1. Introduction

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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.

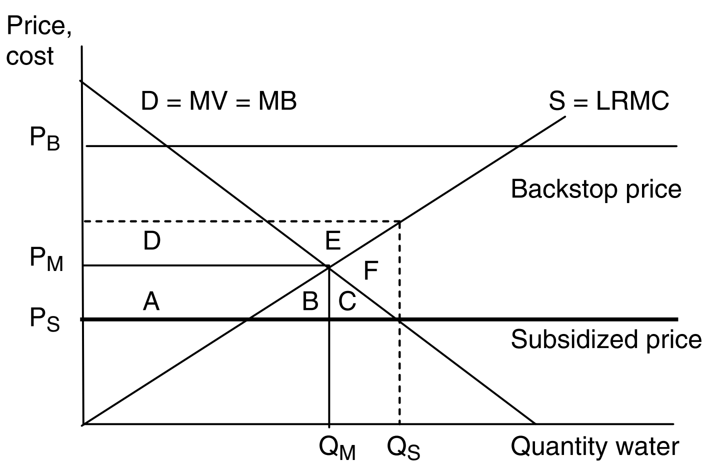

Figure 1.1 Basic case of supply and demand for water

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

<table>
<tr><th></th><th>Irrigation</th><th>Public supply</th></tr>
<tr><td>Africa</td><td>5.1</td><td>1.7</td></tr>
<tr><td>Latin America</td><td>3.1</td><td>5.2</td></tr>
<tr><td>SE Asia</td><td>—</td><td>8.6</td></tr>
<tr><td>W Pacific</td><td>—</td><td>10.9</td></tr>
<tr><td>Asia</td><td>11.4</td><td>—</td></tr>
<tr><td>E Mediterranean</td><td>—</td><td>2.2</td></tr>
<tr><td>Total: non-OECD</td><td>19.9</td><td>28.5</td></tr>
<tr><td>Total: OECD</td><td colspan="2">15.0</td></tr>
</table>

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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