on EU Cohesion Funds (75–85 per cent) plus central government and interest concessions (OECD, 2000). It is estimated that fully covering the whole Greek population demands the construction of 2000 more plants, with a capacity of 500–10 000 p.e. at an estimated total cost of 500 million Euros (Aggelakis et al. 1999). Significant funds have already been invested in these plants. An analysis of the cost structure of operating plants in Greece reveals that high personnel and energy costs are important factors for their economic viability. Conventional activated sludge systems are, contrary to expectations, shown to be less economical, a fact that speaks for the need of improving the design, size and operation schedule of the new plants in the future (Tsagarakis et al., 2003).

Future steps towards full compliance with Directive EEC/271/91 should and will take place within the framework of the recent Water Framework Directive's (WFD) evolving 'legacy'. On a European scale, the WFD prominently offers tools in support of an integrated management of water resources. It represents a major and ambitious piece of European environmental legislation, which will serve, in principle, to substitute around 30 previous Directives (from the beginning of the 1970s) with a direct effect on aquatic ecosystems and biodiversity. The new Directive will provide a much more integrated and strategic (river basin) approach to European water policy, explicitly recognizing the interdependencies between ecological and socio-economic realities.

Economic methods and tools are particularly relevant to this task as investments and water resource allocations must be guided by cost recovery and cost-effectiveness criteria and be in line with 'the polluter pays' principle.[6] The use of CBA in Greek public administration is practically non-existent, though the idea of balancing environmental and economic trade-offs is gaining a momentum in Greek courts.[7]

This chapter presents an attempt to estimate the costs and benefits of clean coastal water in Greece. The study site is the Inner Thermaikos Gulf (Thessaloniki bay)[8] where water pollution and clean up is a real policy issue.

Table 8.1 EU population connected to sewerage system

Country and year	Public sewerage system					Independent sewerage	Of which independent treatment (%)
	With treatment				Without treatment		
	Mechanical	Biological	Advanced	Total			
B 1998	—	22.0	16.1	38.1	44.4	17.3	—
DK 1998	1.6	3.4	84.0	89.0	0.0	10.9	10.9
D 1995	4.1	12.2	72.3	91.5	0.6	7.9	—
EL 1997	32.4	14.4	9.6	56.2	11.3	32.2	—
E 1995	10.6	34.4	3.3	48.3	—	—	—
F 1995	—	—	—	79.0	2.0	—	10.0
IRL 1995	24.0	31.8	1.8	57.6	—	32.0	—
I 1995	2.9	36.1	24.1	75.0	—	—	—
L 1999	—	—	—	93.0	—	7.0	7.0
NL 1996	—	19.6	78.1	97.7	—	2.3	—
A 1998	0.5	17.2	63.7	81.4	0.1	18.5	18.5
P 1998F	17.8	26.0	2.3	46.0	36.0	18.0	4.7
FN 1999	—	—	80.0	80.0	—	20.0	—
S 1998	—	6.0	87.0	93.0	—	7.0	—
UK 1997	12.0	52.0	20.0	84.0	10.0	6.0	—

Source: EUROSTAT.

The chapter focuses, on the one hand, on the cost of treating urban waste water discharges in Thermaikos and, on the other hand, on public willingness to pay to ensure the full operation of the WWTPs that discharge into Thermaikos Bay. The main focus of the research presented here is therefore on improving coastal water quality of the Inner Thermaikos Gulf by improving urban waste water treatment. Waste water treatment is considered a 'water service' according to the WFD. The costs of this service have to be estimated and calculated through to the users of the water service (cost recovery). However, clean coastal water generates a variety of wider social benefits. Water quality improvements of Thermaikos bay are therefore valued based on attitudes and preferences of a random sample of Thessaloniki residents.

Public interest in environmental protection in Greece is often accompanied by lack of confidence in public and private management of natural resources, as information on the environment is often not available or accessible to the public. In the study presented here, a random sample from the public were given the opportunity to express their opinions, points of view, attitudes and preferences towards (the need for) water quality improvement in the Thermaikos bay, with the help of the contingent valuation method. Although an individualistic and quantitative based social research method, this public referendum type of approach is also considered to fit in very well with the 'public participatory' approach advocated by the WFD. The study aimed *inter alia* to estimate public willingness to pay for an increase in four-monthly water rates for cleaning up the Thermaikos bay. Costs and benefits were subsequently compared and conclusions drawn as to the economic efficiency of investing in the improvement of water quality of Thermaikos bay.

The chapter is structured as follows. First, a detailed description of the problem is given before we embark on the analytical evaluation of the WWTP cost estimation. Following this, we describe the design and application of a survey-based elicitation of perceived benefits from a clean Thermaikos Gulf and present the main results. We conclude with a summary and an evaluation of the future of cost and benefits estimation in the Greek water policy arena.

2. THE PROBLEM SETTING

The Thermaikos Gulf is a half enclosed shallow sea basin, with a southern opening to the Aegean Sea, approximately 19 km wide, situated in the north-west of the Aegean Sea. The Gulf is bordered to the east by the Kassandra peninsula, to the west by the coast of Pieria and to the north by the coastline of the city of Thessaloniki (see Figure 8.1). The northern part

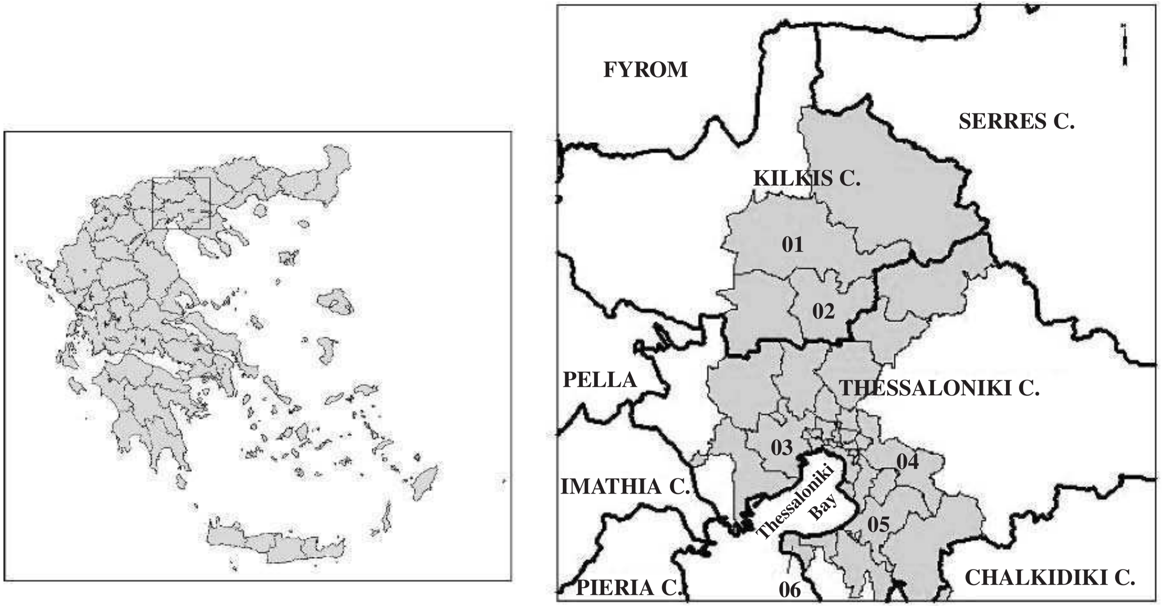

Figure 8.1 *Drainage basin of Thessaloniki bay*

of the Thermaikos Gulf constitutes the Gulf of Thessaloniki. The watershed draining into Thermaikos bay is administratively divided into two counties, Thessaloniki and Kilkis, with 33 municipalities, covering an area of 2027 km^2 and a population of 869 955 inhabitants (Census 2001). Local population density in the watershed reaches 429 inhabitants per km^2 (the Greek average is 83 inhabitants per km^2), mainly due to the presences of the Thessaloniki agglomeration. Parts of the Thessaloniki's municipalities are extremely heavily populated with population density higher than 20 000 inhabitants per km^2, reflecting the important role of the city as one of the major pressures in the area. From 1981 to 1991, the population increased by 10.5 per cent, whereas during the next decade population growth was 5 per cent. The population in the area is continuously growing due to the fertile and highly productive land, increasing industrial development and the consequent demand for labour.

Main sources of pollution contributing to the degradation of the Thermaikos bay include urban sewage and industrial waste, followed by agricultural draining, the discharge of petroleum-based products and other toxic substances. The pollution of the gulf affects all human activities directly or indirectly related to the waters, such as recreation, swimming, sports, living along the coast, fishing, shellfish cultivation, and so on (Karageorgis et al., 2003). Currently, protection of coastal waters is dependent primarily on six operational WWTPs (see Figure 8.1, points 01 to 06) serving 763 250 inhabitants, that is, 88 per cent of the watershed population. Of these, one plant offers primary and two secondary treatment of waste water (extended aeration systems). In 2003, another WWTP[9] will join the existing network serving an additional 4785 inhabitants. Treatment capacity is planned to increase further in the coming years since two major municipalities within the watershed (Halastra and Kilkis), with a total population of 20 240 inhabitants, have already secured the financial funds and publicized a call for tenders for the construction of a new WWTP. That leaves us with a residual of 81 680 inhabitants, who are not connected to a WWTP in the region or, equivalently, an estimated volume of 12.3 million litres of waste water per day[10] discharged without treatment in the Thermaikos Gulf.

3. THE COSTS OF MUNICIPAL WASTE WATER TREATMENT

The costs associated with the operation, modification and improvement of WWTPs are usually determined to a large extent by site-specific conditions and factors. Although efforts have been made to produce reliable cost indicators, the fact remains that 'carrying out cost comparisons between the different extensive techniques at the European level remains very delicate' (EC, 2001, p. 26). Each case should be examined individually, taking into account all the local parameters. In principle, the total cost of a WWTP includes the costs of land acquisition, construction, operation and maintenance (see EC, ibid. and Tsagarakis, 1999). The economic costs of land depend on the required area size, its current and possible alternative future uses and its availability. The construction costs depend on the quality level of waste water treatment required and the capacity of the installation,[11] as well as a number of individual local factors (for example, special site preparations, quality of materials used, tender procedure, housing of process units and so on). The costs of operation and maintenance (O&M) include the costs of:

- Personnel. Total personnel costs depend on the degree of automatization, the size of the installation, the treatment processes and systems, managerial efficiency and so on.
- Energy. Total energy costs depend on the number of kWh consumed per p.e. for different waste water treatment technologies, for example, recycling the biogas produced, automatization etc. Smaller systems usually require more energy in terms of kWh per p.e. than large systems.
- Chemicals. Total costs for chemicals depend *inter alia* on necessary doses, purchasing discounts and so on.
- Maintenance. Total maintenance costs depend *inter alia* on the quantities of spare parts kept in stock, purchasing discount deals and so on.

Additional costs may be incurred in O&M, which cannot easily be put into these categories. These costs include costs related to the necessary infrastructure, building and landscape maintenance, administration, consumables and other expenses.[12] In order to be able to compare annual costs and benefits, the total annual economic cost (TAEC) of the WWTPs in the Thessaloniki watershed are estimated in constant 1999 prices by grouping them into the following three classes:

1. Existing WWTPs. These include both those already in operation (numbered 01–06 in Figure 8.1) as well as the one in Asvestochori-Exochi already constructed and planned to become operational in 2003; their TAEC is estimated on the basis of historical data supplied by the relevant municipalities.
2. Those WWTPs in the municipalities of Halastra and Kilkis where financial means have been secured and a call for tenders already publicized; their TAEC is estimated on the basis of cost data provided by the municipalities concerned.

3. Those WWTPs needed to serve the municipalities (81 680 inhabitants), which are left today without any treatment of their waste water; their TAEC is estimated on the basis of cost coefficients per p.e. for secondary treatment.

Table 8.2 presents the total construction costs and the annual operation and maintenance costs of existing WWTPs.

Table 8.3 presents the total construction costs and the annual operation and maintenance costs of WWTPs in the municipalities of Halastra and Kilkis.

The cost estimates for the third group of WWTPs are essentially dependent on the specific assumptions and cost coefficients used. The municipalities concerned are municipalities with less than 3000 inhabitants each. The extra total capacity needed to support the treatment of these inhabitants' waste water is set at an aggregate capacity for 100 000 inhabitants.[13] For this case, estimated construction cost coefficients average €130 per p.e. and

Table 8.2 Total costs of existing WWTPs (1999 prices, in million €)

Municipality	Population connected	Construction cost (K1)	Annual O&M cost (OM1)
01:Kristoni	1 000	0.19075	0.00626
02: Pedina	1 100	0.15695	0.024394
03: Sindos	745 000	32.25	2.91992
04:Hortiatis	1 900	0.27462	0.030909
05: Thermi	6 000	0.851442	0.040635
06: Michaniona	8 250	2.6015	0.784750
07: Asvestochori-Exochi	4 785	0.26668	0.029954
Total	768 035	36.591942	3.836822

Note: All secondary treatment except 02 (primary treatment only).

Table 8.3 Total cost of WWTPs in Halastra and Kilkis (1999 prices, in million €)

Municipality	Population connected	Construction cost (K2)	Annual O&M cost (OM2)
Kilkis	13 130	2.918112	0.157715
Halastra	7 110	2.08	0.084895
Total	20 240	4.998112	0.24261

operational and maintenance cost coefficients average €6.26 per p.e.[14] Under these assumptions, the construction costs (K3) are estimated at €11.75 million and the operation and maintenance costs (OM3) at €0.57 million (both in 1999 prices). Aggregating the costs for all three classes of WWTPs we have:

- Total construction costs (K1 + K2 + K3) = €53.3 million
- Total (annual) operation and maintenance costs (OM1 + OM2 + OM3) = €4.6 million

The economic lifetime of a WWTP is 30 years for buildings and 10 years for machinery and parts thereof. For this reason the total construction costs are divided into a 'fixed' (FC) component (for example, buildings, technical pre-feasibility studies) and a 'variable' (VC_1) component (parts of machinery such as pumps, motors and so on). The FC are estimated at about 45 per cent of the total construction costs. The annual costs of operation and maintenance (VC_1 and VC_2) represent the other variable cost component. So we have:

- FC ≅ €24 million
- VC_1 ≅ €29.3 million
- VC_2 ≅ €4.6 million

Elsewhere in Europe, some Treasuries (Ministry of Finance/Economic Affairs) prescribe discount rates in risk-free environments, but this is not the case for Greece. As a practical rule, and in order to estimate the present value of the total cost (PVTC) and the TAEC, a 6 per cent discount rate is used based on the interest rate earned, on average, on five- and 10-year state bonds in Greece. Our choice of a 6 per cent discount rate is also backed up by the latest EU guidelines on this matter.[15]

The present value of VC_1 and VC_2 is calculated as usual:

$$\begin{aligned} PVVC_1 &= 29.3 + [29.3 / (1.06)^{10}] + [29.3 / (1.06)^{20}] \\ &= €54.8 \text{ million} \end{aligned} \tag{8.1}$$

$$\begin{aligned} PVVC_2 &= 4.6 + [4.6 / (1.06)] + [4.6 / (1.06)^2] + .. + [4.6 / (1.06)^{30}] \\ &= €67.1 \text{ million} \end{aligned} \tag{8.2}$$

The present value of the total cost (PVTC) is hence equal to:

$$\begin{aligned} PVTC &= FC + PVVC_1 + PVVC_2 = 24 + 54.8 + 67.1 \\ &= €145.9 \text{ million} \end{aligned} \tag{8.3}$$

Finally, the total annual economic cost of all WWTPs in the watershed is calculated from the formula:

$$TAEC = \frac{PVTC}{\sum_{1}^{30} \frac{1}{(1.06)^{30}}} = \frac{146 \text{ mil.€}}{\sum_{1}^{30} \frac{1}{(1.06)^{30}}} \cdot \frac{146 \text{ million}}{14.59} \cdot \text{€}10.0 \text{ million} \tag{8.4}$$

4. ESTIMATING THE BENEFITS OF IMPROVED WATER QUALITY IN THE THERMAIKOS GULF

4.1 Approach

The estimation of the economic benefits was based on a contingent valuation survey. Face-to-face interviews were conducted on site around Thessaloniki bay in June and July 1999. A total of 480 interviews were taken with residents and visitors to Thessaloniki bay, of which 466 were usable. The questionnaire survey was designed to elicit responses in three information categories for each respondent:

1. Attitudes and behaviour towards current water use and knowledge of water quality in Thessaloniki bay.
2. Economic valuation questions regarding water quality improvements in Thessaloniki bay.
3. Socio-economic characteristics of respondents.

Each of these categories incorporated a series of questions, closed and open ended. The structure of the questionnaire is as follows.

The first part of the questionnaire contains questions concerning respondents' opinions about the importance of different social problems, followed by a question specifically about the environmental problems faced by Thessaloniki. These questions are considered important in placing the issue at hand in the context of people's general concerns in their lives (Kontogianni et al., 2001; Langford et al., 1999; 2000). Next, current water use and behaviour was elicited, such as recreational walking, swimming and boating in Thessaloniki bay. In order to assess respondents' familiarity with and knowledge of the subject of the survey, as well as their comprehension of water pollution problems, questions were asked related to their knowledge of the municipal and industrial waste disposal in the Bay, along with a question about the sources of this waste. Respondents were then asked to rate water quality in Thessaloniki bay on a scale ranging from 'poor quality, with untreated waste and garbage' to 'high quality and suitable for swimming', allowing them to express their own perceptions of water quality (Georgiou et al., 1998; 2000).

Following this, respondents were asked how important it was to them personally for Thessaloniki Bay to have cleaner water, and whether they agreed with the statement that the state should invest more in better water quality. Respondents were then asked to state why they felt that water quality in the bay should be improved, as previous research has shown that assessing prior motivation is important in determining respondents' stated WTP amounts (Langford et al., 1999; Mathieu et al., 2000). Finally, respondents were asked whether they would visit the bay more often if the water quality were to be improved, and if they knew of the existence of waste water treatment plants which could, if fully operational, help to achieve this.

The next part of the questionnaire comprised the actual valuation of improved water quality, where respondents are asked first whether they are willing to pay in principle for cleaning up the bay, and, when replying positively, how much they are willing to pay through an increase in their quarterly water rates. The methodology used in this part of the questionnaire follows Bateman and Willis (1999), using an open-ended elicitation method, that is, respondents are given no prompts, but simply asked to state their maximum willingness to pay an amount. Those who refused to pay in principle were asked why they refused, whilst those who responded positively were asked the following WTP question: 'What is the maximum amount that you would like to give through the water rates bill of EYATH for the next 5 years in order to cover the running costs of the waste water treatment plant for these years?'[16]

Following this question, respondents were asked to state their reasons for wishing to participate in funding the running costs of the treatment plant. Respondents were then reminded that they only have limited disposable income to fund the project, and asked to reconsider their maximum WTP amount. A number of standard questions regarding respondent demographic and socio-economic background completed the survey.

4.2 Results

Just over half of all respondents (n = 240) qualified water quality in Thermaikos bay as 'very bad'. Only 19 respondents (4 per cent) replied that they consider the water quality good. Ninety-five per cent of all respondents consider the issue of water pollution in the bay important to extremely important. Only two respondents replied that they do not care about water pollution in the bay. Positive predictors of believing the issue to be 'extremely important' are whether or not someone is a Thessaloniki resident, knowing that there exists a not yet operational waste water treatment plant and whether or not someone is concerned about pollution from the Axios river entering the bay.

There was also a strong positive association between considering water pollution an extremely important issue and being concerned about unregulated construction and development in the area. Moreover, there also exist positive associations between finding the issue of the bay extremely important and being motivated by moral issues surrounding environmental protection and caring about future generations. Those who consider the issue extremely important are also more likely to state an intention to visit more frequently if the quality of the bay is to be improved.

A slightly lower number of respondents not only consider the issue of water pollution in Thermaikos bay (extremely) important, but also are willing to financially commit themselves to solving this problem by paying higher water bills. Almost 70 per cent of all respondents (68.9 per cent) said that they would, in principle, be willing to pay extra water rates to improve the water quality of the bay. The belief that the state or central government should invest in better water quality is a strong predictor of a positive response to the WTP in principle question, suggesting that people are willing to pay higher water rates if they believe that the state will invest in better water quality as well. Being a member of an environmental protection organization also helps explaining a positive response, whilst being unemployed or a student is most likely to result, on average, in a negative reply to the WTP in principle question, suggesting a significant income constraint.

An interesting finding is that fishermen or those respondents involved in the shell fishing industry in the bay are unlikely to say yes to the WTP in principle question, because they feel the state should pay. There exists a strong correlation between those stating they feel financial means should be spent on fishing and shell fishing purposes and those stating that the state should pay for cleaning up the Thermaikos bay. This result was confirmed when debriefing the interviewers about the interview results. Some of the respondents were (shell) fishermen, who have a strong union and lobby trying to convince the state government to provide a cleaner environment and other incentives for shell fishing in the area. Members of hunting clubs were also more likely to say that the state should pay for cleaning up Thermaikos bay.

The feeling that 'investments from the state for better water quality should be increased' is one of the main determinants to reply yes to the payment principle question. This variable is in turn dependent upon a number of different explanatory factors, such as wanting to keep the option open to use the bay in various ways in the future, also for the sake of future generations, and moral stands on having a right to a clean environment. This suggests that belief in the need for more state investment and involvement and ethical-altruistic motivations are linked in this specific water pollution context. However, respondents who claimed that the state should increase spending on improved water quality in the bay also had high levels of knowledge of the sources of municipal pollution around the bay, perhaps reflecting the belief that the authorities should not only pay because of moral and ethical considerations regarding the environment and future generations, but also because those who cause the pollution, including the local authorities, should clean it up (polluter pays principle).

Table 8.4 presents the willingness to pay amounts for the whole sample and the sample broken down according to the different motivations held by people for wanting an improvement in the water quality of Thermaikos bay. The first column states the motivation, while the second and third columns give the number and percentages of the sample of 466 respondents who stated these motivations. The next three columns show the mean WTP associated with these various motivations and their 95 per cent confidence limits, calculated by applying a non-parametric bootstrap to the distribution of WTP values (Efron and Tibshirani, 1993; Langford et al., 1998a). The whole sample mean WTP, presented in the last column, including those who stated a zero WTP or refused to pay in principle, is a €15.22 increase of the quarterly water rates payment.

Looking at Table 8.4, those respondents who are motivated to pay for the sake of future generations stated, on average, the highest WTP amount of

Table 8.4 Willingness to pay amounts categorized according to different motivations

Motivation	Number	%	−95% CI	Mean WTP	+95% CI	Whole sample mean
Swimming	26	5.6	7.88	12.31	16.7	0.69
Fishing	28	6.0	7.18	11.28	15.4	0.68
Bay smells	136	29.2	13.28	15.25	17.19	4.45
General environmental concerns	175	37.6	13.9	15.6	17.3	5.85
Option value?	104	22.3	10.2	12.4	14.65	2.77
Biodiversity preservation	180	38.6	11.99	14.08	16.17	3.56
General moral issue	146	31.3	13.66	15.6	17.45	4.89
Future generations	218	46.8	14.47	15.4	17.42	7.2
Total (excluding zeros)	321	68.9	21.42	22.1	22.74	15.22
Total (including zeros)	466	100	14.21	15.2	16.27	15.22

€7.2 every four months, followed by concerns about the environment in general (€5.85), moral concerns (€4.89) and alleviation of the unpleasant smell from Thermaikos bay (€4.45). These results demonstrate that the stated WTP amounts for the improvement of water quality in Thermaikos bay are based on a complex variety of concerns, which range from practical and use reasons, such as smell and recreational activities, to altruistic and ethical concerns, such as future generations and moral matters.

Looking at other factors, which help to explain the variability found in stated WTP, the predictors of willingness to pay are household income, having young children under five (basically another indicator of the concern for future generations motivation), being a member of an environmental protection organization (basically another indicator related to concerns about the state of the environment in general), knowledge of the existence of a non-operational treatment plant and the belief that industry is the main source of pollution.

Also from respondents' answers to the question why they are willing to pay exactly the amount of money they just stated, it can be seen that the reasons behind respondents' WTP are rather complex and multidimensional. The most common reason heard relates to simply wanting to have a clean sea, which does not smell, but this was very often stated in the context of the city as a whole and wanting to feel good about one's own direct surrounding and living conditions. People were able to see economic possibilities beyond their own personal use, and saw the bay as a potentially positive asset for Thessaloniki as a whole, which was at present not the case.

The different motivations stated by respondents for their willingness to pay for improved water quality provide interesting evidence for individuals responding both as citizens and consumers, following the arguments developed, for example, in Sagoff (1988) and Brouwer et al. (1999). Quality of life was often cited in this respect, and people were quite ready to take moral responsibility and act as citizens as well as consumers. The highest WTP amounts are given for moral reasons and future generations, and many of the comments for stating a specific WTP amount support the view that many respondents are perhaps expressing no more than a moral commitment to help financing an environmental programme (Vadnjal and O'Connor, 1994).

On the other hand, the results provide evidence that people are capable of considering wider environmental, social and ethical issues when considering their WTP in contingent valuation surveys. Of course, not all motivations can be simply classified as altruistic or 'citizen oriented', but even consumer behaviour was seen to be complex. On average, respondents stated a higher WTP to remove the smell, and from the overall results of the survey, it was clear that the immediate visual unpleasantness of the bay provided a strong motivation for wanting the bay to be cleaned up, and respondents were also willing to financially commit themselves to achieve this. However, from the qualitative responses, it appeared that people broaden the issue to include general improvements to the city of Thessaloniki, with the condition of the bay being a particularly prominent image in need of improvement. A beautiful Thessaloniki would help them feel better about themselves, perhaps in a similar way that pledging money for moral and ethical reasons or for future generations can also strengthen self-esteem and self-identity. Respondents also considered the increased options for tourism and visitors, not just themselves, but again for the 'good of the city'.

5. DISCUSSION AND CONCLUSIONS

The implementation of the EU water framework Directive 2000/60 will be dependent on proper implementation by the European member states. The European Commission is therefore developing a common implementation strategy of the WFD economics in collaboration with EU member states and accession countries. The WFD economics bring the issue of cost-efficiency in infrastructure planning to the foreground. In those cases though, where the cost for cleaning up the watercourses are deemed 'disproportionate' unavoidably the issue of quantifying the prospective benefits of clean water becomes apparent. Monetary valuation of water resources also becomes necessary when forgone benefits of clean water have to be included in a full cost pricing policy.

Whereas the cost estimation of investments in preserving water quality seems in the majority of cases to be a data demanding but, nevertheless, straightforward exercise,[17] the economic estimation of benefits is still a controversial topic, much debated within the social sciences research community as well as by policy practitioners. Diverging value judgements and ethical beliefs seem to lie at the heart of the disagreement, but also a different perspective on how environmental decision-making processes (should) look. Balancing costs and benefits of major public investments affecting the environment has also, in Greece, become a cornerstone of a persuasive and effective public policy. The ensuing conflicts are nevertheless not cast in terms of monetized costs and benefit but rather in terms of a 'conflict of constitutional rights', that is, a conflict between free enterprise versus environmental protection. Cast in these terms, the issue falls into a notorious 'legal vacuum' since no trade-offs are allowed between equally strong sanctioned constitutional rights! In practice, the contesting parties usually invoke a number of political, social, financial and cultural arguments, but in the end the constitutional judge is left with the burden of decision.

The present study was accordingly not planned and financed as an explicit input to the real environmental problem-solving in Greece. It represents rather a modest contribution to highlight the potential role that a cost–benefit analysis can play in assisting the process of environmental decision-making in Greece. The controversial nature of economic valuation notwithstanding, we apply in the present chapter the contingent valuation methodology in order to estimate the social benefits from cleaning up the Thermaikos bay. We compare the benefits with the relevant cost by expressing them both in annual equivalents. The estimated benefits per person per four months equal €15.22 or €45.66 per person per year. Projecting this amount to the population of Thessaloniki (750 000 inhabitants) we have a total annual social benefit of €34.245 million. Subtracting the estimated total annual economic cost of operating the WWTPs (€8.652 million) a net benefit estimate of €25.593 million remains. The estimated profitability of cleaning up Thermaikos bay has, however, a number of important characteristics:

- The cost is underestimated in as much as the cost of inlet and outflow networks specifically needed for the connection of municipalities to the WWTPs is not included.
- The benefits are underestimated in as much as the annual benefits per capita are projected to the inhabitants of Thessaloniki only.
- Both cost and benefits are underestimated since agriculture, a major source of nutrients into Thermaikos bay, and mussel farming, a major user of good water quality in Thermaikos bay, have not been included in the survey.

A future application of social benefits estimation in Thermaikos bay should be able to narrow down the range of uncertainty in money estimates of both costs and benefits. Nevertheless, the estimated annual net social benefit of €25.593 million is important, because, it represents a sign of future investment funds that the residents of the Thessaloniki region are willing to pay for securing a better water quality. However, to conclude, we have shown much more than this – WTP has not been expressed as a simple consideration of consumption of the environmental good in question, but as a complex mixture of citizen-based and consumer-based preferences. These include the environment of Thessaloniki as a whole, and may be linked to issues of self-identity and pride in the city, as well as higher moral and ethical considerations. For policy-makers, this means that an acceptable policy for cleaning up the bay need not be based solely on considerations of water quality and extended use of the bay for a variety of activities, but on appealing to citizens about the quality of their local environment and their role in the future environmental quality of Thessaloniki as a whole.

NOTES

1. OECD (2000, p. 131).
2. Ibid. p. 132.
3. See YPEXODE (2002), OECD (2000), UNEP/MAP (2001).
4. As reported in the Ministry for the Environment's web page: http://www.thalassa.gr/2002/index.html.
5. 'The programme provides considerable weight to the treatment of liquid wastes at the national scale, with the construction of waste treatment facilities in settlements larger than 15000 inhabitants' (YPEXODE, 2002, p. 88).
6. See articles 5 and 9 of the WFD.
7. Siouti (2002, p. 32).
8. In the present chapter, the terms 'Inner Thermaikos Gulf', 'Thessaloniki bay' and 'Thermaikos Gulf' are used interchangeably.
9. In the municipality Hortiati.
10. Calculated on the basis of (estimated) waste water production of 150 l/inhabitant/day. This is a conservative estimation, since for large urban centres the number may reach 200 l/inhabitants/day or, including tourism, even 300 l/inhabitant/day (Graziou and Anagnostopoulos, 1999).
11. As a general rule, construction costs are positively correlated with the planned level of treatment and negatively correlated with the planned capacity as a result of scale effects. See Tsagarakis (1999).
12. The operation of WWTPs presupposes obviously the existence of an inlet network in place (or, at least, the transportation of waste water loads to the treatment plant by trucks) as well as the existence of an outfall network for disposing of the treated water to the sea. Estimating the costs of pipelines outfall and inlet networks needed for the proper operation of the WWTPs is a difficult task since it depends on the precise length of the network in every individual WWTP. According to information from construction engineers, construction and maintenance costs of the sewerage network are higher than the construction cost of WWTPs and in some cases much higher, as is for instance the case in the municipality of Halastra. In Halastra the funding of outfall pipelines and inlet networks has been approved recently together with the construction of a WWTP. The construction costs of the WWTP (capacity 12000 p.e.) in 2002 prices amounts to €2.3 million while the costs for the outfall pipelines and inlet networks are as high as €17.2 million.
13. Applications for WWTPs by the municipalities in the region since 2001 concern small-scale plants, approximately serving 2000 inhabitants each. This represents a 22 per cent higher capacity than actual number of inhabitants (81680). Planning the construction of WWTPs with approximately 20 per cent overcapacity is a legitimate assumption according to experts. See EC(2001) and Tsagarakis (1999).
14. See Tsagarakis (1999), Stamou et al. (1995) and Zannou and Kopke (2001).
15. See EC (n.d.) and Florio and Vignetti (2003).
16. EYATH: Water and Sewerage Corporation of Thessaloniki.
17. A review of cost-estimation analysis in water related projects is given in Zanou et al. (2003).

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9. Cost–benefit analysis of the Remedial Action Plan to improve water quality in the Great Lakes in Canada

D.P. Dupont and S. Renzetti

1. INTRODUCTION

The Great Lakes are aptly named. Taken together, they hold approximately 18 per cent of the Earth's freshwater and are the largest body of freshwater on the planet (USEPA, 2002). They supply water to 40 million citizens of Canada and the United States and support billions of dollars of activity by providing water for manufacturing, farming, electrical power generation, commercial shipping and recreation. The Great Lakes also provide many ecological services including fish and wildlife habitat, nutrient cycling as well as playing a role influencing continental weather patterns. Industrial, agricultural and domestic use of the Great Lakes, however, has come at a cost. During the period following the Second World War, the rapid increases in manufacturing activity, population and agricultural production in the Great Lakes watershed led to significant deteriorations in water quality in many parts of the Great Lakes (Environment Canada, 1986).

Initial clean-up efforts began in the 1970s. These met with some success but were criticized for not allowing for sufficient public consultation and participation. As a result, the governments of the United States and Canada embarked on a novel approach to environmental restoration where control of local remedial actions was placed in locally organized committees. Under the Remedial Action Plan (RAP) programme, 42 'Areas of Concern' were identified (including Hamilton Harbour) and a community-based remediation process was initiated for each (Sproule-Jones, 2002). The scale of the RAP programme is significant: remedial efforts have been underway since the mid 1980s and expenditures on sediment remediation and waste water treatment have already surpassed $300 million at Canadian sites and several billion dollars at American sites (International Joint Commission, 2003). Future expenditures needed to attain the RAP programme's objective of restoring all sites' beneficial uses are not known with certainty but are estimated to be $2 billion and $7.4 billion, in Canada and the United States, respectively[1] (International Joint Commission, 2003).

In addition to the role played by representatives of local stakeholder groups, an important feature of the RAP process concerned the identification and assessment of potential remedial actions. As described below, the first stage of each local RAP's efforts was devoted to significant efforts by physical and natural scientists to understanding the dynamics of the processes that led to the degradation of the local ecosystem. This degree of care was not complemented, however, with an equal effort to assess the economic aspects of remedial actions. While all potential remedial actions were costed, the economic value of their benefits was rarely assessed. For the Hamilton Harbour RAP, it is clear that this was the case: 'The overall benefit to the Harbour of the Remedial Action Plan is expected to be substantial because of the many spin-off benefits to the economy and an improved image of the community. A specific study of these economic benefits, while it would be valuable, was not part of this update' (Hamilton Harbour Remedial Action Plan, 2002, Chapter VI, p. 1). As we demonstrate below, the implication of this lack of analysis is that the costs of the remedial actions adopted for the Hamilton Harbour exceed any reasonable estimates of the benefits by a substantial margin. Specifically, we estimate that the present value of the RAP costs are 240 million CAN$ while the benefits are only 68 million CAN$.

The remainder of the chapter is organized as follows. The next section provides a brief overview of Canadian water resources policies and management and, in particular, examines the role played by cost–benefit analysis in environmental legislation and decision-making. The third section describes Hamilton Harbour. The fourth details the costs of remediating Hamilton Harbour. It also assesses the process employed by the RAP to identify and select remedial measures. The fifth section examines estimates of the benefits of remedial efforts. The last section compares the estimated costs and benefits of remediating Hamilton Harbour and discusses our findings.

2. THE USE OF CBA IN CANADIAN WATER POLICY: AN OVERVIEW

2.1 The Institutional Setting of Canadian Water Management

The Canadian constitution divides responsibility for water management between the federal and provincial governments. This division, however, is quite unbalanced as the provinces have the major role in decision-making with respect to the allocation of water resources and the protection of water quality (Field and Olewiler, 2002; Percy, 1988). Federal areas of jurisdiction are limited to interprovincial and international waters, waters on Native Canadian reserves and establishing drinking water quality guidelines. Provinces govern the allocation of water through a permit system whereby withdrawals and instream uses (such as hydroelectric power generation) are allowed only after the issuance of a government permit. Water quality is protected by regulations that set out maximum quantities (or in some cases concentrations) of substances that may be deposited into water bodies. Water quality is also protected indirectly through legislation regarding agricultural waste management practices, commercial forestry practices, land use restrictions and shoreline development. An important feature of almost all provincial laws and regulations governing both the allocation of water and the protection of water quality is the uniform lack of reliance on economic instruments to effect government objectives (Dupont and Renzetti, 1999). For example, provincial governments issue water withdrawal permits at little or no charge to applicants (although the province of Alberta has recently moved in the direction of establishing a limited market for its water withdrawal permits – see Horbulyk and Lo, 1998). In the area of water quality protection, provincial laws follow the 'command and control' approach where polluters' emissions levels are set out in permits. Polluters face the possibility of paying fines if they exceed permitted levels of emissions. A limited number of provinces such as Ontario have extended their water pollution control frameworks by introducing requirements for polluting firms to adopt best available technology (BAT) or best available technology economically achievable (BATEA) to decrease emissions.

In addition to the federal–provincial division of responsibilities, Canadian water management is fragmented along other lines as well. Perhaps the most important division occurs in the provinces' regulation of the allocation of water and the protection of water quality. Despite the fact that these are both primarily provincial responsibilities, their regulation usually is conducted separately by different ministries within the government. The allocation of water resources is usually the responsibility of a Ministry of Natural Resources whose primary focus is on promoting the (sustainable) exploitation of natural resources while the protection of water quality is usually the responsibility of the provincial Ministry of Environment. Furthermore, other facets of provincial water management such as the regulation of municipal water utilities and the regulation of hydroelectric power generating facilities, farms, forestry operations and shoreline land use are allocated to different ministries. In the 1970s and 1980s an effort was made under the federal Canada Water Act to address these fragmentations by establishing river management boards that were to be managed jointly by the federal and provincial governments. The boards were to be a mechanism for integrating all aspects of water management into one agency that would have responsibility for an entire river basin. Unfortunately, most provinces viewed these boards as an attempt by the federal government to usurp their constitutional authority in water management. This distrust undermined the workings of most boards.

This situation of divided and, in some cases, overlapping, jurisdiction over water resources is relevant to the analysis below. Both the federal and Ontario provincial governments have jurisdiction over the Great Lakes. The federal government's role stems from its international treaty obligations under the Boundary Waters Treaty (1909) and from its role in regulating commercial navigation and commercial fisheries. The provincial government's role stems from its jurisdiction over the allocation of water, the protection of water quality (especially for drinking water) and the regulation of commercial enterprises which may affect water quality (such as farms). This shared responsibility for the Great Lakes means that the two levels of government must co-operate closely for programmes and policies to be successful (Sproule-Jones, 2002).

2.2 Economic Analysis in Canadian Water Policies and Legislation

Economic analysis has historically played a small role in Canadian water resources management. None of the most important pieces of federal legislation such as the Fisheries Act (1868), the Canada Water Act (1970), the Canadian Environmental Protection Act (1988) or the Canadian Environmental Assessment Act (1999) require the assessment of the costs and benefits of proposed projects or the government's own initiatives. There are two specific instances, however, where water-related federal legislation comes close to mandating cost–benefit analysis. The first is the requirement found in the Canadian Environmental Protection Act that the government must 'take preventative and remedial measures to protect, enhance and restore the environment' (s.2.1(a.1); Government of Canada, 2003) and, in deciding what actions to undertake, the government is required to consider, amongst other factors, 'the positive economic impacts arising from the measures including those cost-savings arising from health, environmental and technological advances and innovation, among others and . . . any other benefits accruing from the measure' (s.2.1.1(b); Government of Canada, 2003).

The second instance is implicit in the Canadian Environmental Assessment Act. This legislation, which is concerned with the application of environmental assessments to projects under federal jurisdiction, allows the government to specify the issues that an environmental assessment review panel must consider in its appraisal of a project. An example of this occurred when the federal government conducted an environmental assessment of the construction of a dam on the Oldman River in southern Alberta. Details regarding this case are provided in the case below. It is important to remember, however, that the federal government is not required to direct its panels in this way and, as Weick (1993) and Hazel (1999) point out, it has often demonstrated considerable reluctance to do so.

At the provincial level, the situation is largely the same with most provinces' major water-related and environmental laws eschewing economic analysis of proposed projects or government actions as well as reliance upon economic instruments (Percy, 1988). Virani and Graham (1998) note that there is a significant variance in the application of economic analysis of environmental legislation across provinces. As the authors conclude 'some provinces do not use these tools at all, while others apply CBA and CEA extensively to environmental policy' (pp. A1–100). However, those provinces such as Ontario who have applied economic analysis 'extensively' were found to use it primarily as an *ex post* method of assessing the impact of regulations rather than as part of the design and assessment of new regulations.

BOX 9.1 COSTS AND BENEFITS OF THE OLDMAN DAM

The Oldman dam project was intended to provide a number of benefits to southern Alberta including increased water for irrigation, more reliable water supply for local cities and industry and recreational opportunities. Critics of the project argued it was an example of outmoded supply-orientated water management and would only provide marginal benefits to an already subsidized industry while causing significant environmental and social damage.

The federal government was reluctant to anger the government of Alberta (a major supporter of the dam) by invoking the Canadian Environmental Assessment Act to consider the project and only did so under court order. Because of the delays associated with the legal arguments regarding whether the federal government could and should conduct an environmental assessment, the Environmental Assessment and Review (EAR) Board was actually convened after the dam was built.

The federal government ordered the EAR Board to consider, amongst other matters, whether the construction and operation of the dam were justified on economic grounds. Although it did not conduct its own cost–benefit analysis of the project, the EAR Board considered the analysis submitted by the project's proponents and the assessments of that analysis done by both opponents of the dam as well as its own technical experts. Specifically, the board first considered whether the construction of the dam was justifiable on economic grounds and second, given that the dam was already built when the board was convened, whether the operation of the dam was justified given that its construction costs were sunk.

The board concluded that the construction of the dam was not justified on economic grounds. It found the proponent's cost–benefit study flawed in that it erroneously included secondary spin-off effects in its welfare analysis (thereby confusing income redistribution with income creation) and it used an unreasonably low discount rate (thereby favouring a long-lived project such as a dam). In addition, the board found that the operation of the dam could not even be justified on economic grounds. The board argued that the social, economic and environmental costs arising from the dam's operation exceeded the benefits to agricultural operations and municipal utilities. Finally, because of the disputes that had arisen over the proponent's cost–benefit study of the dam, the board recommended that the federal government revise the environmental assessment legislation to require project proponents to justify projects with cost–benefit studies following established guidelines. This last recommendation was not acted upon by the government and the dam is in operation today.

Sources: Federal Environmental Assessment Review Office (1992); Hazel (1999); Weick (1993).

Furthermore, analyses were often constrained by limited data and resources. The case study of Ontario's analysis of regulations to combat mercury pollution that is described in Box 9.2 is a case in point. More recently, however, the Ontario government has enacted the Regulatory Impact and Competitiveness Test as a means of conducting economic analyses of proposed regulations. The tests are supposed to ensure that benefits exceed costs and that the regulations are cost-effective (Virani and Graham, 1998).

Dupont and Renzetti (1999) highlight the shortcomings of provincial water allocation regulations. With the exception of Alberta (Horbulyk and

BOX 9.2 THE COSTS AND BENEFITS OF SEDIMENT REMEDIATION IN THE ENGLISH–WABIGOON–WINNIPEG RIVER SYSTEM

The English–Wabigoon–Winnipeg (EWW) river system is located in a sparsely-populated part of north-western Ontario near the Manitoba border. At the eastern end of the river, a paper mill and chlor-alkali plant discharged mercury into the river for many years. It was estimated that between 1962 and 1971, 11 metric tonnes of mercury were discharged into the EWW river system. These discharges resulted in the contamination of a number of fish species. These fish made up an important part of the diets of members of two small Ojibiway communities (Grassy Narrows, pop. 624 and White Dog, pop. 827) along the river as well as forming the basis for valuable sports fishery. Because of the elevated mercury levels found in these fish, the Ontario government banned all commercial fishing along the river and initiated a food substitution programme in the Ojibway communities.

After process changes at the polluting mills reduced mercury inputs to the system, mercury levels in fish began to decline slowly. The Ontario government investigated remedial efforts that would accelerate the reduction in mercury levels. The two preferred options were removing the contaminated sediment and capping the remaining riverbed with clean clay and sand. The present value of these actions was estimated to be $13.5 million (CAN 1984$). Unfortunately, government scientists were unable to establish with precision the impact that the remedial actions would have on the rate of reduction of mercury contamination levels. As a result, they were unable to estimate when mercury concentration levels would fall below the desired level of 0.5 parts per million for commercially caught species.

This lack of scientific information was a major impediment to estimating the benefits of the proposed remedial actions. In general, it was expected that there would be several categories of benefits including the following:

- reduced health risks to members of local communities;
- increased commercial and sport fishing opportunities;
- increased food fishing opportunities; and
- non-user benefits (to other residents of Ontario).

- Social benefits due to 'reduced social disruption to the Native people of Grassy Narrows and White Dog.' (Donnan, 1986, p. 4).

In addition to a lack of scientific information, there were other challenges confronting efforts to estimate the benefits of remedial actions. Tragically, the native communities' exposure to mercury contamination had been so pervasive that it was thought by medical experts that reducing future mercury concentrations would only avoid damages to future generations. The health benefits of reduced mercury exposure for the existing population were expected to be quite small. As a result of these constraints, researchers were only able to estimate the monetary values for commercial and food fishing. The benefits to reduced health risks and the other categories listed above were not estimated. The present value of the restricted set of benefits was calculated to be only $1.0 million.

The author of the report, while acknowledging the incompleteness of estimated benefits, recommended against remedial efforts,

Financial values of the benefits alone were insufficient to

> offset the costs of remedial projects. A monetary value of at least $12.5 million (in present value terms) would have to be attributed to possible reduced health risks, benefits associated with sport fishing and other amenity benefits in order to justify the cost of the dredging project. (Donnan, 1986, p. 56)

Source: Donnan (1986).

Lo, 1998), all provincial governments provide access to water to self-supplied users at little or no cost and with little analysis of the costs and benefits of proposed withdrawals. Similarly, provincial water quality legislation has allowed polluters free access to the assimilative capacity of receiving waters so long as sewage treatment plants, manufacturing plants, farms and other emitters do not exceed legislated levels of effluents. Most recently, provincial governments have acted to confront the challenges to water quality posed by agricultural run-off and other forms of non-point pollution (Goss et al., 2002). Given the difficulties presented by the costs of measurement, observation and enforcement, most provincial governments have relied primarily upon command and control style regulations that specify appropriate technologies and production processes. In some cases, such as the province of Quebec, these regulations have been backed up with the threat of fines for non-compliance.

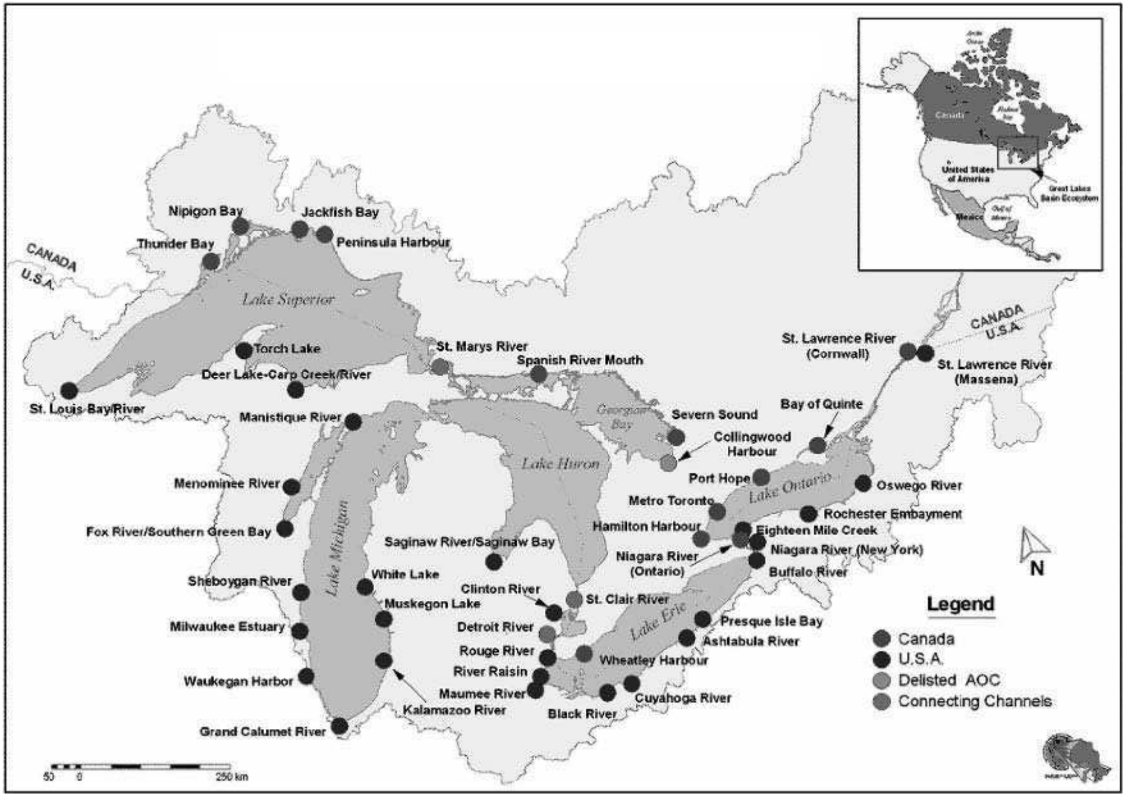
2.3 The Remedial Action Plan Programme and Economic Analysis

The decades following the Second World War witnessed industrialization, rapid population growth and economic development in the Great Lakes basin. In part due to a lack of appropriate government regulation, this growth resulted in serious declines in the health of the Great Lakes ecosystem. This environmental damage resulted in closed beaches due to unacceptable fecal coliform counts, fish-eating advisories, contaminated sediments and significant reductions in fish and wildlife populations. This situation continued largely unabated until the 1970s when Canadian and American governments enacted some of the earliest environmental laws and regulations. The governments' initial responses relied upon traditional 'end of pipe' regulations that set upper bounds on emissions into the Great Lakes. These early efforts had some success and water quality in the Great Lakes began to improve (Environment Canada, 2003). However, governments were criticized for not doing enough and for failing to provide opportunities for public consultation and involvement in decision-making. As a result of these criticisms, governments sought a novel approach to water quality protection in the Great Lakes. This approach, following on pressure in the mid-1980s from the International Joint Commission (IJC),[2] took the form of the Great Lakes Remedial Action Plan (Sproule-Jones, 2002, provides an excellent analysis of the RAP programme).

Remedial Action Plans were developed and implemented at 42 'Areas of Concern' (AOC) on the Great Lakes – one of which was Hamilton Harbour. Figure 9.1 shows the location of the AOCs throughout the Great Lakes. The goal of each RAP was to restore beneficial uses, both ecological and cultural, in the degraded AOC. Each RAP was directed by a committee of officials from federal, provincial/state and local governments as well as representatives of stakeholder groups. These representatives were drawn from industry, environmental groups, academia, and other segments of the public.

The RAPs were instructed by the IJC to carry out their work in three stages. The first stage was to conduct the scientific research needed to assess the state of each AOC's ecological health and the mechanisms or processes responsible for the loss of ecological integrity in these areas. The second stage was to identify, assess and begin to carry out remedial actions. Finally, stage three was to complete the remediation and restore the beneficial uses that had been impaired in the AOC. The last stage would lead to the delisting of the AOC.

An important feature of the RAPs concerned their assessment of potential remedial actions. All of the RAP committees were directed by the appropriate federal or provincial/state government to estimate the capital



Source: Environment Canada website: www.on.ec.gc.ca/water/greatlakes/maps-e.html).

Figure 9.1 Great Lakes' Areas of Concern and operating costs of all remedial measures. In addition, each RAP was expected to be able to demonstrate the ecological impact of each remedial measure in terms of a change in one or more of the impaired beneficial uses. However, there was no directive for the RAPs to estimate the economic value of the predicted change in beneficial uses. As a result, the RAPs did not gather the information needed to conduct a cost–benefit analysis of alternative remedial actions. On the Canadian side, the Ontario government commissioned a limited series of consultant's reports that examined these benefits (Apogee Research International et al., 1990). These reports were constrained in that they examined the values of only those benefits for which each RAP had gathered data. Furthermore, the reports confounded (and most likely overestimated) the measurement of the benefits by improperly combining estimates of the projected welfare improvements arising from increases in local water quality and estimates of increased regional economic activity (as measured by changes in employment levels, tax revenues and aggregate spending) arising from the injection of government funds to support the RAP's clean-up efforts.

3. HAMILTON HARBOUR

Hamilton Harbour is a 40 km^2 embayment located at the western end of Lake Ontario. A watershed of about 900 km^2 drains in the harbour. The harbour is made up of two parts: a 2.5 km^2 shallow area of open water and a wetland (Cootes Paradise) at its western end (see Figure 9.2). There are three factors that reduce the environmental resilience of the harbour and have contributed to it being identified as an Area of Concern. The first is the concentration of heavy industry in the harbour itself (seen in Figure 9.2 as the shaded areas on the southern and western shores of the harbour). The second is the small size of the harbour in relation to its watershed (the ratio of the area of the harbour to the area of the watershed is 1:22.5). The third is the fact that the only outlet from the harbour to Lake Ontario is the narrow Burlington Ship Canal (this can be seen as the thin gap in the landmass forming the western boundary of the harbour). This feature means that, instead of diluting and dissipating inputs to the larger water body, the harbour tends to catch and accumulate them (Ontario Ministry of the Environment, 1988). This feature also implies that actions taken by the Hamilton Harbour RAP will have primarily localized benefits.

In the early 1980s the IJC conducted a large-scale study of the Great Lakes' ecosystem in order to identify the 'hotspots' whose ecosystems were particularly degraded (Ontario Ministry of the Environment, 1988).

Source: Hamilton Harbour Commission website: www.city.hamilton.on.ca.

Figure 9.2 Hamilton Harbour

As part of the scheme to classify areas within the Great Lakes, the IJC developed a list of possible 'Beneficial Use Impairments' and assessed each particular location with respect to this list. In selecting Hamilton Harbour to be included in the list of AOC, the IJC pointed to the following specific impairments:[3]

- restrictions on fish and wildlife consumption;
- degradation of fish and wildlife populations;
- presence of fish tumours or other deformities;
- bird or animal deformities, reproduction problems;
- degradation of benthos;
- restrictions on dredging activities;
- eutrophication with undesirable algae;
- beach closures;
- degradation of aesthetics;
- degradation of phyto-plankton and zooplankton communities; and
- loss of fish and wildlife habitat.

These impairments stemmed from decades of pollution from a number of local sources. The most important of these were the following:

- Industry: Hamilton is a heavily industrialized city and its most important industries are located adjacent to Hamilton Harbour. Factories surrounding the harbour include metal foundries and plants manufacturing metals and metal products, soaps and cosmetics, paints and solvents and chemicals and petrochemicals.
- Waste water treatment plants.
- Non-point sources of pollution including agricultural run-off, storm outfalls and road run-off.
- Commercial shipping and recreational boating.
- Aerial deposition from industry and non-point sources.

Once the major sources of pollution were documented, the RAP committee was able to identify a series of remedial actions. These actions were aimed at reducing or eliminating loadings of pollutants into the harbour, dealing with contaminated sediments, and restoring degraded wetlands and wildlife habitats. Table 9.1 provides a list of the major recommended remedial actions and their respective objectives. It can be seen from Table 9.1 that the actions are a combination of industrial process changes, upgrades to sewage treatment facilities, sediment remediation and wetlands improvements. These actions are complemented with a series of initiatives to educate and involve the public as well as to monitor the impacts of remedial actions.

Table 9.1 Major remedial efforts for Hamilton Harbour RAP

Remedial actions	Objective
Water quality and bacterial contamination	
• Expand and upgrade waste water treatment plants	Reduce loadings of phosphorus, ammonia, suspended solids and bacteria
• Decouple combined sewer-storm water outflows	
Land management	
• Introduce universal residential water metering	Reduce water consumption and flows to waste water treatment plants
Toxics and sediment remediation	
• Change production processes and upgrade discharge water treatment at steel and chemical plants	Reduce loadings of metals and other pollutants
• Remediate contaminated sediments	
Fish and wildlife	
• Maintain, enhance and create fish and wildlife habitats	Restore Cootes Paradise wetland and support fish and wildlife repopulation efforts
Public education, research and other actions	
• Build Marine Discovery Centre	
• Construct shoreline trails	
• Continue monitoring, research and education programmes	

Source: Hamilton Harbour Remedial Action Plan (2002).

4. COSTS OF THE HAMILTON HARBOUR REMEDIATION

Remedial actions in Hamilton Harbour began in 1990 and are projected to continue until 2015. During the period 1990 to 2000, the most important actions undertaken related to process changes at metal foundries and manufacturing facilities, remediation of toxic sediments, improvements to public recreational facilities, and restoration of wetlands. According to RAP documents, total capital expenditures during this period were \$205.38 million (nominal Canadian dollars). The period 2000 to 2015 is also projected to witness significant expenditures, totalling \$645.5 million (nominal

Table 9.2 Costs of major remedial actions for Hamilton Harbour

Activity	Capital cost	Annual O&M cost
Water quality and bacterial contamination		
1990–2000	103.6	NA
2000–15	543.0	
Land management		
1990–2000	1.5	NA
2000–15	10.11	0.777
Toxics and sediment remediation		
1990–2000	53.48	NA
2000–15	64.02	0.048
Fish and wildlife		
1990–2000	14.93	NA
2000–15	8.5	
Public access and aesthetics		
1990–2000	22.83	NA
2000–15	19.7	
Public education, research and other actions		
1990–2000	9.04	NA
2000–15	0.096	1.67
Total	850.81	2.50

Notes: All figures are millions of nominal Canadian dollars. NA = estimate not available.

Sources: Apogee Research International et al., 1990; Hamilton Harbour Remedial Action Plan, 2002; International Joint Commission, 2003 and authors' own calculations.

Canadian dollars). The most important sources of expenditure in this period are expected to be related to upgrading sewage treatment facilities, continuing remediation of contaminated sediments, introduction of universal residential water metering and further expansion of recreational facilities.

Because the cost estimates in Table 9.2 are in current dollars, it is difficult to calculate the real magnitude of resources devoted to remediating Hamilton Harbour. A preferable alternative is to calculate the present value of real expenditures from the perspective of 1990 – when the project began. Doing this, however, requires a few assumptions. First, we assume that the average annual inflation rate between 1990 and 2015 will be 2.0 per cent. Second, we apply a real discount rate of 10 per cent. This rate is chosen as it is the rate prescribed for use in Canadian federal government programme evaluations (Treasury Board Canada Secretariat, 2002). Finally, we assume that the expenditures recorded for the two periods 1990–2000 and 2001–15 were spread evenly over their respective sub-periods. Using these assumptions, it is then possible to calculate what the present value of the stream of past and future real capital and operating expenditures would be over the lifetime of the project. The present value of real capital and operating expenditures is approximately $240 million in 1990 Canadian dollars.

The actions listed in Table 9.1 and costed out here represent a significant and expensive set of initiatives. Before turning to the estimation of the benefits arising from these measures, it is worthwhile noting several features of the remedial actions' costs. First, as the recent IJC report on RAP activities (International Joint Commission, 2003) has pointed out, there has been relatively little explanation of the process employed by the RAP to choose and prioritize individual remedial actions. This lack of information makes it particularly difficult to determine whether the recommended remedial actions are the most cost-effective options available. Another factor inhibiting assessment of the remedial options is that their objectives (see Table 9.1) are stated in quite general terms. The IJC has pointed out that, without specific water quality targets, it is difficult to determine when exactly a remedial action's objectives have been met.

Second, it may be that the significant costs associated with the upgrading of the region's sewage treatment facilities are greater than they needed to be. The RAP documents indicate that the costs of upgrading sewage treatment facilities take into account the effects of expected future population growth on needed treatment capacity (Hamilton Harbour Remedial Action Plan, 2002). At the same time, however, the RAP plans to pursue a programme of installing water meters for all residential water customers. The installation of these meters would facilitate an aggressive water pricing programme aimed at promoting water conservation. There already exists evidence that Hamilton, like most other Ontario cities, underprices both its water and sewage treatment services (Renzetti, 1999). Such a water-pricing programme could limit or even offset the impacts of population growth on water (and, thus, sewage treatment) demands. Hence, taking these price-induced reductions into account in the expected growth rate in water and sewage demands would possibly have allowed the RAP to scale back its planned capital expenditures on upgrading sewage treatment facilities.

Third, the industrial process changes aimed at reducing industrial emissions into the harbour were actually developed under a concurrent Ontario environmental programme titled the Municipal and Industrial Strategy for Abatement (MISA). This programme has been criticized because it established each polluter's required level of abatement by applying BAT and BATEA approaches. There is a substantial amount of evidence that this approach to pollution abatement does not result in the cost-minimizing distribution of abatement responsibilities across polluters (Field and Olewiler, 2002). This is because firms differ in their marginal costs of abatement and efficiency dictates that abatement responsibilities should be allocated so that the marginal cost of abatement is equalized across firms. These observations all suggest that the Hamilton Harbour RAP's remedial actions may not be the least-cost means of achieving its objectives.

5. BENEFITS FROM THE HAMILTON HARBOUR REMEDIATION

Total benefits flowing from the RAP's actions to improve Hamilton Harbour may be divided into two categories: use and non-use values. Use values arise when people receive benefits (that is, enjoy an increase in welfare) through some form of use of the water resource, while non-use values are those that arise when there is neither actual nor planned use of the resource. The latter are sometimes called existence values (Krutilla, 1967). People are willing to pay money in order to secure an improvement in environmental quality or to know that a particular natural resource exists, even though they have neither current nor future plans to avail themselves of the services associated with that resource.

Use values can be further divided into three categories: direct, option and indirect. Direct use values flow to human beings when they benefit from being able to make direct use of improved water quality, either by consuming products that incorporate this higher-quality water or through participation in activities that benefit directly from water quality improvements. Option values arise from the option to make use of an asset in the future (Weisbrod, 1964). In other words, the amount of money that those individuals would be willing to pay to have improved water quality for future, as yet undefined, use. Indirect use values are provided in the form of (freshwater) ecosystem services, such as the downstream benefits of the water purification capacity of aquatic ecosystems. Indirect and option values are sometimes referred to as passive use values (Turner, 1999). Later in this section we provide estimates of some of these values flowing from the various remedial actions undertaken in Hamilton Harbour through the RAP. Prior to discussing the approach used to obtain these estimates, the remedial actions are categorized according to the different definitions of direct, indirect and option use components of benefits.

First, a number of actions have together led to improved recreational fishing opportunities. These actions include: reductions in storm water and industrial effluents leading to better water quality, the restoration of spawning and nursery habitats to support the reintroduction of popular species such as pike and largemouth bass that had been replaced by low-valued carp, the creation of a carp barrier to prevent carp from fouling habitats of desirable species, and the creation of fishing platforms. These improvements provide not only direct use benefits to recreational fishers, but also future benefits to fishers in the region. In addition, a number of the improvements support the provision of ecosystem services, thereby leading to indirect use benefits.

Second, better water quality and facilities to support two new beach areas have improved swimming and windsurfing opportunities in the harbour. These improvements are associated with direct use benefits to swimmers and future benefits to future generations of swimmers.

Third, better water quality and improved access and facilities have enhanced the recreational boating experience in the harbour, thereby creating both direct and future use benefits to individuals interested in recreational boating.

Fourth, restoration of wildlife habitat, the construction of boardwalks, a number of wetlands improvements, and the creation of water-side pathways have improved both access to and the aesthetic experiences of near-shore walkers, cyclists, and birdwatchers. These improvements provide a variety of benefits, including direct use, indirect use, and possible future use, to the inhabitants of the Hamilton Harbour watershed.

Finally, taken as a whole, the remedial actions provide non-use benefits. The clean-up of the harbour ensures the continued existence of a healthy water body for the region. This represents a potentially valuable asset yielding passive use benefits to inhabitants of the region, even if they do not choose to enjoy these benefits actively.

If we are to assess whether the RAP expenditures are likely to have improved aggregate welfare, the challenge is to estimate the economic value of the welfare improvements arising from these changes above. As indicated in the introduction to this chapter, the RAP committee, while recognizing the utility of estimating these values, has not carried out a comprehensive study of the benefits from environmental improvements. As a result, we have attempted to carry out such an exercise *ex post*. While we have some benefit values that are directly attributable to the RAP improvements, we do not have complete information for all categories of benefits. In some cases, we are able to draw upon existing literature to obtain estimates of the likely size of benefits for the Hamilton Harbour RAP. The values from this benefit transfer, however, must be regarded with caution since they are constructed under specific contexts. In some cases, we do not have any estimates for the benefits that might accrue under the RAP improvements.

5.1 Use Values: Direct, Indirect, and Future

The most significant benefits from improvements to water quality are related to enhanced recreational opportunities. Benefit values that are directly attributable to the improvements in Hamilton Harbour are available from a contingent valuation study conducted in 1995 (Dupont, 2003). Using a dichotomous choice framework, a questionnaire described improvements to survey respondents and asked them to vote in favour or against increases to their water bills in order to obtain such benefits. The resulting benefit values are hence willingness-to-pay welfare measures for stated improvements. In total, 713 individuals participated in a general population mail survey, representing a response rate of 63.5 per cent, thereby suggesting a great deal of interest on the part of the residents of the area around Hamilton Harbour.[4] By asking further questions about actual and planned participation, respondents were divided into active users, potentially active (or future) users, and passive users of the harbour. This was subsequently used to estimate the direct, future, and indirect use values.[5]

Direct use values from improved swimming were on average \$57.57, while direct use values for improved recreational boating were \$33.13 and direct use values for improved fishing were \$15.40.[6] Future use values for the three activities were \$32.65 for swimming, \$19.65 for boating and \$30.23 for fishing. All values are in 1995 Canadian dollars and are per household per year.

In order to obtain estimates of the aggregate amount of benefits that might flow to individuals in the Hamilton Harbour area, estimates are needed of the total number of households and the proportion of active and future users in each use category.[7] These household-level estimates must then be aggregated to yield water basin-wide benefit estimates. According to Statistics Canada's 1996 Census, there were 173 120 families in the Hamilton Census Metropolitan Area. Dupont's survey provides estimates of the number of households who have undertaken the recreational activity recently and are considered active users or who plan to do so in the future and are considered future users. These figures from Dupont's survey are used to calculate the proportions of the sample population who are active and future users. Given the representative nature of the sample data, this is considered a reasonable approach. Applying the proportions for active users to the Statistics Canada Census data on households yields the following estimates for the number of active user households in the Hamilton Harbour watershed for each activity: swimming 7444 households; fishing 12 205 households; and recreational boating 22 506 households. Using the proportions for future users yields the following numbers of future user households by activity: for swimming 94 870 households; for fishing 78 077 households; and 70 287 households for boating.

The number of potential or future users is very large especially for swimming and fishing when compared to active users. However, this may be seen as a reflection of the previous damage done to the harbour, which prevented residents from enjoying recreational activities involving close water contact. By multiplying per household values by the total number of active user or future user households, aggregate annual household benefits are obtained for each of the three categories of direct and future use benefits. For swimming the annual direct (active) use benefits are $428 551, for fishing benefits $187 957 while for boating they are $745 624. The corresponding annual future benefits for swimming are $3 097 506, for fishing $2 360 268 and for boating $1 381 140.

In order to obtain the present value of these benefits, we assume that they do not begin until 1995 (at that stage enough of the remediation work had been completed for increased swimming, fishing, and boating opportunities to become significant), and they are assumed to flow up to and including 2015 (when all RAP remedial activities are scheduled to be completed). Finally, we continue to assume a real discount rate of 10 per cent.[8]

Households undertaking recreational activities such as cycling, walking, birdwatching, and windsurfing also benefit from improvements under the RAP. However, there are no Hamilton Harbour specific estimates of the relationship between the value assigned to these activities and changes in water quality. Indeed, improvements to these types of activities have not been much pursued in the benefit valuation literature, although we have been able to find estimates in the literature for two of the activities: birdwatching and windsurfing.

When including estimates produced from a different context, that is, benefit estimates not calculated for the specific environmental improvement being evaluated, a technique called benefit transfer is employed to obtain estimates of values of interest. Benefit transfer or environmental value transfer takes place when estimates of benefits obtained from one location (and within a specific context) are used at the location of interest. The implicit assumption is that the two locations are perfect substitutes.

Generally, two methods have been used when undertaking benefits transfer. First, the mean estimated unit value from one location is applied to the second location with appropriate adjustments made for any currency differences in the case where the locations are not in the same country. Second, the estimated benefit function itself is used to make the transfer; this means that the coefficients from the estimated willingness-to-pay function for the first location are used with the second location's mean values for the explanatory variables. Assuming the relevant data are available, this second approach is preferred. Simply put, more information is being used to make the estimate of the benefit (Pearce et al., 1994). A number of authors have cast doubt upon the usefulness of the benefit transfer procedure and a number of efforts have been made to examine it (Boyle and Bergstom, 1992; Brouwer, 2000; Brouwer and Spaninks, 1999; Downing and Ozuna, 1996). While benefit transfer is frequently used – in part because of the large cost of undertaking surveys to obtain environmental values from the public, it must be done with caution and the estimates regarded as illustrative, rather than definitive.

A review of the relevant literature provides a few examples of estimates for windsurfing and birdwatching benefits. However, they are not easily transferable to the Hamilton Harbour case since the sites are not really substitutable. Wellman and Noble (1997) provide estimates, for example, of the value of windsurfing opportunities in the Corpus Christi, Aransas, and Upper Laguna Madre estuaries of Texas. Using a travel cost approach, the authors estimated that the net willingness to pay for windsurfing was as high as \$828 per trip in US dollars. Similarly, travel cost estimates of the willingness to pay for birdwatching in the area, which is home to over 400 bird species and, thus, one of the premier birdwatching spots in North America, are \$91 US per trip. These numbers come from estimating the travel costs of sports enthusiasts who may come from all over the United States for the opportunity to enjoy a unique experience. For our purposes, they are likely to be overestimates of the benefits obtainable in the Hamilton Harbour area. The reasons for this are twofold. First, the bird and wildlife species and water/wind conditions for windsurfers are replicated in many other locations around the Great Lakes and are, therefore, not unique. Second, birdwatchers and windsurfers are drawn only from the local population.

The only other estimate available for birdwatching comes from a dichotomous choice questionnaire similar to the one done by Dupont for Hamilton Harbour. The study was undertaken in the Vendicari region of Italy in 1994 (Signorello, 1998). Interviewers spoke in-person with 293 birdwatchers and obtained aggregate annual benefits of between 231 680 000 to 274 400 000 lire per year. Using an exchange rate of 1 lire equals 0.000815 Canadian dollars, this works out to between CAN \$190 000 and CAN \$225 000 of dollars per year. This represents the use of the less preferred mean unit value transfer approach since the benefit transfer function was not reported by the author.

So far we have concentrated on the direct benefits from Hamilton Harbour RAP improvements. To a large extent these remedial actions lead to ecosystem benefits such as the provision of habitat for desired fish and wildlife. These ecosystem improvements, in turn, provide indirect benefits to people living in the watershed. These indirect values cannot be disentangled from the previously discussed direct and option values enjoyed by people who derive utility from fishing, walking, and so on. Obviously, other types of ecosystem benefits include what de Groot (1994) calls ecosystem functions or ecological services such as the provision of flood control, climate regulation and water purification. However, the RAP programme was not designed to make improvements relating to these functions. Of these, the water purification aspect might be seen as an important goal, however, the Hamilton Harbour RAP plans did not consider improved drinking water quality as one of its mandates.

5.2 Non-Use Values

Non-use values associated with the RAP improvements are also available from Dupont's (2003) work. Respondents are asked to value improvements and then to indicate whether they plan to participate in the types of activities that benefit from the improvements. Approximately 45.2 per cent of the sample of individuals who answered the questionnaire indicated that they would not participate. Nonetheless, they had a positive willingness to pay for water quality improvements of CAN $12.92 per household per year.[9] Assuming the same proportion of the total number of households according to the Census data that would not participate, this amount is multiplied by 78 250 households. Total annual non-user benefits over all households per year amount to $1 010 990 in 1995 nominal Canadian dollars.[10]

Table 9.3 summarizes the results of these calculations. The aggregate value of all direct uses for which we are able to provide estimates (this being the sum of the swimming, fishing, boating and birdwatching values) is CAN $1.587 million. The aggregate value for potential future use values is CAN $6.839 million. Finally, the aggregate non-use values associated with water quality improvements is CAN $1.011. As indicated above, we assume that these values are enjoyed starting in 1995 and ending in 2015. Using the same inflation rate and real discount rate as in the cost calculations, these figures imply that the present value of the total benefits associated with the RAP's efforts to improve water quality in Hamilton Harbour is CAN $68 million. It is important to remember, of course, that we may be understating the total benefits somewhat, because our estimates of the direct use and non-use values may not capture the economic value added of some improvements to the local ecosystem's functions and services.

We can perform some sensitivity analysis using these data. First, we can extend the time horizon to account for the possibility of benefits accruing beyond 2015 under the assumption that, while the RAP activities are

Table 9.3 Estimated annual aggregate benefits of major remedial actions for Hamilton Harbour

Benefit category	Estimated value
Direct use values	
Swimming	0.429
Fishing	0.188
Boating	0.746
Birdwatching	0.224
Future use values	
Swimming	3.098
Fishing	2.360
Boating	1.381
Non-use values (passive or existence)	1.011
Total	9.437

Note: All figures are millions of nominal (1995) Canadian dollars on an annual basis. With exception of birdwatching, all values are specific to Hamilton Harbour RAP improvements. Birdwatching is estimated using benefit transfer methodology and data from Signorello (1998).

completed by that date, benefits continue to accrue from improved water quality. This does relatively little to the present value of total benefits largely because of the impact of discounting. For example, $1 of nominal benefits earned in 2015 has only a real present value of $0.07 in 1990. As a result, extending the time horizon to 2040 (thereby doubling the length of time for which benefits are accumulated), for example, raises our estimate of the present value of total benefits from CAN $68 million to only CAN $77 million.

Second, we can maintain the 2015 time horizon and account for population growth between 1990 and 2015 and its influence on aggregate benefits. If we assume that the population in the Hamilton Harbour watershed is projected to grow by 1.22 per cent (Statistics Canada, 2001) and that that population growth translates into increases in aggregate water quality-related benefits of the same proportion, then our estimate of the present value of total benefits would rise again from CAN $68 million to CAN $77 million.

6. DISCUSSION AND CONCLUSION

Decades of post-war population growth, economic development and industrialization resulted in significant decreases in water quality in the

Great Lakes Basin. While initial clean-up efforts in the 1970s met with some success, they were criticized for excluding public participation. In the 1980s the governments of the United States and Canada initiated the RAP process to address the most polluted Areas of Concern on the Great Lakes while allowing a more significant role to be played by local community organizations.

The individual RAP projects – of which Hamilton Harbour was one – began by assembling local community members to administer the RAP and by modelling the state of the local aquatic environment by natural scientists. In addition, the costs of recommended remedial actions such as changes to industrial processes, upgrading of sewage treatment facilities and remediation of contaminated sediments were estimated. The economic benefits of the environmental changes expected from these remedial efforts were usually not measured. As a result, a significant shortcoming of the decision-making process used to select specific remedial efforts in almost all RAPs (including the Hamilton Harbour case) was an absence of any effort to compare the costs and benefits of either individual remedial efforts or the scale of those efforts.

As this chapter has demonstrated, the implications of this absence of a rational decision-making process resulted in the selection by the Hamilton Harbour RAP committee of a menu of remedial efforts that, while having positive effects on water quality and the local ecosystem and contributing to the overall improvement in the water quality of the Great Lakes (International Joint Commission, 2003), were characterized by costs exceeding any reasonable estimate of the benefits arising from them. Our estimates of the present value of the costs and benefits relating to the Hamilton Harbour RAP activities over the period 1990 to 2015 are $240 million and $68 million, respectively.

The RAP process represents both a departure from, and a continuation of, Canada's historical approach to environmental remediation and legislation. It represents a departure in that it was the first significant effort by the Canadian government to ensure a substantial degree of public involvement in the design and execution of environmental remediation efforts. In this sense, it would perhaps be appropriate to recognize the development of local decision-making expertise and lines of communication (such as between environmental and industry groups) as investments in local 'social capital'. Our cost–benefit analysis has not made an attempt to value this important feature of the Hamilton Harbour RAP. On the other hand, it represents a continuation of both the federal and provincial governments' failure to employ economic analysis, and specifically cost-benefit analysis, in their approaches to environmental protection and remediation. This feature of Canadian environmental legislation in general and the RAP process specifically means that there are few mechanisms in government decision-making processes that ensure that either cost-minimizing remedial options are designed or that economically efficient policy options are selected. In fact, the localized nature of the RAP process, seen by many as a positive feature, contributed to this difficulty. If a more basin-wide approach had been taken, it is possible that more cost-effective approaches (such as the development of tradable pollution permit schemes to reduce industrial effluents or the development of more efficient cost accounting and pricing rules for municipal water and sewerage utilities) to improving Great Lakes water quality could have been developed.

NOTES

1. At the time of writing, the exchange rate between Canadian and US dollars was 1CAN$ = US$0.73.
2. The International Joint Commission was established by Canada and the United States through the Boundary Waters Treaty in 1909 (for more information see www.ijc.org). The IJC was given the mandate to assist the governments' efforts to regulate water levels (and, subsequently, water quality) in the Great Lakes and other waters shared by Canada and the USA.
3. One of the potential impairments considered by the IJC for each AOC was 'Restriction on drinking water; taste and odour problems'. However, this impairment was not selected by the IJC scientists for Hamilton Harbour.
4. The sample was representative of the general population in terms of demographics. Participants were given a financial incentive consisting of a $2 bill attached to the survey.
5. Clearly, individuals with use values may also hold non-use values. It was not possible to separate these individual components.
6. These numbers are comparable to others obtained for Ontario by other researchers. For example, Usher et al. (1987) calculate that Ontario residents value, on average, the services from all of Ontario's swimming beaches at about $90 per year in 1987 prices. Similarly, Fortin and Mitchell (1990) found that the mean and median payments for beach related water quality improvements in Canada were respectively $62 and $30 per year using 1990 prices. A study conducted by the Department of Fisheries and Oceans Canada in 1997 (Fisheries and Oceans Canada, 1997) of licensed sports fishers in Ontario found that 11 per cent were willing to pay an additional amount of less than $10, while 23 per cent were willing to pay $10, 18 per cent were willing to pay $20, and 8 per cent were willing to pay $30 more to have improved fishing opportunities.
7. This approach may result in a slight underestimate of the total amount of benefits since it does not include any benefits to people living outside the watershed. However, recent work (Georgiou et al., 2000) shows that benefits for water quality improvements fall inversely and rapidly with distance to any improvement site. Furthermore, impacts on water quality will provide little spill-over effect to the rest of the Great Lakes' ecosystem since Hamilton Harbour is self-contained and the narrowness of its shipping channel means that there is relatively little mixing with the rest of Lake Ontario. Thus, it is reasonable to assume very small benefits from the RAP for individuals who do not live in close proximity to the harbour. Exclusion of these benefits is, therefore, unlikely to alter the outcome of the cost–benefit analysis.
8. According to the latest Census results for the Hamilton region, the population has grown by 6.1 per cent over the five year period from 1996–2001. If we take an average annual growth rate of 1.22 per cent and apply this to the household numbers from the 1996

Census, then we can estimate the potential growth in demand for recreational services and, hence, the potential direct benefits associated with a larger population base. We discuss this below.

9. This is a simple average of the non-use values from each of the three questions asked about swimming, fishing, and recreational boating. These non-user values are likely to be an underestimate of the total non-use value since users may also have non-use values included in the direct use values we described earlier. However, since we are interested only in the total benefits and not the magnitudes of the individual components, this does not affect our overall cost–benefit analysis.
10. Another estimate of non-use values arises from enhancements to the ecosystem's ability to support biodiversity. Previous work by Pearce and Moran (1994), summarizing so-called debt-for-nature swaps, has suggested that US $5.00 per hectare is a rough approximation to existence values associated with biodiversity. Applying this value to the approximately 90 000 hectares in the watershed would imply an annual estimate of biodiversity existence values equal to US $450 000 or CAN $675 000. It would be very difficult though to transfer this estimate because of significant differences in the context in which it was derived.

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10. Benefit–cost analysis of regulations affecting surface water quality in the United States

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1. INTRODUCTION

There are a number of departments within the Federal Government of the United States that deal with water quality. The Department of Interior manages the nation's western water resources and hydrological science, primarily though the US Geological Survey (USGS), which collects, analyses, and disseminates information about the quality of the nation's surface and groundwater resources. The Department of Agriculture helps landowners protect their natural resources through its Natural Resources Conservation Service (NRCS), which helps landowners develop and carry out voluntary efforts to improve water quality and reduce upstream flooding. The Department of Commerce includes as part of its mission understanding the benefits of the Earth's physical environment and oceanic resources. This effort is carried out by the National Oceanic and Atmospheric Administration (NOAA), which is responsible for monitoring and forecasting the environmental quality of the nation's coastal and ocean areas, assessing the damage caused by spills in these areas, and protecting the nation's living marine resources. The Department of Transportation establishes the nation's overall transportation policy, which includes enforcing laws relating to the protection of the marine environment, through the Coast Guard (NARA, 2002).

Although these departments may have a hand in affecting the nation's water quality, it falls to an independent agency, that is, not an 'Executive Department', to pass the majority of the water quality regulation in the USA. The Environmental Protection Agency (EPA) was established on 2 December 1970 to permit co-ordinated and effective governmental action on behalf of the environment and to serve as the public's advocate for a liveable environment.[2] The mission of the EPA is to 'protect human health and to safeguard the natural environment – air, water, and land – upon which life depends' (US EPA, 2003a). The Agency's water quality activities are conducted by the Office of Water and represent a co-ordinated effort to keep the nation's waters clean and safe for fishing, swimming, and drinking. This effort includes developing national programs, technical policies, and regulations for water pollution control of the nation's surface, ground, and drinking water supply. It also entails marine and estuary protection, control of polluted runoff, water quality standards and effluent guidelines development, support of regional water activities, development of programs for technical assistance and technology transfer, and training in the field of water quality (NARA, 2002).

The main objective of this chapter is to discuss the role and use of benefit–cost analysis (BCA) in US regulations affecting water quality. This will be done based on five rules which were published during the period 1993–2003 and which have been classified as 'economically significant', that is, rules with an annual economic impact of more than $100 million. The remainder of this chapter is organized as follows. Section 2 summarizes the institutional context of water management in the USA, including the most important piece of legislation related to water quality, the Clean Water Act. Section 3 briefly discusses the institutional embedding of BCA in regulations and executive orders, while section 4 introduces the five economically significant rules affecting surface water quality. Finally, section 5 addresses the costs and benefits of these rules in more detail, while section 6 reflects on the use of BCA in the past and looks forward to its future use in regulatory processes.

2. THE CLEAN WATER ACT

The EPA's regulatory efforts are covered under more than a dozen Federal statutes and laws (Browner 1995).[3] The vast majority of its water-related regulation, however, is passed under the authority of the Safe Drinking Water Act (SDWA) and the Federal Water Pollution Control Act. The SDWA was passed by Congress in 1974 as a way of providing safe drinking water at the tap. The law was amended in 1986 and 1996 to protect drinking water at its source (rivers, lakes, reservoirs, springs and groundwater wells), but is still primarily used to regulate water that passes through a treatment system (US EPA, 1999c). As such, the law is of less relevance to a discussion of surface water quality and will not be discussed further in this chapter. The Federal Water Pollution Control Act was passed in 1972 and set a target of eliminating all pollutant discharges into Navigable Waters by 1985 (Public Law 92–500). The law was amended in 1977 under the moniker by which it is currently known, the Clean Water Act (CWA)

(Public Law 95–217), and has been amended a number of times since then, most recently in 2002 (Public Law 107–303).

In the CWA, Congress established several goals, including three that pertain to water quality: 'that the discharge of pollutants into the navigable waters be eliminated', 'water quality which provides for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water', and 'that the discharge of toxic pollutants in toxic amounts be prohibited'. Congress provided the EPA with several mechanisms to accomplish these goals. However, many aspects of the regulatory process are administered by states that are authorized (by the EPA) to do so. The relevant mechanisms are implemented through National Pollutant Discharge Elimination System (NPDES) effluent permits, which can be based on the following national programs: effluent guidelines or other technology-based standards, water quality standards, or total maximum daily loads (TMDLs).[4]

Most of the regulation associated with the Clean Water Act come from the EPA's Office of Water (OW), which has four main organizations that set and implement policies and regulations: the Office of Wastewater Management (OWM), the Office of Science and Technology (OST), the Office of Wetlands, Oceans and Watersheds (OWOW), and the Office of Ground Water and Drinking Water (OGWDW). While each office has many and varied responsibilities, each office has fairly specific responsibilities with respect to implement the CWA (except for OGWDW, which implements the SDWA). OWM directs and oversees the NPDES program, including issuing permits for Concentrated Animal Feeding Operations, storm water discharges, and combined and sanitary sewer overflows. OST issues effluent guidelines and is responsible for the process of developing, adopting, and approving water quality standards and criteria. OWOW is responsible for watershed programs, including the TMDL program. Each of the EPA's ten regional offices also has a Water Division that works with the OW and with other stakeholders to design and implement policies and regulations.

All point sources of pollution are required to have a permit under the NPDES program. These permits specify pollutants that a point source must control, numerical or narrative limits on those pollutants, and time periods for how often the source must monitor for that pollutant. Additionally, the permit specifies if, within a given month, the source has a maximum allowable daily limit, a monthly average limit, or both. Permits are written by designated officials of authorized states or by the EPA; permits may be written for individual plants or 'general permits' are written for classes of operations. These permits can be technology based or water-quality based. We now describe both bases in turn.

Technology-based permits can be based on national effluent guidelines or on the best professional judgement (BPJ) of the permit writer. For a given industrial facility, if a pollutant is regulated by an effluent guideline for that facility, the NPDES permit limit must be at least as stringent as the limit in the effluent guideline. Effluent guidelines are developed for categories of facilities, such as pulp and paper mills or iron and steel plants, although these categories are usually further subdivided.[5] Whenever a guideline is first promulgated or revised, the CWA requires consideration of limits based on the performance of technologies (although it is important to note that facilities may meet the limit using any technology they choose). There are three types of limits for existing sources (new sources have similar, but potentially more stringent, limits): best practicable technology (BPT), best conventional technology (BCT), and best available technology (BAT). Best practicable technology regulates all pollutants, but BCT regulates only conventional pollutants: biological oxygen demand (BOD), total suspended solids (TSS), fecal coliform, pH, and oil and grease, and BAT regulates toxic and non-conventional pollutants (everything that is not a conventional pollutant). The CWA spells out a number of factors the EPA must consider when setting these limits, including that a technology be available to meet the limits and economic tests that are discussed below. If a guideline is not available for, or does not regulate, a particular pollutant that a permit writer would like to regulate then a BPJ limit is determined by the permit writer. The BPJ limits are based on the best technical data available for a given plant (US EPA, 1996).

In addition to technology-based permit limits, facilities may have additional, or more stringent, limits based on water quality standards.[6] States are required to assign a 'designated use' (such as fishing or swimming) to all water bodies and then assign water quality criteria for protection of each level of designated use. Water quality criteria are numeric pollutant concentrations (and narrative requirements) that designate the maximum allowable in-stream pollutant levels that support a designated use. If any criteria are exceeded (for example, if effluent guidelines do not cover a specific pollutant or technology-based standards do not offer enough control) then facilities can have more stringent limits – targeted so that the relevant water criteria are attained – written into their NPDES permit. However, a more stringent limit is not automatic. For example, if adoption of more stringent limits would result in 'substantial and widespread' economic impacts, then facilities can be granted a variance from the standard (US EPA, 1995c).

Finally, all states are required periodically to assess waters, identify those that are not meeting their water quality standards, and prioritize waters for further action. Some of these waters will require a TMDL, which is the amount of a given pollutant that may be discharged into a water body and still maintain a water quality standard. The TMDLs quantify pollutant sources and set load allocations for both point and non-point sources. Once a load allocation is determined for point sources, limits reflecting these allocations can be written into NPDES permits. States may also (although the EPA may not under the Clean Water Act) require non-point sources to implement best management practices (BMPs) to meet a load allocation.

3. THE INSTITUTIONAL EMBEDDING OF BCA IN WATER POLICY AND MANAGEMENT

In passing regulation, the Office of Water is obligated to design the most effective regulation for the taxpaying public while still meeting the requirements of the CWA. Defining what is meant by effective, however, is a difficult task. In practice, tradeoffs have to be made in designing effective regulation. These tradeoffs can be in the form of equity, distributional, and monetary concerns.

Economists tend to consider tradeoffs by detailing the positive and negative aspects in terms of dollars, a process called monetization. Comparing all of the monetized gains in consumer utility, called benefits, to the monetized losses in utility, called costs, is BCA. Whether the change in utility is associated with a market product (for example the purchase of water treatment equipment) or not (for example, the value of fresh water recreational opportunities), the imputed value to society, or shadow price, is the correct value to monetize (Pearce, 1996). The net benefit (that is, benefits minus costs) gives an idea of the implications of a proposal. While not the only criteria in decision-making, BCA is a useful accounting metric for comparing the desirable and undesirable consequences of regulation (Arrow et al., 1996).

Prior to 1981, the EPA was required in some cases to conduct economic analysis and BCA. However, in 1981, President Reagan issued Executive Order[7] (EO) 12291 that, in effect, required a BCA for all 'major rules'. A major rule was defined as one which had an annual effect on the economy of more than \$100 million, caused a major increase in costs or prices, or had a significant adverse effect on the economy. A rule which exceeds the \$100 million annual effect threshold is commonly called 'economically significant'. This executive order was designed to address the tradeoffs and analyse the impact of the larger regulatory actions and emphasized that agencies should maximize net benefits to society, a change from previous requirements (US EPA, 1987). For each major rule, a Regulatory Impact Analysis was required to account for all of the costs, benefits, and net benefits of the rule, including effects that could not be quantified in monetary terms. Agencies were expected to promulgate rules only if the net benefits outweighed the costs, and the options chosen for the rule were expected to maximize net benefits (President, 1981).

This executive order had a major impact on regulation development since it provided an explicit role for economists and BCA. Requiring benefits to exceed costs, however, seemed to rely too heavily on monetized benefits analysis. In 1993, President Clinton issued EO 12866, which replaced EO 12291 but maintained its ultimate goal of assessing the trade-offs of rules and designing the most effective regulation. Under this new executive order, agencies were expected to promulgate regulation under a reasoned determination that benefits justify its costs. This is a much softer charger than before. In addition, while the executive order also required that the chosen regulatory option maximize net benefits, these net benefits were expected to include potential economic, environmental, public health and safety, distributive, and equity concerns (President, 1993). President Bush has recently amended EO 12866 with EO 13258, but no substantive changes have been made to the requirements for economic analysis (President, 2002).

Additional executive orders and laws have been passed that reflect the desire to measure the effectiveness of regulation. In 1994, EO 12898 required that all agencies assess the environmental justice of regulation by measuring the impact of regulation on minority and low income individuals (President, 1994). The Unfunded Mandates Act of 1995 was passed by Congress in an effort to reduce the imposition of unfunded Federal mandates on states and localities (Public Law 104–4). In 1996, the Small Business Regulatory Enforcement Fairness Act amended the Regulatory Flexibility Act of 1980 and required that agencies assess the impact of rules on small businesses (Public Law 104–13). In 1997, EO 13045 required that the rules be evaluated for adverse impacts on the health and safety of children (President, 1997). Executive Order 13084, issued in 1998, required consultation and co-ordination with American Indian tribal governments on regulations that significantly affected their communities (President, 1998). In 1999, EO 13132 required that agencies encourage states to develop their own policies in preference to federal regulations, and, where possible, allow states to set their own regulatory standards (President, 1999).

All these laws and executive orders were designed to guide the agencies in creating effective regulation. The difficulty is that this is often done by adding yet another criterion by which to judge a regulation, without assisting in trading off the various criteria amongst each other. Ultimately, it is left to the agency to determine if the requirements of these executive orders have been met, but not completely without oversight. In 1983, the EPA issued internal guidelines for performing benefit–cost and regulatory impact analysis (US EPA, 1983). These guidelines were reprinted in 1991 and updated in 2000 (US EPA, 2000). In addition, all major rules must be reviewed by the Office of Management and Budget, which falls under the Office of the White House, providing a check on all major regulation. Finally, there is also congressional review. For example, one of the requirements placed on the Office of Management and Budget is that the benefits and costs of all major rules must be assessed in a yearly report sent to Congress for review (Public Law 106–58).

In this way, BCA plays a very important role in water regulation. It serves as a useful tool to compare the favourable and unfavourable effects of policies. However, a number of critiques have been levelled against the technique (Heinzerling and Ackerman, 2002; Kelman, 1981) and a host of alternatives have been suggested. For example, cost-effectiveness analysis takes the benefits objective as fixed and compares the cost of achieving that objective (Field, 1997). Risk analysis focuses on the risk levels and risk reductions, and risk-risk analysis evaluates the risk tradeoffs of a proposed regulation (Viscusi, 1996). Each of these has a place in the regulatory process, but economists, in general, are comfortable with BCA and its limitations.

Freeman (2000) compares existing estimates of the benefits and costs of the CWA. Although there are no sets of estimates that make comparable estimates of both benefits and costs, one set estimates costs in 1985 of $42.4 billion (1996$) while the 'best' estimate of benefits for 1985 is $22.6 billion, although the high estimate is $44.3 billion. Freeman concludes that the benefits of the act have not outweighed the costs and suggests investigating adjusting requirements where the marginal benefits are 'substantially different' from marginal costs as well as ways to reduce the costs of meeting current requirements. However, we would also argue that the current accounting of benefits in US water quality regulation is less complete than the accounting of costs and that this fact accounts for some of the difference between estimates.

4. ECONOMICALLY SIGNIFICANT WATER QUALITY RULES SUBJECT TO BCA

From 1993 to 2003, the EPA began some type of action for 267 water-related rules (US EPA, 2002c). Of these, only five affect surface water, are economically significant, that is, exceed the $100 million annual impact threshold, and have been published in the Code of Federal Regulations. These five rules are listed in Table 10.1. Although other rules may be

*Table 10.1 Economically significant rules affecting surface water quality**

Title	*Federal Register* publication date	Benefits in millions per year	Costs in millions per year
Effluent Guidelines and Standards for the Oil and Gas Extraction Point Source Category, Offshore Subcategory	4 March 1993	\$30–\$111 (1991\$)	\$134–\$160 (1991\$)
Final Water Quality Guidance for the Great Lakes System	23 March 1995	No aggregate estimate	\$60–\$376 (1994\$)
National Emissions Standards for Hazardous Air Pollutants for Source Category: Pulp and Paper Production; Effluent Limitations Guidelines, Pre-treatment Standards, and New Source Performance Standards: Pulp, Paper, and Paperboard Category; Final Rule	15 April 1998	*Total:* (\$727)–\$1500 *Water portion:* \$12–\$57 (1995\$)	*Total:* \$420 *Water portion:* \$263 (1995\$)
National Pollutant Discharge Elimination System – Regulations for Revision of the Water Pollution Control Program Addressing Storm Water Discharges; Final Rule	8 December 1999	\$671–\$1600 (1998\$)	\$848–\$981 (1998\$)
National Pollutant Discharge Elimination System Permit Regulation and Effluent Limitation Guidelines and Standards for Concentrated Animal Feeding Operations (CAFOs)	12 February 2003	\$204–\$355 (2001\$)	\$335 (2001\$)

Note: * Economically significant rules are those with an annual economic impact of more than $100 million. Year to which price levels refer given between brackets.

designated as 'major' for other environmental and economic reasons, and therefore have a BCA associated with them, these five represent those with the largest economic impact related to water quality and are considered representative of the most comprehensive type of BCA.

The Effluent Limitation Guidelines for the Offshore Oil and Gas Industry, promulgated in 1993 (US EPA, 1993b), established the technology-based standards for this industry, as required under the CWA. The rule was based on an anticipated 759 wells per year from 1993 to 2007, of which, 456 wells per year were expected to produce oil at an average price of $21 per barrel. The total annual cost of compliance, in 1991 dollars, ranged from $134 million to $160 million in the first year, declining to $38 to $94 in 2007. These costs included the cost for the proper disposal of drilling fluids and drill cuttings in the drilling operation; produced water; treatment, work-over, and completion fluid if the well becomes productive; produced sand and slurried particles; and other miscellaneous waste. A potential loss in producer surplus was recognized but not monetized, but since the production effects were not anticipated to affect the world market price for oil, no consumer surplus effects were expected. Very few plant shut-down, job loss, job dislocation, or other general equilibrium effects were anticipated or monetized, and no government sector regulatory costs were estimated.

The total monetized annual benefits of this rule, in 1991 dollars, ranged between $30 million and $111 million. These monetized benefits were exclusively the human health benefits from reduced exposure to lead and other various organic and inorganic carcinogens. Exposure was assumed to occur through consumption of recreationally caught finfish and commercially harvested shrimp in the Gulf of Mexico. Mortality of both infants and adults was monetized using the value of statistical life (VSL) literature (US EPA, 1997a; 2000; Viscusi, 1992). Monetized morbidity values were cost of illness estimates of reductions in children's IQ (Salkever, 1995; Schwartz, 1994), hypertension (US EPA, 1997b), stroke (Taylor et al., 1996), and heart disease (Wittels et al., 1990) (US EPA, 1993a; 1993c).[8]

The Water Quality Guidance for the Great Lakes System was published in 1995 (US EPA, 1995b) and established the water quality criteria for 29 pollutants to protect aquatic life, wildlife, and human health in this area and methodologies to develop criteria for additional pollutants.[9] The total annual costs range from $60 million to $376 million, in 1994 dollars. These costs reflect modification to the compliance decisions of the 7323 direct and indirect, municipal and industrial dischargers. Potential compliance options included operating and facility changes, waste minimization and pollution prevention controls, waste minimization and pollution prevention in conjunction with simple treatment, and end-of-pipe treatment.

There was no discussion of changes in consumer and producer surplus, job losses or dislocation, government sector costs, and macroeconomic effects (US EPA, 1995a).

Aggregate benefits from this rule were not monetized, but rather three case studies were conducted: the lower Fox River drainage, including Green Bay, located on Lake Michigan in north-eastern Wisconsin; the Saginaw River and Saginaw Bay, located on Lake Huron in north-eastern Michigan; and the Black River, located on Lake Erie in north-central Ohio. Monetized benefits for these three areas, in annual 1994 dollars, ranged from \$0.3 to \$8.5 million, \$0.2 to \$7.6, and \$0.4 to \$1.5 million, respectively. The estimated compliance costs were \$3.6 million, \$2.6 million, and \$2.1 million, respectively. The two primary monetized benefits categories were mortality from cancer and improved recreational fishing. The mortality estimates were based on the same VSL literature as in the offshore oil and gas rule discussed above. Valuation of the improved recreational fishing is obtained by scaling the results of a study by Lyke (1993), who valued the consumer's willingness to pay for 'toxic free' recreational fishing if all fish consumption advisories were lifted. Three other monetized categories were non-consumptive recreation, commercial fishing, and non-use benefits. The non-consumptive recreation (nature viewing and hunting) and commercial fishing categories were based on a scaled proportion of the estimated baseline value of the resource. The scaling factor was either one-half or equal to the scaling factor obtained from the Lyke study, but no justification was given for this estimate. The non-use value was equal to one-half of the recreational fishing benefits based on results by Fisher and Raucher (1984) (US EPA, 1995d).

The National Emission Standards for Hazardous Air Pollutants, Effluent Limitations Guidelines (ELGs), Pre-treatment Standards and New Source Performance Standards for the Pulp, Paper and Paperboard industry were issued in 1998 (US EPA, 1998). This action jointly regulated the effluents of this industry into waterways as well as into the air because technological changes to production processes affect both air and water emissions. Under the water portion of this rule, analytic methods were issued for 96 mills in the Bleached Papergrade Kraft and Soda subcategory and the Papergrade Sulfite subcategory. The total air and water related compliance costs of the rule, in 1995 dollars, were \$420 million, with \$263 million of these costs related to the water portion. Mill closures, job losses, decreased shipments, decreased exports, and other direct and indirect effects were considered, but their value was not monetized. No mention was made of government administrative costs.

Total benefits of the air and water rule were estimated to range from negative \$727 million to \$1.5 billion 1995 dollars.[10] The water benefits alone were from \$12 to \$57 million.[11] Three water-related benefits categories were monetized: human health improvements from recreational and subsistence fish consumption, recreational angling benefits, and reduced cost of sludge removal. The human health category was based on a reduction in cancer cases and relied on the same VSL literature as previous rules. The recreational fishing benefits were based on the same Lyke (1993) study used in the Great Lakes guidance. The sludge removal was an original study, conducted for this rule. Sludge disposal is a regulated practice and sludge with higher concentrations of dioxin must be disposed of via land filling or incineration rather than cheaper options such as land application. Because the regulation required lowering the amount of dioxin in sludge from pulp and paper mills, this sludge could be disposed of through land application (US EPA, 1997c).

The NPDES Regulations for Revision of the Water Pollution Control Program Addressing Storm Water Discharges (Phase II) was published in 1999 to augment earlier storm water regulation (Phase I) (US EPA, 1999b). The Phase II regulation was designed to address storm water discharges from small municipal separate storm sewer systems serving less than 100 000 persons and construction sites that cover one to five acres. The total annual cost of this rule, in 1998 dollars, ranges from \$848 million to \$981 million, with the costs subdivided as due to Municipal Minimum Measures, Controls for Construction Sites, and Federal/State Administrative costs. Municipal costs were estimated from a survey of 1600 Phase II municipalities and an estimate of application, record keeping, and reporting labour hours. Construction costs included estimates of best management practices (BMP) costs for both construction site runoff and post-construction site runoff controls. Because the regulation affects small municipalities and construction sites, it was not expected to have any impacts on employment or the national economy.

Annual benefits, in 1988 dollars, were estimated to range from \$671 million to \$1.6 billion. This wide range was partially due to the fact that two different methods were used to compute aggregate benefits. The first method was to use the National Water Pollution Control Assessment Model (NWPCAM) to estimate the water quality, based on four pollutants, for each of 632 000 miles of rivers in the country. A willingness-to-pay value for improvements in this water quality was applied based on a Carson and Mitchell (1993) contingent valuation survey. This survey assessed the public's willingness to pay for moving waters to various classes of recreational use: boatable, fishable, and swimmable. The second method was to combine the estimates of three different programs: municipal measures in fresh waters, municipal measures in marine water, and construction site controls. Municipal measure in fresh waters relied on the same Carson and

Mitchell (1993) survey, but, rather than modelling the river system, relied on state reported storm water impairment and an assumed 80 per cent program effectiveness. Municipal measures in marine waters produced two types of benefits. First, it was estimated that there would be improved recreational benefits from reduced beach closures based on travel cost estimates (Leeworthy and Wiley, 1991; Walsh et al., 1990). Second, cost of illness health benefits were estimated for reduced cases of highly credible gastroenteritis 2 (Mauskopf and French, 1991) and significant respiratory disease (Pope et al., 1995; Tolley et al., 1986). Finally, the benefits of construction site controls were based on a contingent valuation survey of the willingness to pay for erosion and sediment controls in North Carolina (Paterson et al., 1993) (US EPA, 1999a).

In February 2003, the EPA published the NPDES Regulation and ELG for Concentrated Animal Feeding Operations (CAFOs) (US EPA, 2003c). The rule established a mandatory duty for more than 10 000 CAFOs to apply for an NPDES permit and to develop and implement a nutrient management plan. It also established performance expectations for existing and new sources to ensure appropriate storage of manure, as well as expectations for proper land application practices. The total cost of this rule, in 2001 dollars, was estimated to be $335 million. These costs include compliance costs borne by CAFOs and also administrative costs to federal and state governments. The final regulation was not expected to cause significant changes in aggregate employment or the national economic output, and only minor impacts on foreign trade, but more significant losses in employment and output of the animal production sector and other indirect effects were estimated (US EPA, 2002a).

Benefits of the rule, in 2001 dollars, were estimated to be between $204 million and $355 million. Monetized benefits categories included recreational benefits from improved surface water quality, reduced fish kills, improved commercial shell fishing, reduced contamination of private wells, reduced contamination of animal water supplies, reduced eutrophication of estuaries, and reduced costs to water treatment facilities. The recreational benefits accounted for the largest portion of benefits and were estimated using an updated version of the NWPCAM model and the same Carson and Mitchell (1993) contingent valuation survey used in the Storm Water Phase II rule. The estimation of the willingness to pay values, however, was slightly different, and six pollutants, rather than four, were used to estimate water quality. The fish kill analysis was based on travel cost estimates of the lost recreational use value of a dead fish (IEc, 2002). The value of the improved shellfish harvest was an original study conducted for this rule, and was an estimate of the changes in producer and consumer surplus. The benefits of reduced nitrate contamination were based on contingent valuation results (Crutchfield et al., 1997; De Zoysa, 1995; Poe and Bishop, 1992). Reduced cattle mortality was an original study that estimated the market value of cattle lost to contaminated water supplies. The reduced estuary eutrophication was a case study based on the travel cost estimates for visits to an estuary (Kaoru, 1995; Kaoru et al., 1995; Smith and Palmquist, 1988). Finally, the reduced cost to water treatment plants was an original study based on the reduced cost of treatment material from cleaner source water (US EPA, 2002b).

5. COSTS AND BENEFITS REVISITED

5.1 Costs

For these five rules, and in general for most BCA conducted by the EPA, the bulk of the cost analysis is focused on the cost of compliance. This makes sense, given that the largest component of the overall social costs of these rules tends to be the compliance costs borne by the regulated industry. However, these are not the only costs to society from regulation. Increasingly, an effort is being made to recognize that there are social costs associated with overall changes in consumer and producer surplus, opportunity costs associated with government regulatory activity, transitional social costs associated with unemployment and firm closing, and indirect effects on other industries, productivity, investment, and foreign trade. Although the EPA frequently relies on estimates of private compliance costs as an approximation to social costs, this approach may either overstate or underestimate true social costs (US EPA, 2000, ch. 8). In fact, in 1995 Congress passed the aforementioned Unfunded Mandates Act which required a calculation of the impact that legislation with major intergovernmental mandates has on states and local governments (Public Law 104–4).

It should also be recognized that other factors may come into play when assessing the burden that a rule has on regulated entities. The CWA specifies a number of factors, including economic ones, that must be considered when choosing effluent guidelines and setting water quality standards. Setting BPT requires 'consideration of the total cost of application of technology in relation to the effluent reduction benefits to be achieved from such application'. While this may appear to be a cost–benefit test, the interpretation has consistently been that EPA 'effluent reduction benefits' refers to pounds of pollutants and BPT thus requires a cost-effectiveness test. Setting BCT requires a similar comparison, although there are two parts to the BCT test.[12] Setting BAT requires identifying the 'best available technology' that is 'economically achievable' although the CWA is silent on how to determine this achievability. A review of the 16 effluent guidelines that were finalized between 1979 and 2000 found that as many as 20 different measures of economic impact were used to measure achievability (Wheeler and Covington, 2001). Employment losses were estimated in all of these ELGs and facility closures estimated in 14 of them using discounted cash flow or similar methods. In general, however, facility closures seem to weigh more heavily in decision-making than employment losses. This is important to recognize since, from an economic standpoint, facility closures most likely only have a transitional cost associated with them. The resources can be reallocated to other activities. Still, even though it could theoretically be possible for a large number of facility closures to be socially optimal in order to obtain large benefits, only a certain number is considered acceptable in regulation under the CWA. While no firm rule can be established from this analysis, a closure rate of less than 5 per cent of the industry was generally acceptable, while anything greater than 10 per cent was generally unacceptable.

5.2 Benefits

The benefits of these rules cannot be as easily categorized as the costs. In general, environmental benefits can fall into four broad categories: human health (including mortality and morbidity effects), amenities of the environment (for example, taste, odour, and visibility), ecological benefits (including market product, recreation opportunities, ecosystem services, existence and bequest values), and materials damage (Freeman, 2003; US EPA, 2000). Each of these benefits categories contain a multitude of various economic endpoints which can be valued. The common valuation techniques include: market methods (for example, estimate changes in producer or consumer surplus), revealed preference methods (including recreational demand or travel cost models, hedonic studies, averting behaviour, and cost of illness methods), and stated preferences (including contingent valuation and conjoint analysis).

Table 10.2 lists the various benefits categories, economic endpoints, and valuation methods used in each of these five rules. With the exception of the sludge removal, shellfish harvest, cattle mortality, and water treatment cost analyses, all these studies are based on benefits transfer exercises, in which the valuation was not specifically carried out for the regulation, but based on previous studies. This is typical for most surface water regulation, given the limited time and budget associated with rule-making and the constraints associated with collecting original data (Griffiths, 2002). This type of benefits transfer does have its dangers and drawbacks but, if done carefully, can produce a reasonable approximation of the correct economic

Table 10.2 Benefits categories and valuation methods of economically significant rules

Rule	Benefit category	Economic endpoint	Valuation method
Effluent Guidelines and Standards for the Oil and Gas Extraction Point Source Category, Offshore Subcategory	Mortality	Cancer risk	Contingent valuation, Hedonic wage
	Morbidity	IQ < 70	Cost of illness
	Morbidity	Lost IQ point	Cost of illness
	Morbidity	Hypertension	Cost of illness
	Morbidity	Stroke	Cost of illness
	Morbidity	Heart disease	Cost of illness
Final Water Quality Guidance for the Great Lakes System	Mortality	Cancer risk	Contingent valuation, Hedonic wage
	Recreation	Recreational fishing	Travel cost, Contingent valuation
	Recreation	Other recreation	Estimate
	Market product	Commercial fishing	Estimate
	Non-use benefit	Non-use benefit	Estimate
Effluent Limitations Guidelines, Pretreatment Standards, and New Source Performance Standards: Pulp, Paper, and Paperboard Category; Final Rule	Mortality	Cancer risk	Contingent valuation, Hedonic wage
	Recreation	Recreational fishing	Travel cost, Contingent valuation
	Materials damage	Sludge removal	Market method
National Pollutant Discharge Elimination System – Regulations for Revision of the Water Pollution Control Program Addressing Storm Water Discharges; Final Rule	Recreation	Fresh water recreation	Contingent valuation
	Recreation	Estuarine recreation	Travel cost
	Morbidity	HC gastroenteritis 2	Cost of illness
	Morbidity	Respiratory disease	Cost of illness, Contingent valuation
	Ecosystem Services	Erosion/sediment control	Contingent valuation
National Pollutant Discharge Elimination System Permit Regulation and Effluent Limitation Guidelines and Standards for Concentrated Animal Feeding Operations (CAFOs)	Recreation	Fresh water recreation	Contingent valuation
	Recreation	Fish kills	Travel cost
	Market product	Shellfish harvest	Market method
	Morbidity	Private well contamination	Contingent valuation
	Market product	Cattle mortality	Market method
	Recreation	Estuarine eutrophication	Travel cost
	Market product	Water treatment costs	Market method

value (Desvousges et al., 1998). Recently, Smith et al. (2002) have suggested ways in which the estimates of various studies can be combined to avoid double counting of benefits and other problems by conducting transfers in a manner explicitly consistent with utility theory.

In general, it can be said that a wide diversity of benefits categories have been estimated using a number of valuation techniques. The two categories that have produced the largest benefits for these rules are reductions in mortality and improvements in fresh water recreation. Reductions in mortality are based on a fairly broad literature on the value of a statistical life (VSL) (see Viscusi, 1992, and US EPA, 1997a, 2000, for a review of this literature). The concept of VSL has often been inappropriately criticized (Heinzerling and Ackerman, 2002). The VSL is estimated by obtaining the willingness to pay for a small change in risk of death and then aggregating this amount to the total willingness to pay that a collection of individuals, as a group, would offer, *ex ante*, to eliminate one statistical death. This is hence a value to avoid a probability-based risk of death, and should definitely not be confused with the value that society places on any particular individual's life.

The fresh water recreation benefits estimated in the storm water rule and the CAFO rule both rely on the results of a contingent valuation survey by Carson and Mitchell (1993). In this survey waters were classified by their recreational use state: no use support, boatable, fishable, and swimmable. This is sometimes referred to as a water quality ladder, since higher use states are only accomplished by meeting the lower ones (for example, a river that is fishable is also, by construction, boatable, but is not yet swimmable). This was illustrated to the survey participants by showing them a water quality index from zero to 10, with demarcations in water quality where the river moves from one state to the next. Individuals were asked to state their willingness to pay for moving the nation's waters from one recreational state to the next.

The demarcations in the water quality index were developed by Vaughn (1981) based upon the levels of five water quality parameters: biochemical oxygen demand, total suspended solids, faecal coliform, the dissolved oxygen saturation, and pH. The minimum level of each parameter for each use was first determined. Each water quality parameter was then given a quality scale from zero (worst) to 100 (best) and raised to weight, with the five weights summing to one. The product of these exponentially weighted parameters was divided by 10 to get a water quality at each use value. This is a modified version of the National Sanitation Foundation (NSF) index (Mitchell and Stapp, 2000), which has nine water quality measures, including temperature, total phosphate, nitrates, and turbidity. These nine water quality parameters and the 0 to 100 quality scale for each were decided upon through a NSF-sponsored delphi-method survey of 142 water quality experts (McClelland, 1974).

In the storm water rule, water quality benefits were monetized by attributing this willingness to pay for all streams that moved from one use state to the next, using the Vaughn index excluding pH. In other words, the entire willingness to pay was associated with a stream mile only if the stream actually changes use states. This meant that no benefits were associated with streams that improved in water quality, but did not actually change use states. For the CAFO rule, it was decided that partial benefits should accompany small increases in water quality, so a willingness to pay function was estimated based on the original Carson and Mitchell article (1993). Also two additional water quality measures, total phosphate and nitrates, were added to the index.

5.3 Data and Models

The data used for regulation comes from a number of sources. Under Section 308 of the Clean Water Act, the EPA has the authority to ask for records and reports related to pollutant discharges from owners and operators of point sources. In practice, this means that the EPA sends out letters to these sources ('Section 308' letters in agency parlance) requesting information so that it may design regulation. This is the preferred method of obtaining information on industries, since it is comprehensive and the firms must respond by law. In the CAFO rule, however, it was infeasible to send out Section 308 letters, so the benefit and cost analyses were conducted using model firms based on publicly available data. Data on firms compliance with regulation comes from internal EPA datasets, particularly the Permit Compliance System (US EPA, 2003e).

For water quality data, the EPA generally does not systematically collect water samples, but, rather, relies on other sources. Under section 305(b) of the CWA, each state must submit a biennial report of the water quality of all its navigable waters. This is a good source of data, but states vary in the quality of their reports and the comprehensiveness of coverage of their navigable waters. There are some internal EPA datasets on water quality, particularly the Storage and Retrieval System (STORET) (US EPA, 2003f), but these data are reported by states and other groups rather than collected by the agency. Finally, other government agencies, like the USGS and the USDA collect information that can be used by EPA.

Historically, surface water quality has been modelled independently for each rule. More recently, the storm water rule and the CAFO rule both estimated water quality using the National Water Pollution Control Assessment Model (NWPCAM), although different versions of the model were used in each rule. NWPCAM is a publicly available, national-scale model constructed by the EPA and the Research Triangle Institute for policy-making. The earlier version of this model (Bondelid et al., 2000), used for the storm water rule, estimates the four water quality parameters for 632 000 freshwater rivers and streams in the continental US using the EPA's river Reach File 1 system (Bondelid et al., 2000). In this version, pollutants are measured using a first-order decay model. A more enhanced model of NWPCAM (Van Houtven et al., 2001) was used for the CAFO rule. This version includes a more complicated non-linear estimation model for nitrogen and phosphorous and uses a subset of the 3.2 million perennial and non-perennial streams in the EPA's Reach File 3 system.

6. LOOKING FORWARD

The review of these five major water quality rules and the issues surrounding them suggests that the EPA has addressed some fairly complex economic issues in evaluating surface water quality regulation. The valuation of improved surface water quality is required under EO 12866 and is considered a rational procedure to justify public funding in public goods such as water quality (Arrow et al., 1996). It is interesting to note, however, that the passage of a rule did not require a clear-cut indication that net benefits be positive. In two of the five rules, the range of monetized benefits is less than costs and in two others, the range of monetized benefits bounds the range of costs.

There are a number of reasons why policy adoption does not require the BCA to indicate positive net benefits. First, while the EPA is obliged, as a federal agency, to follow the directives of the President's executive orders, they generally are not judicially enforceable while the statutory requirements of the CWA are enforceable. Thus, no suit can be brought against the agency charging that it did not follow EO 12866. Furthermore, that executive order only requires that benefits justify costs, not necessarily exceed them. Second, the description above is limited to monetized benefits. Each of the complete economic analyses for these rules contained sections on the non-monetizable benefits of the rule (for example, US EPA, 2003c, 7239–7242). It is generally believed by proponents of these rules that these monetized benefits may be large and could offset the differential between monetized benefits and costs. Finally, net benefits measure economic efficiency, but it is not the only criteria economists use to judge efficacy. Issues of the distribution of costs and benefits and equity are considered to be an essential part of a complete economic analysis (US EPA, 2000).

While economics in general, and BCA in particular, may not be a statutorily or judicially required decision-making criterion for policy, this does not imply that economics is not a valued contributor. Morgenstern (1997) neatly illustrates the contribution of economics for twelve major EPA regulations. His conclusion is that economics contributed to cost-savings, administrative feasibility, and the probability of compliance for these rules, and the same can be said for the five rules described here. Benefit–cost analysis is a very useful tool for comparing the favourable and unfavourable effects of a policy and can help decision-making in setting regulatory priorities. Economists in the agency have already conducted some relatively sophisticated analyses, but these analyses can always be improved. A number of important issues still face these economists with future regulation.

With the current set of water quality rules in place, additional regulation will have some serious compliance and baseline issues to address. If firms are not currently meeting the provisions of an existing rule, then future rules must carefully evaluate their benefits. Technically, the benefits of improved water quality through full compliance were assumed in the BCA of previous rules and should not be counted again in future rules. This, however, obscures the fact that future rules may produce benefits that the previous rule claimed, but were unable to generate (if, for example, compliance is not complete). Careful specification of the baseline includes assumptions regarding compliance with previous rules, the interaction of the proposed rule with other regulation, and strategic firm behaviour.

Future rules will also have to find new ways to value certain benefit categories. Although the monetized benefits described above form an impressive list, two categories suffer notable weakness. The non-use value from improved surface water quality was addressed in the Great Lakes water quality guidance, but only in an extremely ad hoc fashion. New approaches to measure this type of value must be developed since some environmentalists believe that large benefits are missed from this source. Perhaps more accessible to non-economists than non-use benefits are ecosystem services. These types of benefits have been claimed in many surface water regulations, including the ones above, but generally have not been monetized. The exception is the one component of the storm water rule. Both of these benefit categories are of growing importance as the EPA looks to new regulation. Existing regulation has addressed some of the more obvious benefits of improving surface water quality: health and recreation benefits. Future regulation will have to address some of the more nebulous issues of surface water quality. New techniques will be needed to capture ecosystem benefits like flood control, sea grass production, and biodiversity.

Both these benefit categories suffer from difficulties associate with benefits transfer. In fact, the valuation of ecosystem benefits has been identified as a research priority for EPA and OW because of the scarcity of applicable ecosystem benefits estimates that are suitable for transfer (Mehan, 2003; US EPA, 2003d). As mentioned previously, benefits transfer is the procedure of associating the values obtained in one study to the policy scenario described in the regulation. Guidance exists on how to perform economically valid transfers (Desvousges et al., 1998; US EPA, 2000), but it inevitably opens the valuation exercise up to criticism that the study is not comparable to the policy case. It would, of course be better to conduct an original study, but time and financial constraints often make this impossible (Griffiths, 2002). For example, the Paperwork Reduction Act (Public Law 104–13) requires the EPA to obtain OMB approval if it collects the same information from ten or more non-Federal respondents. This 'Information Collection Request' (ICR) must justify the need for the data, estimate the time and cost burden placed on the public, and must be published in the *Federal Register* twice, with an appropriate public comment period and can take more than 180 days to complete.

National-scale models, like NWPCAM used for the storm water rule and the CAFO rule, have the advantage that a consistent set of assumptions can be made across the country and only one model needs to be run for each policy scenario. These advantages are often critical to allow for new data and information to be incorporated in a timely fashion. Water quality engineers, however, have a tendency to be sceptical of national-scale models (Bondelid et al., 2000). A number of other modelling options either are or will soon be available for future policy-making. The Office of Water has put together a suite of publicly available models under the rubric BASINS. This is designed for more local scale modelling, but includes most of the basic data needed for these models. Additional models available from the Office of Water include AQUATOX, QUAL2E, and WASP6 (US EPA, 2003h). A number of external models are also available both from the private and academic sectors, such as HSPF (USGS, 2003), and from other government agencies, such as the forthcoming USGS SPARROW model (Smith et al., 1997).

Finally, the regulatory approach of the agency is changing. In compliance with the current law, all point sources that discharge into US waters hold NPDES permits and there are effluent guidelines in place to detail the technical requirements of this discharge. To improve surface water quality further, the EPA must control non-point source pollution, which is much more difficult. Under the CWA, point sources are specifically subjected to the NPDES system. Non-point sources, on the other hand, are covered under Section 319 of this act, where there is no measure of direct control, only the indirect control measures of state management programs, technical assistance, grants, and information provision. In other words, the EPA has weak regulatory authority over non-point sources. To improve water quality further, the EPA will have to control non-point source through voluntary measures, trading schemes, and best management practices.

There are, however, reasons to be optimistic that future valuation exercises will rise to meet these challenges. The National Center for Environmental Economics has issued its *Guidelines for Conducting Economic Analyses* (US EPA, 2000) that details many of these considerations and offers suggestions to deal with them. In addition, OMB is in the process of issuing guidance on BCA. The EPA's TMDL program (US EPA, 2003i) in conjunction with their water trading program (US EPA, 2003g) offer a solution to some of the non-point sources issues. As valuation techniques improve, there is every expectation that they will be incorporated further in the regulatory process.

APPENDIX: LAWS AND REGULATIONS

Under the American system, Congress passes the governing laws and the President, as head of the executive branch, implements the laws. A member of Congress will propose a bill which, if approved, is then sent to the President who has the option to either sign it into law or veto it. If signed, the new law is called an 'Act', and the text of the act is known as a 'Public Statute'. The text of the law is then published in the US Code, which is the official record of all Federal laws (Law and Revision Council, 2000).

In general, laws do not include all of the details necessary for full implementation. Congress authorizes government agencies, including the EPA, to implement laws by creating and enforcing regulations, also called rules. Once an agency decides that a regulation is necessary, it typically (with some exceptions) follows certain steps. The agency researches the rule and publishes a proposed regulation in the *Federal Register*. The public then has a period of time during which they may consider the proposed regulation and send their comments to the agency. The agency then considers all public comments, revises the regulation accordingly, and then issues a final rule, which is also published in the *Federal Register*. Twice a year, each agency also publishes a comprehensive report in a Unified Agenda that describes all of the regulations that it is working on or has recently finished.

Once a regulation is completed and has been printed in the *Federal Register*, it is 'codified' by being published in the Code of Federal Regulations (CFR). The CFR is the official record of all regulations created by the Federal Government (NARA and GPO, 2003). It is divided into 50 volumes, called Titles. Each title focuses on a particular area. Almost all of the environmental regulation appears in Title 40. Once the regulation is in effect, the government agencies then both enforce the law and help the public comply with it (US EPA, 2003b).

NOTES

1. The views expressed in this chapter are entirely those of the authors and do not necessarily represent the views of the US Environmental Protection Agency.
2. By law, the head of the 15 Executive Departments are members of the President's cabinet. While the EPA is not an Executive Department, the Administrator of the EPA is still a member of the President's cabinet.
3. For a brief summary of the adoption of laws and regulation in the USA in general, see the appendix to this chapter.
4. Publicly owned treatment works (POTWs), which are designed to handle municipal sewage are also regulated under the CWA. Each POTW is required to meet discharge standards for secondary biological treatment unless more stringent standards are required to meet a water quality standard or TMDL. Some industrial dischargers also send their effluent to POTWs and these discharges are regulated under the pre-treatment program.
5. For an overview of effluent guidelines and a more thorough description of their development see Kahn and Rubin (1989), US EPA (1996, esp. ch. 5), and Caulkins and Sessions (1997).
6. For a thorough description of the next two paragraphs, see US EPA (1994).
7. Executive orders are neither regulations nor laws. They are official documents issued by the President to manage the operations of the Federal Government.
8. Some of the literature cited in this section post-dates the rule-making effort. This was done either because the exact citation used in the rule-making effort could not be located directly or is summarized or incorporated in the reference cited here. In general, however, an effort was made to refer to the same line of literature and the citations are provided to offer the reader some frame of reference for the benefits category valuation.
9. Somewhat more detail on this analysis is provided in Castillo et al. (1997).
10. Negative benefits are a lower bound resulting from the possibility of increased SO_2 emissions. The EPA considered the SO_2 estimates to be less certain than other benefits estimates (US EPA, 1997b; 1998).
11. The EPA also estimated costs and benefits for nine more detailed case studies and extrapolated from these to national estimates. These extrapolated estimates range from \$91 million to \$451 million.
12. The BCT cost test was promulgated to implement language in the Clean Water Act (US EPA, 1986). The first part of the BCT cost test calculates the incremental cost per pound to remove conventional pollutants when upgrading from BPT to BCT. This must be less than \$0.25 per pound (in 1976 dollars). This \$0.25 per pound benchmark is based on the cost that a POTW would have incurred in upgrading from secondary to advanced secondary treatment. The second test requires comparing the ratio of the incremental BPT to BCT cost per pound to the BPT cost per pound. This ratio must be less than 1.29 (that is, the cost increase must be less than 29 per cent). A BCT option must pass both tests to be the basis for regulation.

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11. The costs and benefits of a revised European Bathing Water Directive in The Netherlands

R. Brouwer and R. Bronda

1. INTRODUCTION

The European Commission (EC) is working towards a revision of the current European Bathing Water Quality (BWQ) Directive (76/160/EEC). Current BWQ standards for *escherichia coli* and intestinal *enterococci* will become more stringent, and, contrary to the existing Directive, the identification of effective management measures in the case of non-compliance will play a much more important role besides BWQ monitoring. This is in line with the principles laid down in the Water Framework Directive (2000/60/EC). In its 2000 Communication, the EC states that the revised BWQ Directive should have a greater emphasis on the application of suitable, prompt management actions, without forgetting the fact that water quality objectives have to be met. Under the new scheme, there will be requirements for both compliance with the quality standards and for reaction when these standards are breached.

In The Netherlands, there are over 600 official bathing locations. Non-compliance is currently limited: less than 5 per cent of all the official bathing sites are unable to comply with current standards (Table 11.1). However, the proposed new standards are expected to result in a substantial increase in the number of non-complying bathing sites to more than 30 per cent. Most of these sites (>95 per cent) concern inland waters, only a few are coastal bathing locations. At these sites, measures will have to be taken in order to comply with the new BWQ standards.

In order to support policy and decision-making regarding the revision of the existing BWQ Directive and the setting of new more stringent standards for bacteriological water contamination, the extent and cause of the problem was investigated and measures identified in order to resolve expected future problems with bathing water quality (Brouwer and van Pelt, 2002). The costs and effectiveness of these measures were estimated

Table 11.1 Number of current and future bathing water locations which do not meet existing and future bathing water quality standards

	Inland freshwater locations (561 locations)	Coastal saltwater bathing locations (95 locations)
Current standards	17 (3%)	0 (0%)
Future standards [1]	167 (30%)	3 (3%)

Note: [1] Based on the expected standard of 500 cfu/100 ml for *escherichia coli* and the 95 percentile of the available monitoring data between 1998 and 2000 (with a minimum of 20 observations per location per year).

along with the least costs to achieve the new BWQ standards (Bronda, 2003). In a separate study the socio-economic benefits of the new standards were also assessed (Brouwer, 2003). These costs and benefits were subsequently compared in order to assess the economic efficiency of the new standards in a pre-feasibility cost–benefit analysis (CBA). Although cost-effectiveness analysis (CEA) has been used before as a decision-support tool for water quality management in The Netherlands (for example, van der Veeren, 2002; van der Woerd et al., 2000),[1] more specifically for nutrients and a limited number of metals, CEA or CBA were not previously used in this specific water domain in The Netherlands, that is, looking at bacteriological water contamination.

The main objective of this chapter is to present and discuss the main findings from these studies and their role in the decision-making process. Two important issues highlighted by the studies were, first, the uncertainties surrounding the source-pathway-impact assessment and consequently the cost-effectiveness analysis, and, secondly, the public importance and socio-economic value attached to improved bathing water quality, suggesting that increased investments in BWQ improvements are justified.

The remainder of this chapter is organized as follows. Section 2 details the set-up and main results from the cost-effectiveness study and section 3 the benefits assessment. Section 4 presents the outcome of the CBA and concludes.

2. SOURCES OF POLLUTION, MEASURES AND COSTS

The CEA focuses on the sanitary quality of designated bathing water locations. This quality is negatively affected mainly by faecal pollution originating from human beings and animals. This pollution enters surface water bodies through various pathways. As a first step in the analysis, all possible sources of pollution and pathways were identified. This was done on the basis of a literature review and additionally through expert meetings. The main sources of pollution of bathing water quality deterioration are summarized in Figure 11.1.

A distinction can be made between *point sources* (for example, insufficiently treated waste water discharge from waste water treatment plants (WWTP), untreated waste water from combined storm water overflow (CSO), untreated discharge from food or other organic processing industries), *diffuse human sources* (for example, manure or slurry spread on agricultural land, waste water discharge from boats), *diffuse animal sources* (for example, bird colonies, horses and dogs on beaches) and *external sources*, that is, sources which are located outside the realm of influence of water managers in The Netherlands (for example, pollution from abroad which enters Dutch bathing locations through the rivers Rhine, Meuse or Scheldt or illegal discharges).

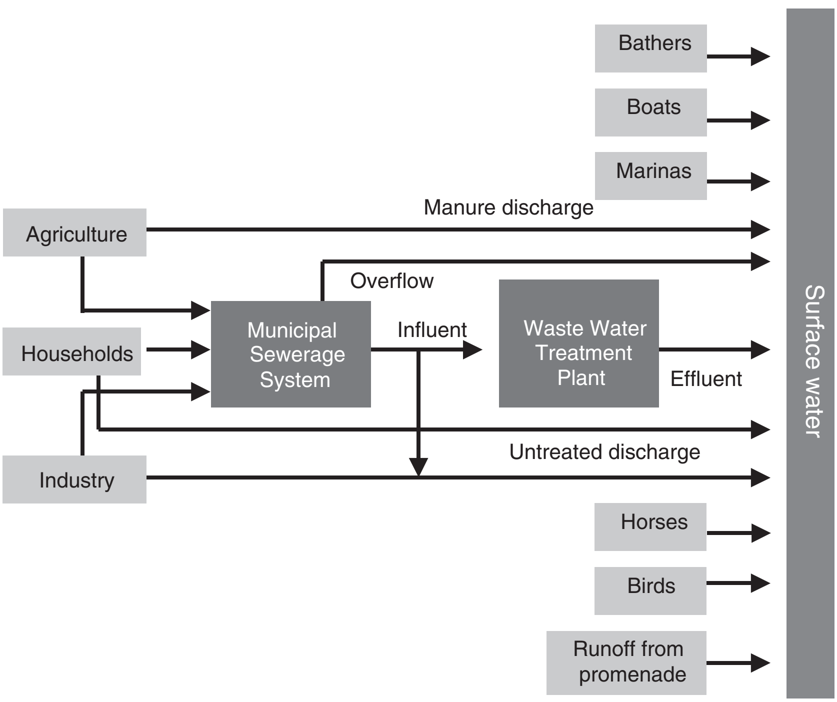

Figure 11.1 Potential sources of bathing water quality deterioration

Pollution can occur directly at the bathing water site (for example, pollution by bathers, horses and dogs on site) or indirectly through various pathways (for example, cattle drinking at a river which is connected to a bathing site, manure or waste water discharged into a ditch which is connected to a river, waste water discharged by commercial and recreational boaters to a system of interconnected rivers and lakes). In the case of indirect pollution, bacteria dilute and dissolve before they reach the bathing site and the receptor (bathers). An important distinction can be made between standing waters at bathing sites (often isolated) such as an isolated lake and flowing waters at bathing sites such as a river or canal. In the former case the sources of pollution are local (on site), whereas in the latter case various off-site (upstream) sources of faecal pollution may play a role. The faster the flowing water, the larger the area of influence. On the other hand, flowing water results in a diminishing concentration of bacteriological contamination. Hence, the location of the bathing site in the water system is of utmost importance when assessing potential sources of pollution and their pathways, making it difficult to come up with generally applicable solutions.

On the basis of the inventory of potential sources of pollution and their pathways, four different types of measures were subsequently identified by the same experts who also identified the potential sources and pathways:

- measures targeted at point sources;
- measures aimed at the elimination or dislocation of discharges;
- measures aimed at changing human behaviour; and
- measures targeted at the pathway of bacteriological contamination.

Examples of the first type of measures are the treatment (enlargement capacity and/or desinfection techniques) of waste water from WWTP, increasing overflow capacity or the individual treatment of waste water. Examples of the second type of measures are the connection of individual households or plants to the sewer system, the dislocation of a WWTP or marina outside the sphere of influence of a bathing site or the construction of non-grazing buffer zones along rivers for cattle. The third type of measures consists, for example, of information and education programmes or signs aimed at changing the behaviour of bathers themselves, recreational boaters or people who walk their dogs near bathing sites or legislation such as the prohibition of the presence of pets or horses at beaches. Finally, examples of the fourth type of measures include hydrological isolation of bathing water or re-freshening bathing water at isolated sites with stagnant water.

In view of the limited time and financial resources available, it was impossible to investigate all bathing sites in detail, which are expected not to be able to comply with future bathing water quality standards (170 in total). Therefore, a sample of 30 sites was selected from the expected non-complying 167 inland and three coastal bathing sites (see Table 11.1). A random sample of 27 locations was taken from the 167 freshwater inland locations, while all three expected non-complying coastal bathing sites were included in the investigation. At each of these 30 sites, the potential sources of pollution were identified with the help of:

1. A previously developed Geographical Information System (GIS) model (Brouwer and van Pelt, 2002), which includes information about the location and pressure exerted by important potential sources such as storm water overflow, marinas, effluent from WWTP and the direct discharge of manure into surface water. The model also shows how different hydrological units in The Netherlands are interconnected and is hence also able to show how pollution in one hydrological unit may affect other units where bathing water sites are located.
2. A questionnaire survey amongst the different water managers responsible for the water quality at the different bathing locations asking them to indicate which sources they believe are responsible for possible bathing water contamination. The results from this survey were compared with the findings from the first step.
3. In those cases where the results from the first and second step did not correspond, follow-up telephone interviews were held with water managers, trying to find out what really is causing the problem at a specific site. In some cases, the outcome of this interview was that a source, which had not been identified in the first step, was added to the list based on the information provided by the water manager. In other cases, the assessment of sources of pollution by the water manager could be dismissed based on available factual information and data about the presence of potential sources.

Two of the freshwater inland locations were excluded from further analysis, because the water managers responsible for these sites did not supply any information about these sites. A third inland location was taken out of the analysis in view of the fact that this location was not officially a bathing water location anymore since 1 January 2002. This means that the assessment and analysis of potential sources, measures and their costs and effectiveness was carried out on a random sample of 27 bathing sites, three coastal and 24 inland locations.

The sources of bacteriological contamination identified at these sites are presented in Table 11.2. An important starting point in the assessment of sources of pollution was that each potential source or pathway is considered a factor of influence unless we are able to prove that they are not. From Table 11.2 it can be seen that the following six sources were identified in more than 30 per cent of all investigated sites:

- waste water from CSO;
- waste water discharge from boats;
- waste water discharged at marinas;
- pets in water or at beaches;
- bathers; and
- bird colonies at or near bathing water locations.

Table 11.2 Results from the assessment of sources of pollution of bacteriological contamination at bathing water sites

Source of pollution	Number of bathing locations at which the source was proven to be a factor of influence	Number of bathing locations at which the source could not be excluded as a factor of influence
Discharge of untreated sewerage	1	2
CSO without measures	7	4
CSO with measures	0	3
Insufficiently treated discharge from WWTP	1	2
Manure from farm	0	1
Manure from agricultural land	0	2
Manure from cattle drinking at water side	0	2
Waste water discharge at marinas	6	5
Waste water discharge from recreational boats	9	3
Waste water discharge from commercial ships	3	2
Pets in water or at bathing water location	6	8
Bathers	12	7
Bird colonies at or near bathing location	9	7
Large international rivers	1	0

The other sources were identified at less than 30 per cent of all the sites investigated. Six potential sources, which were not identified at any location, include discharges from food and organic processing industries, slaughterhouses, non-functioning sanitary facilities at bathing water locations and illegal discharges. Obviously, the latter are difficult to prove as these are not registered or are difficult to observe and were therefore not mentioned by any of the water managers. They may nevertheless play an important role in explaining why sites are contaminated. However, the extent to which illegal discharges play a role is unknown.

The last column in Table 11.2 shows the number of bathing water locations at which specific sources of pollution could not be excluded as a factor of influence, because we were unable to prove that they were non-existent. For instance, untreated sewerage could be demonstrated to be a factor of influence in a third of all the locations investigated, but could not be excluded as a factor of influence in two-thirds of the locations.

The local sources of pollution were linked to the bathing sites' percentile values with which they exceed the expected new bathing water quality standard. This was done in order to be able to get an indication of the weighted contribution of the sources to the overall bathing water problem and hence value the relative contribution of the identified sources to this problem. A source which contributes 10 per cent to a 95 percentile value which is 20 times higher than the standard is, for instance, considered relatively more important than a source which contributes 100 per cent to a 95 percentile value twice as high as the standard.

The weighted contribution of the various sources is presented in Figure 11.2. Important observations from Figure 11.2 are, first, the high percentage (22 per cent) of non-identified sources of pollution. These often include a mix of different sources at or near large surface waters, for which the exact origin of pollution is hard to determine. Second, the large contribution of diffuse sources is remarkable, that is, waste water discharge from recreational boats (13 per cent), bathers (10 per cent) and bird colonies (11 per cent).

Based on the assessment of sources of pollution, adequate sets of measures were identified per bathing location and the costs and effectiveness of these measures estimated. Costs and effectiveness of measures were estimated on the basis of expert judgement, available data sets at DHV Water, the company hired to carry out the cost-effectiveness analysis, and additional field research. The most cost-effective measures identified at the different locations are presented in Figure 11.3. Circulation and (ultraviolet or chloride) disinfection are the most frequently proposed measures (in 11 and 7 per cent of all the sites investigated respectively), followed by the prohibition of pets at bathing locations (in 6 per cent of the cases). The

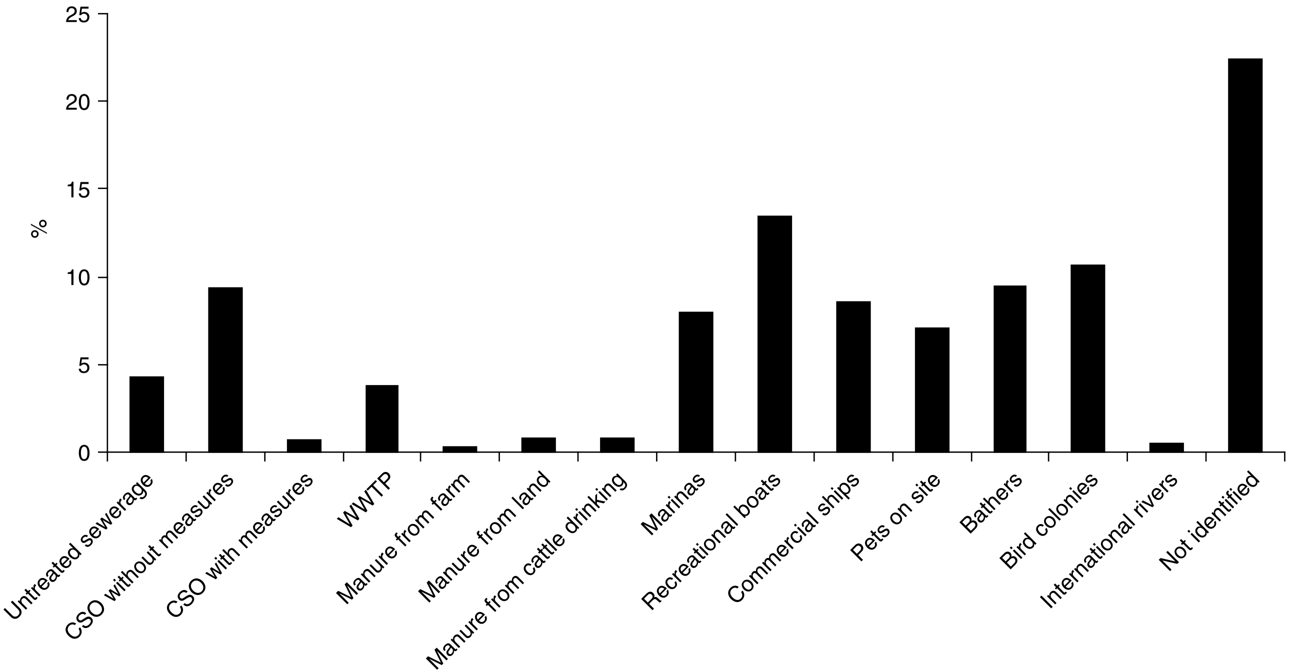

Figure 11.2 Estimated relative contribution of bathing water pollution sources to the overall problem of bacteriological bathing water contamination

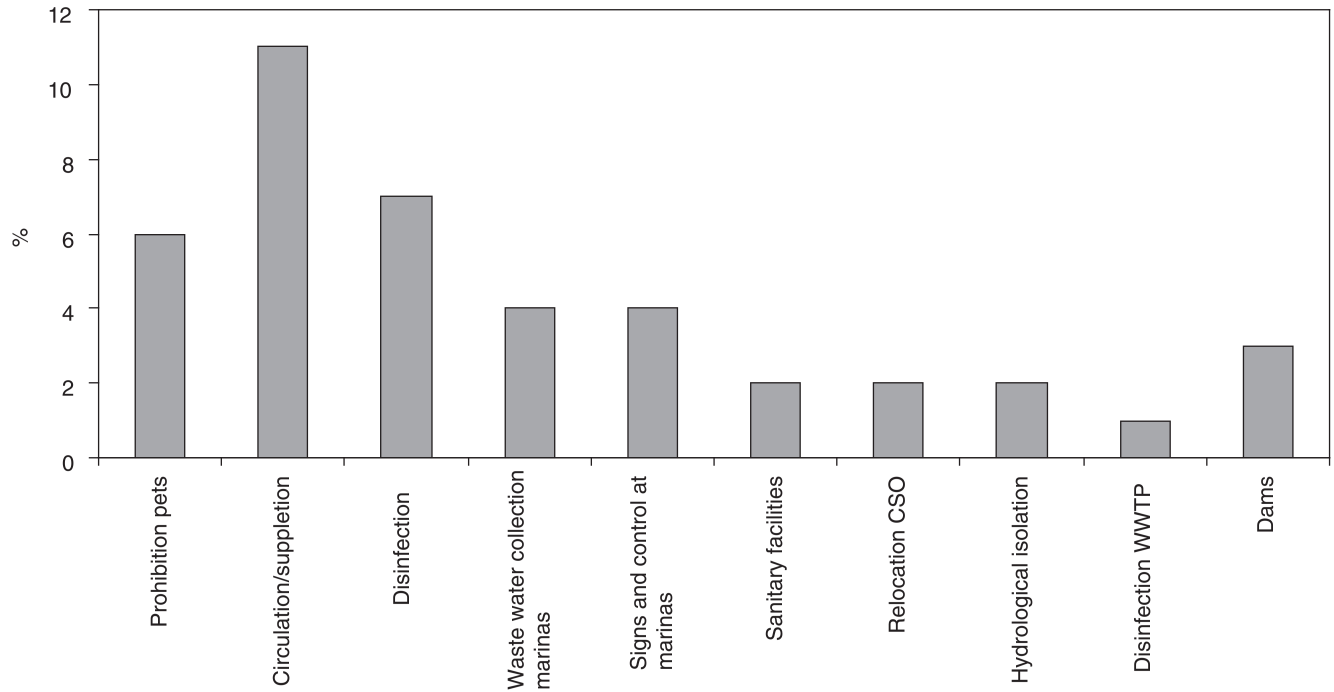

Figure 11.3 *Proposed cost-effective measures at the different bathing water locations and their relative contribution to solving bathing water quality problems at the sample sites (percentage of sites at which the measures were proposed)*

measures presented in Figure 11.3 mainly refer to bathing sites with standing waters, where the most cost-effective measures can be taken. Coming up with cost-effective measures for sites with flowing waters is difficult as these measures often have to deal with a mix of diffuse sources of pollution. The degree of control at these latter sites is very low.

Table 11.3 shows the estimated costs and effectiveness of the proposed sets of measures at the 27 different bathing water locations. The first 24 locations refer to freshwater bathing waters and the last three locations to the three coastal sites. Most of the locations concern sites with flowing waters. Nine of the 24 inland locations (38 per cent) are standing bathing waters (S). It is at these standing waters where measures can be taken most effectively to reduce or eliminate bacteriological contamination: on average at the nine locations 84 per cent (and up to a maximum of 98 per cent at an individual site) of the amount of bacteriological on-site contamination (in cfu/l) compared with 67 per cent (and up to a maximum of 84 per cent at an individual site) in the case of flowing waters.

Table 11.3 shows that for seven of the 24 inland bathing locations (29 per cent), all with flowing water, no measures can be identified. The main reason for this is the complexity and diversity of the sources of pollution and their pathways. It was impossible to identify what exactly causes non-compliance of these six cases and therefore also no effective sets of measures could be identified. In the case of the three coastal bathing water locations, for only one site a set of measures could be identified. In those cases where sets of measures could be identified (75 per cent of the inland sites and 33 per cent of the coastal sites), the total investment costs are €2 million to reduce pollution for the inland bathing water sites and €360 000 for the coastal bathing water location. The corresponding operating costs are respectively almost €360 000 and €23 000 per year.

The costs to reduce one colony forming unit (cfu) per litre range from €0.8 to €8.4 per year for standing freshwaters and €0.2 to €8.8 per year for flowing freshwaters.[2] The unit costs for the single coastal location are €3.2 per year. Relating the investment and operating costs to the number of bathers counted at the sites during the bathing season in 2001, the annual costs per bather can be calculated. These costs vary from one euro per bather per year (location no. 10) to almost 4000 euros per bather per year (location no.18). On average, these costs are €221 per bather per year for standing bathing water and €1051 per bather per year for flowing freshwater. In the case of coastal bathing water, the estimated costs are €10 per bather per year.

In a final step, the estimated costs have been scaled up to a national level. This was done by multiplying the estimated costs for the inland freshwater bathing locations with a factor 7 (167:24), assuming that the 24 randomly selected inland bathing sites are representative for all non-complying

Table 11.3 Overview of estimated investment and operating costs at the different bathing water locations (price level 2002)

Site	Type	Bathers (max/day)	Measures identified?	Investment sum (€)	Operating costs (€/year)	Overall cost-effectiveness (€/year/cfu/l)	Overall cost-effectiveness (€/year/bather)
1	S	750	Y	15 000	54 076	8.42	74
2	S	150	Y	21 000	10 820	0.84	82
3	F	40	Y	35 000	31 340	6.21	871
4	F	40	Y	83 000	40 760	8.66	1 227
5	S	2 000	Y	39 000	34 000	3.32	19
6	F	n.a.	Y	42 000	41 470	1.03	—
7	S	n.a.	Y	210 500	23 400	2.90	—
8	S	n.a.	Y	20 950	10 838	1.38	—
9	S	25	Y	41 900	11 676	3.02	579
10	F	2 000	Y	10 500	250	0.39	1
11	F	n.a.	Y	50 000	—	0.23	—
12	F	n.a.	N	—	—	—	—
13	F	300	N	—	—	—	—
14	F	500	N	—	—	—	—
15	F	100	Y	83 800	13 352	0.98	189
16	F	200	Y	214 000	4 280	1.27	43
17	F	400	N	—	—	—	—
18	F	25	Y	838 000	43 520	8.84	3 975
19	F	25	N	—	—	—	—
20	F	250	N	—	—	—	—

Table 11.3 (continued)

Site	Type	Bathers (max/day)	Measures identified?	Investment sum (€)	Operating costs (€/year)	Overall cost-effectiveness (€/year/cfu/l)	Overall cost-effectiveness (€/year/bather)
21	F	1 000	N	—	—	—	—
22	S	100	Y	40 950	16 638	3.98	194
23	S	n.a.	Y	230 800	4 616	2.95	—
24	S	50	Y	40 950	16 238	1.73	379
Total				2 017 350	357 274		
25	F	2 000	Y	362 800	12 256	3.24	10
26	F	1 000	N	—	—	—	—
27	F	30	N	—	—	—	—
Total				362 800	12 256		

Notes:
Column 1: first 24 sites are inland freshwater bathing water locations; last three sites are coastal locations.
Column 2: S = standing water; F = flowing water.
Column 3: maximum number of bathers counted during the bathing season (n.a. = not available).
Column 4: whether or not a set of measures could be identified based on the assessment of sources and pathways of pollution. Y = yes; N = no.

bathing sites in The Netherlands.[3] In the case of the coastal bathing water sites, all non-complying sites were included in the analysis, even though no set of measures could be identified for two of the three sites and hence also no costs could be estimated. This results in a total estimated investment sum of €14.5 million and annual operating costs of €2.5 million.

3. BENEFIT ESTIMATION OF BATHING WATER QUALITY IMPROVEMENTS

Improving bathing water quality is expected to have significant and substantial recreational benefits. The estimated number of people swimming at non-complying sites (based on the proposed new BWQ standards) on a hot summer day is about 125 000. Most importantly, the health risks of bathing in open waters are expected to be reduced by 50 per cent. Currently, one in every 10 bathers runs a risk of getting one or more of the following health symptoms when bathing water quality standards are not met: infections to eyes, ears and throat, and stomach upset (*gastroenteritis*) such as diarrhoea. Meeting the proposed new bathing water quality standards means that the health risks of bathing are reduced to one in every 20 bathers. The above mentioned health risks are especially high when swimming, for example, during a hot day directly after heavy rainfall causing storm water overflow at or near bathing locations (that is, discharge of excess rainwater together with untreated sewer) or when swimming in standing waters during a hot weather period with increased algae blooms.

Public perception and valuation of improved bathing water based on the new proposed BWQ standards was assessed based on a large-scale contingent valuation (CV) survey. In December 2002 a questionnaire consisting of 45 questions and based on Dillman's (1978) 'total design method' was sent to 5000 randomly selected households in The Netherlands. In the questionnaire, households are asked about:

- their bathing behaviour (how often, where);
- their perception of bathing water quality in The Netherlands (distinguishing between freshwater and coastal waters);
- whether they ever got ill after swimming in open water and whether they saw a doctor for this;
- whether they are aware of and are informed about existing BWQ standards;
- how they feel about being unable to swim in open water during the bathing season;
- how urgent and important they believe improving BWQ is;

- to what extent they are able to relate the information provided in the questionnaire about the new proposed BWQ standards and the reduced health risks to themselves;
- whether they are willing to pay additional taxation in order to improve BWQ in The Netherlands and hence reduce the health risks involved;
- their demographic and socio-economic background; and
- their ability to answer the willingness-to-pay question based on the information provided.

More than 1500 questionnaires were returned (response rate of 31 per cent). Based on the information provided about respondent demographic and socio-economic background (age, household size, education, income), it was concluded that the sample was representative for the whole of The Netherlands. Sixty per cent of all respondents indicated that they swim in open waters in The Netherlands. A quarter of those who said they never swim in open waters gave water quality as their main reason. Fifteen per cent do not like swimming or cannot swim, almost 10 per cent only swim in public swimming pools, and another 10 per cent said he or she is either too old or claims the water is too cold. More than half (55 per cent) of those who said they do swim in open waters, mentioned coastal locations as their most important bathing site. On average over the past five years, respondents swim on eight days per year. During the 2002 bathing season, respondents indicated to have swum on between six and 10 days. Thirteen per cent of all respondents indicated they have suffered from symptoms of poor bathing water quality such as eye, ear and throat infections and diarrhoea. Forty per cent of these people went to see a doctor with these symptoms.

A remarkable finding is that people perceive coastal water quality and inland freshwater quality as significantly different. The quality of coastal bathing water is perceived higher than the quality of inland freshwater. The same applies when asking respondents how dangerous they believe swimming in coastal and freshwaters is for their health (that is, health risks as a result of water quality, not drowning risks as a result of, for instance, currents or collisions with surfers or boats). Coastal waters are judged safer than inland freshwaters. On the other hand, almost 40 per cent of all respondents believe that inland water quality has improved over the past 10 years compared with 20 per cent who believe the same about coastal waters. However, most respondents (45 per cent) said they do not know whether coastal bathing water quality has improved or deteriorated over the past 10 years against 33 per cent in the case of inland bathing water quality.

A third of all respondents feel that they are being insufficiently informed about bathing water quality. Half of all respondents feel they are being sufficiently informed. Eighty-five per cent of all respondents said that they know that there exist standards for BWQ in The Netherlands. A majority of 60 per cent of all respondents indicated that they feel bad if they are unable to swim during the bathing season as a result of bad water quality and are willing to pay in principle to improve BWQ in The Netherlands and hence reduce the health risks involved. A quarter of all respondents would not mind if they are unable to swim and are also not willing to pay to improve BWQ, while 15 per cent are neutral and do not know whether or not they are willing to pay for improved BWQ.

Those who were not willing to pay were asked why not. The most heard reason why people were not willing to pay was that the polluter should pay, followed by reasons like 'I never swim in open water', 'the current situation is good enough' and 'I don't believe that the money will be spent on improving BWQ'. Reasons like the latter (mistrust that the money will be spent on what it is intended for) are indicative of what are usually called 'protest bidders' in the CV literature. A large amount of protest to the WTP question can seriously invalidate the research. Thorough pre-testing is an essential prerequisite to produce valid research results in CV studies. In this study, a total of 138 protest bidders were detected, that is, 8 per cent of the total response. This is considered a reasonable result. Combined with the fact that a majority of 62 per cent indicate that they have no problem answering the WTP question and 75 per cent of all respondents claim that the information provided in the questionnaire is sufficient to answer the WTP question, this supports the validity of the CV survey.

Those who replied positively to the willingness-to-pay question were subsequently asked whether they would be willing to pay every year a specific amount of extra money in general taxation in order to improve BWQ and hence reduce the health risks involved. Twelve different money amounts (also referred to as 'bid levels' in the CV literature) were used in a dichotomous choice format. These bid levels, ranging from €1 to €200 per year, were based on extensive pre-testing of the questionnaire and randomly allocated to the randomly selected households. It was furthermore emphasized in the questionnaire that this amount of money will be used exclusively to fund the additional costs of measures to improve BWQ and reduce the health risks involved. The cumulative probability function is shown in Figure 11.4. As expected, the probability of saying 'yes' to a specific bid amount decreases as the bid level increases.

Mean WTP can be estimated based on different statistical methods. Here we present the most conservative estimation result, which was based on a linear-logistic regression analysis (see Brouwer, 2003, for more details).

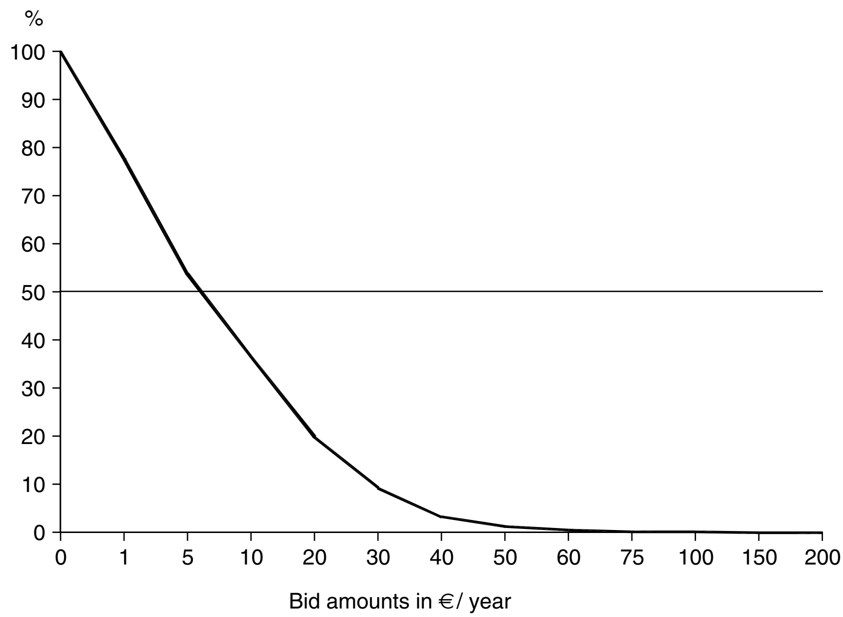

Figure 11.4 *Cumulative probability function of 'yes' replies to the WTP question*

Table 11.4 *Mean WTP values for users and non-users (price level 2002)*

Category	Mean WTP (€/household/year)
Whole sample population	35 (3.6)
Users (bathers)	41 (3.8)
Non-users (non-bathers)	22 (6.6)

Note: Standard error between brackets.

The estimated mean WTP is €35 per household per year. A distinction can also be made between mean WTP for people who bathe in open water in The Netherlands and people who do not bathe in open water, usually referred to as users and non-users respectively (see Table 11.4). As expected, non-users are willing to pay, on average, less than users, but, even so, they are still willing to pay a substantial amount of money (just over €20 per household per year).

Aggregating the overall WTP estimate across the whole population that benefits from improved BWQ (6.9 million households[4]), this results in a total economic value of €242 million per year. Aggregating the user value (€41/household/year) across those in the population who actually bathe in open water in The Netherlands (users) (60 per cent of the 6.9 million households in The Netherlands), we get a total economic value of €170 million per year.

4. CONCLUSIONS AND RECOMMENDATIONS

If we compare the estimated least costs to achieve the new proposed BWQ standards presented in section 2 (€3.3 million per year) with the estimated benefits in terms of public WTP for improved BWQ and hence reduced health risks presented in section 3 (€170 million per year), it becomes immediately clear that the annual benefits exceed the estimated annual costs. Discounting the estimated costs over a period of 20 years at the prescribed 4 per cent discount rate results in a total cost figure of approximately €50 million. Discounting the estimated benefits over the same time period at 4 per cent yields a total benefit of €2.4 billion, which is almost 50 times higher than the estimated costs. It was furthermore estimated that approximately 125 000 bathers are protected on a hot summer day at sites, which are expected not to be able to comply with the new proposed BWQ standards. Based on these findings the conclusion is quickly drawn that it is economically efficient and wise to improve BWQ and reduce the health risks involved.

However, the pre-feasibility cost–benefit analysis carried out here is surrounded by a number of uncertainties, requiring careful interpretation of the results found. Perhaps the most important source of uncertainty is the reliability of the existing monitoring results and the extent to which non-complying bathing sites face structural or incidental problems of bacteriological contamination. The monitoring data used as the basis for the assessment of future non-compliance of sites is based on two-weekly measurements at the more than 600 sites in The Netherlands. At each site one sample is taken every two weeks. Hence, the reliability of the prescribed monitoring practices is doubtful to say the least. Important factors, which may have caused non-compliance with BWQ standards, including weather conditions, are not taken into account. It is therefore impossible to assess the nature of non-compliance, that is, structural or incidental as a result of, for instance, heavy rainfall and storm water overflow the night before the sample was taken.

Another important source of uncertainty is the complex diffuse nature of bacteriological contamination of bathing water, especially flowing waters. The estimated least costs to achieve the new proposed BWQ standards only refer to cost-effective measures that can be taken at about two-thirds of all the non-complying sites (mainly isolated standing waters). In a third of all cases, mainly flowing water systems, no effective set of measures could be identified due to (1) the diffuse nature of the sources of bacteriological contamination (either no source could be identified at all or a mix of diffuse sources were expected to be responsible for non-compliance) and (2) sources which are located outside the sphere of influence of the responsible water manager, such as bacteriological contamination from abroad. More in-depth research is needed to identify which sources exactly are underlying the BWQ problems and to what extent the problem is a structural and not merely an incidental one, in order to be able to identify adequate measures. Moreover, the effect of algae and viruses on BWQ was not considered in the study and neither was the cost-effectiveness of closing non-complying bathing sites.

In this latter case, more research is needed regarding the effect of closure on the number of swimmers visiting these sites (and the possibilities they have to visit other sites nearby) and the economic revenues lost in the associated recreation sector. In another non-published study, it was estimated that the annual loss of income in retail and catering business and marinas at coastal bathing sites could add up to €5 and €8 million if BWQ standards are not reached (assuming no reallocation effects and full employment).

NOTES

1. Cost-effectiveness analysis has started to play a much more prominent role in Dutch water policy more recently after the introduction of the Water Framework Directive, which explicitly asks for the selection of cost-effective programmes of measures. Efforts are under way to set up databases, which should enable policy and decision-makers in the near future to assess the cost-effectiveness of different measures aimed at the reduction of the emission of water pollutants.
2. Investment costs are translated here into annual capital costs.
3. The extent to which the selected sample sites are representative for the whole population of in the future non-complying bathing water locations was also examined in so far as possible on the basis of available information. Important criteria were (1) the nature of the source(s) causing bathing water quality deterioration, (2) the number of bathers visiting the locations, (3) the physical characteristics of the bathing water location (that is, current or standing water systems) and (4) the geographical location of the sample sites.
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12. Cost–benefit analysis of improved bathing water quality in the United Kingdom as a result of a revision of the European Bathing Water Directive

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1. INTRODUCTION

In the last few decades, both the general public and policy-makers have become increasingly concerned about sewage discharges to coastal bathing waters in the European Union (EU) and the consequent risks to public health (CEC, 2002, House of Lords, 1994–95). The public health risks of sewage discharged into coastal marine waters are derived from human population infections. The sewage contains various micro-organisms that have been shown to be pathogenic and the causative agents of several human diseases. The main risk faced by people bathing in sewage contaminated water is in increases to minor morbidity such as gastrointestinal and upper respiratory tract ailments.

The European Commission (EC) Bathing Water Directive of 1976 (CEC, 1976) sets out standards for designated bathing waters which should be complied with by all member states. This has been one of the first and most important elements of European Water Policy. The 1976 Bathing Water Directive reflects the state of knowledge and experience of the early 1970s, in respect of its technical-scientific basis, the managerial approach and the involvement of the public. Recently changes in science and technology as well as in managerial experience have obliged the Commission to consider revision of EU environmental legislation where appropriate. Further legislation has thus been proposed on more than one occasion by the EC in the form of revisions to the 1976 Directive (CEC, 1994; 2000; 2002).

However, policy-makers and regulators face a number of dilemmas in the area of coastal bathing water health risk policy. Whilst sustained year-on-year bathing water improvements have been obtained as a result of the many investments in new sewage treatment schemes, more recently, further improvements are becoming increasingly difficult to deliver, largely due to the role of diffuse non-sewage sources of faecal bacterial pollution from catchments draining to the coast. Furthermore, there is a question mark over the level of protection to be afforded against minor illness acquisition by EC standards. The costs of tightening these standards are very expensive and the health gain associated with any tightening is likely to be measured in terms of self-limiting and minor illness, such that there is a question as to whether any expenditures on sewage clean-up represent effective and efficient use of resources. Regulators and governments have to balance the public desire for better environmental quality with the economic impact of policy changes on both water bill payers and the financial health of water companies. Furthermore, any new policy must be compatible with EU Water Policy, which has been completely restructured by the adoption of the Water Framework Directive and which provides a coherent managerial framework for all water-related EU Legislation.

The central purpose of this chapter is to conduct an economic cost–benefit investigation of the EC Bathing Water Directive revision. In particular it seeks to consider the question of whether the revision is worthwhile in terms of the economic benefits of coastal bathing waters complying with it, or whether the resources required to afford compliance would be used more efficiently to achieve other societal goals. The economic benefits are estimated using a contingent valuation study (Mitchell and Carson, 1989), which considers a bathing water quality improvement scenario based on a revised Directive. The focus is on the public's willingness to pay for particular bathing waters to comply with such legislation and, by implication, on the public health benefits afforded to individuals and society. These economic benefits are compared to the costs of implementing changes to bring bathing waters up to the required standard.

2. THE BENEFITS OF THE EC BATHING WATER DIRECTIVE REVISION

Public policy decisions on coastal bathing water health risks influence health by reducing the adverse human health effects expected from bathing in sewage-contaminated coastal waters. The focus of such policy decisions is on the requirement of coastal bathing water quality to achieve compliance with certain minimum water quality standards – the EC Bathing Water Directive Standards. These standards serve as appropriate 'acceptable' health risk standards and are measured in terms of bacterial quality criteria that are indirectly related to adverse human health effects. The human health effects from bathing in sewage-contaminated coastal waters primarily consist of minor morbidity impacts. These human health effects have a number of economic consequences. As such it is possible to estimate economic welfare (benefits) measures of the effect of public policy decisions on 'acceptable' coastal bathing water health risks. The economic consequences of the adverse health effects from bathing in sewage-contaminated coastal waters include:

1. Medical and care-giving costs – such as out-of-pocket medical expenses of the affected individual (or family), the opportunity costs of time spent in obtaining treatment, plus costs paid by insurance, and so on. The individual may also be unable to undertake some or all normal chores and thus require additional special care-giving and services not reflected in normal medical costs.
2. Work loss – this includes lost personal income, plus lost productivity (irrespective of whether the individual is compensated or not – whilst some individuals may be paid sick pay and hence not perceive any income loss, sick pay is nevertheless a cost of business and in this respect reflects lost productivity).
3. Other social and economic costs – these include lost opportunities for enjoyment of leisure activities, discomfort or inconvenience (pain and suffering), anxiety, concern and inconvenience to family members and others. In addition, individuals may engage in defensive and averting expenditures and activities associated with attempts to prevent the health impacts.

The medical costs plus work loss (consequences 1 and 2) constitute the measure of welfare known as cost of illness (COI). Since this measure does not include other social and economic costs it will not reflect the total welfare impact of an adverse health effect. The maximum willingness to pay to reduce the risk of the health effect is, however, a comprehensive measure of welfare. It reflects all the reasons an individual might want to avoid an adverse health effect, including financial and non-financial concerns. Furthermore WTP reflects expectations rather than realized damages and, since public policy decisions on 'acceptable' coastal bathing water health risks are *ex ante* decisions (based on expected reductions of adverse human health effects), then WTP is again considered appropriate.[1]

Whilst the main focus of policy decisions on acceptable coastal bathing water health risks is obviously on public health protection, such policy decisions may nevertheless have other additional benefits as part of the policy package, which impact on social welfare and hence have an economic value. These additional benefits include increases in tourism and employment, possible ecological impacts, and other aesthetic and amenity improvements. Tourism expenditures by beach visitors (for example, food, accommodation, shopping, and so on) and employment increases from any increase in tourism are sometimes perceived as benefits since they may be very important for the development of regional coastal economies. However, they are likely to be transfers, that is, the activities would have taken place elsewhere in the country, and hence there is no net increase in spending across the country. Although they can legitimately be added to an economic impact analysis, they should not be included in a cost–benefit analysis since they do not represent net economic gains (Loomis and Helfland, 2001).[2] Other benefits related to marine and wildlife ecology, aesthetics, amenity and non-use improvements can all be considered legitimate components of the total economic value of coastal bathing water health risk policies and hence should be included in benefits assessments of those policies.

In organizing and presenting measures of the benefits of coastal bathing water health risks policy there are a number of other issues, which need to be considered. First, it is frequently necessary to choose between options that differ in temporal patterns of benefits or that differ in their duration. Using a rate of time preference (discount rate) the streams of benefits (or costs) can be adjusted to yield discounted present values. Second, estimates of benefits will not be known with certainty. Some data and models will be likely to introduce substantial uncertainties into the estimations of benefits. Numerous assumptions are often made in deriving the benefits estimates and therefore the conclusions drawn in cost–benefit analysis will be sensitive to the degree of uncertainty present and the assumptions that were made. Reporting the uncertainty of the data, the assumptions used, and how the uncertainty and assumptions affect the results are thus important components of the presentation of the benefits of policy.

The present study sought to estimate willingness to pay for bathing waters to comply with a revised EC Bathing Water Directive and hence, whilst the primary focus is with public health benefits, alternative motivations, stemming from the additional benefits mentioned above, may also find some expression in the WTP values being expressed. A contingent valuation (CV) study was designed to estimate the economic benefits associated with compliance of *all* beaches in the Anglian water region (37 beaches in total) with a revised EC Bathing Water Directive. The study comprised of an identical CV survey questionnaire undertaken at two coastal and one urban location in East Anglia, and was, wherever possible, designed to correspond to the NOAA ‘Blue Ribbon’ panel guidelines (Arrow et al., 1993) on conducting CV studies.[3]

A contingent valuation survey requires that the change in the provision of the good that respondents are being asked to value is communicated and understood by them. A procedure to elicit respondent's values is then required (elicitation method), as well as a mechanism by which respondents are told that they will have to pay for the change in provision (payment vehicle). One needs to be confident that respondents are actually valuing the specific change in provision and not some other more general change. These elements are usually contained within an information statement, a valuation scenario and questions, and debriefing questions. The elicitation method used in this study was a referendum style payment principle, followed by an open-ended WTP question. The payment vehicle used was an increase in water rates per year, which although problematical (owing to the fact that visitors to the coastal location may be from outside the charging area) was nevertheless considered to be the most likely way of financing any bathing water improvements.

The information statement was designed to inform respondents about sewage contamination of bathing water and the subsequent possible health risks from bathing, as well as the existing EC bathing water standards. In this respect they were informed of the current status quo regarding the standard of bathing water quality and associated risks of illness associated with most beaches in the region. This information stated that although most beaches in the region pass the existing Directive, the health risks associated with beaches, which satisfy the standard are as follows:

> out of every 1000 bathers,
> 51 will suffer from vomiting, diarrhoea, indigestion or nausea accompanied by fever;
> 20 will suffer from respiratory illness such as sore throat, runny nose, coughing;
> 54 will suffer ear ailments, and
> 24 will suffer from eye ailments.
> Some bathers may suffer more than one of these illnesses at the same time.

Respondents were then asked to consider the introduction of a new standard, which should result in further reductions in risks to health at those beaches that satisfy the new standard. They were told that in order for all beaches in the Anglian region to achieve compliance with the new standard, extra expenditure in the form of higher water rates may be required. Respondents were then asked a payment principle question, with those agreeing to the principle being asked a further open-ended WTP amount question. A budget constraint reminder was given prior to the payment principle and WTP amount questions. In addition, prior to the WTP amount question, a reminder was given that respondents already pay for sewage treatment in order to ensure compliance with the existing directive, and therefore the benefit of the new standard is in terms of further reductions in risks to health at those beaches that comply with the new standard.

In describing the proposed new EC Directive standard, it was not possible to define the specific health risk probability reductions associated with compliance (since at the time scientific evidence on this was limited). In this respect the contingent commodity being offered was implicitly framed in terms of a change between two perceived 'publicly acceptable' health risk levels. The first associated with the existing directive and the second in terms of the revised Directive. Hence, although the framing of the contingent commodity is very much in terms of public health concerns, the reliance on respondents perceiving the changes in health risks means that there is scope for them to incorporate additional benefit motivations (other than just public health risk reductions) into their valuations. Given the use of a change in perceived 'publicly acceptable' health risk levels, it was decided explicitly to examine the variation in people's perceptions regarding this change. Prior to the valuation questions, therefore, respondents were asked to state what they themselves expected in terms of proportional health risk reductions (in terms of incidence of illness) from the new EC standard relative to the existing EC standard.

The survey was administered using in-person interviews. The sample of respondents was chosen at random amongst the population of visitors to Great Yarmouth and Lowestoft beaches. A partially stratified (according to house-type areas) sample was chosen amongst the population of household residents in the city of Norwich. The sampling strategy was such as to obtain a varied sample rather than a true cross-section in order to investigate the effects of demographic and social factors.

2.1 Summary Results

The total sample size was 616, of which 230 interviews were carried out at Great Yarmouth, 189 at Lowestoft and 197 at Norwich. The socio-economic composition of the three location sub-samples and visitor type composition of the beach based questionnaire samples (Great Yarmouth and Lowestoft) was examined. Socio-economic composition was similar, except for statistically significant differences between samples in the mean number of household residents, the percentage who were members of an environmental organization, and the percentage of Anglian Water ratepayers. Norwich had a significantly lower mean number of household residents, whilst environmental organization membership and number of Anglian Water ratepayers were significantly higher in Norwich than in the other two sample sites. The composition of visitor types was also quite different, with the main group of respondents in Great Yarmouth being holiday-makers, whereas at Lowestoft the composition is more evenly distributed between holiday-makers, day-trippers and local residents. This difference in composition may have implications for some of the results presented subsequently.

Table 12.1 provides a summary of the responses to the payment principle valuation questions for each of the three site samples, as well as according to visitor type (for the two beach-based survey samples). In nearly all cases a majority of respondents were in favour of all beaches in the Anglian region having to comply with the new EC standard even if it cost their household some money in extra water rates. The highest rates in favour were found at the urban sample location (Norwich), whilst the beach-based sample locations had somewhat lower rates in favour. There was in fact a statistically significant relationship between payment principle responses and site location. High rates in favour were also found amongst the holiday-makers and day-trippers in each of the beach-based survey samples, and whilst it may be argued that this may be partly due to free-riding by those holiday-makers and day-trippers living outside of the region (who hence can avoid the higher water rates charges), the fact that a higher rate in favour was observed in the urban sample location (where free-riding is not possible) suggests this is not the case. For the beach-based samples, whilst holiday-makers and day-trippers have higher rates in favour than the local residents, there is not a statistically significant relationship. In any case, the higher rates are as expected since holiday-makers and day-trippers are likely to make more use of the water in terms of bathing activity. The lower rates in favour amongst the local residents in each of the beach samples are possibly due to lower incomes amongst this group.

Analysis of why people voted against the payment principle found that the main reasons were to do with not being able to afford to pay, not living in the region, and having problems with the payment vehicle – people felt that they paid enough taxes already and objected to 'profiteering' by the privatized water utilities.

Respondents who answered positively to the payment principle were asked the open-ended willingness to pay question. Table 12.2 presents a summary of the mean WTP amounts found for each of the three site samples, as well as the combined sample, according to respondents'expectations regarding the reductions in number of illnesses achieved by compliance with the revised Directive. Respondents were asked whether they believe the revised Directive results in reductions in the number of illnesses by 25, 50, 75 or 100 per cent. These mean WTP values are aggregated for the English and Welsh population using 2002 prices and converted to net present values using a 25-year time frame and discount rates of 6 per cent

Table 12.1 *Outcome of the payment principle question*

WTP for increase in water rates?	Great Yarmouth				Lowestoft				Norwich	All sites
	All sample	Holiday-makers	Day-trippers	Local residents	All sample	Holiday-makers	Day-trippers	Local residents		
% ‘No’	32.2	25.9	32.1	51.2	27.5	26.3	19.7	39.3	16.24	25.6
% ‘Yes’	64.8	70.2	66.0	48.8	70.9	73.7	78.9	58.9	82.74	72.4
% ‘Indifferent’	2.2	3.1	0.0	0.0	0.5	0.0	0.0	1.8	0	1.0
%‘Don’t Know’	0.8	0.8	1.9	0.0	1.1	0.0	1.4	0.0	1.02	1.0
% Total	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0
Sample size N	230	131	53	41	189	57	71	56	197	616

Table 12.2 Summary of mean WTP and net present value of the benefits of the revised EC Bathing Water Directive

Directive and benefits considered	Study site	Mean WTP per household per year (£)		Aggregate WTP per year[2] (£2002 Million)	Total net present value (£2002 million) (25 years, 6% discount rate)	Total net present value (£2002 million) (25 years, 3.5% discount rate)
		Study year prices	2002 prices[1]			
Revised Directive	Great Yarmouth	27.8	30.5			
irrespective of expected	Lowestoft	31.6	34.7			
reduction in illnesses	Norwich	49.1	54.0			
	All sites	36.1	39.7	953	12 914	16 258
Expected reduction 100%	All sites	50.7	55.7	1 337	18 114	22 803
Expected reduction 75%	All sites	42.1	46.3	1 110	15 044	18 939
Expected reduction 50%	All sites	28.0	30.8	738	10 006	12 596
Expected reduction 25%	All sites	26.6	29.2	700	9 491	11 948

Notes:
1. Figures from year that original WTP derived (either 1995 or 1997) are adjusted by GDP deflators (UK Treasury figures) to give 2002 prices.
2. Aggregate WTP for England and Wales is found by multiplying the household WTP figures by the number of English and Welsh households = 24 million.

and 3.5 per cent. The benefit aggregations make the assumption throughout that the WTP values are representative of the WTP values of the English and Welsh population at large. It is acknowledged that the various samples cannot be considered to be highly representative of the population. In addition, it should be noted that the CV studies used to generate the benefit estimates only covered improvements at a small proportion of the total number of bathing waters in England and Wales, and hence the estimates may possibly be underestimates of countrywide improvements. In order to work out the aggregate WTP for the English and Welsh population per year the relevant mean WTP value is multiplied by the number of households in England and Wales, currently equal to 24 million.

Mean WTP is highest in Norwich (£54 per household per year in 2002 prices), followed by Lowestoft (£35 per household per year, again in 2002 prices) and finally Great Yarmouth (£30 per household per year in 2002 prices). Aggregated over the whole of England and Wales, this results in a total WTP of almost £1 billion, irrespective of the expected reduction in illnesses. Taking into account the expected reduction in illnesses, the aggregated WTP varies between £700 million for a 25 per cent reduction and £1.3 billion for a 100 per cent reduction.

2.2 Value Function

Multivariate statistical analysis (Goldstein, 1995; Langford et al., 1999a) of the payment principle responses and WTP amounts established that there were notable differences in variables explaining the WTP and payment principle responses. Whilst standard neoclassical economic factors (as signified by 'personal characteristics' such as income, education, and so on) were used to try to explain stated WTP and payment principle responses, other factors derived from cultural theory (Douglas, 1982; Douglas and Wildavsky, 1982; Rayner, 1992; Schwarz and Thompson, 1990; Thompson, et al., 1990) and a modified social learning theory approach (Bandura, 1977; Wallston, 1992) were also taken into account. The survey questionnaire thus contained questions according to six separate categories as follows:

- *Views of nature (world views)*. These questions attempted to ascertain respondents' underlying beliefs about the environment, and their worldviews in general. Respondents' views of nature, or 'myths of nature' as proposed by cultural theorists were elicited. This led to the construction of variables describing how respondents viewed the natural world from the point of view of it being adaptable to pressures (ADAPT), controllable by expert management (EXPMAN), fragile and vulnerable to pressures (FRAGILE), and unpredictable in the way it responds to pressures (UNPRED). Respondents were asked to assign values, on a five-point Likert scale, indicating the extent to which they agree or disagree with these views (1= disagree strongly, 3 = neither agree or disagree, 5 = agree strongly).

- *Knowledge and experience*. These questions enquired about respondents perception of their awareness of risks to health from polluted bathing waters (AWARENESS), whether they had heard of the current EC standard (HEARDSTD), and whether they themselves or a member of their family had been ill as a result of swimming in polluted bathing waters (ILLNESS).
- *Self-efficacy*. Respondents were asked if they felt personally capable of making a decision about the new EC standard (CAPABLE), whether the decision should be left to experts (EXPERTS), and whether public consultation should be courted on the issue (PUBCON).
- *Expectations*. Respondents were asked about whether they believed that their participation in the survey would have an important input into the decision-making process (IMPINPUT), if the implementation of a new EC bathing water standard was realistic in practice (REALISTIC) or whether the success or failure of a new EC standard would be largely a matter of chance (CHANCE). Participants were asked if they trusted the government to implement the new EC standard (TRUSTSTD). Respondents were also asked to estimate what decrease in health risks (as a proportion of existing risks) they would expect from a new EC standard (EXPRED).
- *Importance values*. These questions related to the importance to the respondent of the new EC standard, both personally (IMPPERS) as well as to the nation (IMPNAT), and whether the trustworthiness of government in implementing EC directives was an important issue to the individual (TRUSTIMP). Participants were also asked if the proposed EC standard was something that particularly interested the respondent (INTEREST). Finally, respondents were asked to rate on a Likert-type scale (1= not important, 5 = very important) how important it was in terms of their health that the bathing water at beaches in the Anglian Water region should pass the new EC standard (IMPHEA), as well as how important they thought action on a set of coastal environmental problems was (ISSUES).
- *Personal context and characteristics*. Each individual was asked a set of questions about their sex (SEX), age (AGE), income (INCOME), level of education (EDU>16, that is, educated beyond age 16), whether they had young children or not (CH<10), or were members

of various environmental groups (ENVGROUP), and leisure interest groups such as the Surfers against Sewage pressure group (INTGROUP).

A summary of the multilevel modelling results is given in Table 12.3. For the payment principle question, there was a positive association with belief in expert management. Several of the personal characteristic variables were

Table 12.3 Multilevel modelling results for WTP in principle and WTP amounts

Category	Variable	Payment principle	WTP amounts
Views of nature	ADAPT		
	EXPMAN	++	
	FRAGILE		
	UNPRED		−−
Personal characteristics	SEX		
	AGE		
	INCOME		
	CH<10	++	
	EDU>16	+++	++
	INTGROUP	+	
	ENVGROUP	+	
Self-efficacy	CAPABLE	−−	
	EXPERTS		−−
	PUBCON		−−
Expectations	EXPRED	−−−	++++
	IMPINPUT	+++	
	TRUSTSTD		
	REALIST	++++	
	CHANCE		
Values	INTEREST		
	TRUSTIMP		
	IMPPERS	+++	
	IMPNAT		+++
	IMPHEA	++	++
	ISSUES		
Knowledge and experience	ILLNESS		
	AWARENESS		
	HEARDSTD		

Note: +/− = $p < 0.10$, ++/−− = $p < 0.05$, +++/−−− = $p < 0.01$, ++++/−−−− = $p < 0.001$.

also significant, namely, having higher education, having children and being a member of an environmental interest group. Belief in being capable of making a decision was interestingly negatively correlated with a positive response, suggesting that those who refused to pay believed they were incapable of making a decision. Three expectations variables were significant, with belief that the respondent was having an important input into the decision-making process and belief that implementation of a new standard was realistic being positively correlated with a positive response. This is important, as it suggests that saying 'yes' to the payment principle is to a degree dependent on belief in the action being offered and the perceived importance of the contingent valuation study in determining benefits. Those who were willing to pay something had lower expectations of the reduction in risk, suggesting that those who wanted a greater reduction in risk were objecting to the payment principle question. High personal importance value was also associated with saying 'yes', as was importance to personal health, suggesting that more immediate personal concerns were determining the response to the payment question. None of the variables to do with knowledge or previous experience were significant predictors.

Unpredictability of nature was the belief associated with lower WTP amounts (of those who were willing to pay anything at all). This supports other results (Langford et al., 1999b; Marris et al., 1998) that those with a more fatalistic outlook, believing that industry and government act largely out of self-interested motives, are less willing to commit themselves to institution-based improvements. Out of the personal characteristic variables, only higher education was associated with higher WTP (income was not significant). However, higher WTP amounts were positively associated with the size of the expected reduction, suggesting that WTP amounts were more based around what people would like for their money than with income constraints in this case. Willingness-to-pay amounts were also negatively correlated with both public consultation and experts taking decisions. These two explanatory variables were not highly correlated in the model ($r = -0.20$), perhaps surprisingly, but negative associations with both may suggest a preference for the status quo, rather than potentially expensive public consultation or further expert analysis. Importance to personal health was a predictor of higher WTP, as was perceived importance to the nation. Again, none of the knowledge and experience variables were significant.

We can see from the value function analysis that the interpretation of explanatory factors is not straightforward. It appears that, although standard personal characteristics are important, other factors must also be taken into account. Such factors extend the range of variables considered to be important in determining stated preferences in contingent valuation.

3. COSTS OF THE EC BATHING WATER DIRECTIVE REVISION

The benefits of coastal bathing water health risk policy have to be compared with the opportunity (or social) costs of policy in order to yield a measure of the net changes in social welfare. The opportunity costs of policy are the value of the goods and services lost by society resulting from the use of resources to comply with and implement the policy, and from reductions in output. In general, the opportunity costs of coastal bathing water health risk policy consist of five components that must be included in social cost analyses (EPA, 2002). These include:

- Real-resource compliance costs – these are the direct costs associated with: purchasing, installing and operating new pollution control equipment; changing relevant production processes by using different inputs or different mixtures of inputs; capturing the polluting wastes and selling or reusing them.
- Government regulatory costs – these include the monitoring, administrative and enforcement costs associated with regulation.
- Social welfare losses – these are the losses in welfare associated with the rise in the price (or decreases in output) of goods and services that occurs as a result of policy.
- Transitional costs – these include the value of resources that are displaced because of regulation-induced reductions in production and the private real resource costs of reallocating those resources.
- Indirect costs – these other costs include the adverse effects policies may have on product quality, productivity, innovation and changes in markets indirectly affected by the policy.

The challenge in developing an estimate of the social costs of coastal bathing water health risk policy is to consider the markets being affected by the policy, assess the available data and analytical methods, and adopt an analytical approach that will yield an estimate suitable for use in CBA. As was the case in organizing and presenting measures of the benefits of coastal bathing water health risks policy, it is also necessary when considering social costs to take account of the issues of discounting and sensitivity analysis.

Coastal bathing water health risks policy essentially relies on standards-based controls that mandate a level of performance intended to achieve the health objective. These controls are mainly in the form of ambient water quality standards indirectly related to health risks. It should be noted that such standards-based controls are not the only regulatory and non-regulatory policy approaches available. Others include incentive-based controls and voluntary actions taken to reduce risks (Baumol and Oates, 1988; Pearce et al., 1994). Nevertheless, coastal bathing water health risks have predominantly been dealt with solely through the use of standards-based controls. Previous empirical estimates of the compliance costs associated with coastal bathing water health risks policy exist, unlike the benefits estimates for which there were no previous figures available. These two previous estimates of compliance costs are now considered. The first set relates to the cost compliance assessment (CCA) that was commissioned by the UK Department of the Environment and given in evidence to the 1995 House of Lords Select Committee on the European Communities Enquiry, which considered the EC's 1994 proposal to revise the 1976 EC Bathing Water Directive (House of Lords, 1994–95). The CCA was carried out in relation to designated bathing waters and based on the provision of suitable engineering and sewage treatment facilities to meet the limit values of the indicative parameters set by the 1994 proposed revision. The facilities that were necessary were based on judgements made by the consultants responsible. Whilst the resulting estimates were intended to be strategic and were not based on actual feasibility studies, they were produced in consultation with the UK water companies and the National Rivers Authority (precursor to the UK Environment Agency). The CCA required the evaluation of costs associated with four possible scenarios. Scenario A_{1994} is the Commission's 1994 proposal, which introduces a mandatory standard for faecal streptococci and an enterovirus standard. Scenario B_{1994} is the existing Directive made more stringent by making mandatory the standards that are presently the optional *Guideline* standards. Scenario C_{1994} is the Commission's 1994 proposal except for the omission of the more stringent enterovirus requirement. Finally, Scenario D_{1994} is the existing directive plus a new mandatory standard for faecal streptococci.

The second set of cost compliance figures relates to a second cost compliance assessment report commissioned by the UK Department of the Environment, Food and Rural Affairs in response to the EC's 2000 proposal to revise the 1976 EC Bathing Water Directive (Cascade Consulting, 2002). The assessment examines the costs of three scenarios for upgrading bathing water quality. These are all based on increasingly stringent levels of faecal streptococci that correspond to WHO's microbiological assessment categories for bathing waters (WHO, 2001). Scenario C_{2000} is equivalent to the current mandatory EU standards, while Scenario B_{2000} is roughly equivalent to the current *Guideline* EU standard. Finally Scenario A_{2000} is the strictest standard in the WHO's classification categories. Table 12.4 shows the indicative parameters and their respective limit values associated with each of the seven revision scenarios. The range of improvement measures

Table 12.4 Cost compliance assessment – Directive revision scenarios

Proposed Directive revision scenario	Indicative parameter	Limit values
A_{1994}	FS	400cfu/100 ml
	ENT	0 pfu/10 l
B_{1994}	TC	500cfu /100 ml[1]
	FC	100cfu/100 ml[1]
	FS	100cfu/100 ml[3]
C_{1994}	FS	400cfu/100 ml
D_{1994}	TC	10 000cfu/100 ml[2]
	FC	2000cfu/100 ml[2]
	FS	1000cfu/100 ml
A_{2000}	FS	<40 cfu/100ml[4]
B_{2000}	FS	40–200 cfu/100 ml[4]
C_{2000}	FS	201–500 cfu/100 ml[4]

Notes:
1. 80 per cent of samples should not exceed this level.
2. 95 per cent of samples should not exceed this level.
3. 90 per cent of samples should not exceed this level.
4. 95 per cent percentile.
5. FS – faecal streptococci; ENT – enterococci; TC – total coliform; FC – faecal coliform.

required included the installation of ultraviolet (UV) disinfection systems, the upgrading of combined sewer overflows to reduce spill frequency, and the reduction in agricultural point and diffuse pollution.

The cost figures relate to the eight affected water companies (excluding Northern Ireland and Scotland) and it is thought that the impact of the standards on the cost estimates in some water company areas might be twice the national average (House of Lords, 1994–95).

The two sets of cost compliance figures relating to the 1994 and 2000 proposals for revising the EC Bathing water directive are shown in Table 12.5, using net present values, based on 2002 prices, a 25-year time frame and discount rates of 6 per cent and 3.5 per cent.[4] As can be seen, the figures vary considerably depending on the particular scenario considered. As expected, the strictest scenarios under each set of revision proposals (scenario A for the 1994 and 2000 revisions) are the most costly, ranging between £3.1 billion and £7.5 billion at the current prescribed discount rate of 6 per cent. Two of the scenarios (B_{1994} and B_{2000}) from each set of revision proposals, both relate to the same *Guideline* standard of the current

Table 12.5 Net present costs of the EC Bathing Water Directive revision scenarios (aggregated for English and Welsh bathing waters)

Proposed Directive revision scenario	Capital cost £2002 million[1]	Operating cost £2002 million/pa[1]	Total NPC £2002 million (25 years, 6% discount rate)	Total NPC £2002 million (25 years, 3.5% discount rate)
A_{1994}	1971–5096	84–180	3111–7539	3406–8171
B_{1994}	1370–3173	60–120	2184–4802	2395–5223
C_{1994}	529–1322	24–48	855–1974	939–2142
D_{1994}	24–48	0	24–48	24–48
A_{2000}	590	500	7365.18	9119.18
B_{2000}	280	230	3396.58	4203.42
C_{2000}	2.9	0.5	9.68	11.43

Note: Costs adjusted where necessary by UK Treasury gross directive product (GDP) deflators to give 2002 prices.

Directive and hence serve as a cross-check of the credibility of the two cost compliance assessments. It is interesting to note that, although the individual capital and operating cost figures for scenario B appear to diverge somewhat between the 1994 and 2000 figures, the net present cost (NPC) figures are very similar (the figure for B_{2000} is about the mid point of the range given for B_{1994}). The total net present costs for the guideline standard range between £2.2 billion and £4.8 billion.

4. CONCLUSIONS

The cost estimates for the various revision scenarios in Table 12.5 can now be compared with the various benefits estimates for the different estimation scenarios in Table 12.2. It would appear that the benefits of a revised Directive outweigh the costs of even the most stringent of the revision scenarios, irrespective of respondents' expectations regarding reductions in the number of illness from compliance. Given the fact that the benefit estimates may even be conservative underestimates (since they may only cover improvements at a small proportion of the total number of bathing waters in England and Wales with certainty), it seems likely that the benefits will outweigh the costs even allowing for any sources of imprecision in their estimation. It is acknowledged that there may be problems over the representativeness of the samples in the CV study, such that the benefits' estimates are somewhat biased, though on balance it is felt that this is unlikely to make any substantial difference to the finding of positive net economic benefits associated with bathing water pollution control.

The central purpose of this chapter was to conduct an investigation into the economics of revising the bathing water Directive. Estimates of the costs of bringing UK bathing waters up to the existing standards vary, but it is believed that by 2005 somewhere in the region of £9.5 billion will have been spent. Despite these expenditures, a systematic evaluation of benefits has only recently been undertaken, though no nationwide assessment of physical morbidity impacts has been carried out. Further improvements in terms of a tightening of legislation and standards have now been proposed and there is considerable debate as to whether any further expenditures on sewage clean-up represent effective and efficient use of resources. The CBA undertaken here, although based on a number of assumptions with a significant degree of uncertainty attached to them, shows that a further tightening of standards and consequent clean-up of bathing waters is certainly warranted. This finding is qualified by a number of important lessons and insights regarding the fact that policy-makers need to be informed not only of the economic costs and benefits of bathing water clean-up, but also about the reasons why people will or will not pay, and how much they pay. These reasons involve a consideration of the various beliefs, attitudes and values that individuals hold about themselves, society, institutions and the environment, and how these different subjects and objects operate in relation to one another. Attempts at redrafting and successful implementation of a revised EC Bathing Water Directive must take account of their motivations, concerns and expectations in this respect.

NOTES

1. See Kuchler and Golan (1999) for a more detailed discussion of why the WTP approach is generally more appropriate than the COI approach in measuring welfare impacts.
2. Unlike a cost–benefit analysis, which rests its conclusions exclusively on comparison of social benefits and costs, an economic impact analysis examines the distribution of many different economic impacts.
3. It should be noted that these are guidelines for conducting studies that are to be used as evidence in natural resource damage litigation cases in the USA. There is some debate as to their appropriateness for more academic research, for example Fischhoff (1997) argues that the NOAA panel's recommendations to some extent stifle research and exclude examination of interesting areas of work. For this reason, whilst some effort was made to conform to these guidelines whenever possible, it was not an overriding objective to do so.
4. At the time of writing 6 per cent is the rate of discount used by the UK Treasury in its 'Green Book', though it is thought that this is likely to change to 3.5 per cent in the next revision.

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13. Cost–benefit analysis of large-scale groundwater remediation in France

J.-D. Rinaudo and S. Loubier

1. INTRODUCTION

Since the industrial revolution, the development of economic activities has exerted significant pressures on groundwaters through diffuse and point source pollution. Diffuse industrial pollution is mainly related to atmospheric pollution, which contaminates rainwater and soils and, ultimately, groundwater. Groundwater point source pollution, the focus of this chapter, generally results from leakage from tanks, waste dumps such as urban and industrial landfills, mining waste dumps and spoil heaps or accidental spills caused by transport accidents, fire and so on. Contaminants found in groundwater are mainly volatile organic contaminants (VOC), such as dicholroethylene, trichloroethylene and tetrachloroethylene, vinil chloride and benzene. Other contaminants commonly present include heavy metals, polycyclic aromatic hydrocarbons (PAH) and oils.

One of the main characteristics of industrial point source pollution is that they often remained undetected for decades. The actors responsible for the pollution (for instance, leakage of buried chemical storage tanks) were either not aware of the pollution or they did not report the pollution to the competent authorities. Their impact is thus frequently discovered long after the pollution actually took place, typically when the pollution plume reaches a drinking water well, generating an economic damage for a third party. Given the long time that usually passes between the pollution event and its detection, the contaminated area may be very large and the costs of possible remediation measures significant.[1]

In France, the number of reported cases of pollution plumes generated by ancient industrial sites has increased during the last two decades, as the monitoring networks progressively extend. As it is not always possible to identify the exact origin of the pollution or the pollution comes from 'orphan industrial sites', the cost of groundwater remediation projects is usually born by public actors.[2] Given the high costs of such projects and the limited financial resources available, economic considerations should play a key role in prioritizing between polluted sites.

The allocation of scarce public funds should be based on a comparison of the costs and the benefits of groundwater restoration for each site and action should only be undertaken where benefits outweigh the costs of restoration.[3] Also the target level of groundwater restoration can be informed by a cost–benefit analysis (CBA). However, until recently such economic analyses, when carried out, only focused on financial costs, that is, the direct expenditures associated with the implementation of the remediation measures and the direct financial expenditures avoided by such measures (benefits). Other non-monetized costs caused by the often diffuse effects of pollution, usually affecting less organized segments of society, are generally not considered.

The CBA methodology and its practical implementation is now rapidly evolving in France. In a recent paper, which focuses on the restoration of polluted industrial sites, Guelton (2002) shows that CBA is increasingly used to choose the level of restoration effort, taking into account the indirect benefits. In the water sector, this development can partly be attributed to the implementation of the Water Framework Directive (WFD), which reinforces the role of economic analysis in the decision-making process (Kallis and Butler, 2001). The WFD requires that member states (MS) implement the necessary measures to achieve or restore the good ecological and chemical status of all water bodies by 2015. It allows MS to define derogation for water bodies for which the good status cannot be achieved by 2015. Derogation either consists of extending the deadline (six years, renewable only once) or of the definition of less stringent objectives. The derogation can be justified either by proving that the restoration is technically not feasible or that it would entail costs that can be considered as disproportionate compared to the benefit they generate.[4] Thus, CBA becomes more and more part of the toolbox of river basin planners.

This chapter focuses on the use of CBA carried out in an area extensively polluted and for which policy-makers and stakeholders are debating the necessity to justify derogation. The chapter's main objective is to illustrate the main challenges when carrying out CBA of large-scale groundwater remediation projects in a real policy context. The focus will mainly be on the assessment of the associated benefits though, as these are usually the most difficult to estimate in practice.

The remainder of this chapter is organized as follows. The next section presents the case study, including the groundwater pollution problem and the restoration measures which have been implemented since the discovery of the pollution. Section 3 briefly discusses the set-up of the CBA and the underlying assumptions. The fourth section focuses on the benefits of groundwater restoration, which are assessed in monetary terms in section 5. The results are then discussed in a concluding section, highlighting the uncertainty surrounding the results. Finally, we stress that the discussion initiated by the CBA is probably the most important outcome of the study. Cost–benefit analysis appears to be a powerful support tool for discussion, the effectiveness of which is likely to be improved if implemented in a more interactive and participatory context.

2. CASE STUDY DESCRIPTION

2.1 General Description

The case study presented in this chapter involves a highly polluted area of the upper Rhine valley alluvial aquifer. This aquifer is a transboundary water body, which extends over 4200 square kilometres in Germany and France (see Figure 13.1). With a reserve of approximately 45 billions cubic

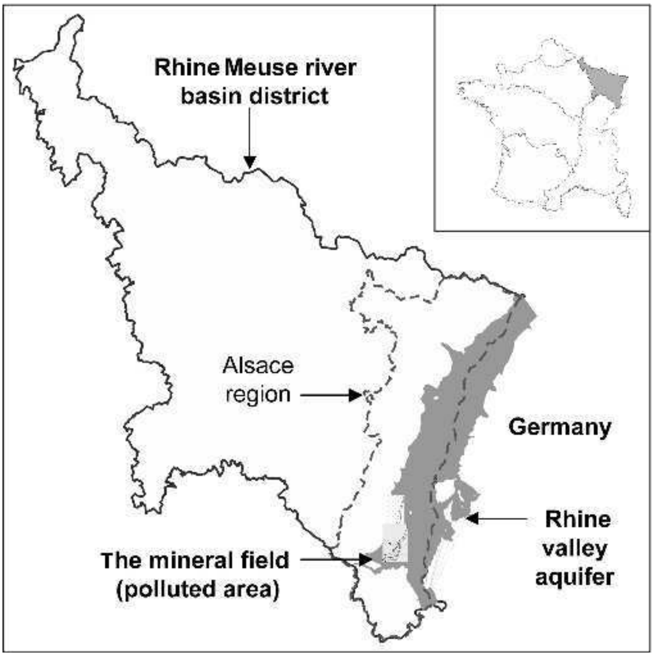


Figure 13.1 Location of the case study

metres of water, equivalent to approximately half of the volume of the Lake Geneva, this aquifer is one of the largest fresh water reserves in Europe. Groundwater from the Rhine alluvial valley is used to meet 75 per cent of the drinking water needs and about half of the industrial water needs. More than 3 million inhabitants of the Alsace Region (France) and the Land of Baden-Württemberg (Germany) directly depend on its quality.

Although usable for drinking purposes without prior treatment in most locations, groundwater has progressively been affected by diffuse and point source pollution. Nitrate concentrations exceeding 50 mg/l have been reported at a large number of monitoring points. Pesticide pollution is also reported. High concentrations in volatile organic contaminants (VOC) have been detected downstream of certain industrial areas, and a large area is affected by chloride pollution, originating from the potash mining industry, on the French and German side of the aquifer. In this case study, we will focus on this mining pollution in the French Alsace region, close to the industrial city of Mulhouse.

2.2 Pollution Levels

Potash ore extraction in the region of Mulhouse in Alsace started in the early years of 1910. The ore extracted from the mines was processed close to the mining site in order to separate the potash from its impurities and salt (sodium chloride) content. Residuals from the ore processing plants were piled up in huge waste dumps. Seventeen of such waste dumps, characterized by high salt contents, were established in the potash mining fields after 1910. Two of them are still active (the mine is supposed to close in 2004). Overall, it is estimated that approximately 18.5 million tons of salt have been deposited in the waste dumps.

For decades the waste dumps have been leached by rainfall. Rainwater percolates through the waste dumps and its salt concentrations increases, frequently reaching 200 g/l of chlorides. It then infiltrates the aquifer, which has progressively been contaminated by this (point) source of pollution. As a result, two huge pollution plumes have progressively extended over time following the flow lines in the aquifer. Nowadays, chloride concentration in groundwater can reach a maximum of 50 g/l in certain areas close to the waste dumps. Huge quantities of salt have already been dissolved by rainfall, but, unless additional measures are implemented, the remaining quantities will continue to leach from the waste dumps for the next 180 years (Chabart and Elsass, 2001).

A water quality monitoring network, established in the late 1970s and progressively extended since then, has been used to map the pollution plumes.[5] The latest maps produced in 2000 show that the plumes extend over approximately 40 kilometres. Chloride concentration exceeds 100 mg/l across approximately 187 square kilometres and 200 mg/l across more than 80 square kilometres. Because of the relatively higher density of saline water, deep layers are more affected than surface layers (Chabart and Elsass, 2001).

2.3 Protection and Remediation Measures

The public authorities became fully aware of the extent of the pollution problem when drinking water wells located several kilometres downstream of the mining sites were contaminated by the salt, in the mid-1970s. Three types of measures aiming at preventing any further degradation of the aquifer were subsequently implemented and progressively intensified. These measures are briefly discussed below.

- Between 1976 and 1985, a number of water wells were drilled directly downstream of the mining deposits in order to intercept the salt after it infiltrates in the aquifer. The salt and water pumped from these wells, most of which are still functioning today, are transported to the river Rhine through a pipeline.[6]
- After 1989, a number of waste dumps characterized by a low salt content (less than 33 per cent) have progressively been covered by a geo-membrane and/or a vegetal cover in order to make them watertight and reduce infiltration. Other waste dumps (containing more than 33 per cent salt) have been artificially dissolved through intensive leaching with high pressure water guns (accelerated dissolution). In these cases, the highly concentrated water is recovered through a system of ditches and drains and transported to the river Rhine with the help of a pipeline. As a result, the pollution sources are increasingly controlled and the inflow of salt into the aquifer is reduced. However, in view of the fact that this pollution-dismantling programme will not be completed before 2010, salt continues to flow into the aquifer.
- More recently, a number of deep wells have been installed across the pollution plumes (several kilometres downstream of the sources) in order to prevent an extension of the polluted area and to remove salt from the aquifer. The total installed pumping capacity is approximately 3500 cubic metres per hour. Water extracted is discharged into the river Rhine using the above mentioned pipeline. The Mining Company estimates that between 10 to 15 per cent of the salt content in the aquifer is removed every year with the help of these pollution-removal wells.

Using figures provided by the Mining Company and public agencies, we estimated the total financial costs of the measures implemented between 1976 and 2001 at more than €67 million, consisting of €27 million investment costs and close to €40 million of operation and maintenance costs. Approved additional measures, which will be implemented between 2002 and 2010, will cost another €44 million (€30 million investment costs and €14 million operation and maintenance costs).

3. SET-UP OF THE CBA AND COST ESTIMATION

Although the financial costs are significant, the decision to protect groundwater in the potash mining fields of Alsace has not been based on – and not even informed by – an economic analysis. The level of investment decided upon in the 1970s was largely determined by the budget constraints of the actors responsible for the funding of the restoration programme.[7] A careful analysis of the long-term costs and benefits of groundwater restoration was not carried out. The long-term financial costs, including investment, operation and maintenance costs were not even estimated at that time.

Groundwater restoration in the potash mining fields was established as a key water management objective in 1995, through a local decree defining drinking water quality standards as a target for the entire aquifer. This objective was stated again in the Water Management Master Plan (SDAGE) of the Rhine Meuse Basin District, which was drawn up by representatives of water users, elected politicians and government agencies in 1997. However, again economic analysis was not used to support the decision.

The demand for an economic assessment of the groundwater restoration measures only emerged in 2002, as a result of the WFD requirements. Public authorities realized that, despite significant efforts being made to restore water quality, 'good status' could probably not be achieved by 2015. A derogation would therefore have to be justified using the 'disproportionate cost argument'. To justify derogation, it was considered necessary to demonstrate that the costs of the additional measures would largely exceed the benefits that could be derived from an accelerated clean-up scenario. For this purpose, a pre-feasibility CBA was carried out.

In the CBA a reference scenario, which assumes that the measures already implemented in the year 2002 will be prolonged as long as chloride concentrations do not fall below 250 mg/l in the entire polluted area, is compared with an accelerated clean-up scenario, which will restore the target concentration of chloride in the entire area by 2015. The associated costs are those linked to the construction and operation of the additional clean-up equipment. The benefits equal the welfare increase for various users likely to benefit from an accelerated restoration of groundwater quality.

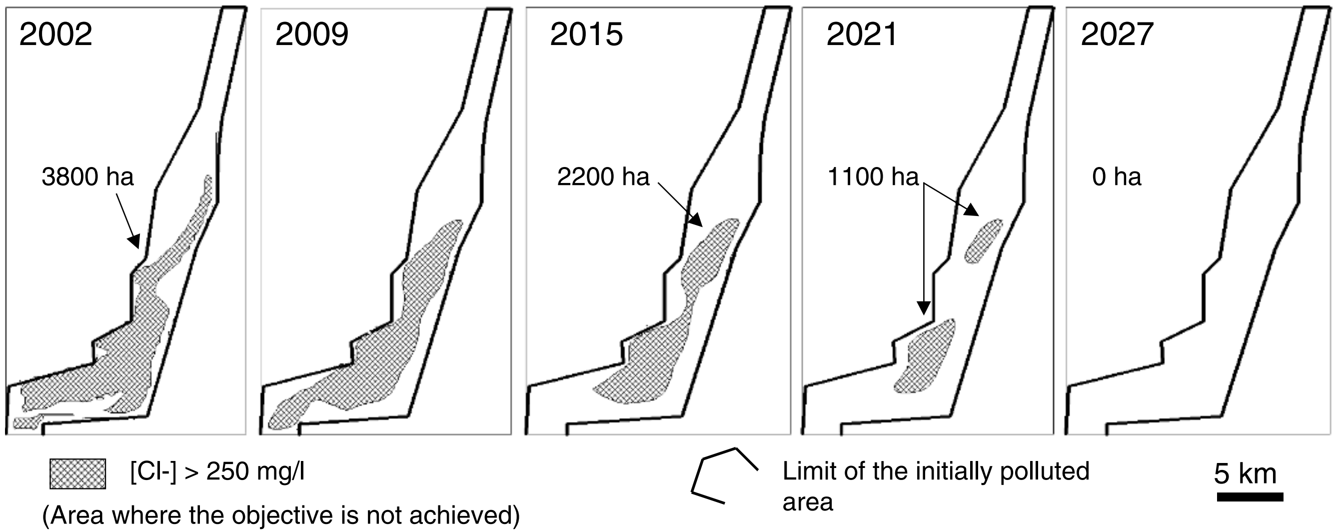
The reference scenario corresponds to the so-called 'baseline scenario' in the WFD. It assumes that the pumping wells will stay in operation as long as the target concentration is not achieved and the continuation of the programme of encapsulation of the pollution source is acceptable to all the financial partners. A simple hydrodynamic model is developed to assess the effectiveness of this reference scenario.[8] The model is developed for the eastern pollution plume only and it is run to simulate the development of the chloride concentration of this plume. This development is assumed to be similar for the western plume. The simulation results (Figure 13.2) show that with this reference scenario, the chloride concentration will fall below 250 mg/l in the entire polluted area by 2027. In 2027, approximately 96 per cent of the salt present in the aquifer in 2002 will have been removed.[9]

The accelerated clean-up scenario assumes that additional pumping wells will be installed in 2002. Using the same simulation model, the number, capacity and location of pumping wells are calculated so that the target concentration of chloride will be achieved in the entire aquifer by 2015. The estimated additional pumping capacity needed is 2400 cubic metres per hour.

The investment required (construction of additional wells, new pipeline to transport the brine to the Rhine and so on) as well as the additional operation and maintenance costs (energy, labour, spare parts) to achieve this objective are estimated using the technical information provided by the mining company. Concerning the Eastern plume, the total additional investment cost is roughly estimated at €15 million and the additional operation and maintenance costs at €1.5 million at a 3 per cent discount rate.[10] Extrapolating these results to the Western plume, the total costs of the accelerated clean-up scenario are estimated at €33 million.

4. A TYPOLOGY OF THE BENEFITS OF GROUNDWATER QUALITY RESTORATION

In general, the benefits of (accelerated) clean-up scenarios for groundwater consist of the economic damage costs avoided when groundwater quality is restored (more rapidly). This approach hence considers the costs, which are borne, now and in the future, by society due to groundwater quality deterioration. These costs will be (partly) avoided if groundwater quality is restored.



Source: Rinaudo et al. (2002).

Figure 13.2 Development of the chloride concentration in the reference scenario

In this study, one of the main challenges was to assess the damage costs that could be avoided in the case of an accelerated clean-up. As shown in Figure 13.3, the benefits of groundwater quality restoration increase much more rapidly in the case of the accelerated clean-up scenario (bold line) than in the case of the reference scenario (dotted line). The additional benefits generated by the accelerated restoration scenario are represented by the triangle OAB.

In principle, four different types of avoided damage costs (or clean-up benefits) can be distinguished (according to who pays for them).

The first type of avoided costs relates to the costs born by economic agents using water as input in their production process (agriculture, industry, drinking water suppliers). The deterioration of groundwater quality reduces the profitability of economic activities using water as an input, either because it negatively affects the quality (and therefore the price) of the final product or because it leads to an increase of production costs, for example, because of the need to use alternative inputs or the use of different and more expensive technologies. In agriculture, for instance, the presence of pollutants in water used for irrigation may prevent the cultivation of certain crops or have an impact on the quality of the crops. In industry, groundwater pollution may increase the production costs if, for instance, water used in the production process has to be treated before it can be used or if the pollution is responsible for the deterioration of the equipment (for instance, corrosion). Groundwater pollution may also force actors from the

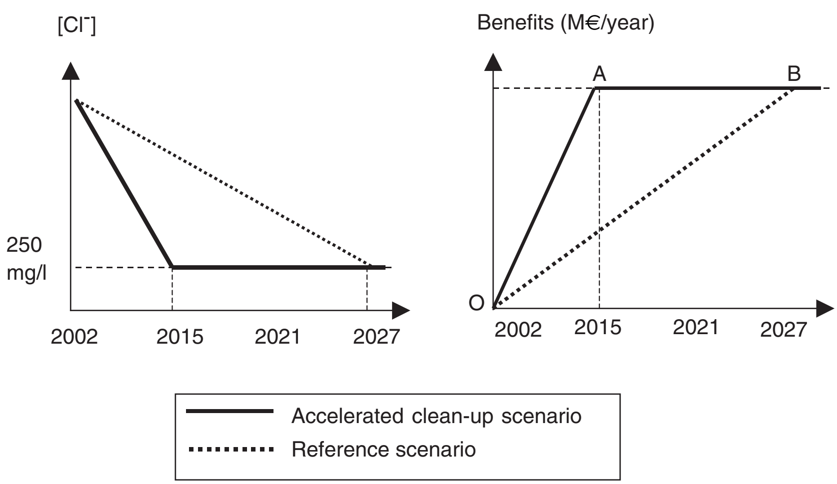


Figure 13.3 Development of chloride concentration and related benefits in time

drinking water sector to invest in the construction of water treatment units (aeration towers, activated carbon filters, reverse osmosis and so on) to drill new wells or to hook-up to a neighbouring water utility (construction of pipeline) from which water is purchased.

The second type of avoided costs are those born by agents using water as a final consumption good (usually households). With over 63 per cent of the French population receiving drinking water that comes from aquifers (Cour des Comptes, 2002), its deterioration may generate fear, anxiety and a loss of trust in the safety of tap water. This may consequently result in an increase of the consumption of bottled water or the installation of purification devices. Although this fear is not justified in most cases, it nevertheless generates additional expenditures for households, which can be considered as a pollution damage cost.[11]

The third type of avoided costs includes all costs born by indirect groundwater users. The deterioration of groundwater quality may have severe negative impacts on ecosystems, such as wetlands, forests and rivers. In certain cases, this may result in a deterioration of the functionality of these ecosystems, which may in turn generate costs for the actors using these ecosystems as a support for production or other activities (for example, recreational fishing, fish farms, tourism based activities and so on) or benefiting from the natural goods and services provided by these ecosystems (for example, natural purification, flood control, landscape amenity and so on). This is the case, for instance, for rivers which are dependent on groundwater recharge in the summer or which are severely affected by eutrophication when groundwater is polluted by nitrates and phosphates (EEA, 2003).

The fourth type of avoided costs relates to the reduction of the so-called non-use value of groundwater when it is polluted. This non-use value includes so-called bequest and existence values that households may attribute to environmental assets (such as groundwater stocks) irrespective of their current or future use.[12] This cost (reduction in non-use value if groundwater is or remains polluted) can only be measured through contingent valuation studies designed to assess public or stakeholder willingness to pay (WTP) for groundwater protection or remediation irrespective of the use of groundwater (see, for instance, Rozan et al., 1997).

5. ASSESSMENT OF THE BENEFITS OF (ACCELERATED) GROUNDWATER REMEDIATION

The benefits of groundwater clean-up were identified through consultation of key experts and stakeholders involved in water management and water use in the area. We selected experts within government administrations (Regional Environment Protection Agency, Forest and Agriculture Department, Department of Health and Sanitation Affairs), local governments (District Councils, Regional Council), environmental protection non-governmental organizations (NGOs) and two research centres. Municipalities concerned by the pollution, professional organizations representing agriculture and industry and the Mining Company were also consulted based on semi-structured interviews. The respondents were first asked to describe the type and extent of the damage that had been generated by groundwater pollution in the past. They were then asked to identify those damages, which they believed are reversible. Finally, the accelerated clean-up scenario was presented and respondents were asked to describe the expected benefits associated with this scenario. The information obtained was then cross-checked with written documents. In particular, archives from the Rhine Meuse River District Agency and Conseil Général du Haut Rhin were investigated in order to assess and double check some of the financial figures quoted. The results are presented in the next sub-sections.

5.1 Benefits Accruing to Consumers through Costs Variations for the Utilities

Drinking water utilities and their customers were consistently across respondents mentioned as an important sector or water agent, which has suffered a significant cost as a result of pollution from the past. Although several drinking water wells located in the pollution plumes were abandoned between 1910 and 1975 and no written evidence about the damage to these wells was available, we did manage to find written evidence about two specific cases where water wells were contaminated after 1990.

The first concerns the public drinking water utility Syndicat des Eaux de Ensisheim-Bollwiller et Environs (EBE). The six wells operated by this utility, supplying water to seven municipalities and 17 800 inhabitants, were affected by the pollution from the late 1980s. The chloride concentration of the water pumped from these wells ranges between 350 and 800 mg/l (with significant fluctuations over time). In response to the pollution, the utility constructed a pipeline in 1990 to import water from the neighbouring municipality of Guebwiller. The imported water is mixed with the polluted water in order to meet the EU Drinking Water standards (the average dilution rate being 50 per cent). The water purchased by EBE from Guebwiller is much more costly than the exploitation of the original polluted resource, since it is produced from surface water (and hence treatment is needed) and has to be transported over a long distance. The price of water delivered by EBE to its consumers has therefore been raised by €0.35/m^3. Knowing that EBE utility sells approximately 630 000 m^3 per year, the total additional costs for the consumers equal €12.4 per inhabitant per year.

According to the simulated development of the chloride concentration, the accelerated clean-up scenario would allow EBE utility to stop purchasing water from Guebwiller in 2012 instead of 2024 as in the reference scenario. Assuming that the change in water demand is proportional to the population increase (0.5 per cent per year), the implementation of the accelerated clean-up scenario would generate a cost saving of €1.8 million for the customers of EBE utility (at a 3 per cent discount rate).

The second case is the public drinking water utility supplying drinking water to the 90 700 inhabitants of the city of Colmar and surrounding municipalities. Two of the four wells of this utility (Neuland wells) have been affected by pollution since the end of the 1980s, with chloride concentration fluctuating between 200 and 270 mg/l. Since the wells are located in the tail of the pollution plume, the situation is expected to worsen in time if no additional clean-up measures are taken. In response to the pollution, the Colmar utility has modified its distribution network in order to mix water from these polluted wells with the water abstracted from two other wells (Dornig wells). In addition, polyphosphates are added to the water before distribution in order to form a protection layer inside the pipes and limit corrosion provoked by the presence of chloride.

However, this solution is not thought to be sustainable as the two other wells, also located in the plume trajectory, are expected to be contaminated soon too. The utility is therefore planning to create new wells in a nearby forest area or to construct a treatment plant to remove chloride from groundwater pumped from the wells (reverse osmosis technology). For the first option, the investment needed to construct new wells and connect them to the network is estimated at €10 million. The operation costs will remain unchanged. Regarding the second option, the investment cost is estimated at €4.6 million and the operation and maintenance costs of the treatment plant will cost an additional €4.35 per inhabitant.

The choice for or against one of these options is highly dependent on the groundwater management scenario. In the reference scenario, chloride concentrations are likely to exceed the drinking water standards until 2020, making the treatment option slightly more expensive than the new wells option (respectively €10.7 and €10.0 million – at a 3 per cent discount rate). In the accelerated clean-up scenario, the existing wells could be used again without any treatment in 2008. The option to treat water for six years will therefore be cheaper than the construction of a new well (respectively €7.2 and €10.0 million). The net benefit of the accelerated clean-up scenario is

Table 13.1 Alternative options and their costs for the Colmar drinking water utility and their customers under two water management scenarios (in € million, 2002 prices)

	Reference scenario	Accelerated clean-up scenario
Year in which groundwater quality will be restored	2020	2008
Option 1: new wells	Preferred option	Rejected
Investment costs	10.0	10.0
Operation costs	0.0	0.0
Total costs	**10.0**	10.0
Option 2: treatment plant	Rejected	Preferred option
Investment costs	4.6	4.6
Operation costs	6.1	2.6
Total costs	10.7	**7.2**

equal to the difference between these two options, that is, €2.8 million (Table 13.1).

5.2 Benefits Accruing to Private Households

Although water distributed by the two drinking water utilities (EBE and Colmar) complies with the European Drinking Water Standards, the chloride pollution reinforces the general decline of consumer confidence in tap water quality and safety. The number of households who are as a result purchasing bottled water is expected to increase. In other words, the chloride pollution generates additional expenditures for households, which can be considered an indirect cost of the pollution. This assumption is confirmed by the results of a survey conducted ten years ago in the region by Stenger and Willinger (1998).[13] These authors found that 88.8 per cent of the households were purchasing bottled water; 67 per cent declared never to drink tap water; 42 per cent could identify sources of pollution affecting groundwater quality; and approximately 78 per cent perceived pollutants as a serious health risk.

In order to be able to quantify these damage costs, we make the following assumptions:

1. The chloride pollution is only perceived as a problem by the customers of the two drinking water utilities EBE and the one in Colmar (that is, the inhabitants supplied by Colmar and EBE utilities in 2002).

2. Approximately 80 per cent of this population drinks bottled water. The average daily consumption of bottled water is half a bottle (0.75 litres) per person at a cost of €0.25 per person.
3. For one-third of this population, the main motivation to purchase bottled water is a lack of trust in the quality of tap water. Five to 10 per cent of this lack of trust can be attributed to the chloride pollution, while the remaining 90 to 95 per cent is linked to other pollutants such as nitrate and pesticides, fear of terrorism and so on.
4. The households who purchase bottled water because of the chloride pollution would stop doing so if the chloride pollution problem was solved.
5. The accelerated clean-up scenario will solve the pollution problem in 2008 instead of 2020 in the case of Colmar and in 2012 instead of 2024 in the case of the EBE utility. During these periods, the population growth rate is 0.5 per cent per year.

Using a 3 per cent discount rate, the benefit of the accelerated programme ranges between €1.2 and €2.3 million.

5.3 Benefits Accruing to Agriculture

Groundwater is intensively used for crop irrigation by the farming sector. According to the experts and stakeholders interviewed, the high chloride concentrations found in the case study area generate two major types of costs for the agricultural sector. The high concentrations accelerate the corrosion of irrigation equipment and reduce the quality and/or the yield of the crops irrigated with polluted water.[14]

5.3.1 Corrosion damage cost

Depending on the chloride concentration, the lifetime of irrigation equipment (tube-well and pumps, irrigation pipes, centre pivots) can be reduced by a factor 2 to 10, especially when made of steel. The cost of corrosion can be assessed by comparing the theoretical life time of various equipments given by the constructor (noted d) with the actual life time reported by professional farming organizations (noted d'). For a given investment noted I, the annual financial cost (AFC) is:

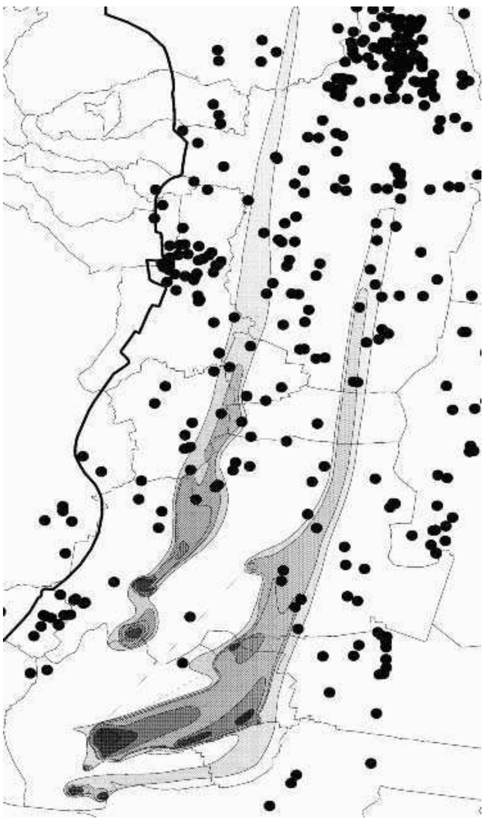
$$\mathrm{AFC}_d = \frac{\alpha I(1+\alpha)^d}{(1+\alpha)^d - 1} \tag{13.1}$$

where α is the discount rate (Loubier et al., 2001).

The annual financial loss (AFL) due to rapid corrosion equals:

$$\mathrm{AFL} = \mathrm{AFC}_{d'} - \mathrm{AFC}_d = \frac{\alpha I[(1+\alpha)^{d-d'} - 1]}{[(1+\alpha)^d - 1][(1+\alpha)^{d'} - 1]} \qquad (13.2)$$

For instance, for a centre pivot covering 30 hectares, the investment sum is €55 000, the average lifetime 20 years and the actual lifetime seven years. Applying the above reasoning, the cost of corrosion equals €5131 per centre pivot per year. Repeating the same approach with all the tube-wells located within the pollution plume (Figure 13.4), the corrosion costs of wells and pumps is estimated at €680 per well per year.[15]



Source: BRGM and Région Alsace.

Figure 13.4 Location of agricultural wells used for irrigation (dark dots) and the extent of chloride pollution

To assess the total costs of corrosion associated with the two water management scenarios, we assume that: (1) the demand for irrigation water will not change during the coming two decades; (2) the groundwater polluted area will linearly decrease in time at a rate depending on the scenario (−7.7 per cent per year for the accelerated clean-up scenario and −4 per cent for the reference scenario); and (3) the irrigation wells and pivots are homogeneously distributed in space. The total corrosion costs for the reference scenario are €622 000 against €373 000 for the accelerated clean-up scenario. The damage avoided by accelerating the clean-up is therefore equal to the difference between these two values, that is, €249 000 (at a 3 per cent discount rate).

5.3.2 Impact on crop quality and yield

High value-added crops such as tobacco and vegetables cannot be grown in areas where groundwater is characterized by a high chloride concentration. Irrigation of tobacco crops with polluted water leads to the formation of coloured marks on the leaves, which reduces their market value. It also reduces the combustibility of the leaves. As a result, farmers who are interested in cultivating tobacco (a labour demanding crop), are forced to produce crops with a lower value-added such as cereals and maize. This causes a loss of income equal to the difference between the gross margins of maize and tobacco (respectively €1100/ha and €8000/ha) multiplied by the area where tobacco could be grown. It appeared, however, after consultation of various experts, that the size of this estimated area is only 20 hectares, taking into account soil constraints, the availability of production rights[16] and the number of farms with a high labour potential (tobacco cultivation requires a lot of labour and cannot be grown by all farmers). The total annual income loss is thus equal to €136 000.

To assess the loss of income associated with the two water management scenarios, we assume that (1) the relative profitability of tobacco compared to other crops will not change over time, and (2) the area affected by chloride concentration will decrease linearly in time. The total loss of income is estimated at €1.4 million in the case of the reference scenario and €862 000 in the case of the accelerated clean-up scenario (at a 3 per cent discount rate). The difference (€576 000) represents the net benefit of the accelerated clean-up scenario.

5.4 Benefits Accruing to Industry

Some of the experts interviewed reported that the extension of the pollution plume has forced many industries to abandon their own water well, mostly because of the corrosion problems caused by high chloride concentrations.

The corresponding costs consist of the costs of getting connected to the drinking water infrastructure (fixed costs) and the additional (variable) costs of volumetric water units used (tap water generally being much more expensive than water pumped from a private well). However, due to the historical dimension of the pollution, the experts interviewed could not identify the companies involved, some of which may even have moved or disappeared in the course of time.

The interviewees also reported that many industries have installed small treatment units to purify water and remove chloride. However, treatment units have sometimes also been installed in areas where the chloride concentration is below 200 mg/l (in particular in the region of Mulhouse and Colmar). Also, they have not always been installed solely in response to the presence of chloride: in many cases, the treatment is part of technological constraints linked to the industrial process itself. Here, too, the experts were not able to distinguish between industries, which have been forced to install treatment plants in response to the presence of chloride and those, which have not. In both cases, the damages caused by the chloride pollution are irreversible and the measures taken represent sunk costs. The only benefit of an accelerated clean-up scenario would be that the water purification could be stopped in certain companies. Given the lack of any data, this benefit could not be assessed further and quantified in monetary terms.

5.5 Environmental Non-use Benefits

Scientists, government and environmental protection organizations agreed that the pollution of the aquifer in the potash mining field has not caused any serious damage to ecological resources such as fish stocks and wildlife habitats. This can be explained by the fact that the polluted river beds are significantly higher than the water table. As a consequence, rivers tend to recharge the aquifer and there is no flow of water (and pollution) from groundwater to surface waters. Locals report that forest areas located very close to the waste dumps have been affected by chloride in view of the fact that the roots of trees are able to get into contact with high chloride concentration groundwaters. However, this involves only a few hectares of forest and the economic loss is considered negligible. We therefore assume that groundwater quality restoration will not yield any significant (direct or indirect) ecosystem benefits.

In the potash mining fields, as in many other mining regions in Europe, the public did not pay any specific attention to the pollution of groundwater until the late 1960s. In particular, it seems that chloride pollution and land subsidence, the other major negative impact of mining, were long considered a necessary undesirable side effect and assumed inevitable by the people whose jobs were directly dependent on this activity.[17] This perception changed with the decline of the mining activity and its significance in the local economy. Several of the experts and stakeholders consulted stressed that the aquifer is nowadays considered by the local population as an important component of the regional natural heritage. This changed perception towards groundwater management issues can partly be ascribed to information and awareness campaigns organized by environmental protection NGOs, such as Alsace Nature, the Association for Environment and Nature in Alsace (ARIENA) and the Association for the Protection of the Alsatian Aquifer (APRONA), with the active support of the regional authorities.

The results of a household survey conducted by Stenger and Willinger (1998) confirm the statement above. They found that over 90 per cent of the households interviewed were able to identify some of the major risks of groundwater pollution and 43 per cent could mention sources of pollution. The study also demonstrated, using contingent valuation techniques that inhabitants attach a significant value to the aquifer and were willing to pay in 1993, on average, €93 per household per year on top of their water bill to prevent any further degradation of the resource. Using the same methodological approach, Rozan et al. (1997) interviewed another sample of households, located in the same region, whose drinking water supply did not originate from groundwater. They showed that these non-user households also have a positive WTP to protect the aquifer of €52 per household per year in 1995. This WTP is considered by Rozan et al. as a proxy of the existence value of the aquifer.[18] The results from these two surveys are used here to assess the economic non-use value of the aquifer.

In order to calculate household WTP in 2002 prices, the 1995 values are adjusted using the general consumer price index.[19] Moreover, we assume that the socio-economic value of environmental goods is an increasing function of the level of wealth of a society (Horowitz, 1996). This means that instead of reducing the discount rate to take into account future environmental costs, we prefer to take into account the increase in relative prices. In other words, we suppose that the non-use groundwater value is rising at a rate β equal to the long-term average economic growth, whereas the value of other goods is assumed constant over time.

Considering a 1 per cent growth rate and a 1.1 consumer price index between 1995 and 2002, the 2002 household WTP to preserve the aquifer ($WTP_{NU;2002}$) is estimated at €61 per household per year.[20]An accelerated clean-up of chloride pollution would increase the non-use value of the aquifer (consisting of bequest and existence value). To estimate this benefit in monetary terms, we consider that only the population located in the cities and villages located close to the polluted area is aware of and concerned with this problem. This population is assumed to attach a non-use value to the Alsatian aquifer in its present condition of zero.[21] The welfare of this local population would be increased by €61 per household per year in 2002 if the aquifer was not polluted, which leads to a total non-use benefit of the accelerated clean-up scenario equal to:

$$B_{nu} = \sum_{t=2015}^{t=2027} \frac{WTP_{NU;2002} {}^{*} N}{1+\alpha} \left[\frac{(1+\beta)+(1+\delta)}{1+\alpha} \right]^{t-2002} \qquad (13.3)$$

where B_{nu} is the non-use benefit of the accelerated clean-up scenario
t is the year (from 2015 to 2027)
$WTP_{NU;2002}$ is taken as a proxy of the non-use value of the aquifer in 2002
N is the number of households affected by the problem in 2002
α is the discount rate
β is the annual rate of increase of the relative value of environmental goods
δ is the annual population growth rate.

Assuming that between 5 and 10 per cent of the Alsace population is affected by the chloride groundwater pollution ($30\,000 < N < 60\,000$ households), a 3 per cent discount rate, a 1 per cent long-term growth rate, a 0.5 per cent annual population growth rate and an economic non-use value of €61 per household per year in 2002, the total non-use benefit of the accelerated clean-up scenario ranges between €17.6 and €35.2 million.

6. DISCUSSION AND CAVEATS

In this chapter, we tried to illustrate the problems and challenges when using CBA in the context of groundwater remediation projects. As is generally the case for water-related CBA applications, the estimation of the socio-economic benefits of groundwater quality restoration is the most difficult and challenging part of the CBA. This is why this chapter focused especially on the benefit side and to a lesser extent on the cost side. The benefits of an accelerated clean-up scenario for one of the largest freshwater aquifers in Europe were assessed and where possible quantified using limited available information and data. The pre-feasibility CBA's main purpose was to initiate and support the discussion and decision-making process about possible derogation in the context of the recently adopted and implemented European WFD.

The net present value of the total sum of benefits ranges between €24 and €43 million, depending on the assumptions made, especially with regard to the calculation of possible non-use benefits and household averting

Table 13.2 Net benefit estimation of the accelerated clean-up scenario compared to the reference scenario (in € million, 2002 prices) assuming low and high estimates for some of the benefit estimations

		Low estimate	High estimate
Benefits	Households (utilities consumers)		
	EBE utility	1.8	1.8
	Colmar utility	2.8	2.8
	Households averting behaviour	1.2	2.3
	Agriculture		
	Corrosion	0.3	0.3
	Crop losses	0.6	0.6
	Non-use benefits	17.6	35.2
	Industry	NA	NA
	Total	24.3	43.0
Costs	Investment	30.2	30.2
	Operation and maintenance	3.1	3.1
	Total	33.3	33.3
Net Benefits		−9.0	+9.7

Note: NA: not available.

behaviour (see Table 13.2). Taking into account the investment, operation and maintenance costs of the accelerated clean-up scenario, the net benefits vary between −€9 and +€10 million.

The CBA results suggest that the government authorities can either justify a derogation or not in their River Basin District Water Management Plan, which they have to draw up according to the provisions specified in article 4 of the WFD. The CBA highlights the high dependency of the results on household WTP for groundwater protection. As shown in Table 13.2, the estimated non-use benefits represent approximately 70 to 80 per cent of the total benefits, depending on the assumptions made.[22] This dependency is all the more serious because of the fact that the WTP amount was estimated with the help of a contingent valuation study conducted 10 years ago and the assumptions made to adjust this amount and aggregate it across the relevant population of beneficiaries are very simple.

Another important assumption regards the discount rate. Since most of the benefits are spread over a long period of time, due to the inertia of groundwater systems, the CBA results are highly dependent on the choice of the discount rate and the development of relative prices. This is especially true in this case study where the costs are incurred in the initial stages

Table 13.3 Estimates of the range of net present values of the net benefits of the accelerated clean-up scenario (in € million, 2002 prices) using various discount and growth rates

		Discount rate		
		$\alpha = 3\%$	$\alpha = 5\%$	$\alpha = 8\%$
Growth rate	$\delta = 0\%$	−13 / +2	−20 / −9	−23 / −17
	$\delta = 1\%$	−9 / +10	−17 / −4	−21 / −14
	$\delta = 2\%$	−4 / +20	−14 / +3	−19 / −10

(the construction of the clean-up infrastructure), while the flow of benefits are distributed over a long time horizon – a characteristic of nature restoration projects.

This dependency and impact of assumptions on the outcome of the CBA is illustrated in Table 13.3. In Table 13.3, the estimation of the net present value of the net benefits is shown using three different discount rates and three different growth rates for the future development of relative prices for environmental goods and services. Table 13.3 shows that the CBA outcomes can either lead to the implementation of the accelerated clean-up programme of measures or to a demand for derogation if high discount rates are used and a low future relative price of environmental goods and services (and hence non-use value).

The choice of the discount rate is often an important determinant of the economic efficiency of water related projects. Moreover, recommendations from different stakeholders involved in the decision-making process to use a specific discount rate may sometimes differ significantly. Dubgaard et al. (2002, p. 33) illustrate this point with three examples. In Denmark, the Environmental Research Institute and agencies under the Ministry for the Environment recommend a discount rate of 3 per cent in social CBA, while the Ministry of Finance recommends a discount rate within the range 6–7 per cent. A similar discrepancy is reported in the USA where the Environment Protection Agency recommends to use a discount rate of 2–3 per cent, whereas the American Office of Management and Budget recommends a standard discount rate of 7 per cent. Finally, Norway employs a discount rate varying between 3.5 and 8 per cent, depending on the risks concerning the returns from the project.

A final caveat relates to the uncertainty surrounding the simulated effectiveness of the reference and accelerated clean-up scenario. The hydrogeological model, which was developed for this case study, is a very simple tool, which relies on specific assumptions related to the area's geology and hydrology (for example, porosity, permeability). Close monitoring of chloride pollution could reveal unexpected developments of the pollution plumes in the future. This uncertainty is, however, characteristic to groundwater related problems and groundwater remediation projects in general. The uncertainty will increase in those cases where the pollutant–soil interaction is unknown or the transfer time of the pollutant from the top soil to the groundwater aquifer is difficult to assess.

In those cases where it is not possible to numerically estimate the effectiveness of alternative scenarios, the analyst is forced to rely on expert judgement and advice, which may increase the uncertainty attached to the CBA results. It is important to emphasize that the quality of the CBA is highly dependent on the quality of the knowledge and information about the natural processes involved and that CBA applied to groundwater management issues requires a multidisciplinary approach.

Despite the caveats listed above, CBA is a very useful and valuable tool when preparing and informing groundwater policy and decision-making. It facilitates a common understanding and representation of the relevant issues at stake. Our experience is that the process of conducting a CBA, involving stakeholders and experts at different stages of the study, is a much more important element in the whole decision-making process than the actual outcomes of the CBA. When carrying out a CBA or any other analysis, inevitably a number of simplifying assumptions and choices have to be made related to norms, values, target and reference scenarios, the allocation of rights and so on. Discussing these choices and assumptions with key stakeholders may be even more fruitful – in terms of debate and discussion – than the result of the analysis itself. For example, the use of the results of the contingent valuation carried out by Rozan et al. (1997) were put up for discussion with different experts and stakeholders. Some actors felt that asking citizens how much they would be willing to pay to prevent further degradation of groundwater a priori assumed that the polluters own certain 'pollution rights' of the natural resource, which was considered contrary to the polluter pays principle.

A major achievement of the analysis was hence to make explicit the different benefits and costs, relying on pooled and shared knowledge of lay stakeholders and professional experts.[23] This interactive and participatory approach ensures that significant external effects are not forgotten or neglected or certain impacts are not overestimated.

Finally, CBA helps identifying the nature of the transfers between segments of society associated with the different options under consideration. By doing so, it informs the decision-makers and stakeholders involved of the welfare distribution effects and the social acceptability of alternative policy options.

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NOTES

1. Remediation is mostly carried out through 'pump and treat' schemes: water is pumped out of a polluted area, treated (using activated carbon filters, oxydation processes, membranes, biological treatment) and then either discharged in surface water or returned to the aquifer through injection wells. This entails significant energy and maintenance costs as the scheme is generally functioning for long periods of time. Other *in situ* clean-up techniques such as bio-remediation with bacteria or artificial aeration of groundwater (venting) can be implemented when the volume of water to be treated is not too large (see Lallemand-Barres, 1995).
2. The fact that groundwater clean-up is frequently undertaken by public actors is not a specific feature of the French institutional context. In the USA for instance, a specific fund, the *Superfund*, was created in 1986 to pay for site clean-up when parties who caused the contamination could not be found or could not pay for the clean-up themselves (see Spofford et al., 1989).
3. Because actions are undertaken by public institutions subject to budgetary constraints, the choice between a set of restoration projects should be made progressively from low to high cost–benefit ratios until the budget for groundwater protection is exhausted.
4. Although the text of the WFD does not give a clear definition of disproportionate cost, the guidance document produced by the WATECO European Working group and endorsed by the European Water Directors, specifies that the decision to consider costs as disproportionate should be based on a cost–benefit analysis (WATECO, 2002).
5. This network now counts 463 monitoring points used to measure the evolution of the salt concentration at different depths of the aquifer.
6. The discharge of salt in the river Rhine has no noticeable ecological impact. The discharge is in line with the maximum discharge allowed by the Bonn Convention.
7. All actors involved in the funding of the restoration measures belong to the public sector: the Potash Mining Company of Alsace (MDPA), the Rhine Meuse Water Agency, the Alsace Regional Council and the state (through its regional administrations).
8. For a detailed description of this model (development, calibration and simulation) see Rinaudo et al., 2001.
9. It is however possible that pockets of intense pollution, trapped in the lower section of the aquifer, remain after 2027. The model is too simple to be able to account for such localized phenomena. It is assumed here that the presence of such pockets of pollution can be neglected.
10. The operation and maintenance costs depend on the discount rate. Using a discount rate of 5 per cent, the operation and maintenance costs amount to €2.3 million, while the total costs are €2.9 million at a discount rate of 8 per cent.
11. For case studies assessing the extent of the damage costs for households, see, for example, Abdalla (1994), Spofford et al. (1989) or Traoré et al. (1999).
12. See chapter 9 in Pearce and Turner (1990) for a detailed presentation of the concept of total economic value.

13. Eight hundred and seventeen residential households were interviewed in this study in the spring of 1993.
14. Some farmers have started proceedings against the Mining Company, asking for financial compensation for the deterioration of their irrigation equipment. In most cases, the dispute was settled through a negotiated arrangement involving financial compensation.
15. Assuming that: (1) the lifetime of tube wells is reduced from 40 to 20 years and of pumps from 15 to 7.5 years; (2) the cost of a 20 metres deep tube-well and a pump are respectively €20 000 and €3000 and (3) approximately 50 wells are affected by the pollution in 2002.
16. The tobacco industry distributes production rights (or quotas) to farmers. This represents a significant constraint limiting the area where tobacco can be cultivated.
17. Similar attitudes were reported in Spain (Loredo et al., 2001).
18. See Rozan et al. (1997, p. 15).
19. National Institute for Statistics and Economic Studies at www.insee.fr.
20. $WTP_{NU;\,2002} = WTP_{NU;\,1995}\frac{I_{2002}}{I_{1995}}(1+\beta)^{2002-1995} = 52 \times \frac{223.1}{203.5} \times 1,01^{7} = 61$
21. A negative value would also be possible, that is, if the public wishes to be compensated for the current deteriorated state of the groundwater aquifer.
22. It is worth mentioning here that, although we used a different framework for assessing use values, the ratio between use and non-use value found in this study is close to the 60 per cent found by Rozan et al. (1997) in their contingent valuation study.
23. See Rinaudo and Garin (2004) for a discussion of the benefits of combining lay and expert input in water management planning.

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14. Cost–benefit analysis and efficient water allocation in Cyprus

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1. INTRODUCTION

The scarcity of water resources in both arid and temperate countries alike is one of the most pervasive natural resource allocation problems facing water users and policy-makers. In arid countries this problem is faced each day in the myriad of conflicts that surround its use. Water scarcity is a fact with which all countries have to become increasingly involved.

Water scarcity occurs across many dimensions. First, there is growing demand for water in residential, industrial and agricultural sectors stemming largely from population and economic growth. Second, supply-side augmentation options have become increasingly constrained and restrictively costly in many countries. In combination, demand growth and supply-side interventions have stretched current water availability to its hydrological limits. In addition to these quantity constraints, the limits to the assimilative capacity of water resources for human and industrial waste have been reached in many places, and the quality of freshwater has been degraded (Winpenny, 1994).

In turn, water scarcity has become an important constraint on economic development, which has resulted in fierce competition for scarce water resources between economic sectors that rely upon it (Winpenny, 1994; World Bank/EIB, 1990). Water scarcity is important for sustainability in economic development as well, on account of the many associated environmental/watershed services. In the face of hydrological constraints, the focus of current thinking in water resource management is on the allocation of scarce water between competing demands (Dublin Conference, 1992; UKWIR, 1999; Winpenny, 1994).

How is it possible to allocate water between its many competing uses, all of which depend on water for their existence? Clearly, water resources are necessities for many of the most important goals of every society. First, water is a necessity for human existence. The absence of clean drinking water and sanitation leads to health problems, whilst the lack of access to/property rights for water resources per se is a significant dimension of poverty. Water is also an important input to economic activities and can be seen as both a production and consumption good (Young, 1996). Furthermore, water is a public good contributing to recreation, amenity and general environmental and watershed values as an input to ecosystems and habitats. How can it be possible to balance such crucially important, but competing uses?

The fact is that a balancing of these uses must be accomplished, and the mechanism for doing so must be carefully constructed. The existing overlay of complex hydrological, socio-economic and property rights/legal environments (in many if not most jurisdictions) predisposes water resources to open access appropriation within the watershed and the consequence of negative environmental and economic externalities (for example, the degradation of wetlands and coastal fisheries, depletion of aquifers and loss of watershed services). In short, the combination of the arbitrariness of the prevailing property rights structure for water resources in most jurisdictions and the failure of markets to capture the value of many watershed services necessarily imply that the prevailing distribution of water within most societies is not likely to be the most desirable one.

In what follows, a 'watershed economics approach' is proposed which is composed of two important stages. In the first stage, economic valuation techniques are used to establish the economic value of the competing demands for surface and groundwater, incorporating where necessary an analysis of water quality. The valuation exercise allows the balancing of demands based upon the equi-marginal principle to achieve economic efficiency. In the second stage, a policy impact analysis is proposed, which addresses issues of social equity and the value of water for environmental/ecological purposes. The analysis is undertaken within the confines of the watershed, the most natural unit for the analysis of water allocation and scarcity since it determines the hydrological links between competing users and thus the impacts of one user upon another. The methodology is encapsulated by a case study of the Kouris watershed in Cyprus.

The methodology described here and the case study which accompanies it provide an example of how CBA of water policy can be structured in the context of the watershed. They show how the multifarious constraints faced by watershed managers and policy-makers can be usefully evaluated and traded off using the principles of CBA and the important considerations of environment linkages and equity.

2. METHODOLOGY

In this section we outline the methodology we propose for application to the underlying problem of watershed management. This methodology is based on (1) the identification of the appropriate unit for management; (2) the agreement of the objectives of water allocation; (3) the evaluation of the various attributes of water demand within that unit; (4) the identification of optimal water resource allocations relative to objectives; and (5) the assessment of the impacts of the proposed reallocation.

2.1 The Appropriate Management Unit

The watershed is a natural unit of analysis for addressing the balance of supply and demand for water, and the issues of efficiency, equity and sustainability, for a number of reasons. First, the aggregate availability of water resources, including sustainable yields is bounded by the hydrological cycle of the watershed. Second, the interaction between different water sources (for example, groundwater and surface water) is confined by the watershed. Third, the demands for water interact within the watershed and the hydrological impacts of one water user upon another and upon environment, that is, externalities, are defined by the watershed. For these reasons, an understanding of the hydrological cycle in the watershed area in question is a prerequisite for the determination of efficient, equitable and sustainable water resource allocation.

2.2 The Objectives of Water Allocation

Given the natural water resource constraints, there is a clear need to address the pattern and growth of water demands in order to address the imbalance. The methodology proposed provides the policy-maker and planner with a transparent approach to balancing the competing demands for water subject to the natural constraints. The approach is based on the comparison of the economic value of water in different sectors, in terms of quantity and quality, in comparable units of measurement. The overall objective of public policy is to maximize societal welfare from a given natural resource base subject to those valuations. The key objectives of public policy in the allocation of resources are as follows:

- *Efficiency*. Economic efficiency is defined as an organization of production and consumption such that all unambiguous possibilities for increasing economic well-being have been exhausted (Young, 1996).

For water, this is achieved where the marginal social benefits of water use are equated to the marginal social cost of supply, or for a given source, where the marginal social benefits of water use are equated across users.

- *Equity*. Social welfare is likely to depend upon the fairness of distribution of resources and impacts across society, as well as economic efficiency. Equal access to water resources, the distribution of property rights, and the distribution of the costs and benefits of policy interventions are examples of equity considerations for water policy.
- *Environment and sustainability*. The sustainable use of water resources has become another important aspect in determining the desirable allocation of water from the perspective of society. Consideration of intergenerational equity and the critical nature of ecological services provided by water resources provide two rationales for considering sustainability. In addition, the *in situ* value and public good nature of water resources should enter into water allocation decisions.

2.3 The Evaluation of Water Demand

For physical, social and economic reasons, water is a classic non-marketed resource. Even as a direct consumption good, market prices for water are seldom available or, when observable, often are subject to biases such as subsidies, taxes and so on. Similarly, environmental and ecological water values are rarely explicitly marketed and priced. Thus the economic value of water resources is seldom observed directly. The balancing of demands to resolve the resource conflicts described above requires the identification and comparison of the benefits and costs of water resource development and allocation among alternative and competing uses. In addition, water management policies have widespread effects on the quantity and quality of water within a watershed and the timing and location of supplies for both in- and off-stream uses. In general, these impacts have an economic dimension, either positive or negative, which must be taken into account in policy formulation. Again, the value of these impacts is seldom observed directly.

Fortunately, economists have refined a number of techniques to value water resources and address the balance of demands and evaluate the impacts of water management policy. The first step towards the evaluation of economic benefits requires the identification of the demands for the resource. Water is needed for all economic and social activities, so the evaluator is faced with the problem of identifying a multi-sectoral demand curve. The dimensions of demand include municipal and industrial, agricultural, tourism and environmental (recreation, amenity and ecological).

The valuation of each of the identified demands usually calls for a different approach for the following main reasons:

- the specific economic and hydrological context;
- data availability; and
- because the use of the resource is sector specific.

The residential and tourist sectors exploit the use value of water and use it as a consumption good. The agricultural sector derives use value from water as an input in production. The value of water related environmental goods can be a use value or a non-use value, including so-called existence value.

The valuation techniques allow the estimation of the following desirable parameters:

- *Marginal value of water*. The efficient balance of demands from a given source is found where the marginal value (benefit) of water is equated across users. In any given context, efficiency is achieved where the marginal value of water is equated to marginal social cost.
- *Price elasticity of demand (PED)*. This measures the responsiveness of demand to price changes. It characterizes the demand function and tells the policy-maker the extent to which prices must change to cause demand to fall to a particular (for example, efficient or sustainable) level.
- *Income elasticity of demand (IED)*. This measures the extent to which the demand for water varies with income. It tells the policy-maker whether water is a necessity or a luxury good and provides one way in which to assess the fairness of pricing policies. In combination with PED, IED can be used to estimate welfare changes resulting from policies.
- *Marginal or average willingness to pay for public goods (WTP)*. This estimates the strength of demand for water as an environmental good. This determines in part the efficient environmental allocation of water.
- *Marginal willingness to pay for quality changes of common access resources*. This parameter estimates the value of quality attributes of the resource, which are particularly important if the resource is used as a productive input.
- *Risk parameters*. Measurement of preferences towards risk and uncertainty. Useful for establishing policies, which reduce the impacts of risk on consumer groups occasioned by reason of variability in water availability.

2.4 Balancing Water Demands in the Watershed

The outputs of the demand analysis allow the determination of the economically efficient allocations of water resources. The first element of an economically efficient allocation is the equi-marginal principle. This prescribes that each use of the water resource should achieve the same benefit from that water at the margin. In short, if water is more heavily valued at the margin in one sector than another, then it should be reallocated towards that sector until equality is achieved. The second element of the economically efficient allocation is that aggregate water resources are allocated efficiently where the marginal social benefit of their use is equated to the marginal social cost of supply.

One option for achieving an economically efficient water allocation is the use of the instrument of water pricing, where water is uniformly and universally charged at the marginal social cost of supply. This has the following implications. First, competing demands will each make use of the supply until its marginal benefit is equated with marginal social costs of supply (the equi-marginal principle). Note that this implies that every use must receive an equal marginal benefit from water resources in the optimum. The second implication is that aggregate demand for water will expand until the marginal benefit is equated with the marginal social cost of supply (aggregate efficiency). Demand is hence endogenous and managed within this model. The third implication is that the key to the success of the policy is the determination of the appropriate marginal social cost of supply and the marginal benefits to environmental uses.

2.5 Deriving Policies from the Methodologies – Policy Impact Analysis

There is a second phase to the water allocation methodology that follows from the consideration of the implementation of the conclusions from the first. The discussion here has largely been phrased in terms of the use of water pricing as the appropriate allocation mechanism, but this need not necessarily be the best or most appropriate instrument for allocating water in every context. There are many different approaches to enable the efficient allocation of water resources – pricing, marketable permits, even auctions (Dinar, 1996; Easter et al., 1999; Winpenny, 1994). Ultimately, the particular context must be considered for the feasibility of the various instruments, and the policy-maker must determine the most appropriate allocation mechanism within that context.

Second, it is crucial to note that an economically efficient allocation need not necessarily be an equitable or sustainable one. Additional analysis is required to assess the distributional impacts of the allocation recommended by the equi-marginal principle. The hydrological impacts of the allocation need to be assessed in order to assess whether the various demands are compatible within the existing watershed. Finally, the continued provision of basic environmental services within the watershed needs to be considered. In summary, the watershed needs to be double-checked for unforeseen externalities and for missing markets for watershed services to ensure that intra- and inter-temporal efficiency is achieved and that equity and sustainability considerations are properly considered.

The methodology can therefore be thought of as two complementary stages, the first stage to ascertain economically efficient water allocations and the second stage consisting of a policy impact analysis. The overall evaluation strategy, applied to the case study of the Kouris watershed in Cyprus, is shown in Figure 14.1.

3. CASE STUDY

The following study illustrates how the economic watershed appraisal methodology described above has been implemented in Cyprus. The Kouris watershed is used as an example of a watershed with multiple water use conflicts. The valuation process for the sector demands in Cyprus and the policy implications are described.

3.1 Water Supply in Cyprus

Cyprus is an arid island state situated in the north-eastern Mediterranean (Figure 14.2) in which renewable freshwater resources are highly constrained. The hydrological cycle of Cyprus is characterized by spatial and temporal scarcity in water quality and quantity. Eighty per cent of the rainfall is lost through evapo-transpiration, the remaining 20 per cent can be considered as the available annual water resources in Cyprus.

A number of different water supply investments and interventions have been made in Cyprus. In addition to surface water dams and groundwater exploitation, these included recycling, desalination and even evaporation suppression, cloud seeding and importation of water. The most significant investments have been those contributing to the Southern Conveyor Project (SCP). This scheme forms an interconnected water supply system, which allows the transfer of water resources throughout the southern part of the island and also to and from the capital, Nicosia. The scheme was designed to supply water to irrigated agriculture and residential areas, alleviating the spatial and temporal scarcity of water supplied in the country. The SCP effectively links all groundwater and surface water sources from the

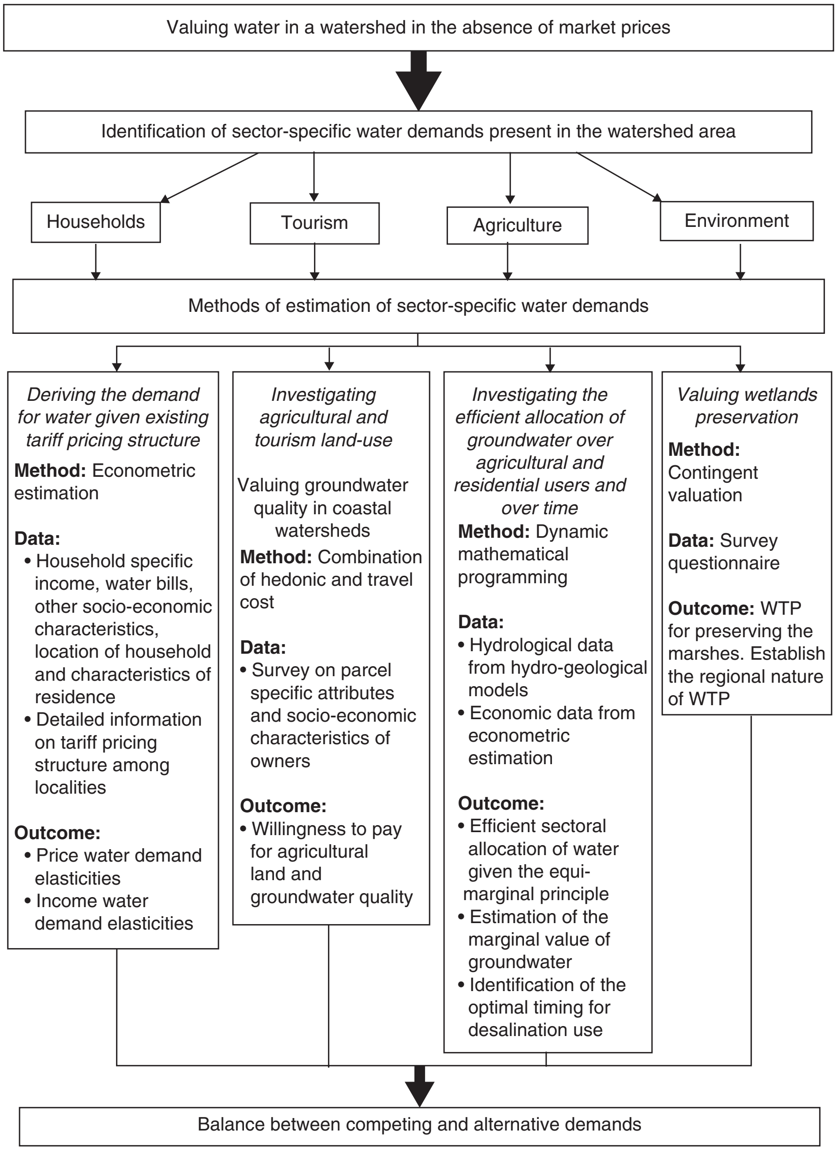
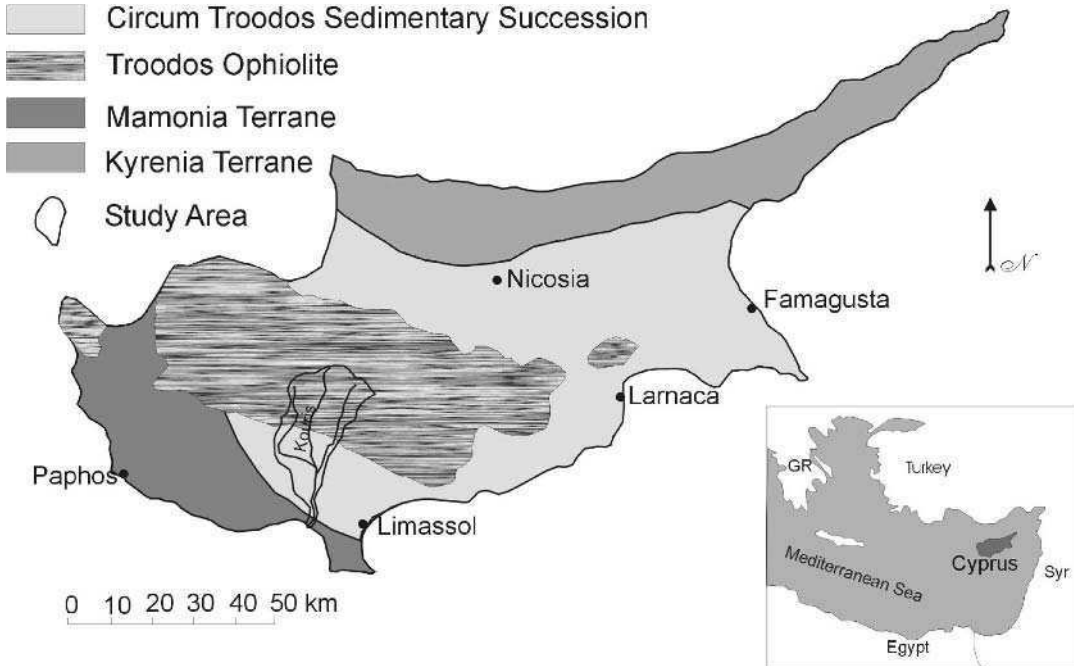


Figure 14.1 Overall evaluation strategy applied to the Kouris watershed in Cyprus



Source: Boronina et al. (2001).

Figure 14.2 The Kouris catchment in Cyprus

Diarizos river (near Paphos) in the west to Paralimni (south of Famagusta) in the East. As a result, the management of the individual catchments in Cyprus has become of national importance and consequence (World Bank, 1996).

Currently all aquifers are exploited beyond their safe yield, with the excess of use over natural recharge estimated to be 40 million m^3/annum. The possibilities for additional exploitation of surface water have been largely exhausted and this has necessitated the consideration and/or use of costly unconventional sources such as desalination, recycling and evaporation suppression.

3.2 Water Demand in Cyprus

The sectoral demand for water is shown in Table 14.1 for the three major water schemes in Cyprus. It can be seen that approximately 75 per cent of current water use is in irrigated agriculture. The majority of the remaining demand involves urban areas, tourism and industrial demands.

There is a distinct seasonality to the demands for water from these water-consuming sectors. Urban demands are clearly higher in the tourist season, whilst the demands for agriculture also vary according to the growing season. Economic growth has averaged 6 per cent over the past 15 years, driven largely by the annual growth in the tourist sector (up to 10 per cent per year). There has also been economic growth in the industrial sector. Under current government plans, the irrigation sector will be expanded in the coming years, having grown at a rate of 2.2 per cent over the period 1980–92. Coupled with an expected population growth of 0.9 per cent per year and rapid urbanization, these will place further pressure on water resources in the years to come.

Table 14.1 Water consumption in the major water schemes in Cyprus (in millions m^3 in 1994)

Water scheme	Urban, industry and tourism	Irrigation	Total
Southern Conveyor System	42.7	45.9	88.6
Paphos System	4.2	23.2	27.5
Khrysokhou System	0.4	6.3	6.7
Other	8.1	84.5	92.6
Total	55.4	160.0	215.4

Source: Adapted from World Bank (1996).

Price is a significant determinant of water consumption. The consumption of water resources by irrigated agriculture is subsidized up to 70 per cent of the average unit production costs (World Bank, 1996). Current pricing strategies in urban areas differ significantly between municipalities, but generally involve substantial cost recovery.

3.3 Water Balance and Property Rights

Given the spatial and temporal variability of water resources and demands, the water balance varies from one watershed or water scheme to the next, and from one year to the other. The scarcity of water resources in Cyprus is thus characterized by extreme fluctuations over time and space of water supply and demand. Of the water schemes shown in Table 14.1, the SCP has the least favourable water balance (World Bank, 1996). The SCP caters for 40 per cent of the aggregate demand, 80 per cent of all urban demand and 25 per cent of all agricultural demand. It is the deficit of surface water flow, which causes the main shortfall. However, given the yearly fluctuations in precipitation and the resultant surface flow, the scarcity and the severity of the deficit varies from year to year.

The negative water balance is reflective of the interaction of supply and demand, and the underlying distribution of the right to control resources. Agriculture is clearly the largest water consumer. Table 14.1 shows that the major water schemes all have significant irrigation components, and indeed the primary motivation for the development of some of these projects was to maintain water supply for expansion of irrigation. The deficits in the water balance illustrate a conflict in resource management stemming from an absence of co-ordinated control of water use and the balancing of those demands with supply in a manner consistent with the underlying hydrology. The rights to water stem from government control. Non-governmental schemes consist of many scattered, small individual and communal schemes, like those using groundwater from the Kiti aquifer and the upper reaches of the Kouris catchment. The rights to groundwater resources are largely common property or open access here, despite the provisions of the 1946 Well law.

In addition to these water users, direct diversions from surface water flows, mainly in the Troodos mountains, including the Kouris catchment, for use by individuals and communal irrigators, account for 150 million m^3 per year of total resource availability. Surface water is also subject to open access, and farmers have the rights to construct irrigation schemes and use surface water (World Bank, 1996). Urban water resources are largely supplied by public schemes such as the SCP, but also by localized commissions from groundwater and surface water schemes.

In summary, although the government has the responsibility for monitoring and protecting water resources, this responsibility is divided between many institutions, resulting in a fragmented regulatory framework (Grimeaud, 2001). A brief overview of the institutional and legislative background to water policy in Cyprus is given in the appendix to this chapter.

3.4 The Kouris Watershed

The current water balance in the SCP and the overdraft of groundwater resources are indicative conflicts between resource use and the natural constraints of water supply that have arisen under the current water management environment. The current extent of resource use is clearly unsustainable and there is nothing to guarantee that the benefits or social welfare derived from water resources are maximized or well distributed under the current pattern of water demand.

The conflict can be illustrated with the help of the Kouris watershed. The Kouris watershed covers 300 km^2 in the south-west of Cyprus (see Figure 14.2). The watershed contains storage dams with a total capacity of 180 million m^3 and provides much of the surface water for the SCP. The largest single storage dam is the Kouris Dam, with a capacity of 115 million m^3. The water users within the watershed are many and disparate and their property rights to water vary. In the upper reaches of the watershed, agricultural users extract groundwater and divert surface water for irrigation purposes under a common property arrangement. Downstream, water is diverted to storage dams for distribution to the main urban centres and to other irrigation schemes via the SCP. In the lower reaches of the watershed, surface water feeds into the coastal wetland areas, which provide a habitat for indigenous wildlife and migratory bird species.

It is widely believed that the uncontrolled growth of private and communal water use in the upper reaches of the Kouris watershed has contributed to reduced surface flows for the SCP (World Bank, 1996). Given the inter-basin transfers that the SCP allows, this watershed issue is of national consequence. Furthermore, the storage dams of the SCP have reduced the freshwater resources reaching the coast and feeding wetlands. There is concern that this has caused damage to the habitats important to migratory species. The management of water resources and conflicts within the watershed is not co-ordinated and the balance between these dimensions of demand within the Kouris watershed has not been met. There is a need for a new approach to water management in Cyprus, which takes into consideration the pertinent contextual factors: (1) imbalance of growing demand and exhausted/costly supply; (2) growing environmental costs and issues of sustainability; (3) watershed-level water management and river basin districts; and (4) fragmented legal and institutional framework.

In short, the unregulated interplay of water-using agents acting in their own interests has led to conflicting demands within the watershed. The management of water resources has not taken a watershed approach, has been uncoordinated, and the balance between demands within the Kouris watershed has not been met. As a result the water balance for the SCP is in deficit and, given the expected sectoral growth, is likely to worsen in the coming years, whilst environmental impacts go largely unchecked. The development of conventional water sources has proved insufficient for securing water resources in the face of extreme climatic conditions and the options for supply augmentation are nearly exhausted and only available at high cost. An integrated approach is needed.

4. EVALUATION OF WATER DEMAND IN CYPRUS

4.1 Residential Household Water Demand

An analysis of residential water demand from the SCP was undertaken. Water demand was calculated from expenditure data and knowledge of the tariff structure in each of the localities. As in most European countries and in the USA, Cyprus water utilities choose among three types of pricing schemes (uniform, decreasing and increasing block rates) in their attempt to use the price of water as a management tool to influence its use. The government-controlled part of Cyprus is divided into 37 water authorities, each having its own tariff structure. The adoption of an increasing block tariff structure and differences in the application of this pricing policy across water authorities give rise to substantial water price heterogeneity on the island.

Economists have attempted to shed some light on the consequences of the choice of the pricing structure by paying attention to demand estimation. However, opinions concerning the appropriate methodology for estimating water demand models differ. Estimation under a block pricing structure requires appropriate modelling to account for the choice of both within and between block consumption. Earlier studies of water demand ignore the peculiar features of the presence of block rates and perform empirical estimation using *ex post* calculated average prices. More recently, investigators combine marginal price and the so-called Nordin's difference variable (in the case of multiple tariffs, this variable is the difference between the total bill and what the users would have paid if all units were charged at the marginal price) in empirical models of residential demand.

We estimated a model consistent with fundamental principles of the economic theory of consumer behaviour (such as adding-up, price homogeneity and symmetry). The choice of the Quadratic Almost Ideal Demand System (QUAIDS) model reflects the fact that it belongs to the family of rank-3 demand systems, the most general empirical representation of consumer preferences that satisfies integrability. We use a rank-3 demand system for two reasons. First, we estimate demand for water using individual household data. Lower rank demand systems are unable to capture the non-linear income effects pertaining to these data. Second, we need a demand system that satisfies integrability (that is, the ability to recover the parameters of the indirect utility function from empirical demand analysis), because we plan to analyse the welfare implications of alternative water pricing policies on empirical grounds. We consider the ability to evaluate the welfare implications of alternative water-pricing policies particularly important, given the significance attached to equity and the strong political objections to water price reform in Cyprus based on political-economic arguments.

The theoretical model described above is applied to individual household data from the 1996/97 Family Expenditure Survey (FES) of Cyprus. This allows estimation of the price and income elasticities of residential demand for water in Cyprus, the marginal value of water in the residential sector and evaluates the welfare effects associated with changes in the water-pricing system. Empirical results show that the current water-pricing system is progressive, but inefficient in the sense that it introduces gross price distortions resulting in deadweight loss. The regional difference, in particular, introduces a substantial price heterogeneity that cannot be justified on the basis of efficiency or equity criteria. It cannot be justified on efficiency grounds, because it is difficult to imagine that on a small island like Cyprus such large regional differences in price can reflect differences in supply costs. The regional price heterogeneity can also not be justified on equity grounds, because we found that large water consumers pay a lower average price per cubic metre than users consuming smaller amounts of water.

The empirical analysis suggests that the marginal value of water in the residential sector is £CY0.45/m^3. The price elasticity of water demand ranges between -0.4 for households in the lowest and -0.8 for households in the highest 10 per cent income distribution (see Table 14.2). This means that the demand curve for water is downward sloping and highly responsive to price changes for high-income water users. This suggests a strong role for price as a demand management tool. Budget elasticities for water, which reflect the responsiveness of the proportion of income spent on water to income changes, and hence income elasticity of demand (IED), are also shown in Table 14.2. The fact that the budget elasticities are always less

Table 14.2 Estimated price and budget elasticities of household water demand

Elasticity	Income group percentiles					
	Bottom 10%	11–25%	26–50%	51–75%	76–90%	Top 10%
Budget	0.25	0.22	0.23	0.30	0.35	0.48
Price	−0.79	−0.69	−0.60	−0.56	−0.50	−0.39

Source: Hadjispirou et al. (2002).

than 1 implies that water is, as expected, seen as a necessity. However, the value increases with income, suggesting that an increase in income for high-income households leads to a greater increase in the proportion of income spent on water. This can be explained by the fact that higher-income groups use more water for water-intensive luxury goods such as swimming pools and gardens with lawns.

The analysis showed that the existing and regionally defined heterogeneous increasing block pricing system introduces gross price distortions that cannot be justified on the basis of efficiency considerations. In the case of residential water use, price can play a role in a demand management scheme designed to tackle the growing fresh water problems in Cyprus. Such an approach, however, should take into account the distributional impact of alternative price regimes. Any major water price reform is bound to have effects on the welfare of individual consumers, In other words, there will be winners and losers, and therefore there will also be a need to consider how to deal with potential hardship caused by the water price reform.

4.2 Agricultural Water Demand

An agricultural production function for groundwater users was estimated econometrically from which the marginal productivities of the inputs as well as the effects of each of the inputs on risk could be derived. Risk considerations are necessary in the understanding of the agricultural sector's use of water. Public policy should consider not only the marginal contribution of the various inputs to the output, but also the marginal reduction in the variance of the output.

In the estimated production function, fertilizers, manure and pesticides (FMP) inputs, as well as water, had a significant and positive effect on expected profit. These FMP inputs and water exhibit decreasing marginal returns. Water and FMP and labour and FMP appear to be complementary inputs. Water and FMP are risk increasing inputs, but at a decreasing

Table 14.3 Estimated risk premiums and marginal input productivity

Parameter	Water			Fertilizer			Labour		
Average risk premium (% of expected profit)	18			19			17		
Impact on variance of profit (other inputs constant)	Positive and decreasing			Positive and decreasing			Negative and decreasing		
	Citrus	Veg	Cereal	Citrus	Veg	Cereal	Citrus	Veg	Cereal
Marginal productivity by crop (CY£)	0.59	0.21	0.14	0.72	0.55	—	0.17	−0.32	0.25

Source: Groom et al. (2002).

rate. On the contrary, labour appears to decrease the variance of profit at an increasing rate (see Table 14.3).

Crop-specific production functions are found to be statistically different and have better explanatory power than a general agricultural production function in the Kiti region. This indicates that crop specific policies will be more efficient than policies, which do not differentiate among crops. In addition, for all crops fertilizers and pesticides exhibit higher marginal contributions than either water or labour.

Farmers exhibit moderate risk aversion and are willing to pay approximately one-fifth of their expected profit to achieve a situation in which the profit received with certainty leaves them as well off as the uncertain expected profit. No heterogeneity in risk attitudes is observed across the farming population, so policies introduced to reallocate risk do not need to differentiate between specific types of farmers. This is reasonable given the fact that the agricultural region under consideration is small and there exists almost no variation in the accessibility of economic resources, services and information.

4.3 Environmental Water Demand

As the standards of living increase in Cyprus, so water demand for recreational purposes also increases. Furthermore, water may have a use value, but also a non-use or existence value. People who are willing to pay for water and wildlife preservation can be local residents who live near a wetland, for example, but also people who care about its preservation and live far away from it. In a separate study, the willingness to pay (WTP) for environmental goods that are dependent upon freshwater resources, that is, wetland ecosystems, which provide an important habitat for migratory bird species in Cyprus, was estimated.

Possible non-use values were estimated using the contingent valuation (CV) methodology in the context of water provision for migratory bird species (Swanson et al., 2001). The valuation scenario used was a real one: without regional co-operation, a migratory bird species, which uses wetlands in Cyprus and the UK as a habitat and migratory stepping stone, the white-headed duck, will be threatened with extinction. Those surveyed were asked to express their preferences for the provision of water to endangered species under co-operative and non-co-operative funding scenarios. Econometric analysis of the survey responses demonstrated that there exists a positive WTP for the provision of local water to the endangered species of £10 per household per year. It is further demonstrated that there is an increased WTP of £10 plus an extra £5 per household per year for the local allocation of water to the endangered species if other states along the migratory route make similar choices (the co-operative scenario).

4.4 Optimal Groundwater Management

This study looks at the particular issues of optimal groundwater management and the allocation of groundwater between competing agricultural and residential demands. Optimal allocation of groundwater is a multi-stage decision process. At each stage, for example, each year, a decision must be made regarding the level of groundwater use, which will maximize the present value of economic returns to the basin. The initial conditions for each stage may be different due to changes in either the economic or hydrologic parameters of the basin under consideration. However, in most of the dynamic models employed in the groundwater literature the resource is modelled as a stock to be depleted in a mining era before moving to a stationary state era. Implicit in these models are the assumptions of fixed economic relations and/or exogenous rates of change through time.

More complex and realistic representations of increasing resource scarcity incorporate opportunities for adaptation to rising resource prices. That is, in the long run, shifts away from water intensive production activities, adoption of new techniques or backstop technologies, substitution of alternative inputs, and production of a different mix of products offer rational responses to increasing scarcity. To model these, economists have developed the technique of multi-stage optimal control in the context of groundwater mining for agricultural production. Our study employs this technique to describe the chronological pattern of groundwater use by different economic sectors (residential and agriculture) in order to define the optimal quantity of the resource that should be produced when the available backstop technology (that is, seawater desalination) is adopted at some endogenously defined time. Included in a control model this type of adaptation strengthens its ability to describe economic processes associated with natural resource depletion. The additional information can further inform public policy decisions concerning natural resource allocation among economic sectors, optimal timing of adoption of an available backstop technology and definition of the optimal quantity of the resource to be produced by this technology for each of the different users.

Moreover, our model takes into account common property arrangements for groundwater resources that lead to dynamic externalities in consumption. These externalities are associated with the finite nature of the resource, pumping costs and the use of groundwater as a buffer against risk. Our study focuses upon the common use of the Kiti aquifer and addresses the scarcity rents generated by agricultural and residential demand for groundwater. The optimal allocation between agricultural and residential sectors is simulated based on hydrological parameters and the corresponding optimal unit scarcity rents are calculated. The optimal scarcity rents are compared to those that emerge under the simulated myopic common property arrangement, the difference reflecting the common property externality, allowing us to assess the benefits from optimal groundwater management, through, for example, more adequate and incentive groundwater pricing.

Our results suggest that in the presence of a backstop technology the effect of the dynamic externality in groundwater consumption is not particularly strong on the social welfare of the economic sectors using groundwater. This is an intuitive result, because it suggests that when the scarcity of the resource is reduced due to the presence of a backstop technology, welfare gains from controlling resource extraction are not significant for any practical purposes. However, in the absence of a backstop technology and continuous natural recharge, the effect on welfare from managing groundwater extraction is significant. A huge welfare improvement is derived from controlling extraction as compared to myopic exploitation of the aquifer (see Table 14.4).

Finally, an alternative methodology, the distance function approach, is employed to estimate the scarcity rents of the Kiti groundwater using more applicable behavioural assumptions for agricultural firms. Distance functions have a number of virtues, which make their use attractive when the environment under which firms operate is regulated and/or firms are

Table 14.4 Welfare and welfare improvement under the optimal control and common property regime

Regime	Backstop	Welfare	Welfare improvement
Optimal control	Available	£170.360 m	
Myopic	Available	£162.621 m	3.8%
Optimal control	Not available	£110.510 m	
Myopic	Not available	£25.9610 m	409.4%

Source: Koundouri (2000).

inefficient due to a lack of incentives faced by their operators. In particular, the first virtue of distance functions is that they do not necessarily require price data to compute the parameters. Only quantity data is needed. Secondly, distance functions do not impose any behavioural hypothesis (such as profit maximization or cost minimization). They allow production units to operate below the production frontier (that is, to be inefficient) and they also allow derivation of firm-specific inefficiencies. Thirdly, duality results between distance functions and the more conventional cost, profit and revenue functions provide flexibility for empirical applications.

The key extension of this research compared with existing theoretical literature is that, if cost, profit or revenue function representations are precluded, the restricted distance function provides an excellent analytical tool for estimating unobservable shadow prices of *in situ* natural resources (produced and used as inputs in production processes of vertically integrated firms). The data used in this research were based on the Production Surveys conducted by Koundouri and Xepapadeas (2003; 2004) for the years 1991, 1997 and 1999. Our analysis focuses on a sample of 228 agricultural farmers located in the Kiti region. The data set consists of a balanced panel composed of the same 76 farmers over the three years of the survey. Estimation suggests that firm specific efficiencies are increasing over time. The average technical efficiency for agricultural firms in the sample increased rather rapidly from 0.47 in 1991 to 0.78 in 1997 and finally to 0.94 in 1999, where a coefficient of 1.0 would represent a firm at the frontier of efficiency.

The reported increases in the technical efficiency of agricultural firms can be attributed to the major restructuring of the agricultural sector in the last decade in an attempt to harmonize the Cypriot agricultural policies with those of the European Union (EU) in the light of Cyprus accession in the EU. Alternatively, increases may indicate the existence of technological progress in the agricultural sector, which is not accounted for in our empirical model (which assumes constant technological change). These are the

Table 14.5 Resource rents under the optimal control and common property regime

Component of social cost	Optimal control (£Cy/m^3)	Common property (£Cy/m^3)
Groundwater pumping cost	0.31	0.31
Scarcity rent/marginal user cost	0.20	0.0097
Marginal social cost of groundwater	0.5*	0.32

Note: * Cost of the backstop technology desalination.

first estimates of the efficiency of the Cypriot agricultural sector and, as a result, there is no scope for comparison at present. The key outcome of this empirical application, however, is that estimated technical firm specific inefficiencies present in agricultural production technologies, suggest that cost minimization is not the relevant behaviour objective in irrigated agriculture in Cyprus. This result provides support for the use of the distance function approach to derive resource scarcity rents.

The unit scarcity rent of *in situ* groundwater estimated by the distance function is approximately equal to zero (0.0097 CY£/m^3) under the myopic common property scheme (Table 14.5). This is approximately 20 times less than the same value under optimal control. This comparison indicates that agricultural producers in the region are not paying the full social cost for groundwater extraction. This implies that under common property, externalities arise as current users of the resource are only paying the private cost of their resource extraction. As a result, the resource's scarcity value remains unrecognized (Koundouri, 2003). This pattern of behaviour is consistent with perfect myopic resource extraction, which arises because of the absence of properly allocated property rights in groundwater and is consistent with the results found in a study on WTP for groundwater quality.

A hedonic analysis of WTP for improvements in groundwater quality was also undertaken. Groundwater quality may affect the productivity of land used for cultivating crops. Where this is so, the structure of land rents and prices are expected to reflect these environmentally determined productivity differentials. Hence, by using the collected data on land rent or value for different properties, we tried to identify the contribution of fresh groundwater quality to the price of land and therefore WTP for groundwater quality.

Based on this approach, the estimated marginal value of groundwater quality as far as reduced salination is concerned is statistically insignificant and equal to £CY1.07 per hectare of land (Koundouri and Pashardes, 2002).

The statistically insignificant small marginal WTP for improvements in groundwater quality may imply that groundwater extraction is myopic, for instance, because of free-riding. This is expected to be an artefact of the non-existence of properly allocated property rights in a common-pool aquifer.

Another explanatory factor for the low marginal WTP for groundwater quality may be a substitution effect of farmers changing their land use to the more lucrative tourism industry. Tourism utilizes other existing water sources than groundwater.

5. BALANCING THE COSTS AND BENEFITS OF AN OPTIMAL WATER ALLOCATION AND THE CORRESPONDING POLICY IMPACTS

An optimal allocation of scarce water resources in Cyprus requires a careful balancing of the various values of water within the catchment area. In Cyprus, the preferred method for implementing the optimal allocation was through the development of a uniform water-pricing scheme where each water user is charged the same price. Hence, water pricing for residential, agricultural and environmental uses was taken into consideration and based on the marginal social cost of water supply. The marginal social cost of water supply was estimated at £CY0.45/m^3 by the Water Development Department in Cyprus, using the average incremental cost methodology and reflecting the long-run marginal cost of water provision based on the national resources required for its provision. The marginal social cost of water equals the opportunity cost of providing additional water for different purposes in Cyprus rather than providing other socially demanded goods and services (such as health services or education services) on the island. The marginal social cost of water provided to the charged residential and industrial sectors should also reflect the opportunity costs of losing water to the uncharged (public good) sector. The analysis of the value attached by the public to water allocation for the preservation of wetlands as a habitat and stepping stone for endangered migratory bird species has demonstrated that there exists a positive WTP for these non-priced and water-dependent environmental resources. The marginal social cost of water charged to residential and industrial sectors should in fact reflect the costs of all opportunities forgone as a result of a specific allocation of water.

However, there may also be other important factors, which have to be taken into consideration under this optimal allocation, such as equity (the impacts on lower income groups), risk (the impacts on variability) and hydrology (the impacts on conjoint users). The household demand study showed that the current increasing block pricing system introduces price distortions, which cannot be justified, either on efficiency or equity grounds. In terms of efficiency the current tariff system cannot be justified, because the same water resource supplies all locations at similar marginal social costs. Since large water consumers pay a lower average price than small water consumers, the current tariff system can also not be justified on equity grounds. Although a shift towards a uniform marginal cost pricing system will eliminate the deadweight loss of the current system, its benefits will be distributed in favour of the better off households. As such the water-pricing policy can be considered inequitable. The impact of water availability on the variance in producer profitability was also analysed, showing how the current and an alternative water allocation affects the welfare of risk-averse agents. We discovered that water has a positive, but decreasing effect on the variance of profit. Other things remaining equal, this implies that although additional water use increases the output and profit on average (that is, positive marginal productivity), it simultaneously increases the risk associated with the produced output. The analysis shows that the population is risk averse, and therefore additional water use may be welfare reducing.

An important concern was furthermore related to a possible reduction of agricultural subsidies and the expected impact that this may have on employment. However, the estimated production function showed no significant complementarity between labour and water input, indicating that a change in water use will not have any effect on current employment in the agricultural sector.

The logic behind treating the watershed as the most appropriate management unit is that the interactions of the physical elements of hydrology and geo-hydrology and water demand can be addressed in a coherent way and can guide policy. However, thus far the interdependent nature of surface water and groundwater, and the wider impacts that demand for one of these resources has upon the other, has been largely ignored in our study. Any policy impact analysis should consider the interdependency between and conjoint use of groundwater and surface water and their mutual impacts. Excessive groundwater pumping reduces, for example, surface water flows downstream and hence the available water for the sectors located there. On the other hand, groundwater pumping may contribute to surface water flows through return flows, increasing the importance of the timing of resource flows. Seasonal pricing could be used to ensure water availability to downstream users in line with their seasonal preferences.

Finally, the proposed allocation of water needs to be backed up by legislative change. Water legislation in Cyprus is characterized by a piecemeal approach (see the appendix to this chapter). The quality of freshwater resources is dealt with in several laws, depending on resource type and specific water use. Moreover, both water quality and water quantity aspects are dealt with through several different instruments, in particular in the case of groundwater. A more integrative approach is expected in the near future as a result of the implementation of the European WFD.

In sum, the foregoing has provided an example of how CBA of water policy can be structured in the context of the much discussed hydrological unit: the watershed. The methodology and case study combine to show how the constraints faced by watershed managers and policy-makers can be usefully evaluated and traded off using the principles of CBA and how, using economic analysis in conjunction with legal and hydrological backdrop, important considerations on environment and equity can be incorporated into the policy-making process.

APPENDIX

Brief Overview of the Institutional and Legislative Background to Water Policy in Cyprus

Generally speaking, the institutional arrangement to protect freshwater resources in Cyprus is characterized by a fragmented, piecemeal approach, involving the Council of Ministers, the Water Development Department of the Ministry of Agriculture, Natural Resources and the Environment, the Department of Labour of the Ministry of Labour and Social Insurance and the Medical and Public Health Services Division of the Ministry of Health (Grimeaud, 2001).

Water legislation in Cyprus has three main components:

1. *Legislation on the protection of freshwater resources of surface and groundwater*. The 1991 Control of Water Pollution Law [69/91]. Regulations include: 1992 and 1995 Regulations on Application for a Licence relating to Waste Disposal; the 1993 Regulation on the Prohibition of Discharges; the 1996 Order on Measures for the Protection of Underground Waters and a Code of Good Agricultural Practice.
2. *Specific legislation on groundwater*. The 1928 Government Waterworks Law and the 1946 Wells law. This arranges, *inter alia*, that groundwater which has not yet been subject to abstraction and exploitation [as well as waste water] falls under state ownership.
3. *Legislation on water supply*. This aims at maintaining an appropriate level of water quantity in certain sensitive aquifer areas and at providing

consumers with tap water in sufficient quantity. There are three main pieces of legislation: (a) 1955 Water Development and Distribution Law, (b) 1964 Water Supply law and (c) 1951 Water Supply law.

The implementation of legislation concerning the protection of water resources is the responsibility of two different ministries. The Ministry of Agriculture, Natural Resources and the Environment sets Environmental Quality Standards, grants permits for discharges and enforces all provisions related to pollution from industrial sources, while the Ministry of Labour and Social Insurance is in charge of monitoring and compliance with permit conditions. Legislation regarding water supply is implemented by Water Development Committees, which are established to promote the conservation of water resources, to develop the use of those resources and to co-ordinate water supply distribution. Moreover, they may also regulate the use of water and prevent waste discharges. The Council of Ministers designates water shortage areas for which permits have to be obtained prior to the construction of wells or the exploitation of surface water.

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15. Cost–benefit analysis, water scarcity and sustainable water use in Spain

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1. INTRODUCTION

This chapter presents an overview of the use and usefulness of economic analysis, in particular cost–benefit analysis (CBA), in decision-making in Spain. An attempt is made to provide a historical overview of the use of CBA, including recent developments linked to some of the major water plans in Spain. The use of CBA can be traced back to the economic development plans presented in the early 1960s, which were implemented by the Ministry of Development. Its application today is often linked to tariff setting and compliance with European funding requirements.

The economic analysis required by the European Water Framework Directive (WFD), adopted in 2000, has renewed interest in the economic analysis of water use and water policy, and presents a number of challenges in relation to the conventional analyses carried out so far in Spain. The discussions surrounding the use and usefulness of economic analysis in the major water plans in Spain today provide us with helpful insights into the types of problems faced by countries with severe water scarcity. Spain has a long history of policies trying to solve water scarcity and to ensure that water supply is not a limitation for economic development.

In this chapter, we furthermore address the main methodological issues, which arise when using economic analysis in Spain. It is shown that the economic analysis is embedded in a political-institutional context, which largely determines how specific methodological issues are dealt with in practice. Issues such as which alternative options are considered, which costs and benefits are analysed, when and where additional benefits and costs are included, and how much effort is put into the valuation of non-market benefits, are more than just methodological or technical questions. They are part of the public debate and public decisions regarding how much effort should be invested in economic valuation.

Various institutions in Spain have carried out CBA research related to water use and water projects. In most cases, these studies are partial as they often do not assess the full costs and benefits or they ignore the complementary relationship between nature conservation and water use for socio-economic purposes. An example of this latter type of analysis is presented as a case study in this chapter for one of the most important natural areas in Europe, that is, the Doñana protected natural areas. The case illustrates how conservation is made compatible with human use and what the economic values involved are. The latter is considered an important key to ensure the success of the implementation of the European WFD in Spain.

Hence, this chapter's main objective is twofold. First, an overview is presented of the political-institutional embedding of CBA in water policy in Spain (section 2). Second, the case study is presented, illustrating the typical type of economic analysis carried out in Spain to assess the trade-offs between water scarcity, alternative water use and nature conservation (section 3).

2. THE INSTITUTIONAL CONTEXT OF ECONOMIC ANALYSIS IN WATER POLICY

In this section, an overview will be presented of the use of economic analysis, in particular CBA, to appraise water projects and programmes in Spain. First, a brief historic overview will be presented, including the present use of CBA in Cohesion Funding and major European Regional Development Fund (ERDF) funded projects, followed by a discussion of the use of economic analysis in the 2001 National Hydrological Plan and the 2002 National Irrigation Plan.

2.1 Historic Overview of the Use of CBA in Water Projects

Economic analysis of investment projects in Spain started in the 1960s, initiated by the Ministry of Development. The Ministry of Development prepared and implemented development plans. The first plan analysed covered the period 1964–67 and focused on some of the colonization plans, including irrigation projects, which served to transform the economy of some of the poorest areas of Spain (mainly the south and west of Spain, for example, Plan Badajoz).

Following the advice of a 1962 World Bank report, this plan emphasized the need to look into the 'capital–output' relationship of water projects. Economic analysis was later during the 1970s carried out for major water infrastructure projects of the Ministry of Development. Cost-effectiveness analysis was applied in projects for domestic water supply in view of the difficulty of valuing the social benefits from these projects.

In 1980, the Ministry of Development – by then the Ministry of Public Works and Urban Planning – published several guidance documents on CBA for investment, irrigation and flood defence projects (Table 15.1). These guidelines built on prior experiences regarding project evaluation in the 1970s and were published with the aim of establishing common criteria for the application of CBA. These project evaluations were led by the so-called 'Inter-Ministerial Commission for Evaluation'.

Irrigation and water infrastructure projects were subject to CBA and a financial analysis. A financial analysis of the costs of new infrastructure was carried out in order to be able to calculate the possible tariffs to be charged to its users. Subsidies were considered in the analysis according to the applicable law (for example, 1985 Law and Regulations). For irrigation projects, a social CBA was accompanied by a financial analysis from the perspective of the farmers in order to ensure that the benefits brought about by the project allowed them to pay the tariffs imposed by the agency implementing the project. This requirement is still in place under the Law for Agricultural Development and Reform (Decree 118/1973 of 12 January 1973). Cost–benefit analysis was not used much in the late 1970s for major water infrastructure projects, until the European Commission reinstalled the requirement to use CBA for Cohesion Funding and major ERDF projects.

Nowadays, the European Commission regulations for ERDF projects[1] and Cohesion Funded projects[2] require the preparation of a CBA when major water projects are involved. In Spain, CBA has been applied by River Basin Authorities (RBAs), central government departments, regional governments, municipalities and water companies according to a methodology issued by the Ministry of Public Works and Urban Planning (MoPU, 1980), which includes the valuation of non-market benefits from projects. So far, 150 projects have been analysed and submitted to the Cohesion Fund for financing over the period 2000–06, and 20 major projects to the ERDF. The average size of these projects is about €20 million.

The CBA methodology used was based on the evaluation of the first round of projects submitted for Cohesion Fund funding and existing European guidance documents, distinguishing between different types of projects.[3]

The government considers it important to include the valuation of non-market benefits in CBA. Including these benefits, the internal rates of return range between 7 and 10 per cent (Table 15.2). Cost–benefit analysis is mainly used to comply with European regulations rather than to serve as a tool for decision-making and to help decision-makers to choose between

Table 15.1 1980 CBA guidances of the Ministry of Public Works and Urban Planning (formerly the Ministry of Development)

Project type	Costs*	Benefits	Sensitivity analysis	Other effects
Irrigation	*Non-periodic costs* ● Direct costs (project preparation, investment costs) ● Associated costs (private and public investment costs) *Periodic costs* ● Direct costs (operation and maintenance costs) ● Associated costs (operation and maintenance costs of private and public agents)	– Value of future additional agricultural production – Flood control – Improved drainage – Water supply – Electricity production – Recreational benefits	– Variability of main cost items (for example, labour costs) – Future development of (inter)national prices of agricultural products – Average yields of crops and development in time – Interest rate and opportunity costs of capital	*(only need to be identified)* – Income redistribution effects – Employment effects – Trade balance effects – Transport effects – Environmental effects (recreational, erosion, water and waste production, landscape changes)

Table 15.1 (continued)

Project type	Costs*	Benefits	Sensitivity analysis	Other effects
Flood prevention	As above	– Value of directly avoided damages (material, physical) – Value of indirectly avoided damage (loss of life, quality of life, functioning of the economic system) – Value of additional produced goods – Other beneficiaries: supply to population and irrigation, electricity production	– Variability of main cost items (for example, labour costs) – Development of future prices of goods and services generated by the project and the value of damages avoided – Interest rate and opportunity costs of capital	As above and – Reduction of costs for other users of the infrastructure – Impact on other activities in the area

Note: * Standard recommendations applied, for example, regarding the use of opportunity costs of labour and constant (market) prices excluding taxes and subsidies and so on.

Table 15.2 Project appraisal for Cohesion Funding and major ERDF projects

Type of project	Estimated costs	Reference situation	Time horizon	Estimated benefits	Internal rate of return
Water supply infrastructure	Investments costs O&M costs	Compared with status quo	25–50 years	WTP for water, public health effects from improved water supply, improved quality of life, increase in economic activity, avoided purification costs, energy savings, reduction of water abstraction from overexploited aquifers	7–10%
Waste water (major sewage collectors and treatment plants)	Investments costs O&M costs (energy, chemical products, sludge removal, repairs, personnel)	Compared with status quo	25 years	Reduction of pollution through discharges to the aquatic environment* Health benefits Other qualitative benefits (improvement of fish life, visual impacts)	7–10%
River regeneration	Investments costs O&M costs	Compared with status quo	25 years	Reduction of erosion, reduction of salinization, flood prevention	7–10%
Irrigation	Investments costs O&M costs	Compared with status quo	50 years	Productivity increase, changes to more valuable crops, increase of land values	7–10%

Notes:
O&M: operation and maintenance.
*Regulation 849/1986 of the public water domain costs the pollution content of the discharges produced in one year by 1000 inhabitants at €6611.

alternative projects. It also reflects the problem of non-market benefits valuation, which has received limited attention in Spain, even though this issue was already identified early on in the 1970s in the project appraisal guidance documents.

In recent years, CBA of main infrastructure projects, such as dams for irrigation or domestic water supply alternatives in river basins, has been subject to public debate regarding some of the difficulties associated with the lack of European Union consensus on an applied environmental valuation methodology. Some of the issues raised, for example by Arrojo et al. (2002), include the difficulty to value functions of water which cannot be broken down into separate components or which cannot be compensated with money or the difficulty of valuing irreversible changes which affect future generations and so on.

Other issues affecting the results of CBA of water-related projects in Spain and so on:

- delays in project investments and the underassessment of expected investment costs;
- associated delays in benefits, which affect the expected internal rate of return;
- the estimated life time of projects;
- the calculation method for benefits from irrigation transformation projects;
- the calculation method for benefits from urban water supply projects;
- different interpretations of the use of opportunity costs of energy, for example, including or excluding technological progress;
- the calculation method for the additional net benefits of increased activity in agriculture;
- the development of agricultural product prices in the future;
- the way irreversible socio-economic effects are taken into account in flooded areas;
- the appropriate discount rate;[4] and
- the analysis of demand for irrigation and urban drinking water.

In some cases, CBA of major water projects has evolved into a more complex multi-criteria analysis, including stakeholder consultation about the weights attached to different costs and benefits.

2.2 The 2001 National Hydrological Plan

Perhaps the most comprehensive recent use of CBA has been in the context of the 2001 National Hydrological Plan (NHP). The CBA carried out in the context of the NHP investigates the main costs and benefits associated with the proposed investments, including an analysis of the expected water demand – for urban and agricultural uses – and the expected economic benefits from the implementation of the plan. The various aspects of the CBA were reviewed by several experts and representatives of stakeholder groups. This has led in turn to an interesting and fruitful discussion about a number of economic issues which, up to now, had only been of marginal interest to decision-makers.

The political and institutionally embedded debates around the NHP concluded that there is a real need to develop more appropriate and different methodologies for the analysis of water projects. They reflect the changes going on in the context of water policy in Spain over the last 20 years, towards what has been called a 'mature water economy'. Overall, the contribution of agriculture to gross domestic product has decreased, whereas the value attached to environmental goods and services has increased at the same time. A recent study suggests, for instance, that prices for forestland internalize a substantial capital value of owners' environmental self-services (Campos and Mariscal, 2003; Campos et al., 2005). This has led to greater attention to the marginal impact of new water infrastructures and the importance of the distributional consequences of policy measures. The main issues following the discussions about the NHP include:

1. The need for a better understanding of urban and irrigation water demand. Increasing demand for water needs careful consideration and cannot be taken for granted or based on extrapolation. Different scenarios have to be considered in the CBA, especially when facing important changes in the economic structure and context, which affect, for example, economic margins of agricultural products and hence the irrigation area and the crops cultivated. In this context, the following two issues are worth noting:
   (a) The empirical evidence about the relationship between price levels and levels of water use is not conclusive in Spain. Some studies found no or an insignificant relationship between water use and the costs of obtaining groundwater or surface water. This may partly be due to the low price levels. Furthermore, the technical efficiency of water use and cultivated crop types is often related to other variables, such as soil quality, tenancy type, age of the irrigation infrastructure or other social variables.
   (b) Water demand for domestic use may be inelastic in the short run, but not in the long run, and this has to be taken into account when analysing the profitability and viability of a project, and the valuation of benefits, over a longer period of time.

2. The importance of comparing the 'right' alternatives. The question which alternative options to consider when trying to achieve a proposed (beneficial) objective is important. Not only technical solutions should be considered, but also social and economic options. The consideration of a wider set of alternatives may change the overall structure of the project costs and benefits. For example, stabilizing agricultural rents may be achieved through different options, including water demand management (Azqueta, 2001). Often the objective is to achieve a complex set of environmental, social and economic objectives simultaneously. Hence, the set of alternative options and the analysis of their costs and benefits has to take into account not only how they provide economic benefits to water users, but also how they affect other environmental and social objectives at the same time (moving beyond the identification of mitigation measures as in environmental impact assessment – EIA).

In comparing alternative options, several authors have emphasized the need for careful consideration of the costs and benefits involved, and how they are calculated. The calculation of the economic benefits of water projects, such as increased margins of agricultural crops, especially needs to be reviewed in the light of existing simple standard methods based on average values. When valuing such benefits, it is important to consider to what extent average values are appropriate in view of the fact that the net value of water in agriculture is often not constant. The net marginal profit decreases as the amount of water used increases (Uche et al., 2002). Understanding the characteristics of agricultural water use is therefore essential. Not all marginal gains derived from increased water use may be attributable to water since there may be other elements in the agricultural production function which may be exchangeable for water (García Mollá, 2000). The impact of new projects on net margins requires specific information about local conditions and circumstances, such as crop types, property structure, irrigation characteristics and meteorological information, and their relationship with crop yields and costs. When analysing the agricultural benefits, it is important to also consider possible income differentials and the extent to which employment opportunities are distributed across the areas affected (Arrojo et al., 2002; Azqueta, 2001). Finally, sensitivity analysis may help to include uncertainty related to changes in Common Agricultural Policy (CAP), World Trade Organisation (WTO) negotiations, EU tariffs, energy prices and so on. Any of these changes can substantially affect the structure of the expected benefits (Horne et al., 2002; Sumpsi, 1998).

Decisions affecting the water environment need to include an analysis of the environmental costs and benefits involved. Valuing or considering only those water uses for which market values can be calculated may be insufficient in a changing context with greater environmental awareness. Furthermore, compensation to those made worse off needs to be based on the benefits forgone by those made worse off (Azqueta, 2001). This may also require the valuation of non-mitigated environmental damage (Horne et al., 2002). In the context of proposed changes, the valuation of environmental impacts may not be sufficient for decision-making (valuing compensation) if it does not also incorporate non-market valuation. Box 15.1 presents a unique example from Navarra.

BOX 15.1 CALCULATING THE ECONOMIC VALUE OF WATER IN NAVARRA

In 2002, the Directorate of the Environment of the government of Navarra carried out an assessment of the economic value of the natural resources in Navarra. The quality of the natural environment in Navarra, especially the biodiversity, is among the most appreciated in Spain. However, the pressures on the natural environment and biodiversity are severe in Navarra. The region is located very close to the major international transportation routes in Spain and faces rapid industrial development. The regional Directorate of the Environment aimed to develop an instrument to facilitate negotiations between environmental and other socio-economic (sector) interests, including the internalization of environmental costs in decisions taken by the transportation and industrial sectors. This work is unique in Spain because of its scope and its foreseen direct use as a policy instrument.

The estimation of the economic value of the water resources in Navarra was one of the outcomes of this valuation process. With the help of this economic value the regional government hoped to be able to charge a water price to water users, which includes both financial and environmental costs. The analysis also accounted for the opportunity costs of downstream water users as a result of water quality deterioration. This was valued as the increase in costs in adequate drinking water quality supply, an increase in purification costs, reduced irrigation water quality for an increasingly demanding market and a reduction of the diversity of ecosystems and the services these provide. The valuation of aquatic ecosystem services is probably the greatest novelty in this case.

These services were valued with the help of the contingent valuation method, the replacement costs of riverside vegetation and the opportunity costs of reducing biological risks through reduced irrigation. The government of Navarra estimated the price of water including environmental costs at €0.05/m^3 compared to the current water price of €0.007/m^3.

Some of the issues raised by the academic community in relation to the valuation of natural resources in Navarra include the fact that many ecosystem functions were not valued, the arbitrary indirect estimation of use values, the adequacy of the travel cost calculation method to value recreational services and the validity of the willingness to pay approach to value non-market functions compared to other choice methods.

2.3 The 2002 National Irrigation Plan

The National Irrigation Plan (NIP), approved in 2002, incorporates an economic analysis of water use in agriculture. The NIP analyses the value of existing and potential crops in existing and potential irrigated areas. Included in the analysis are different elements of the production function, such as the income generated by specific activities or crop cultivation, the associated employment and the prices paid for crops. The economic indicators used to assess the economic impact of the NIP in different (irrigated or not) agricultural areas include gross income, total gross production, gross and net disposable margin, gross and net value added, economic rents from capital, economic yields and profits from different activities. These indicators were calculated on the basis of information about 2.83 million irrigated and 5.94 million non-irrigated hectares. The areas were considered a representative sample of production units in each of the agricultural zones.

This information served to analyse the agricultural benefits from the NIP and evaluate proposals of the irrigation plan itself and proposals of other agricultural programmes from other public bodies, including regional governments, aiming to improve the efficiency of water use by farmers. According to the NIP, the selection, valuation and prioritization of actions in actual or potential irrigated areas should consider the expected economic returns to the investment, the impact on employment and other variables such as energy use. Box 15.2 presents a regional example from Andalucia.

BOX 15.2 OPTIMIZING THE ECONOMIC VALUE OF IRRIGATION WATER IN ANDALUCIA

In Andalusia, the irrigated area has increased by 50 per cent in the last 20 years. This development has largely been caused by the extension of the areas with olive trees in Jaen and strawberry and citrus production in Huelva. As a result of this increase in irrigated area, the government of Andalusia asked itself the following two questions. First, what is the economic value of water used in agriculture? Second, in view of the structural water scarcity and the difficulty of increasing water supply, which alternative options would help to maximize the economic value of water across different user groups, especially during droughts, taking into account possible employment effects of changes in water use?

In order to answer these two questions, the regional government characterized the existing agricultural holdings in Andalusia in 1999 by crops (olive trees, cotton, sunflower, wheat, beetroot, maize), types of soil and water use. The government furthermore considered the restrictions of CAP and analysed different scenarios of price evolution and the impact of watering during drought periods. The results showed which options would be favoured by the regional government under these different scenarios to maximize the economic value of water use, taking into account at the same time other criteria such as employment, social structures and continuation of agricultural production in the area.

3. THE ECONOMIC USE VALUE OF THE DOÑANA[5]

3.1 General Description

In this case study, we will focus on the analysis of the economic value of activities related to water use in the Doñana National and Natural Parks, both with a total surface of 105 000 hectares, one of Europe's most important remaining protected natural water and wetland areas (Figure 15.1). Contrary to the general perception that the Doñana is purely a natural pristine reserve, important human activities take place in the 'Doñana National Park' and the 'Natural Park of the surroundings of Doñana'.[6] Doñana National and Natural Parks cover approximately 58 100 hectares of marshland and cropland and 46 800 hectares of forest (Campos and López, 1998).

Figure 15.1 Location of the Doñana National Park in Spain (Andalusia)

The communities living in the Doñana area have always been involved in a variety of productive activities. They generate income from the forest (eucalyptus and pine trees), livestock (breeding of cows and horses), recreational use, agriculture (strawberry and rice cultivation mainly) and fisheries (free access fisheries and aquiculture). Traditionally, the main use of the marshlands consisted of rice cultivation (since 1929). Today, also strawberries are grown in the sandy soils. Fishing and shellfish collection are probably the least well-known economic activities in the Doñana, but they are the area's oldest economic activities and the largest source of income. The Doñana is hence an important source of livelihood for many people.

Over time, the types of activities have changed, especially fishing and agriculture. Overall, the activities have intensified, resulting in serious development pressures, including the draining of the marshlands. Until the 1980s there were proposals for tourist development and road construction in an area of 800 hectares (from Huelva to Sanlúcar de Barrameda). In addition, there were proposals to increase the irrigated area to 100 000 hectares. If carried out, this would have meant the drying up of the wetlands and the extraction of more fossil water from the aquifer.

Alternative conservation options were proposed, arguing for the continuation of existing economic activities on which much of the wildlife depends, but excluding large scale intensification of economic activities and a reduction of the proposed irrigated area to 25 per cent. The conservation alternative was implemented with the support of the public authorities in Spain and financial support from the European Union, who paid for the economic costs associated with avoiding the loss of biodiversity and cultural heritage.

In the case study presented here, we analyse the economic use values and generated incomes associated with the implemented conservation alternative. Another alternative would have been to stop existing economic activities, such as agriculture and pasture management, and completely abandon the Doñana. However, it was believed that in that case the natural spaces would become fragmented and isolated areas, unviable for wildlife without the support of human activities.

Hence, the case study starts from the assumption that the protection of the natural values found in the Doñana can only be maintained and sustained if these natural values are considered in the context of the economic uses that traditionally have been part of the activities in the area. The conservation of the natural values depends on the maintenance of the economic use values of the natural resources found in the area. The economic activities carried out in the Doñana are in some cases favourable to nature conservation whereas in other cases they pose a serious threat. The results from this case aim to provide decision-makers with information about the economic value of what is considered 'reasonable conservation with use'. The case study's main objective is to illustrate the possibility of having economic use of natural resources, which is compatible with the conservation of highly valued natural reserves. In the next sub-sections, we will briefly address the main sources of income of the water users in the Doñana.

3.2 The Economic Value of Rice Crops

The marshlands in the Natural Park of the surroundings of the Doñana are used for rice production. The rice production is an important source of income, but also provides an important feeding place for birds and a habitat for crayfish. Nowadays, rice production takes place on approximately 5000 hectares of marshland in the Natural Park. In 1996, one hectare produced, on average, 8900 kilogram of rice. The operating income obtained from rice cultivation was €10.5 million in 1996.[7] Rice cultivation provides permanent employment to 273 people per year. For the owners of the land and the rice, the value of the operating margin was about €1723 per hectare.[8]

The cultivation of rice in the Doñana requires a lot of water. Water consumption was, on average, in 1996 8000 m^3/ha. The price paid by farmers for water is, on the other hand, low (€0.0007/m^3 litres). The abstraction of groundwater has a negative impact on the water table of one the aquifers in the area, which also happens to be susceptible to seawater (saltwater) intrusion. Furthermore, also the use of chemicals is expected to have a negative effect on the wildlife in the Doñana. Although there is no information available about the fertilizer and pesticide balance in the area, we do know that farmers spent, on average, €220 per hectare on these products in 1996.

3.3 The Economic Value of Strawberry Crops

The cultivation of strawberries is relatively new in the Doñana. The cultivation of strawberries has had, however, an important social effect, because of the crop's high economic margins. The total area used for strawberry crops is small (only around 300 ha), but the additional impact on the aquatic ecosystems through water abstraction and infiltration from and to the aquifer is significant and substantial.

The operating income as a result of strawberry production was €7.1 million in the period 1996–97. Almost 600 people are employed annually on a temporary basis in strawberry cultivation. The operating margin of the cultivation of strawberries is almost seven times higher than the operating margin of rice, namely €7484 per hectare. This high margin explains the rapid expansion of strawberry cultivation in the area and the pressure exerted to increase the cultivated area (mainly by transforming forest). Strawberries and rice are, economically speaking, the crops which yield the highest value added.

Strawberry cultivation contributes to the continuous trend of increased groundwater use and the lowering of the water table beyond a critical level. Other negative environmental impacts are related to the discharges of fertilizers to the aquifer, the accumulation of plastic materials covering the crops in the ground and landscape changes.

3.4 The Economic Value of Free Access Fisheries

The Doñana has always provided opportunities for free access fishery, because of its beaches, the continental platform, the Guadalquivir river and its tributaries and its lagoons. Public awareness about the need to protect the re-productivity of fish and to prevent the capture of young fish was already raised through medieval regulations for the Guadalquivir river (revised in 1512), which were drafted by the 'Fishermen University of Sevilla'. Nowadays, commercial fishing is still an important sector, but yields are decreasing. A distinction can be made between commercial fishing in the Guadalquivir river, in continental waters and off the beach. The caught species include elver, wedge sole, common prawn, triple-grooved shrimp (in the river), eel and crayfish (in the inland waters), and wedge shell (on the beach).

During the period 1996–97, the commercial fleet in the Guadalquivir river consisted of 157 boats with 314 fishermen. Gross operating income was €1.9 million in that period, that is, approximately €12 100 per boat. Compared with the income levels the mainly self-employed fishermen are able to earn in alternative employment (that is, opportunity costs of labour) and subtracting the consumption costs of fixed capital, commercial fishing in the Guadalquivir river generates negative incomes. In the case of fishing in the continental waters, the gross operating income in that same period was €0.81 million. Also these fishermen are self-employed and face negative gross margins. Fishing for red crab is economically speaking the most lucrative activity.

Finally, wedge shell collectors or 'coquineros' are another group of fishermen. They earn a gross annual income of €0.23 million. Also this activity results in negative operating margins. However, the 'coquineros' work part time and the collection of shells is hence just one part of their annual income.

There are three main reasons why the above mentioned fishing activities in the Doñana continue even though the incomes generated are often negative and the fishermen could get higher-paid alternative employment. First, commercial (shell) fishing is usually carried out part time and allows fishermen to carry out other activities as well. Second, some fishermen are retired and rely on other sources of income too. Third, (shell) fishing has a long history in the region and is to some extent culturally determined.

3.5 The Economic Value of Aquaculture

In the case of aquaculture ('fish farming') the situation is different. Commercial aquaculture involves mainly five species: gilthead, sea bream, European sea bass, grey mullet, eel and common prawn. Aquaculture is a relatively new activity in the area, although there exists an older tradition near the salt flats. The required space, the extensive nature of the activity and the activity's attraction to the bird life in the Doñana are the main characteristics of this economic activity.

In 1997, the operating income was €1.39 million, of which 75 per cent consisted of labour income. The operating margin was €0.34 million in that same period, which equalled €109 per hectare. The economic value added would have been twice as high (approximately €245 per ha) if there would have been no losses to birds living off the fish grown on the fish farms. In turn, the presence of these birds has a beneficial effect on recreation in the area (bird watching).

3.6 Total Economic Use Value of the Doñana

The results discussed in the previous sections are summarized in Table 15.3. The data refer to the two parks in the Doñana in 1997 (the National Park and the Natural Park of the surroundings of the Doñana). Also shown are the results for other economic activities in the Doñana, such as pine and eucalyptus cultivation, livestock and recreation.

Although these activities depend on the availability of water, too, and can be considered 'water users', it is especially the activities discussed before, which are the most important water users in the Doñana. These latter activities account for approximately 85 per cent of all the employment and income generated in the area.

The total operating income (net value added) equals €26 million (€253 per hectare) and the total operating margin almost half of this, that is, €13 million (€121 per hectare). These values are partial values, because not all economic activities and (their effect on) environmental functions have been valued, including biodiversity. It is especially this latter value why the Doñana has been nominated as a national park and Ramsar site.

Table 15.3 Total operating income and margin in the National and Natural Park of the Doñana in the period 1996–97 (in thousand euros)

Economic activity	Employment (working days)	Revenue	Intermediate expenditure	Labour costs	Total costs	Operating income	Operating margin
Pines	36 515	2 730	224	1 233	1 458	2 506	1 272
Eucalyptus	5 132	1 336	477	174	651	859	685
Cows	4 173	627	553	150	703	75	−76
Horses	0	194	181	0	181	13	13
Recreation	12 545	1 282	293	801	1 094	990	188
Rice crops	75 000	13 911	3 375	1 921	5 296	10 536	8 615
Strawberry crops	162 000	12 114	5 001	4 868	9 869	7 113	2 245
Fisheries	119 210	3 331	372	3 582	3 954	2 959	−623
Aquiculture	33 200	3609	2 216	1 052	3 268	1 393	341
Total	447 775	39 134	12 692	13 781	26 474	26 444	12 660

4. CONCLUSIONS

In this chapter, we have tried to illustrate the way economic analysis and in particular CBA is embedded in the political-institutional context of water policy in Spain. The review of the use and usefulness of CBA showed that CBA was incorporated in the early development plans in the 1970s to improve the economic rationality of promoting the development of poorer areas in Spain. More recently, renewed public and scientific interest in the economic rationality of water plans is reflected in the discussions surrounding the implementation of the National Hydrological Plan. At regional level, governments are increasingly interested in assessing the impact of water projects on the total economic value of their natural resources and identifying viable and sustainable options, which maximize the use value of scarce water resources.

The lessons emerging from the discussions in the past are important for the implementation of the recently adopted European WFD in Spain. In this context, careful attention has to be paid in CBA, for example, to:

- the development of future water demand, especially agricultural and urban water use, the associated pressures and impact on water quality and ecological status;
- the associated responses to these pressures and impacts, including the adoption of water demand management measures;
- the effectiveness of water pricing as a specific demand management instrument in different environmental and political-institutional contexts, taking into account questions related to both economic and social (welfare distribution) issues; and
- the assessment of environmental costs and benefits associated with different alternative options to stimulate sustainable water use and good ecological status.

The application of CBA in Spain is usually rather partial. The case study presented in this chapter is a good example of the type of economic analyses carried out in Spain for water-related projects. The full costs and benefits of alternative options are not taken into account. The economic analysis is used to ensure that the commercial use values of water use justify the policy proposals. More recently this has included social and environmental benefits. In the context of the WFD, conservation objectives are of paramount importance and the economic analysis hence needs to ask the question how this new objective can be achieved and thereby enters the political economy domain.

In a situation of severe water scarcity and a complexity of historically, culturally and institutionally evolved multiple water claims and rights, as is the case in many Mediterranean countries, sustainable water resource management and nature conservation cannot happen without some level of commercial use as well. Economic multi-functionality with environmental regulation of the productive use of natural reserves is considered the right strategy, maximizing the total economic value of these areas. In today's policy context in Spain, it is therefore important to point out the need to put more effort into the analysis of environmental costs and benefits of water-related policies and programmes, especially those which are not valued by the market. Until now, there has been little attention given to this issue by different institutions, with some exceptions such as Navarra.

Finally, when faced with irreversibility (for example, the loss of endangered species) and the trade-offs between conservation and commercial uses in the context of water scarcity, the suitability of CBA may be limited. The question of efficiency is in those cases often not the main issue and a comparison of the costs and benefits of the 'conservation alternative' and the 'development alternative' not relevant. What is relevant in that case is the economic cost of avoiding irreversibility. The decision of conserving a unique habitat may need to be treated in the context of social acceptability of the conservation costs and not of social preferences. Irreversibility options hence usually need to be considered beyond the analysis of welfare gains and losses. However, once the conservation decision is taken, it still remains possible to investigate and make explicit the economic value of the maintenance of the traditional water resource uses, which are compatible with the conservation values.

NOTES

1. Article 16 (2) of council regulation 2082/93 defines major projects as those whose total eligible cost equals €25 million for infrastructure investments and €15 million for productive investment.
2. Article 10 (5) of the Council Regulation (EC) no. 1164/94 establishing the Cohesion Fund explicitly requests an appraisal of the medium-term economic and social benefits (of projects), which shall be commensurate with the resources deployed and states that an assessments shall be made in the light of a cost–benefit analysis.
3. Cohesion fund projects include water supply to population centres, sanitation and waste water treatment projects. The ERDF projects include new infrastructures of water supply to population centres and economic activities (measure 3.1), improvements of efficiency in existing infrastructure and in water use (measure 3.2), sanitation and waste water treatment (measure 3.5), environmental actions in coastal areas (measure 3.5) and protection and regeneration of the natural environment (measure 3.6).
4. The discount rate often used is the one derived from the 1985 Spanish Water Law and Regulations (established at a time of high interest rates). Nowadays, however, a negative real interest rate seems more appropriate.
5. This section is based on Campos-Palacín and López-Linage (1998).

6. Note that a distinction is made between the 'Doñana National Park', consisting of a total of 50 800 hectares and including the 'Doñana Biological Reserve' covering almost 6800 hectares, and the 'Natural Park of the surroundings of Doñana', consisting of 54 200 hectares, including the 'Guadiamar Biological Reserve'.
7. Operating income equals here net value added, that is the difference between the market value of the produced quantity and the costs of intermediate inputs and the value of fixed capital consumption (depreciation).
8. The operating margin is calculated by subtracting the total production costs from the value of total production. Contrary to the calculation of the operating income, labour costs are included in the total costs subtracted from the value of production.

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16. Cost–benefit analysis of urban water supply in Mexico City

G. Soto Montes de Oca and I.J. Bateman

1. INTRODUCTION

Shortcomings in the water supply service in large urban areas of developing countries are a critical problem affecting millions of people (ICWE, 1992; UNDP, 1990; WHO et al., 2000). Almost half of the world's population live in urban areas, and most population growth is taking place in the developing world (United Nations, 1995). The enormous volumes of water and extensive infrastructure required to fulfil urban water demand have frequently exceeded the ability of government to provide secure supplies, and have also created severe environmental problems (Drakakis-Smith, 2000; Hardoy et al., 1992; Munasinghe, 1990; Serageldin, 1994).

Governments in developing countries often subsidize water supplies, typically in an attempt to achieve social and health benefits for the low-income households forming the large majority of the urban population. However, a perverse result can arise if the benefits of subsidized water accrue primarily to wealthier households receiving reliable services, with poorer households benefiting in a less than proportionate manner because they have irregular or non-potable water supplies and have to purchase water from other, non-subsidized sources. When this is the case, the drain on government revenues represented by the subsidy can hamper its ability to expand and improve the service provided to the urban poor. The importance of increasing investment in new infrastructure, as well as for the operation and maintenance of the current system, is highly recognized and has become a key political issue. The general consensus at the international level remains that the necessary resources need to come from domestic consumers (Brookshire and Whittington, 1993; World Bank, 1991).

The criteria to evaluate the water supply projects in developing countries initiated a redefinition process since the late 1980s. International donors and other multinational institutions started to discuss the efficiency of the water supply projects financed by donors and national governments up to that moment (Brookshire and Whittington, 1993; McPhail, 1993; Singh et al., 1993; Whittington and Swarna, 1994; Whittington et al., 1990; 1991). According to Whittington and Swarna (1994), water supply projects in developing countries frequently failed, because the projects were not evaluated with an economic rationality. This rationality should consider the application of cost–benefit analysis (CBA) to assess whether the estimated benefits and costs of the proposed change in the water supplies would justify in economic terms a policy intervention in the provision of the service. As Brookshire and Whittington (1993) argue, the fact that international organizations have defined water as a basic right and that it is difficult to value benefits of improved water supplies in both physical and monetary terms resulted in insufficient analysis of benefits. While the cost of the project was given by the amount of the required investment, its benefits presumed to be the collection of water sales. Yet, because of the water prices subsidies, financial rates of return on water projects are typically low and projects must be justified on other grounds. The failure of several projects and the struggles of the water authorities to improve the service conditions have led to the recognition that as part of the project appraisal it was important to analyse the potential beneficiaries, their preferences for a specific level of service, and their willingness and capacity to pay for the level of service provided. In the context of developing countries, such information is necessary to ensuring that poor households will have access to the project services and to know whether and to what extent cost recovery can be achieved (ADB, 1999).

Based on this revised paradigm to evaluate water supply projects, a number of recent studies have been undertaken to estimate the benefits of the projects by calculating people's willingness to pay (WTP). Two approaches have dominated: the estimation of households' compensatory strategies for coping with the inefficient services experienced and the measurement of the households' WTP, frequently through application of the contingent valuation (CV) method (Whittington and Swarna, 1994). Overall, the available evidence indicates that the amount that households are willing to pay for improved water services varies widely. Poor households without good service may often be willing to pay a great proportion of their incomes for improvements to that service. For example, McPhail (1993) found that households in five small cities of Morocco were willing to pay, on average, 5 per cent or more of their total household expenditures for water. Goldblatt (1999) also found for two informal settlements in Johannesburg that the households' WTP to connect to the water system was 5 per cent of their income. Zerah (1998) calculated that the cost of coping with water supply unreliability was about 15 per cent of the households' monthly income in Delhi.

The economic analysis of water services has in most cases been undertaken in rural communities. The available evidence for urban areas is still very limited, often concentrated on peripheral areas of the city without water connections (Saleth and Dinar 2001). The problem of undertaking this type of analysis in large urban areas of developing countries is highly complex, owing to the cities' heterogeneity in terms of service conditions and socio-economic characteristics. Although in some large cities the water supply network is sufficient, the service standards can often vary drastically from one area to another as a result of engineering, geographic or urbanization problems. In socio-economic terms, practically all cities present a remarkable diversity, with a relatively clear distinction between different income groups, and low-income inhabitants representing up to 60 per cent of urban populations.

The use of CBA in developing countries should consider particular aspects related to issues such as income inequality or diversity of service conditions. These aspects should be reflected in the CBA results by identifying the benefits of different groups. Being aware of the distributional aspects of the WTP of households with different characteristics may reflect greater credibility on the study accuracy and acknowledgement of the problem complexity. Also important is presenting the results in a straightforward and logical process. As has been recognized in the literature, if decision-makers can understand the information and observe the obvious implications of the study results, the probabilities of influencing the decision-making process will increase (Colvin, 1985).

This study is concerned with the measurement of Mexico City households' benefits and costs for two water supply scenarios, both predicated upon a do-nothing situation in which supply quality deteriorates. While one scenario offers maintenance of current supply conditions, the other offers an improvement over those current levels. The main objective is to illustrate the usefulness of the WTP approach in an urban water supply context. The study results are used to undertake a CBA to evaluate policy alternatives to improve the service performance in the city and we used the study results to observe the reactions of a number of key decision-makers about the usefulness of this information for the policy-making process. This chapter proposes that the CBA and WTP results can be used as an important input for proposing changes in water tariffs. We argue that presenting economic information to decision-makers can significantly enhance the decision-making process, particularly when that information considers differences across consumers with diverse characteristics and ability to pay for essential water services.

2. THE WATER SUPPLY SERVICE IN MEXICO CITY

Mexico City is the second biggest city in the world. The almost 20 million inhabitants of the Metropolitan Area of Mexico City consume 65 cubic metres per second (m^3/s) of potable water. This water is principally extracted from the local aquifer (71 per cent), with the remainder being piped from external sources. The bulk of this water is used for domestic purposes (67 per cent) and the rest for commercial, service and industrial purposes (DGCOH, 1997; INEGI, 1999).

Although almost all the population (97 per cent) have a piped water connection to their homes, several neighbourhoods suffer rationing of the service and poor water quality standards, particularly in the peripheral areas of the city (INEGI 1999). About 1 million inhabitants experience supply rationing in the main entity of the city, the Federal District, with some receiving water every other day, while others are supplied as infrequently as once each week (Reforma, 30/3/2001).

The Federal District, the core entity of the Metropolitan Area of Mexico City, houses almost half of the total city's population (8.5 million people). The local government is the monopoly provider and operator of the water supply service. Water prices for domestic use are heavily subsidized. Prices are based on the increasing block tariffs structure. Behind this price structure is the idea of using cross-subsidies so that high-income households and enterprises pay more, because they use more water than the poor households (Boland and Whittington, 1998). There is evidence that in developing countries this has led to a situation where a significant proportion of households end up paying an artificially low first block price (Baumann et al., 1998).

According to the local authorities, the approximate direct cost (given by the budget allocated to the involved water institutions in the Federal District) of one m^3 of water is 9 pesos (at 2002 prices), and the marginal cost might reach 15 pesos. However, the average tariff for one m^3 of water for domestic uses is 2 pesos, contrasting with the 12–13 pesos charged for non-domestic uses (CADF, 2001). In addition to this substantial subsidy for the domestic service, the authority's invoicing and enforcement capacity is weak, particularly for domestic consumers. Only 52 per cent of the supplied water is invoiced and just 33 per cent is actually paid (INEGI, 1999). The invoicing system remained highly inefficient for decades, leading people to adopt a culture of non-payment, which was reinforced by the low water prices, lax billing practices and the lack of effective programmes to create public awareness. The cross-subsidy phenomenon of a small sector of the consumers is evidenced in the fact that 80 per cent of the total resource collection comes from the non-domestic sector (CADF, 2001).

During 2002, the government spent 7 billion pesos on water supply, but recovered through tariffs just 3.2 billion pesos (Reforma, 7/4/2002). This deficit has limited the local government's ability both to improve the current service in those areas with water scarcity problems and to invest the necessary resources to guarantee sustainability of supply.

The issue of sustainability is far from being one of mere environmental rhetoric. The sheer volume of the city's water demand has caused a considerable reduction in the available water resources in the region. The local aquifer is already experiencing critical overexploitation and the external sources that currently make up the shortfall in the city's water provision are being employed at maximum capacity. Owing to political dispute with the neighbouring local government, the Federal District government has been constrained to use the same volumes of water to supply the still growing population since 1996. This water resource scarcity has pushed the government to improve efficiency by implementing a programme to repair pipe leaks and install water meters. This programme has reduced the pipe water leaks from approximately 40 per cent in 1997 to 33 per cent in 2001. Water meters have been installed in about 70 per cent of the households, which has already presented some positive results in terms of reducing water consumption per inhabitant and improving collection performance (CADF, 2001). However, these actions are still small in comparison to the magnitude of the ongoing supply problem.

Projections for the year 2010 estimate that an additional 18.2 m^3/s of water will be required in the city in order to fulfil the current water deficit and the expected population growth (SMA, 2000). Different policy alternatives have been proposed to address this problem, with schemes which aim to increase water sources and to decrease water consumption (DGCOH, 1997; SMA, 2000). Yet, while there has been certain assessment of supply-side issues, there is almost a complete absence of information regarding the nature of domestic demand, in particular consumers' WTP for changes in water supply characteristics. As such, it is considered vital to undertake a CBA to underpin any long-term sustainable policy for the city.

In terms of the evaluation of investment projects, the Law of Public Buildings of the Federal District only requires the application of an environmental impact assessment (EIA) (GDF, 1998). However, EIA only estimates the likelihood of adverse environmental consequences; it does not offer any information about the economic convenience of accepting or rejecting a policy or project (Bateman et al., 2002). The use of CBA in the Mexican context has been very limited and is typically only considered for interactions with international agencies. In general, the water management policies in the city have remained isolated from the use of economic techniques to evaluate investment projects. Providing information based on a

CBA to assess future water policies is an important area that needs to be explored.

3. MEASURING WTP FOR WATER SUPPLY CHANGES

To determine whether households are prepared to pay for the proposed changes in service provision, the CV method was used. This method is based on survey techniques in which a member of the household is asked a series of structured questions designed to determine the maximum amount of money that the household is willing to pay for the proposed change in service provision (Whittington and Swarna, 1994). During the last decade, the CV method has been broadly applied into the assessment for water services in developing countries, particularly in rural areas contexts (Brookshire and Whittington, 1993).

The contingent market created aimed to elicit WTP for the implementation of a long-term programme, which would eradicate the risk of water shortfalls in the future. We tested two scenarios via a split sample design. Both scenarios started from a baseline risk of water shortfalls over the next decade. The first scenario (termed the maintenance scenario) was presented as a programme that would avoid this risk and ensure that the current service level would be maintained. The second scenario (termed the improvement scenario) was a programme that would achieve the goals of the first and in addition improve the service conditions. In accordance with best practice guidelines (Arrow et al., 1993; Bateman et al., 2002) a dichotomous choice elicitation method was used throughout.[1]

The sampling strategy was defined to represent regional differences in the service standards, largely a factor of the diversity of the water sources used to supply the city. Three zones were chosen: west, north-centre and east (see Figure 16.1). In general terms, the west zone is considered to have a good service level, because most of the water coming from external sources enters through this area. In this zone, high-income neighbourhoods are located, though some recently urbanized poor areas are also present. The north-central zone has more heterogeneous service standards with low water pressure problems, since it relies more on limited external sources and local wells. Low- and medium-income neighbourhoods dominate this zone. Finally, the east zone suffers from frequent water shortfalls, because fewer wells are located here, creating the need to transport water from other localities. This zone is the most populated in the Federal District and concentrates families with one of the lowest income ranges (DGCOH, 1997; INEGI, 2001).

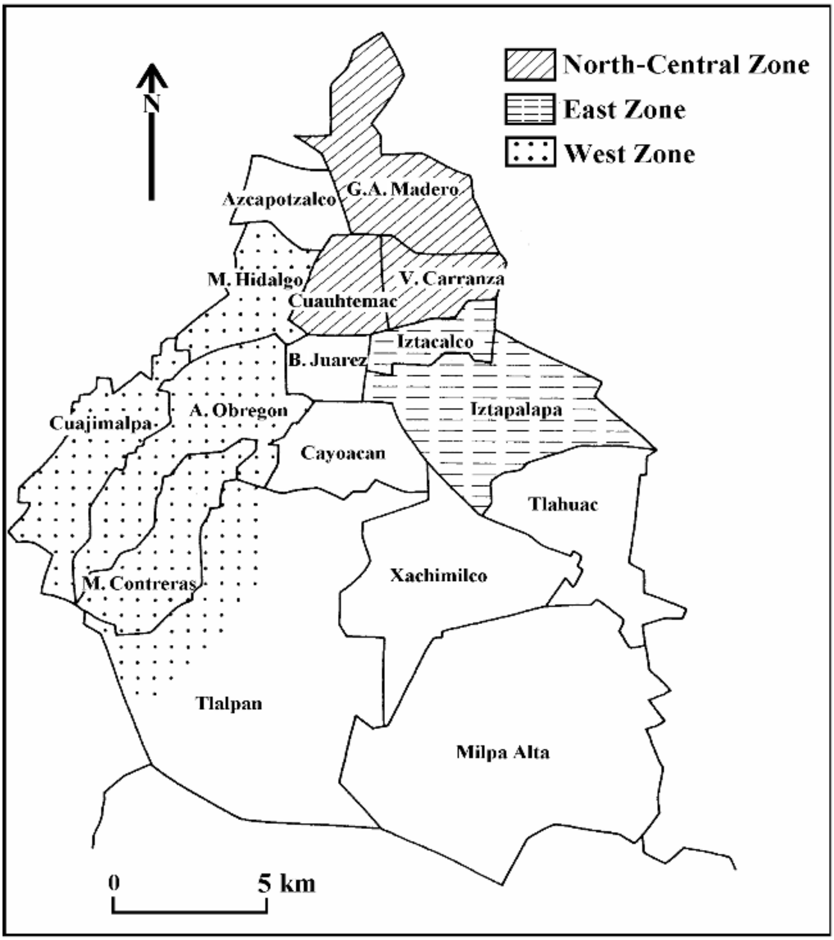

Figure 16.1 The Federal District and the study zones

A random telephone survey (Ethier, 2000) was applied to 1424 households. Some 714 households were interviewed regarding the maintenance scenario and 716 households faced the improvement scenario. The telephone survey was based on the area telephone codes, using the random digit dialling method (RDD) (Frey and Oishi, 1995). The survey was applied over a closed period of 20 days, including weekends, during November and December 2001.[2]

4. SURVEY RESULTS

4.1 Heterogeneity of Service Conditions and WTP Trends

Survey findings confirmed that the quality of water services is highly heterogeneous and differs significantly across the overall study area. A distinct trend in services levels is discernable with households in the lower income north-central and eastern zones suffering more frequent water shortages. Closer inspection showed a significant negative correlation ($p < 0.05$) between the frequency of water shortages and household income levels.

This uneven distribution of water shortages is also reflected in reported averting behaviour. Households in the eastern zone, where service interruptions are most frequent, have a significantly greater water storage capacity (via cisterns) and consume more bottled water than those in other areas. Indeed well over 90 per cent of households in the eastern zone consume bottled water rather than relying upon tap water for drinking purposes. Although the cost of these averting measures was not investigated, we can infer that the respondents in the east and/or those suffering frequent water shortfalls or receiving poor water quality have higher expenditures arising from reported averting measures.

These geographical and correlated socio-economic gradients in the distribution of water supply problems and related averting behaviour are reminiscent of the findings of previous studies such as those reviewed above. Given these relationships we would expect that evidence of corresponding gradients in WTP would also be observed. Multiple regression analysis of WTP responses revealed that such expectations were indeed fulfilled. As reported elsewhere (Soto Montes de Oca et al., 2003), when we control for other influences upon WTP,[3] analysis confirms highly significant relationships ($p < 0.05$) between stated WTP and both household income and the frequency of water shortages. As both of these factors are highly correlated they cannot be included simultaneously within a single regression model of the WTP data and so the relationship with income is emphasized here as it is both the stronger and more readily interpretable of the two.

The nature of the relationship between WTP and household income (with higher levels of income being in turn associated with lower levels of water shortage) differs revealingly between the maintenance and improvement scenarios. Although the location of WTP distributions was roughly similar across scenarios (with mean bi-monthly WTP levels of approximately 250 pesos per household) the scale of those distributions differed markedly across households. While, for both scenarios, higher household income is associated with higher WTP, the range of values is much broader for the maintenance than for the improvement scenario. This results in an interesting change in the implied ranking of projects across households, with poorer households valuing the maintenance project lower than the improvement scenario and vice versa for richer households. This eminently logical result reflects the fact that poorer households currently endure low levels of service, which they already address through a variety of averting expenditures. Therefore, for such households the improvement scenario offers very considerable gains over the status quo and is therefore highly valued. Conversely, rich households have relatively little need for further improvements in their already good levels of supply and are willing to pay much higher amounts to ensure that the status quo is maintained.

The relative benefits of the maintenance and improvement schemes are therefore intimately linked to the status quo position of households in terms of their (highly correlated) income and water service levels. This relationship is illustrated in Figure 16.2. Here the horizontal line represents a water supply quality continuum ranging from the lowest possible service quality level (denoted L) to the highest possible level (denoted H). The current position (status quo) enjoyed (endured) by any given household is denoted C and can lie at any point between L and H (in Figure 16.2 we place this point roughly in the middle of the continuum purely for illustrative clarity). The water supply improvements provided by the maintenance scenario are therefore given by the distance LC, while those for the improvement scenario will be given by the distance CH. We can also describe corresponding WTP for the two schemes as the amounts $\$_M$ and $\$_I$ respectively. Given the observed association between income levels and service quality the we know that for poorer households, C is nearer to L (low service quality) and thus they benefit more from the improvement project than from the maintenance scenario (put simply, for poorer households $CL < CH$). Therefore, for poor households $\$_M^P < \$_I^P$. For richer households, C is nearer to H such that $CL > CH$. Consequently, as we observe in our WTP study, for rich households $\$_M^R > \$_I^R$ (although for both scenarios the WTP of rich households exceeds that of poor households).

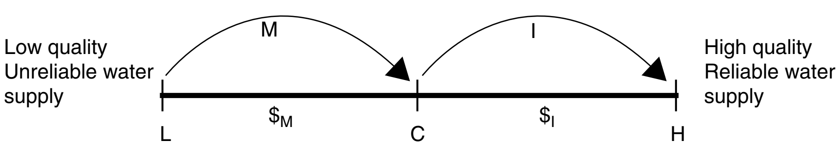

Figure 16.2 Benefits of the maintenance and improvement scenario

4.2 Aggregate WTP

There are different approaches to generalizing the WTP amount from a sample to the respective population. As Bateman et al. (2002) explain, the issue is how to aggregate the individual valuations, since different approaches can have substantial effect upon the size of such measures. Traditionally, CBAs based on CV information have calculated the aggregate WTP by simply multiplying the mean WTP by the total number of households in the population. In this case, when using the estimated mean WTP, the aggregated WTP per year is 3.0 billion pesos for the maintenance scenario and 3.5 billion pesos per year for the improvement scenario.

However, relying solely upon the whole sample mean WTP does not recognize any differences between high- and low-income households. Decision-makers may well wish to consider the WTP of households relative to their income levels in order to evaluate the ability of payment of different groups and the type of benefits that each group values most. This information may become an input to targeting subsidy policies more accurately, so as to enhance distributional equity relative to the current strategy of subsidizing everybody equally, irrespective of income or service quality.

The CV design adopted elicits information of different income groups regarding their ability and WTP. By using the estimated mean WTP of different income groups, we obtained the aggregate WTP of each group of households by income categories. We multiplied the number of households within each income range by their respective estimated mean WTP. If the government decided to adjust the tariffs following this approach, the amount of resources that could be collected total 4.2 billion pesos annually for the maintenance scenario and 4.0 billion pesos for the improvement scenario. As explained above, while the scale of the WTP distributions differs substantially between the two scenarios, their location and hence means are relatively similar as, consequently, are their aggregate values.

Given the spatial nature of the distribution of WTP for the two scenarios, we can envisage a programme whereby an improvement scheme is implemented in poor areas while a maintenance scheme is implemented in rich areas. The aggregate benefits of this combined ‘reliable service programme’ are calculated by multiplying the number of poor households by their WTP for the improvement scheme while the number of rich households is multiplied by their WTP for the maintenance scenario and summing these amounts. Using this approach, the reliable service programme is valued at 4.8 billion pesos annually.

4.3 Cost–Benefit Analysis

The CBA compared the aggregate WTP figures of the population with the annual available budgets and the estimated investment required to improve the service in order to examine the potential implications for the future of Mexico City's water policies. Table 16.1 presents the annual cost budget for the water sector alongside the estimated aggregated WTP amounts.[4]

The authorities reported that the total costs of the service provision for 2002 are about 7 billion pesos. The Treasury expected to recover approximately 3.2 billion pesos, of which about 80 per cent comes from the industrial and service sectors. The supplied subsidy is approximately 3.8 billion pesos, directed essentially towards the domestic sector (CADF, 2001; Reforma, 7/4/2002). The aggregated WTP figures of about 4.0 billion pesos show that the government would be able to withdraw the subsidies and use this money for other purposes if it was decided to invoice the money that the households reported to be willing to pay for maintaining the services.

Table 16.1 Aggregate annual costs and benefits of applying the maintenance, improvement and reliable service options in Mexico City (billion pesos; 2002 prices)

Category	Current available resources	Required resources to modernize the service	Potential resources from households' WTP
I. Allocated cost budget for the water sector	7.0		
a) Collection from consumers	3.2		
• Domestic households	0.6		
• Non-domestic sector	2.6		
b) Subsidy	3.8		
Additional cost budget required to modernize the service		2.0	
II. Aggregate WTP for different scenarios			
Aggregate WTP for maintained services			4.2
Aggregate WTP for improved services			4.0
Aggregate WTP for reliable services			4.8

Current revenues from the domestic sector are around 600 million pesos, that is, seven times lower than the WTP reported by the households.

The water authorities estimated that an annual budget of 9 billion pesos would fund modernization of the system (GDF, 1998; Reforma, 7/4/2002). This means that the government needs approximately an additional 2 billion pesos annually to improve the service in a substantial manner. The aggregated WTP for the improvement scenario is twice the budget that the authorities need for the service modernization. The aggregated WTP of 4.8 billion pesos for the 'reliable service programme' would provide the necessary resources for the service modernization, as well as the possibility of reducing the subsidies by about 2.8 billion pesos, which equals 70 per cent of the current subsidy flows. This information was used as input in interviews with local and federal decision-makers in which we tried to obtain their reactions about the implications of these results for future water supply policy in Mexico City.

5. DECISION-MAKERS' PERSPECTIVES REGARDING THE STUDY RESULTS

Following the WTP survey, findings were presented via one-to-one interviews with decision-makers in an attempt to gauge their views regarding water tariffs and their reactions to the WTP results. Thirteen decision-makers were interviewed to investigate their reactions to the research results. Of these, the majority were from local and federal government, the remainder being drawn from the private and non-government sectors. As such, the decision-makers interviewed represent the key water management institutions in the Federal District and metropolitan area. In the Federal District, the interviewees were officials of the two offices in charge of the service operation and administration, that is, the Directorate General for Hydraulic Construction and Operation (DGCOH) and the Federal District Water Commission (CADF), which recently merged into Mexico City's Water Commission. At the local government level, a representative of the congress (which is responsible for defining and approving water tariffs in the city) was interviewed, as was an official of one of the private companies working in the service provision (SAPSA). At the federal level, officials from both national and regional offices directly involved in water management policies in the Metropolitan Area of Mexico City (MAMC) were interviewed (specifically these were an official from the National Water Commission [CNA] and an official from the Regional Office for the Valley of Mexico [GRAVAMEX]). Finally, the other interviewees were influential representatives of the private sector (ANEAS) and the non-governmental sector (Union of

Environmentalists Groups). Except for the CNA and SAPSA, the heads of all these institutions were interviewed. Some of them are directly authorized to take decisions in future policy changes, while others held positions that give them influence in the policy outcomes.

The interviewed decision-makers believed the results presented to them, especially because the results were logical and in general accorded with their expectations. General points of agreement included:

- that even low-income households would be prepared to pay higher water charges in return for an improved service;
- averting measures to cope with the service unreliability constitute important costs to households with poor service standards. They recognized that these costs could be the same or even exceed what people reported to be prepared to pay; and
- the response of different groups of households to the two scenarios showed coherence – high-income groups would pay more because they are happy with their current services, while low-income groups would do so because they want an improvement.

One NGO representative felt that the CBA result was encouraging, because the aggregated figures showed that enough resources can be generated to finance the service improvement. This contrasts with the more conservative position of a local official, who argued that any increment in the tariffs, regardless of its magnitude, would impact the service collection substantially resulting in increased rates of non-payment of water bills.

The potential uses of the study results were also discussed. A couple of interviewees recognized that present policies are possibly the result of insufficient information, prevailing ideology and politics. In this respect, one of the key local decision-makers said that the information provided by the WTP study injected a new perspective into a fluid decision arena and should form the basis of future discussions between relevant water decision-making bodies. Similarly, other interviewees pointed out that positions could be changed as a result of this new information.

However, an important question is to what extent the information really will influence decision-making regarding current water supply services and tariff setting. After considering the study findings, practically all the interviewees stated that water tariffs should be re-evaluated. The general consensus was that households with good services, mostly relatively well-off (medium- to high-income) families, should be not subsidized. A majority of interviewees argued that all households should face increased water tariffs, the level being linked to the nature of supply changes and the socio-economic circumstances of households. However, two influential local officials, directly responsible for water tariff policies, held a more conservative position. While one conceded that poorer segments of the city population might indeed be willing to pay extra money for maintaining or improving water supply services, he argued that such increases would be unfair. His argument was that a great proportion of the population is poor and increasing the service price will further deteriorate their living conditions. The other official felt that a market approach was not suitable for defining water supply policy, especially not in a context where many of those affected by such policies have low income levels.

Another challenge mentioned by many of the interviewees was the actual collection of the resources. Four principal concerns emerged among decision-makers:

1. The competence of the government to design a successful programme of payment collection. This competence is related to the authorities' capacity to convince the population of the necessity of increasing the water prices and also their capacity to enforce payment.
2. Scepticism about whether actual WTP would correspond to stated amounts.
3. The perception that the local authorities may not be willing to increase the water prices due to political reasons. One interviewee mentioned that the authorities or politicians have no incentive to increase the water prices as this does not raise votes and might even result in them not getting re-elected.
4. The consideration that a re-evaluation of water-pricing policies would depend on a broader government agenda. Although all sides concede that water policies are in need of reform, one local official said that the solution was not straightforward, because this would require a major effort on the side of the government, which needed to be evaluated in the light of a more integrated government agenda.

Overall, the interviews showed that decision-makers took the information provided by the WTP survey seriously and in general did not question its findings or value as inputs to the decision-making process. However, concerns regarding political pressures and the tariff collection powers of authorities constitute important caveats to the simplistic implementation of such findings.

6. CONCLUSIONS

The problem of the water supply service in Mexico City encompasses mutually reinforcing concerns regarding efficiency, equity and non-sustainability.

The government's central strategy is to improve the administrative invoice system and reduce water loss from pipe leaks, but this seems unlikely to make a difference to areas with current scarcity problems. An integrated approach is required to collect the necessary resources to improve the service in large parts of the low-income areas of the city, and to reduce the consumption levels in areas with good services. The limited available information indicates that the magnitude of the investments required is huge and the available resources are very limited. Increasing the water tariffs for domestic uses appears as the most obvious alternative to finance future investments. Evaluating the allocation of scarce resources with an economic rationality and investigating how much money households would pay for protecting and improving the service is an important aspect to be considered.

In many developing country contexts, including that of Mexico City, the evaluation of water projects has relied on the costs side of the project and the benefits have not been integrated. This chapter has presented a CBA based on primary information about households' WTP for service protection and the cost that the government has estimated as necessary to improve the service conditions in the Federal District.

The inequality issue is a major concern in developing countries. Scarce attention is paid to the wider implication of having large urban areas where population is compounded by highly heterogeneous conditions in terms of service characteristics and socio-economic profiles. This study has revealed the importance of this diversity, not only with regard to the ability to pay, but also the type of benefits that households with different service conditions value most. Thus, it was found that poor households give great value to securing reliable services, while more wealthy households, which tend to enjoy better services already, are willing to pay high amounts to avoid service deterioration.

We propose that the distributional aspects of the WTP results could provide important information within an urban context where there is unequal income and service distribution. By simply using the WTP of different income groups to aggregate the figures, the benefits of a policy change can be estimated more precisely. Integrating these results within a CBA shows that for any of the management scenarios the government would be able to collect the necessary resources for service modernization and, in the extreme, to reduce the subsidies by about 70 per cent. By doing this, the government could still subsidize the service, but primarily to compensate households with lower income levels. Moreover, the revenue would give the government the financial capacity to deal with the service problems at a city-wide level, which is threatening to become a major problem in the coming years.

The discussion about the alternatives to redefine water-pricing policies in urban developing countries contexts is still limited. In some cases, it has been recognized that a certain level of subsidization needs to be maintained (Foster et al., 2000). Defining appropriate water prices is a complex process, because many objectives related to aspects of marginal benefits, revenue to cover the service provision, allocation of costs among different users, and provision of incentives should be considered (Hanemann, 1998). The objective of this research is not to define water prices for Mexico City. However, the WTP information shows the ability and willingness to pay of different groups of households for different changes in the service provision. In large cities, the institutional capacity of the authorities is usually sufficient to allow for the development of more sophisticated tariff structures. What is often lacking is sufficient information as a basis to differentiate tariffs across market segments. Differentiated tariffs, which recognize WTP of households by neighbourhoods or zones or subsidies targeted to specific households, are feasible alternatives. Some of these options have been implemented by different developed and developing countries (Briscoe, 1996; Gomez-Lobo, 2001; OECD, 1987). As has been observed in the past, when tariffs are established with the main objective being that everybody can pay for the service, a great proportion of households actually pay prices considerably lower than their ability and WTP. On the contrary, if the water prices are increased to cover the full price of the service provision, many households could not afford this payment or the burden to their household would be too onerous. The problem, therefore, is defining water tariff structures that charge the real cost of the service provision to the wealthier group of households which are able and willing to pay for it, while providing the service to the rest of the poorer households, recognizing their ability and WTP. This would require a redefinition of subsidy policies to make them more efficient, which would automatically increase the service revenues.

The extent to which the economic assessment information can influence decision-makers perspectives was also addressed. We observed that a number of decision-makers found the study results logical and reasonable given the characteristics of the city, particularly with regards to the WTP of different income groups in the two scenarios. The information seemed to give them a different perspective on the problem, where they could recognize some of the problems of the present service performance and the service pricing system. This gives an indication that decision-makers have had insufficient and/or imprecise information about the consumers' preferences. However, other authors have recognized the limited influence that CBA and WTP information have had in the policy-making arena (Hanley, 2001; Pearce, 1998; UNDP, 1999). The challenge is to make policies that reflect the apparently obvious adjustments required for water tariffs. This is related not only to the fact that decision-makers are exposed to this type of information, but also to other issues such as the establishment of an effective debate between political leaders and administrators, and/or the capacity of the institutions to respond to the information.

Throughout this chapter we have observed how the WTP and CBA information has allowed for the observation of the existence of economic and policy opportunities to give water a more realistic value. Although effective implementation of realistic water tariffs will not be achieved easily, it offers the best prospects for ensuring service maintenance in those areas with relatively good service standards, for improving the service in areas with problems, and hopefully rationalizing the consumption of this regionally scarce yet vital resource.

NOTES

1. Full details of the CV survey and its findings are given in Soto Montes de Oca (2003).
2. A total of 5108 telephone calls were made, from which 2908 are considered eligible respondents. A response rate of 49 per cent was achieved.
3. Further relationships were observed and full results are reported in Soto Montes de Oca et al. (2003).
4. Obviously, current available budgets may not equal the real total economic costs of maintaining and improving the city's water supply services. However, no further detailed information about costs was available. We assume here that the required current and future resources for funding the water supply services reflect real economic costs and are constant in time. The same applies for the estimated aggregate WTP amounts.

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