Economic appraisal of environmental regulation

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Scottish Environment Protection Agency 2007

ISBN 10 - 0114973423
ISBN 13 - 978 011497342 1

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Acknowledgements

We owe an enormous debt of gratitude to all the participants and their institutions for their participation in an informative and enjoyable conference we held in Edinburgh during September 2006. The papers contained in this volume are a result of their efforts; thank you one and all. Additionally we would like to acknowledge the contributions of a few other people; Pierre Strosser (ACTeon) is a wise and thoughtful economist who has worked with us in various roles over the last five years. When he agreed to help us in the organisation and management of the Edinburgh conference (with the assistance of Josselin Rouillard), we knew it would be a productive and stimulating event, and so it proved to be. Chapeau Pierre!

Hazel Arnott in our office in Stirling acted as our administrator and fixer in the run up to and throughout the conference, a role she undertook quietly and efficiently as always. Thank you Hazel. Finally, thanks to our colleagues across the Scottish Environment Protection Agency, too numerous to mention, who provide inspiration, challenge and enthusiasm in almost equal measure.

JLR & EW

Introduction

In September 2006 economists at the Scottish Environment Protection Agency organised a workshop in Edinburgh on assessing costs and benefits of regulatory actions. 30 people from 10 countries participated in the workshop and this book is the result. Over those two days in September it became obvious that there were different perspectives on how to deal with the economic issues we face yet many of the preoccupations were definitely common. Sharing this experience more widely appeared like an absolute necessity then. And what better realisation than a collection of papers by the speakers to do so!

This publication is an important tiny step forward. It does not aspire to be at the leading edge of a revolution in the approach environmental economists adopt in valuing regulatory or policy actions. The breadth and range of topics tackled and the diversity of experience of different countries and institutional structures provides a rich overview of what is currently being done to assess costs and benefits of regulatory actions.

The Water Framework Directive provides the focus for a number of the papers. It is probably the single most significant piece of European environmental legislation being implemented at the moment. Different pieces of legislation and different drivers form the basis of the remaining papers in the collection, giving a good overview and an opportunity to compare and contrast approaches across national boundaries and across regulatory frameworks.

The first group of papers concerns the application of costs and benefits assessment, is tackled in three papers, and the geographical scales developed (Scotland, USA and the European Commission) provide an outstanding overview of the subject. Without rehashing the textbook analysis of costs and benefits the three papers give a real insight into the demands of such assessment in practice.

The Controlled Activities Regulation regimes, developed in Scotland, are a novel example, in Europe, of the implementation of the Water Framework Directive. Most member states' implementation of the Water Framework Directive has been through modification of existing legislative systems. In Scotland, a completely new suite of regulation has been developed replacing existing regimes. This process and the considerations behind it are presented by Rebecca Badger and Dougie Johnstone.

This is followed by a fascinating insight into how things work on the other side of the Atlantic Ocean. In many respects the United States is still thought of as the place where the most innovative things happen in the field of environmental regulation. While this may not be universally accepted the overview of issues in US regulatory analyses provided by Richard Morgenstern gives an insight into what might be thought of as the state of the art of empirical analysis of environmental regulation. For those of us in Europe Richard provides a compelling and useful benchmark to keep in mind when looking at what is being done in Europe.

Turning to Europe, Craig Robertson's review of how impact assessments are carried out for European regulation shows how the potential consequences of new or proposed legislation are carefully considered before national implementation is required. Clearly, this is not to say that the system is perfect but shows how decisions made in the Commission are subject to systematic review and analysis prior to implementation. It shows too the continuing process of engagement and involvement being adopted in the Commission that can only lead to greater inclusivity and ultimately better decision- making.

The available concepts, methods and tools to assess environmental costs and benefits can be difficult, even for economists. Part two presents three papers that illustrate these concepts very effectively. The methodologies developed and applied in France to assess the demand for costs and benefits in the Water Framework Directive context is presented by Patrick Chegrani. In his paper Patrick sets out the three stage methodology developed by the French Ministry of Ecology.

A Spanish perspective is to be found in Salvador del Saz-Salazar's analysis of a case study on the economic valuation of use and non-use values of a wetland carried out in the Juncar basin. Salvador sets out for us how the difficult issues of use and non use of environmental assets can be incorporated into useful analysis.

Approaching the subject from a different perspective, Colin Green's paper provides a very interesting account of his reservations about the theoretical (economic) underpinnings of the Water Framework Directive and its implementation. Colin's central contention is that the neo-classical assumption of perfect competitive markets tends to fatally simplify the issues faced by economists. Once this hypothesis is gone, as, he argues, is the case in real-life water economics, analysis suddenly becomes much more complicated and concepts such as prices and cost recovery have to be handled with great care.

Part three introduces the next step of the challenge: to confront policy demand and policy supply. Two papers about how this is carried out at the European level provide an insight into the issues and problems.

The first paper, by Uffe Nielsen, explores how costs and benefits are integrated in European impact assessments and provides an extension to the paper in part one by Craig Robertson on European Regulatory Impact Assessment. The second paper, by Jakub Koniecki, provides an insight into the lessons learnt from the comparison of *ex ante* and *ex post* assessments of regulations which reminds us of the important truth that things do not always turn out the way we expected.

Theory on its own can be quite sterile without some illustrations showing how legislation and implementation work when applied to real life situations. In part four we have two fine examples from the United States and Europe to provide lessons from experience. Firstly, Cynthia Manson presents the lessons learnt from the Hazardous Waste Combustion Standards in the United States in which the consequences of "rulemaking" are dissected and analysed in an approach that adds value to econometric information. Secondly, Paul Watkiss et al. show how costs and benefits have been assessed for the European Air Pollution Policy (CAFE) by adopting an impact pathway approach to assign and manage the flow of data and analysis. Their analysis leads to a set of useful conclusions for others seeking to track the costs and benefits of broad policy areas.

Part five, the last part of the book, is a comprehensive review of the way costs and benefits have been integrated in modelling the Rio Grande, presented by Frank Ward. This analysis at a wide basin scale leaves us with a compelling case for making use of the tools of cost-benefit analysis in exploring environmental regulatory decisions and decision-making.

The papers presented in this volume give an insight into the complex but nevertheless powerful techniques available to analysts in making sense of and getting to grips with the complex problems faced by those making decisions that affect the environment. We hope that the clarity of the examples will be helpful in aiding understanding of these approaches by economists and non-economists alike.

Our view is that environmental protection is, quite rightly, attracting more and more public attention, and policy makers demand that high quality economic analysis play a crucial role in defining environmental priorities and policies, and will drive what will be done in Europe and across the world. In organising our conference we were fortunate in attracting a great diversity of talented and thoughtful people with a shared interest in doing this work. The profiles of the authors, from economists working within environment protection organisations to academics via staff from the European Commission, has proved, for us at least, an outstanding way to get an objective view of what is currently being done and what could, maybe, be done in the near future to improve our understanding of economic consequences of environmental protection legislation. As an added bonus the contributions collected here give us a better understanding of the implications of how economic information is presented to the policy making (non-economist) community that will help make our analysis more immediately useful.

It is important to us that this process does not stop with this publication. As we found, organising a conference with a diversity of participants from different countries to discuss the issues they face is an excellent way to advance our knowledge and improve our practice. Hopefully, this collection of papers gives a flavour of why we feel the September conference was a huge success. Thank you to all of the authors whose contributions we greatly appreciate and to the participants and audience members who challenged and advanced our thinking so very much.

Jean Le Roux and Evan Williams[1]

[1] Scottish Environment Protection Agency (SEPA)

Part One

Assessing the costs and benefits: the policy demand

The Controlled Activities Regulations (CAR)

Rebecca Badger and Dougie Johnstone, Scottish Environment Protection Agency (SEPA)

Abstract

This paper describes the Controlled Activities Regulations (CAR). CAR is a new regulatory regime which has been introduced in Scotland to permit the Scottish Environment Protection Agency (SEPA) to regulate activities that impact upon the water environment in order to implement the EU Water Framework Directive (WFD).

The paper introduces and describes the role and functions of SEPA and the principles according to which it implements its duties as the environmental regulator in Scotland. The challenges faced by an environmental regulator are described in general terms.

The Controlled Activities Regulations are described and the approaches that have been adopted by SEPA to manage the general challenges faced are detailed. The paper examines the economic tests required under CAR and the WFD and describes the approach being taken by SEPA to carrying out these tests.

Introduction

The Scottish Environment Protection Agency (SEPA) is an Executive non-departmental public body which operates at arm's length from the government in Scotland (the Scottish Executive - now known as the Scottish Government).

SEPA was created in 1996 and is the national environment regulator for Scotland, employing 1,100 staff and regulating about 8,000 premises (this number is likely to increase with new regulatory activity under the Controlled Activities Regulations – see below). SEPA has 21 offices which are distributed across the Scottish mainland and islands[2].

The EU Water Framework Directive (WFD)[3] was passed in December 2000. The WFD sets a framework for the management and protection of Europe's water environment. The

2 http://www.sepa.org.uk/

3 Directive 2000/60/EC of the European Parliament and of the Council establishing a frame work for the Community action in the field of water policy, for full text of WFD see: http://ec.europa.eu/environment/water/water framework/index_en.html

Water Environment and Water Services Act (2003) (WEWS)[4], which transposed the WFD into Scots law, made SEPA the competent authority for implementing and ensuring compliance with the WFD. WEWS also introduced new powers for SEPA to control abstractions, engineering works and impoundments in the water environment which had previously been largely unregulated in Scotland.

SEPA vision and principles for regulation

As the national environment regulator SEPA is responsible for regulation in a number of areas. These include waste, radioactive substances and air as well as water. SEPA aspires to be an excellent regulator and has a number of principles according to which it operates its regulatory regimes. An excellent environmental regulator should be able to protect the environment from damage with confidence but at the same time should facilitate and drive innovation by regulated parties while commanding their respect and offering good value for money. A poorly performing environmental regulator, on the other hand, would only react after environmental damage had taken place, would contribute towards the stifling of innovation and waste public and private money.

In practice what this means is that SEPA has a tough job to do. On the one hand, it has to prevent business, industry and private individuals from damaging the environment and the main tool available to SEPA to achieve this is environmental regulation. However, environmental regulation really only kicks in when environmental damage has taken place, at which time prosecutions are possible. SEPA much prefers to be able to work with those it regulates to improve understanding about how their activities impact on the environment and drive them to innovate and therefore minimise adverse environmental consequences. SEPA is also very aware of the potential adverse consequences to which environmental regulation might give rise if it stifles economic development.

So, at the same time as protecting the environment, SEPA has to understand how different businesses operate and try to help them to adapt to incorporate environmental awareness into their day to day operations.

The challenge faced by regulators is eloquently expressed by Malcolm Sparrow (2000) in the following quote:

[4] http://www.opsi.gov.uk/legislation/scotland/acts2003/20030003.htm

"Regulators, under unprecedented pressure, face a range of demands, often contradictory in nature: be less intrusive – but more effective; be kinder and gentler – but don't let the bastards get away with anything; focus your efforts – but be consistent; process things quicker – and be more careful next time; deal with important issues – but do not stray outside your statutory authority; be more responsive to the regulated community – but do not get captured by industry." [5]

SEPA therefore has to take account of business efficiency and effectiveness in the way it implements regulation.

It has to implement regulations in a proportionate manner so that large amounts of effort are not wasted on making small environmental improvements. SEPA has to make decisions that are transparent and accountable to those that are affected by decisions. Furthermore SEPA has to operate in a way that improves the awareness of regulated parties about their environmental impact and encourages them to follow best practices.

An industry survey[6] which was carried out in 2002 showed that SEPA was performing well in terms of its aspiration to be an excellent regulator. Around 80% of those responding to the survey agreed that SEPA staff were professional and understood their industry; that SEPA staff were approachable and able to respond quickly and efficiently to any concerns; and that SEPA kept industry well informed about changes to regulatory practices. Between 60% and 70% of respondents felt that SEPA licence conditions were fair and reasonable, that SEPA offered good value for money and that SEPA communicated in a straightforward manner. Only 56% of respondents, however, felt that SEPA took account of industry views in responses to consultations. So, although SEPA is performing fairly well, there is room for improvement. The introduction of the new Controlled Activities Regulations provided an important opportunity for SEPA to establish, from the outset, a regime for the water environment that would allow many of the required improvements to be made.

The Controlled Activities Regulations – a new regime

The Water Environment and Water Services Act 2003 implemented the WFD in Scotland and provided the enabling legislation for introduction of the Controlled Activities Regulations 2005 (CAR)[7]. These regulations replace those operating under the Control of Pollution Act 1974 (CoPA), which were previously used by SEPA to control point source discharges to the water environment.

5 Malcolm K Sparrow (2000), The Regulatory Craft: controlling risks, solving problems and managing compliance, Brookings Institution, US.
6 http://www.sepa.org.uk/pdf/board/agency/2002/papers/5802.pdf
7 http://www.opsi.gov.uk/legislation/scotland/ssi2005/20050348.htm

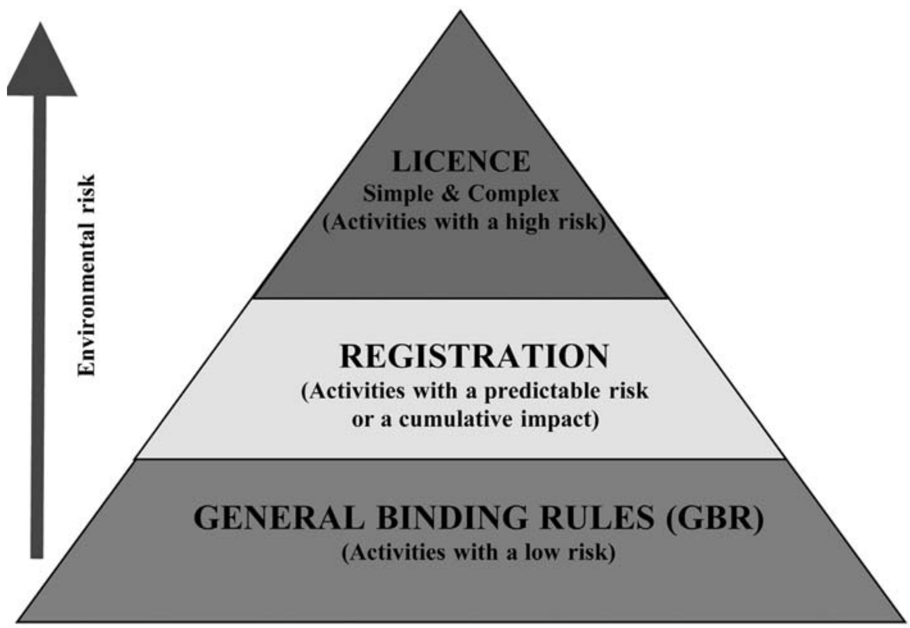
CAR also provides SEPA with new powers to control abstractions and impoundments in the marine and freshwater environment and engineering works in the freshwater environment (engineering works in the marine environment continue to be controlled through another regime operated by another regulator). Prior to the introduction of CAR, abstractions and impoundments in the marine and freshwater environments and engineering works in the freshwater environment were uncontrolled.

CAR came into force on 1 April 2006 although a period of transition meant that some aspects of CAR became operational more quickly than others. Although CAR introduces new controls on activities which were not previously controlled, there are many features of the regime which attempt to make the regulations as sympathetic to the needs of operators as possible. Not least almost all activities are now controlled by the same regulator (SEPA) and the same regulatory regime (CAR) now applies to most activities.

Under CAR, all point source discharges, abstractions, impoundments and engineering works require an authorisation. It is an offence to carry out an activity without such an authorisation. The regulations contain very few exemptions and the definitions of the activities that are covered by the regulations are very wide ranging. So CAR provides extensive powers for SEPA to protect the environment but a key challenge therefore lies in fulfilling these responsibilities in a reasonable and proportionate manner.

A proportionate and risk-based regime

CAR provides three different tiers of regulation, as illustrated in Figure 1 below.
Figure 1 – CAR: a proportionate and risk-based regulatory regime



Three different types of CAR authorisation are possible and the appropriate type is dependent on the likelihood of the controlled activity giving rise to environmental damage. Activities with the lowest risk of environmental damage are controlled through general binding rules (GBRs). SEPA does not require registration by operators of low risk activities and does not impose charges, but is able to prosecute if operators do not comply with the appropriate GBRs. Activities that have a predictable risk of giving rise to environmental damage, or for which cumulative impact might cause damage, are controlled through registrations which impose general activity-based restrictions and for which SEPA only charges an application fee.

Only those activities that have a high risk of giving rise to environmental damage, and for which site specific controls are necessary, require a licence. SEPA imposes application charges for all licences and subsistence charges for certain licences. Simple or complex licences are possible depending on the nature of an operator's impact on the water environment.

Under the previous CoPA regime, licences were routinely required for all SEPA authorisations. Although under CAR many more activities in the water environment are controlled, the nature of the control is proportionate to the risk of the activity causing damage to the environment. CAR is also a one-stop regime for controlling almost all activities in the marine and freshwater environment and it is operated by one regulator. These features will result in the consistent application of the regime across different types of activity, operator and water environment.

<u>Recognising the social and economic importance of the water environment</u>

Two of the key default objectives of the WFD are to achieve good ecological status in all water bodies by 2015 and not to allow any deterioration in the status of water bodies. The WFD does however allow for a balance between protecting the water environment and securing the sustainable use of this resource for the purposes of economic and social development to be achieved in setting the environmental objectives for water bodies. Several exemptions from the default objectives of the WFD and CAR are therefore allowed for within the legislation.

Two exemptions from the default objectives of the WFD and CAR are informed by economic information, as follows:

- New activities which cause a deterioration in the status of the water environment are allowed if:
  a. All practicable mitigation measures are taken to minimise the impact on the water environment;
  b. There are no significantly less environmentally damaging ways of providing the benefits of the new activity; and
  c. The benefits of the new activity to public health, public safety or sustainable development outweigh the costs of the new activity to the water environment.

- Alternative objectives to the achievement of good ecological status by 2015 (e.g.: achievement of good ecological status by 2021 or 2027 or alternative environmental objective to good ecological status) are allowed if meeting this target is deemed to be :
  a. Technically infeasible or
  b. Disproportionately costly.

Part c. of the first of the above tests and part b. of the second of the above tests both essentially involve a cost benefit assessment. For new activities this involves an assessment of whether the benefits of a new activity outweigh the costs of the new activity. For existing activities which need to reduce their impact on the water environment, the assessment is whether the benefits of achieving good ecological status by 2015 are proportionate to the costs.

<u>The SEPA approach to cost benefit tests in CAR</u>

The usual approach taken by environmental economists to cost-benefit assessment is to translate all impacts to monetary terms and then to take the costs (negative impacts) away from the benefits (positive impacts) to provide a net cost benefit figure. If the net cost benefit figure is positive then a course of action is deemed to be economically efficient and should be implemented whereas a negative cost-benefit figure would indicate economic inefficiency and that a course of action should be avoided.

The main difficulty with this approach is that, because of market failure in the provision of environmental goods, many of the impacts resulting from environmental regulation have no recognised monetary value. Although techniques do exist to assign monetary values to all impacts, their implementation is time consuming and, in many cases, may not result in an improved basis for decisions to be made.

SEPA is likely to have to make a large number of exemption decisions in its implementation of CAR and employs only 1.5 (full-time equivalent) economists to work on WFD and CAR issues. It would therefore be unrealistic, impractical and disproportionate to expect full monetisation and full economic cost-benefit assessment to be carried out for every exemption decision. SEPA has therefore developed a framework which can be used by non-economists to describe the impacts of a course of action so that a judgement can be made (also by non-economists) as to whether the action is justifiable or not.

The framework involves describing any significant impacts that a particular course of action might have. The impact areas that might be relevant to a particular decision were based on those used in environmental and social impact assessments and are listed in Table 1.

Most decisions will not need impacts in all of the areas listed to be taken into account. Decisions should be based on a proportionate amount of evidence about impacts and more information should never be collected than is necessary for a reasonable judgement to be made.

<u>Table 1: Impacts to be taken into account in CAR exemption decisions</u>

Economic	**Social**	**Environmental**
Direct – on economy or on operator	Health	Water environment
Indirect – third party businesses	Safety	Biodiversity
	Recreation	Landscape
	Nuisance	Energy/climate change
	Vulnerable/disadvantaged groups	Built heritage
		Earth heritage
		Waste and resource use

SEPA has developed detailed guidance about how the significance of impacts in each of the areas listed in Table 1 should be described and how decisions about weighing up impacts should be made.

It is therefore anticipated that the vast majority of decisions will be possible without monetisation of impacts being required. If a decision cannot be made on the basis of objective descriptions of impacts in qualitative and quantitative terms, then guidance will be made available by SEPA about how monetisation should be carried out.

The above framework for exemption decision making will be familiar to non-economists, it will provide a transparent justification for the decisions which are taken, and it is proportionate in that more information than is necessary to make a reasonable judgement should not be required.

Conclusions

This paper has described the main functions of the environment regulator in Scotland and the principles according to which SEPA aspires to fulfil these functions. These principles include a desire to protect the environment while commanding the respect of those that are regulated and not stifling innovation or economic development. SEPA's approach to implementation of these principles is investigated in more detail in the context of general implementation of the Controlled Activities Regulations and specifically the approach to economic assessments within this regime. The paper shows that the Controlled Activities Regulations offer a proportionate and risk-based approach to environmental regulation. The economic tests contained within the regulations are described in more detail to illustrate how apparently complex tests can and need to be carried out in a practical manner to produce outcomes that are transparent and accountable to all parties that are interested in them or affected by them.

Recent Issues in U.S. Regulatory Analyses

Richard D. Morgenstern

Introduction

In the United States official interest in subjecting economic analyses of new environmental, occupational, and related regulations to formal economic assessments can be traced back at least three decades. As former chief of the U.S. Office of Management and Budget (OMB) Charles Schultze noted the Office of Management and Budget (OMB) became "the lobby for economic efficiency" as a result of "Quality of Life Reviews," "Inflation Alerts" and other regulatory requirements instituted during the 1970s.[8] Subsequent Executive Orders by Presidents Reagan and Clinton significantly increased the emphasis on economic efficiency by mandating that cost-benefit analysis (CBA) be conducted as part of a required Regulatory Impact Analysis (RIA) on major federal regulations, defined as those with expected annual costs and/or benefits of $100 million or more. Currently, RIAs are routinely performed by federal agencies and usually play an important role in the regulatory process.

Despite the general recognition of the benefits of some form of pre-regulatory analysis, criticism abounds on many fronts. One particular area of contention concerns the accuracy of the RIAs themselves. While any single RIA may miss the mark, the claim is sometimes made that systematic biases are at work, with some people believing that costs are routinely overestimated, and others claiming that underestimates are the norm. Similar disagreements are also heard about the accuracy of the benefits. While finding bias in the cost estimates from industry (or environmental) sources is perhaps to be expected, the existence of systematic errors in cost estimates prepared by the regulatory agency itself has potentially significant implications for resource allocation. If costs are regularly overestimated, thereby making potential new regulations appear more costly, rulemakings would generally favor the selection of less stringent emission control options (and, conversely, if costs are consistently underestimated). Large discrepancies could lead not only to bad decisions, but would misrepresent the true burden of regulation on society and undermine public confidence in the regulatory process.

The most direct way to study the accuracy of pre-regulatory estimates is to compare them to the actual outcomes once the regulation is implemented. Unfortunately, *ex*

[8] Charles L. Schultze, The Public Use of Private Interest, Washington, D.C. the Brookings Institution, 1977.

post analyses of the costs (and other outcomes) of regulations are surprisingly uncommon, especially in comparison to the large number of *ex ante* studies. The present paper reviews the recent literature on the accuracy of cost estimates prepared for RIAs with an eye to identifying broad conclusions about the nature and direction of any biases. Portions of the paper are drawn from earlier work I did with two colleagues, Winston Harrington and Peter Nelson (Harrington, et al., 2000). Following this introduction, section II provides background on the regulatory review process in the U.S. Section III considers some conceptual issues relevant to the *ex ante*-*ex post* comparisons. Section IV examines a number of retrospective studies that have attempted to compare *ex ante* estimates of costs (and benefits) with the outcomes observed after the regulations were implemented, including the extent of any bias as well as possible explanations for the findings. Section V draws some broad conclusions from these reviews and offers directions for future research.

Background

CBA is a technique intended to improve the quality of public policy decisions, using as a metric a monetary measure of the aggregate change in individual well-being resulting from a policy decision. Individual welfare is assumed to depend on the satisfaction of individual preferences, and monetary measures of welfare change are derived by observing how much individuals are willing to pay or give up in terms of other consumption opportunities. This approach can be applied to non-market "public goods" like environmental quality or environmental risk reduction as well as to market goods and services, although the measurement of non-market values is more challenging. When measurement of such non-market values is not possible, analysts may resort to cost-effectiveness analysis (CEA), a less ambitious approach in which a policy outcome (e.g., a specified reduction in ambient pollution concentration) is taken as given and the analysis seeks to identify the least-cost means for achieving the goal, taking into account any ancillary benefits of alternative actions. Every CBA has at least one CEA buried inside.
To its adherents, the advantages of CBA (and CEA) include:

- transparency and the resulting potential for engendering accountability;
- the provision of a framework for consistent data collection and identification of gaps and uncertainty in knowledge;
- the development of metrics for both the beneficial and adverse consequences of alternative regulatory approaches, allowing those alternatives to be compared to one another (CEA); and

- with the use of a monetary metric, the ability to aggregate dissimilar effects (such as those on health, visibility, and crops) into one measure of net benefits.

Most economists would acknowledge that CBA does not incorporate all factors that can and should influence judgements on the social worth of a policy and that individual preference satisfaction is not the only criterion. Nevertheless, most would also argue that rigorous CBA can elucidate for a broader audience how various regulatory choices are expected to work and who is likely to be affected. At a minimum, it is widely believed that CBA can play a useful informational role in the decision-making process.

From an economist's perspective, the conduct of CBA is primarily limited by measurement problems—how choices based on preferences permit one to infer economic values in practice. The state of the science of measuring such economic values is quite active. Estimates of the willingness to pay for reductions in mortality and morbidity risks, for avoiding environmental damage to recreational opportunities, and for avoiding visibility degradation are the subject of much ongoing research. Issues of a higher order stalk the estimation of non-use values, and a variety of mostly empirical concerns have left material damage poorly understood. Estimation of the costs of reducing environmental damages, often thought to be relatively straightforward, can be as challenging as estimation of the benefits.

Each year the federal government issues thousands of final regulations that are said to impose large costs and generate even larger benefits on the various actors in the U.S. economy. Most of these effects are concentrated in a small number of regulations, making it both possible and sensible to subject such rules to rigorous analysis. Since the issuance of the Regan Executive Order (E.O. 12291) in 1981, it has been a requirement of the rulemaking process that so-called major regulations -- those with 9-digit estimated annual benefits or costs -- must undergo an RIA. A subsequent executive order by President Clinton (E.O. 12866) made only modest changes and left in place the key components of regulatory benefit and cost estimation as well as OMB review.[9] In the decade prior to 2005, across the entire U.S. government, there was an average of fewer than 10 social regulations per year for which complete RIAs were prepared. Almost half of those RIAs involved rules developed by the U.S. EPA.

The RIAs in which the economic analyses are embedded often undergo considerable changes before being finalized as a result of critical scrutiny by agency and OMB staff, and by commentary from stakeholders. Nearly 30 years of experience has led to an informal list of "best practices" for RIAs. According to Hahn et al. 2000, these include:

[9] In March 2007 President Bush issued E. O. 13422, which expands OMB's jurisdiction to include review of guidance documents issued by federal Agencies. It also requires that regulatory Agencies provide a written rationale for new regulations; and to provide estimates of aggregate annual costs and benefits of all regulatory activities in the Agencies' plan. However, none of these changes affect the basic analytical requirements for CBA contained in E.O. 12866.

- the use of clear and consistent baseline assumptions;
- the evaluation of an appropriately broad range of policy options, including alternatives to new regulation;
- transparency in the use of assumptions, data and models, the comparison of alternatives, and the reporting of results;
- appropriate treatment of discounting future benefits and costs and accounting for the cost of risk-bearing;
- the use of probabilistic analyses and other methods to explore the robustness of conclusions;
- the identification of non-monetary or non-quantifiable aspects of a policy and the potential incidence of all effects; and
- the use of benefit and cost measures that are grounded in economic theory (measures of willingness to pay and opportunity cost).

One result of the growing economic sophistication of RIAs over the past several decades is that they have increased in length and become more technically oriented documents, leading perhaps to a certain sacrifice in transparency. Interestingly, one of the earliest examples of a rigorous and quite transparent RIA by EPA involves the use of lead in gasoline as an octane booster. In March 1985, following publication of the RIA, the agency promulgated a rule to slash the use of lead in gasoline by more than 90 percent. The rule was supported by a quite detailed CBA which indicated that lowering lead levels in gasoline would measurably reduce a number of health and welfare effects associated with the addition of lead to the gasoline pool.

The health effects were clearest for children but new evidence suggested that adults might reap even larger benefits related to reduced blood pressure levels and the associated cardiovascular effects. The rule was also predicted to reduce "mis-fueling" of certain vehicles and increase the fuel efficiency of the auto fleet. Overall, the monetized value of the benefits exceeded costs by a factor of three-to-one even without consideration of the potential gains for adults. When the adult gains were included, the benefits exceeded the costs by more than ten-to-one. As chronicled by Nichols (1997), the rigor and transparency of the RIA were critical to the adoption – more than two decades ago -of this innovative and, arguably, quite stringent regulation to slash the use of lead in gasoline.[10]

[10] As noted by Nichols (1997), the EPA's heavy reliance on the CBA may have been coloured, in part, by the desire of some senior managers to demonstrate that such analyses could be used to support as well as critique proposed regulatory actions.

Analysis versus decision-making

In thinking about the conduct of RIAs, it is useful to consider in detail the role these studies play in actual decision-making. In that context, it is important to distinguish between the analytic and decision-making components of rulemaking. While the two are closely related, they are not one and the same. In fact, the differences between the two components were clarified in E.O. 12866. Specifically, this Order replaced the stipulation contained in E.O. 12291 that benefits "outweigh" costs with a requirement for "a reasoned determination that the benefits…justify the costs." Further, agencies were mandated to "include both quantifiable measures…and qualitative measures of costs and benefits" and "to select those approaches that maximize net benefits (including potential economic, environmental, public health and safety, and other advantages; distributive impacts; and equity) unless a statute requires another regulatory approach."[11] In effect, E.O. 12866 embraces social welfare considerations that may not be easily quantified, such as public health and distributional impacts, and rejects the idea that quantified cost-benefit analysis provides a rigid criterion for decision-making.

Thus, E.O. 12866 is consistent with the views of those economists who see cost-benefits analysis as a "tool" rather than a strict rule for decision-making. As Nobel Laureate Kenneth Arrow and others have written,

…[In] many cases, benefit-cost analysis cannot be used to prove that the economic benefits of a decision will exceed or fall short of the costs…. [But it] can provide illuminating evidence for a decision, even if precision cannot be achieved because of limitations on time, resources, or the availability of information. (Arrow et al. 1996, 5)

Arrow et al. also note that agencies may want to consider other factors in their decisions, such as equity within and across generations, or they may want to place greater weight on particular characteristics of a decision, such as irreversible consequences. They recommend that when the expected cost of regulations far exceeds the expected benefits, agency heads should be required to present a clear explanation justifying the reasons for their decisions.

Some critics are concerned that even this attenuated process places too much emphasis on CBA and, more generally, on economic efficiency in the decision-making process. At the same time, other critics focus almost exclusively on the importance of quantifying

[11] E.O. 12866 1(a), 3 CFR at 638-39 (1995).

costs and benefits, and argue for a best option that, they believe, can maximize overall social wellbeing.[12] Although these critiques continue to have traction in some circles, it is fair to say that the view articulated by Arrow et al. (and embodied in E.O. 12866) is consistent with the current mainstream thinking in the U.S.

Regardless of whether the focus is on rigorous CBA or not, the true influence of RIAs on regulatory outcomes is not fully understood, even among experts. Indeed, it may well be that the regulatory processes would have reached a similar outcome if no CBA had ever been prepared. However, the effect of a CBA on the regulatory outcome is not the only, and may not be its most important, influence on the regulatory process. Twenty years ago an EPA report (USEPA 1987) listed four specific areas—besides supporting regulatory decisions—where the RIA influenced development of regulations:

- guiding the development of the regulation;
- adding new alternatives;
- eliminating non-cost-effective alternatives; and
- adjusting alternatives to account for differences between industries or industry segments.

The RIA requirement also has been credited with making upper management at regulatory agencies – as well as the general public - more aware of the implications of new regulations.

Conceptual issues in making *ex ante ex post* assessments of regulatory costs

Reflecting increasing concerns about the accuracy of cost estimates of environmental and occupational safety regulations, the OMB observed that "industry representatives and think tanks assert…that [government] estimates understate costs…while public interest groups and Federal agencies generally assert…that [government] estimates overstate costs" (OMB 1998). Beyond such anecdotal statements, however, there is a paucity of evidence on the overall accuracy of the cost information that is generated by and available to regulators.

Interestingly, there are seeming ideological differences in the types of evidence used to address these questions. Those who believe costs are underestimated often refer to the costs of an entire program or legislative initiative. Superfund is the poster child for this view. Critics argue that the program, originally designed to clean up Love Canal and a few other large sites, expanded its scope and became a "behemoth, towering over American environmental policy" (Cairncross 1993). Other critics have focused on the

12 Miller (1989).

discrepancy between the initial, often grandiose objectives of U.S. environmental laws, e.g., the Clean Air Act (1970), and the more modest progress toward meeting those objectives.[13]

Another argument made by those who believe costs are understated is that the *ex ante* estimates leave out some important cost categories, e.g., regulatory-induced job losses, claims on management attention, discouraged investment, and retarded innovation. In contrast, those who believe costs are overestimated prefer to look at the direct costs of complying with specific regulations. The most often cited example involves reductions of sulfur dioxide emissions mandated under the Clean Air Act Amendments (1990). In that case, the huge discrepancy between the early cost estimates and recent allowance prices is taken as evidence of a problem of systematic overestimates (e.g., Browner 1997).

The most direct way of assessing systematic errors in regulatory cost estimates is to compare *ex ante* estimates with actual costs, determined *ex post*. Unfortunately, *ex post* studies of the costs of regulation are quite scarce. Rulemaking agencies generally lack both a legislative mandate and a bureaucratic incentive to perform such analyses. In fact, the conduct of *ex post* studies may detract from an agency's mission by using limited resources and by generating outcomes that may prove embarrassing. Not surprisingly, most detailed *ex post* studies have been carried out by independent researchers.

Defining regulatory cost estimates

Although the notion of "regulatory cost estimation" may appear straightforward, in practice it is anything but. The hard part is to identify just what it is that ought to be measured. Making *ex ante*-*ex post* cost comparisons involves more than just determining what is spent; care is also required to ensure comparability in what is being purchased. To shed light on the conceptual issues it is useful to dig for deeper meaning of the terms "cost" and "estimates".

Cost: To determine the cost (or benefits) of a regulation, one must compare conditions in a world with the regulation to conditions in a world without it. To produce *ex ante* estimates, both the "with" and the "without" scenarios must be modeled; they cannot be observed. For the *ex post* calculation, the world with the regulation is observed, but the counterfactual is not. To produce an *ex post* estimate, one must determine the

[13] The National Ambient Air Quality Standards, for example, were originally thought to be achievable within a decade. Yet, even today we still are still unsure how, when, or even if the original goals will be met.

actual outcome empirically, and compare it to a hypothetical baseline with the status quo ante. The definition of baselines is thus somewhat arbitrary, depending as it does on the analysts' beliefs on what would have happened without the regulation. Thus, regulatory cost estimates can hardly escape being to some degree hypothetical whether they are made *ex post* or *ex ante*.

To an economist, the cost of a good or service is generally defined as the maximum value of the opportunities foregone in obtaining that good or service.[14] Table 1 reproduces with minor alterations taxonomy of the costs of environmental regulation developed by Jaffe et al. (1995) moving from the most to the least obvious. Harrington et al. (2000) added a column to the right, indicating whether costs in each category are typically part of *ex ante* estimates developed by regulatory agencies. The list is topped by the capital and operating expenditures associated with regulatory compliance. Such activities are typically carried out and paid for by the private sector, although some activities fall on state and local governments (e.g., drinking water) and some on the federal government (e.g., compliance expenditures of TVA and Bonneville Power Administration). These capital and operating costs are routinely considered in regulatory cost analyses.

A few other cost categories are occasionally addressed in the *ex ante* analyses. Some of these categories are shown under "other direct costs." They are particularly noticeable in analyses of automobile regulations, and they sometimes show up as negative costs. Thus, an important element in the estimates of the cost of standards for new motor vehicles is the improved fuel economy and reduced maintenance requirements attributable to the introduction of computerized fuel injection, a technology that provides many engine benefits besides lower emissions (USEPA, 1993). Of course, not all the "other" costs considered in RIAs are negative. For example, the I/M cost analysis counts the cost of motorists waiting in queues at testing stations. Also, adverse effects of regulation on workers have occasionally appeared in RIAs.

In contrast, the other categories in the cost taxonomy, including government administration of environmental statutes and regulations, some of the other direct costs, general equilibrium effects, and transition costs are not generally considered in regulatory cost estimates. For one thing, often it only makes sense to speak of these costs with respect to regulation in the aggregate rather than for specific regulations. The cost of administration of environmental statutes is usually omitted because of a joint

14 More precisely, the cost of a regulation is equal to "the change in consumer and producer surpluses associated with the regulation and with any price and/or income changes that may result."(Cropper and Oates 1992, page 721)

cost allocation problem; besides, the government's costs are thought to be small relative to those of the private sector. For individual regulations focused on a single sector - and thus not involving many spillover impacts from one sector to another - one can say on a priori grounds that general equilibrium effects are likely to be de minimis. The principal reason other costs are excluded is the lack of credible information or insufficient analytical resources to apply whatever data or models do exist. Thus, additional management resources or disrupted production is plausibly important, but no *ex ante* estimates have been prepared.[15]

[15] There have been some attempts to measure these costs *ex post*, at least indirectly, such as Gray and Shadbegian (1993) and Joshi et al. (1997) in the steel industry, and Morgenstern et al. (1998a) for a set of 11 industries. These studies estimate cost functions to examine the effect of reported abatement expenditures (as measured by PACE) on total cost. The other direct costs are positive if and only if the coefficient on the pollutant abatement expenditure variable is positive. While the Joshi et al. (1997) study finds multipliers up to 12, Morgenstern et al. (1998) estimated the multiplier to be 0.8, suggesting that the other direct costs are more than offset by savings elsewhere in the production process. This may indicate the joint cost aspect of some environmental spending. Of course, this analysis can only be done for fairly large aggregates of regulations, for that is the only way the *ex post* compliance expenditure data are reported.

Table 1:
A Taxonomy of Costs of Environmental Regulation

Cost category	Counted in RIA?
DIRECT COSTS	
Private Sector Compliance Expenditures	
Capital	Yes
Operating and maintenance	Yes
Public Sector Compliance Expenditures	
Capital	Yes
Operating and maintenance	Yes
Government Administration of Environmental Statutes and Regulations	
Monitoring	Rarely
Enforcement	Rarely
Other Direct Costs (including negative costs)	
Legal and Other Transactional	Sometimes
Shifted Management Focus	No
Disrupted Production	No
Waiting time	Sometimes
Intermedia pollutant effects	Sometimes
Other natural resource effects	Sometimes
Changes in maintenance requirements of other equipment	Sometimes
Worker Health	Sometimes
Stimulation of innovation in clean technologies	No
INDIRECT COSTS	
General Equilibrium Effects	
Product Substitution	No
Discouraged Investment	No
Retarded Innovation	No
Transition Costs	
Unemployment	Sometimes
Plant closures	Sometimes

Source: Harrington et al., as adapted from Jaffe et al. (1995).

Estimates: In evaluating the quality and usefulness of a regulatory cost estimate, it is important to keep in mind who is making the estimate and what its purpose is. Before a regulation is adopted, information about response options and costs may be asymmetrically distributed; potentially regulated parties generally have better information about alternatives for meeting requirements than regulatory agencies and advocacy groups. At the same time, however, industry cost estimates may be too high if firms do not fully anticipate cost-saving measures they may discover once resources are directed to the task of compliance.[16]

Part of the difficulty of making cost comparisons is that actual outcomes can deviate from predicted ones in so many ways that it is not easy to know what is comparable. For example, errors could arise in estimating per plant regulatory emissions costs or, alternatively, in calculating the emission reductions likely to be achieved. The expected emission reductions, in turn, depend on knowledge about the per plant reductions and the number of plants actually in operation. Errors in any of these factors could bias the outcomes. Depending on one's evaluation criterion, this could lead to mis-estimation of any of the relevant outcomes: the quantity of emission reductions achieved, unit pollution reduction costs, and/or total costs.

A word about economic incentives: The foregoing discussion was written primarily with traditional command and control regulation in mind. Increasingly, though, modern environmental regulation makes use of economic incentive approaches.

Ex ante estimation of outcomes is just as important for economic incentives, but there are some differences in the uncertainties encountered. Although economic incentives can take a myriad of forms, the focus here is on the pure quantity and price instruments, i.e. marketable emission permits and emission fees.

In a marketable emission permit system, what is specified beforehand is the aggregate emission reduction; plant-specific emission reductions are uncertain.[17] The costs are uncertain *ex ante*, both at the margin and in total, and market simulation models are used to estimate *ex ante* costs. *Ex post*, we observe a market-clearing permit price, which can be taken, with some qualifications, as the marginal cost of abatement.

For an emission fee, the marginal cost – that is, the plant-level abatement cost – is specified *ex ante*, so there is very little uncertainty about what the marginal cost will

[16] The hypothesis that environmental regulation triggers innovation that can offset some or all environmental compliance costs was initially proposed by Porter (1991) and supported by Porter and van der Linde (1995). For a counter-view see Jaffe et al. (1995) and Palmer et al. (1995).

[17] This is in direct contrast to the typical CAC regulation, where one fixes beforehand the emission reductions required from each plant, but the total emission reductions and the marginal cost of the regulation may not be known with certainty.

be. The uncertainty is in the slope of the marginal abatement cost curve. Thus the *ex ante* estimate that is of most interest is the quantity of emission reductions. If demand is more responsive than predicted, then a given fee will result in more emission reductions than expected. The total cost under this assumption will be greater or less than expected, depending on one's assumptions about the shape of the demand curve. For example, assuming linear demand, total cost will be greater than anticipated and average cost will be as expected. Despite the higher total cost, most observers would regard this case as a pleasant surprise, for the higher total cost is more than offset by the larger-than-expected emission reductions.

Accuracy: In order to make these comparisons, of course, a criterion is needed to define when the *ex ante* estimate of total cost (or unit cost or quantity) is accurate or is an under- or overestimate. It is tempting to equate the *ex ante* and *ex post* estimates with "forecasts" and "actuals," but that terminology overlooks the fact that the knowledge of the *ex post* situation is decidedly imperfect and in some ways, perhaps, little better than the knowledge of the situation *ex ante*.

As noted, the quality of the *ex ante* cost estimate is limited by three basic uncertainties: what are firms currently doing, what firms will do in response to the regulation (and what it will cost), and what firms would have done without the regulation (and what that would have cost).

The first of these items is in principle knowable *ex ante* but in practice is usually not known very well. The second and third items are hypothetical, based on economic and process-analysis models, discussions with industry experts and perhaps analogies from other industries.

The *ex post* cost estimate must deal with the same uncertain elements but from a more favorable position. It can be no worse than the *ex ante* estimate, because it has more information to draw on.[18] In addition, the very process of implementation and enforcement generates a great deal of information, not only about the responses of firms to the regulation but about the situation prior to implementation. This means that the *ex post* estimate will in all likelihood be much closer to the "truth." Thus, even though there is a lack of precision in the *ex post* estimate, one can be reasonably sure that it is more precise than the *ex ante* estimate. Thus it is plausible to use the former to judge the quality of the latter.

18 There remains the possibility of bias in the *ex post* study, but as almost all the case studies examined by Harrington et al. (2000) were prepared by academic experts without an interest in the outcome, that possibility is minimized.

Findings

To date, three sets of estimates have been developed using the approach outlined in the previous section. The first set of estimates, developed in 1999, is based on a literature search conducted by myself and two colleagues on *ex ante* and *ex post* cost comparisons in the field of environmental and occupational regulation (Harrington et al. 2000).[19] The second set of estimates, developed in 2005 by the U.S. Office of Management and Budget, updated our original work by adding more recent studies and expanding the coverage to regulations not focused solely on environment, and occupational health and safety (OMB, 2005). In their analysis, OMB also attempted to consider more fully the accuracy of the benefits calculations. The third set of estimates is based on a review of the OMB analysis developed in 2006 by Harrington with a particular focus on the selection criterion used to create the OMB sample (Harrington, 2006).

In our original study we asked colleagues for help in locating *ex ante*-*ex post* comparisons in both the peer-reviewed and the grey literatures. We received numerous replies to the effect that "plenty of studies are out there; you shouldn't have any trouble." In the end, we were able to identify only two dozen studies that were genuine before-and-after comparisons of regulatory costs: nine of the rules were issued by the EPA, seven were issued by the U.S. Occupational Safety and Health Administration (OSHA), and seven were based on actions by the state of California or by foreign governments.

Not surprisingly, the most important constraint on case selection was the availability of an *ex post* study of regulatory costs. While it is not possible to know for sure, we do not believe that the selection criterion introduced any particular bias to our cases.

Problems of comparability among the different *ex post* analyses prevented us from performing a strictly quantitative analysis. Accordingly, we developed a qualitative approach. We labeled an *ex ante* analysis as "accurate" if the *ex post* estimated costs fall within the error bounds of the *ex ante* analysis and if they are between 25 percent higher and 25 percent lower than the *ex ante* point estimate. Three outcomes are compared: the quantity of emission reductions achieved, unit pollution reduction costs, and total costs. The quantity of emission reductions achieved reflects the net effect of the relevant quantity-related factors, i.e., the number of firms or agents subject to regulation, and the estimated emission rates with and without regulation.

19 Harrington, Morgenstern, Nelson. 2000.

Unit pollution reduction cost outcomes generally refer to costs per unit of emissions reduced (over the relevant range), although other margins can be important in individual cases.[20] Perusal of Table 2, which summarises the results for the individual rules, reveals a consistent tendency across all subcategories – EPA, OSHA, state and foreign - to overestimate both total costs and pollution reductions.

Table 2: Case Study Results

	Accurate	Overestimate	Underestimate	Unable to Determine
All Regulations				
Quantity Reduction	9	9	4	2
Unit Cost	7	12	5	0
Total Cost	5	12	2	5
Federal Regulations				
Quantity Reduction	7	9	1	0
Unit Cost	6	6	5	0
Total Cost	4	10	2	1
EPA Regulations				
Quantity Reduction	4	4	1	0
Unit Cost	3	3	3	0
Total Cost	3	4	1	1
OSHA Regulations				
Quantity Reduction	3	5	0	0
Unit Cost	3	3	2	0
Total Cost	1	6	1	0
State and International Regulations				
Quantity Reduction	2	0	3	2
Unit Cost	1	6	0	0
Total Cost	1	2	0	4
Regulations using Economic Incentives				
Quantity Reduction	3	1	4	0
Unit Cost	1	7	0	0
Total Cost	2	4	0	2

Source: Harrington et al. 2000.

[20] In pesticide regulation, for example, the relevant margin is costs per acre. For the inspection and maintenance (I/M) rule, costs can be usefully expressed both as costs per unit of emissions or costs per vehicle.

Overall, pollution reductions were overestimated in nine of the *ex ante* analyses examined and underestimated in four of them. In nine cases, the quantity predictions were judged to be about right. The per unit costs of regulations were even more likely to be overestimated; in twelve cases per unit costs were overestimated, while they were underestimated in five cases. Total costs were overestimated for twelve rules and underestimated in just two cases. Similar patterns are evident when the results for the different agencies are examined separately.

In contrast, when the focus is on per unit costs the outcome is quite different. For rules promulgated by either EPA or OSHA there is no clear evidence of mis-estimation of per unit costs. Cost overestimates are evident for state and international rules, as well as for those using economic incentives although, as noted, there is considerable overlap between these two categories.

Of the 24 rules in the sample, seven involved economic incentives, primarily in the form of emissions trading. It is reasonable to ask, therefore, how the accuracy of the cost estimates of these rules compared to the accuracy of the other estimates. Interestingly, in all seven cases the agencies overestimated actual costs.

In its 2005 Report to Congress on the Benefits and Costs of Federal Regulations OMB updated and extended our earlier work (OMB, 2005). Their sample includes a total of 47 cases, almost twice as many as included in our earlier study. Some of the additional regulations are from agencies we did not consider, including from the National Highway Traffic Safety Administration, and the Nuclear Regulatory Commission. In addition, OMB included several studies related to EPA or OSHA rules that appeared since the completion of our paper. An issue with the OMB analysis is their selection of a particular subset of regulations to include in their analysis. As it turns out, their choice of rules has important implications for their findings.

The principal criteria used by OMB to score the regulations are total benefits, total costs, and the benefit-cost ratio. Where they are calculated, the benefits are defined as the monetized effects of the regulation if applicable; if the benefits are not monetized, then they are the quantitative effects of the regulation. OMB's bottom line is the change in benefit-cost ratio. If it is overestimated, that means the performance of the regulation is not as good as was predicted in the *ex ante* study.[21]

[21] For some rules in the OMB sample, either benefits or costs are not quantified at all. For these situations it was assumed that the missing estimates (costs or benefits) were accurate for the purposes of calculating a change in benefit-cost ratio.

The key finding of the OMB study is that both benefits and costs are more likely to be overestimated than underestimated, but the benefits overestimates are larger than the cost overestimates. As a result, they claim the benefit-cost ratio is overestimated far more often than it is underestimated, i.e. the predicted performance of the regulation is better than its actual performance. By comparison, Harrington et al. also find that both the total costs and total effects of regulation are overestimated, but we found no apparent bias in the estimate of unit costs or cost-effectiveness.

A recent paper by Harrington reviewed the new OMB analysis in some detail and finds that their conclusions are strongly influenced by the highly selective sample they chose to examine (Harrington, 2006). Specifically, Harrington reviewed the regulations in the OMB list with an emphasis on the regulations they excluded from the sample. The excluded regulations involve several (but not all) appliance standards developed by the U.S. Department of Energy, as well as a select number of fuel, vehicle emission, and pesticide regulations developed by EPA. Harrington conducts a quite detailed assessment of the individual cases and finds no obvious explanation why OMB excluded these particular regulations from their study. In his analysis, Harrington adds back in the excluded study to develop new aggregate estimates.

Overall, Harrington demonstrates quite convincingly that by using a more complete sample he can reproduce the original results obtained by Harrington et al. Using the results from 60 case studies, including 26 not included in the OMB estimate, he finds no bias in estimates of benefit-cost ratios. There are slightly more underestimates of the benefit-cost ratio in the entire sample, and slightly more overestimates in the sample with pesticide regulations excluded. This conclusion – that estimated net benefits are reasonably accurate - is at odds with the OMB conclusion, which finds a surplus of overestimates. Harrington's calculations, along with those developed by OMB, are shown in Table 3.

Table 3. Benefit-Cost Ratios: Summary of Revised OMB Results with New Cases Added

	Accurate	Over	Under
OMB			
In validation chapter	11	22	14
Excluding all Gianessi (1999) pesticide cases	2	4	6
Excluding remaining contested cases	0	1	0
Net OMB	9	17	8
Added cases			
OSHA health studies (asbestos and vinyl chloride)	1	0	1
DOE appliance standards (Dale et al. 2002)	0	1	4
Mobile source fuel regulations (AS)	2	0	1
Mobile source vehicle emissions regulations	0	0	1
Unadjusted pesticide cases (Gianessi (1999))	4	3	9
Net added cases	7	4	16
Final tally	16	21	24
Excluding pesticide cases	12	18	15

Source: Harrington (2006)

While no single reason can explain the quite consistent result that costs appear to be overestimated by regulatory agencies, Harrington et al. identified a number of plausible explanations.

The first is technological innovation, often considered to be the primary reason for overestimation of costs. Of course, this is not a universal tendency, since agencies are also sometimes over-optimistic about the costs and effectiveness of new technologies. Four other explanations also seem to apply, at least in some cases. These include "quantity errors," including both baseline and compliance issues, plus three explanations associated with the procedural and methodological practices of rulemaking: changes in the regulation after the cost estimate is prepared; use of maximum cost estimates (often required by law); and asymmetric error correction, wherein industry consistently provides information to correct overestimates and, perhaps, less so for the case of underestimates. These hypotheses are not mutually exclusive; each can be true for different rules, and in some cases may each apply to the same rule. Some lead to cost overestimation, while others may lead to either over- or under-estimation.[22]

[22] For a further explanation see Harrington et al. 2006.

Conclusions

Patterns of over- or under-estimation of regulatory costs (or benefits) can lead to inefficient decisions for society as a whole, and can undermine public confidence in the regulatory process. Fortunately, as nations gain more regulatory experience, the ability to assess the accuracy of pre-regulatory estimates is enhanced.

In fact, the debate over whether the costs of environmental regulatory programs are under or over-estimated is really two debates, and it is important not to confuse them. One issue is whether all the cost elements are being included in the estimates. Many observers argue that important cost elements are omitted, and if they were included they might swamp the costs that are now included. These omissions include the diverted management attention, the innovations that weren't made because of the resources devoted to complying with environmental regulations, as well as the general equilibrium adjustments that ripple through the economy when resources are diverted from one use to another. Not only are such cost elements difficult to measure, they are also virtually impossible to attach to individual regulations. Thus, it is not surprising that those who argue that important costs are ignored in the regulatory process tend also to be those who point to the costs of environmental regulation in general, rather than to the costs of individual regulation.

The second issue is the one considered in this paper: whether the cost elements that are estimated during regulatory procedures contain systematic errors, and if so, what the implications are for regulatory policy. Unlike concerns about the completeness of the cost estimates, this issue is amenable to empirical test. To resolve it, some experts call for more intensive scrutiny of the procedures used by regulatory agencies to collect information, choose rulemaking alternatives, and evaluate costs. Hahn (1996), for example, argues that "without a detailed evaluation of each regulatory analysis and Federal Register preamble, it is difficult to say how individual analyses are likely to be biased (page 224)". Scrutiny of rulemaking procedures and the methodologies in the *ex ante* analyses used to support regulations are important tasks. However, they are not sufficient to determine the accuracy of the *ex ante* studies.

What is needed, in addition, is a systematic comparison of *ex ante* estimates generated by RIAs to *ex post* assessments of the same rules. Based on the evidence reviewed here, a pattern of overestimation of costs is apparent. Yet, it is frustrating that the available data on *ex post* costs are so limited, especially when so many observers, from all points of view, clamor for it.

One difficulty is that there is no consistent source of funding for the required studies. Regrettably, *ex post* studies are not routinely done and appear largely for idiosyncratic reasons, often dependent on the initiative of individual researchers.

Ex post studies also must contend, typically, with serious problems of data acquisition and interpretation. Compliance cost data are often considered business confidential by regulated firms, whose participation in cost studies is very likely to be voluntary anyway. And even if the data are obtained, there are likely to be difficult issues of joint cost allocation.

Beyond the problems of inadequate funding and disincentives for the private sector to reveal *ex post* information, government-erected barriers to the collection of data may also be significant. For example, the Paperwork Reduction Act of 1995 and the associated regulations severely limit government-supported studies from collecting data from firms and individuals. Even if one accepts the tentative finding of this review, namely that pre-regulatory estimates tend to overstate the true costs of regulation,

one must also recognize the fragility of that conclusion. One step toward acquiring more and better data to validate – or, perhaps, contradict – the findings reported here would be to relax the restrictions imposed by the Paperwork Reduction Act. Less than 20 percent of the government-imposed burden of data collection is related to matters other than tax collection (GAO 2006). With more and better data research could then examine the specific agencies, the types of regulations, and the particular methodologies most likely to generate inaccurate results. Surely, the benefits to a democratic society of more reliable information on the true burden of regulation, as well as the benefits of that regulation, outweigh the small additional costs of expanding the data collection process.

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Impact Assessment in the European Commission

Craig Robertson

Introduction

Any responsible public official would look at you with incredulity if you were to suggest that policy or programme initiatives were prepared without careful and reasoned consideration of the potential effects of the action being proposed.

In the dozens of training sessions and seminars on impact assessment held inside the European Commission (EC) during my more than three years working in the Better Regulation and Impact Assessment Unit of the EC's Secretariat General – at which I was called upon to set out the EC approach to impact assessment – it was often the case that one or more Commission officials would listen (occasionally with impatience) to my outline of the logical, analytical steps to be followed in preparing a Commission impact assessment, and then state that this was simply common sense and existing practice. I feel confident in asserting that a similar response would be likely in analogous settings in all tiers of public administration.

The prospect of legislative scrutiny and/or public and media attention means that governments at state, sub-state or supranational levels should expect that their officials will have prepared an initiative in the light of the best available evidence of possible impacts, and that this evidence should be available when a political choice is being made on whether or how to proceed. So-called "evidence-based policy-making" is not meant to replace a political choice with a technocratic calculation. Rather, the intention is to increase the "responsibility" of politicians to think and consider before acting. That so much of what is proposed by governments only sees the light of day following some form of analysis of who and what will be affected, and to what extent, is perhaps what makes it all the more striking and disconcerting when we witness a clear case of a poorly considered or rushed intervention. Sometimes the main driver of this rush to act is not too difficult to discern, with the desire from politicians to respond rapidly to loud statements of public or stakeholder concern often providing the stimulus for action (the 1991 UK Dangerous Dogs Act is often cited in this regard).

So, sometimes the simple application of common sense, possibly backed by some limited, formal requirement to look at impacts in one or more specific areas, is enough to ensure that politicians in government are able to base their decision on sufficient "evidence" of what the desired and undesired outcomes are likely to be. However, in other instances, this is not enough.

The development of (integrated) Impact Assessment in the European Commission

Although there was little to suggest that the protestations from participants at the EC impact assessment training sessions about their professional approach to policy-development, including a sound analysis and exposition of pros and cons, strengths and weaknesses, were not based on fact, the EC decided in 2002[23] to introduce a standardised approach to the assessment of the potential impacts of its major legislative and policy-defining initiatives. In place of the sectoral and partial analyses (business, gender, environmental, etc), which had been developed in a piecemeal fashion in the preceding years, and applied by Commission departments with varying degrees of enthusiasm and attentiveness, the new approach would consist of certain standard procedures and a set of logical, analytical steps which would be universally applicable across the Commission. The steps to be followed are:

- **Problem definition**: Is there widespread agreement that a problem exists? Is it getting worse or better? Who is affected and to what extent? What actions have already been taken or are in the pipeline?
- **Objective setting**: What is the desired impact? Can it be linked back to the problem? Are the objectives consistent with wider objectives, such as the Lisbon or Sustainable Development Strategies?
- **Developing options**: What alternative options are available to feasibly tackle the identified problem and meet the objectives?
- **Impact analysis**: What are the direct/indirect, positive/negative economic, social and environmental impacts of each of the identified policy options? Who would be affected and how? What are the risks and uncertainties associated with each of the alternative options?
- **Compare the options**: Present a clear comparison of the different options in terms of their respective impacts, etc.
- **Plan for monitoring and evaluation**: Identify core indicators to assess progress in meeting the key objectives, and develop broad outlines for future monitoring and evaluation of the intervention.

[23] COM(2002)276final available at http://ec.europa.eu/governance/impact/docs/key_docs/com_2002_0276_en.pdf

The **collection of data** and solicitation of **input from stakeholders** can take place throughout the process.

It replaced the previous sectoral analyses with an integrated approach, whereby impacts of a range of alternative options would be examined across their broad economic, social and environmental dimensions. The evidence of potential impacts, set out in a balanced and proportionate way, would be clearly presented to the political decision-makers in a standard reporting format, to allow them to make the decision on how to proceed. As the 2002 Communication clearly stated:

Impact assessment is an aid to decision-making, not a substitute for political judgement. Indeed, political judgement involves complex considerations that go far beyond the anticipated impacts of a proposal. An impact assessment will not necessarily generate clear-cut conclusions or recommendations. It does, however, provide an important input by informing decision-makers of the consequences of policy choices.[24]

The 2002 Communication also sets out the key political drivers which lay behind the development of the new integrated approach, with reference made to the 2001 Göteborg (on Sustainable Development) and Laeken (on Better Regulation) European Councils, and to the Commission's White Paper on Governance[25]. The aim was to systematise the analysis of the potential economic, social and environmental impacts of the Commission's most significant initiatives in such a way as to allow politicians to see where trade-offs may be necessary, and to identify alternative approaches to tackling an identified problem, which may minimise negative, undesired impacts while at the same time allowing the key objectives to be met. In preparing the analysis of these potential impacts, the lead Commission department would be encouraged to coordinate with other policy areas within the Commission and to ensure that the views of stakeholders were taken into consideration.

The "principle of proportionate analysis"

In the initial two years of operation, a two-step approach was followed. On the basis of Preliminary Impact Assessments (PIA), prepared for all initiatives included in the Commission's Annual Policy Strategy and its annual Legislative and Work Programme, a decision would be taken as to which of the initiatives warranted further analysis in an Extended Impact Assessment. This approach was seen as a useful aid in determining the degree of analysis required. Indeed, the "principle of proportionate analysis" was, and

24 Ibid, p. 5
25 European Council Conclusions and link to the White Paper can be found at http://ec.europa.eu/governance/impact/key_en.htm

remains, central to the Commission's approach. Essentially it is based on a realisation that when applying impact assessment to a wide variety of policy initiatives – ranging from specific proposals for Regulations or Directives through to broad, policy-defining Communications or White Papers - the degree of detail and level of analysis will need to be commensurate with the actual likely impacts. The greater the anticipated scope or degree of impacts, the greater the expectation for an extensive and detailed analysis. However, the potential for other, non-impact assessment related factors to come into consideration in determining which initiatives should be chosen for Extended Impact Assessment was one element that led the Commission to change the approach with effect from 2005. Since then, there has been a blanket requirement that all initiatives featured in the Commission's Legislative and Work Programme (CLWP) would be subject to a proportionate impact assessment[26].

An important stage in the elaboration of the impact assessment is the preparation of the so-called "Roadmap", in which the work already undertaken for the impact assessment is set out, and the orientations and timetable for the remaining work is provided. These Roadmaps, which were also introduced in 2005, are made publicly available once the CLWP has been adopted by the Commission, meaning that stakeholders and other interested parties have an opportunity to review what the Commission has already done and to plan any input that they may have for the remainder of the impact assessment process. In addition to the obvious increase in transparency that the introduction of this new element brought, it was also hoped that the input received from external parties would provide guidance on the degree of analysis likely to be expected in any single impact assessment. If the received contributions indicated that the subject area was particularly sensitive, or if there was clear disagreement over the data or science surrounding the issue, then it would be important for the lead Commission department to ensure that all relevant elements were considered as part of the assessment.

A further check to help ensure that impact assessments drawn up by the lead Commission department are as comprehensive as they ought to be is to be found in the requirement that an Inter-Service Steering Group (ISSG) be established for all Commission impact assessments. Although the ISSG concept had been in place from the earliest days of the impact assessment approach in the Commission, there was no systematic pressure applied on departments to establish such a group. However, the ISSG element was given much greater weight as part of the overall review of the impact

[26] With a few, well-defined exceptions for Green Papers, initiatives under the Social Dialogue procedures, and those for implementing international agreements. Further exceptions were granted for 2007 to include regular progress reports.

assessment approach in late 2004 and early 2005, with the default position that all cross-cutting impact assessments be guided by an ISSG, and imposing on the departments a requirement to provide justification when no group is planned[27]. A useful argument to deploy in those cases where the lead Commission department was, for one reason or another, reluctant to establish an ISSG, is that not doing so could lead to problems and delays when the draft proposal and impact assessment are submitted to formal Inter-Service Consultation. Initiatives due for adoption by the College of Commissioners need to undergo this procedure, whereby the lead Commission department seeks the opinions of other Commission departments on the draft proposal and the impact assessment. Essentially the Inter-Service Consultation process allows other areas of the Commission, including the Secretariat General and DG Budget, to insist on amendments or improvements to the submitted documents. In some serious cases, it could lead to one or more departments effectively blocking agreement on the proposal and impact assessment at the level of officials. The next stage in resolving any such problems would be the political (cabinet) level. Although there is a possibility that reservations about the quality or completeness of a Commission impact assessment may come to be pushed to one side when wider political questions come to the fore, Commission departments are often reluctant to take the risk that their proposal will be held up or blocked in the Inter-Service Consultation process.

Notwithstanding the mechanisms and checks that had been put in place to help guide the lead Commission department in determining how deep and detailed it ought to be with its impact assessment, it has been argued (and with some justification) that some EC impact assessments have failed to examine the potential impacts to a suitably proportionate extent. The Commission response has been to establish an internal Impact Assessment Board (IAB). The IAB, which started its work under the direct authority of the Commission President in late 2006, consists of five senior level Commission officials – chaired by the Deputy Secretary General – who are drawn from the Commission departments with the closest links to the three dimensions of analysis required in a Commission impact assessment (economic, social and environmental). The IAB's role is to examine draft impact assessments before they are submitted to Inter-Service Consultation, to offer an opinion on the quality of the draft and provide suggestions for possible improvements. This is also likely to include guidance on the required depth of analysis in any single case. The IAB only started to issue opinions in the spring of 2007, so it is difficult to assess the extent to which it will be sufficient in tackling the criticisms that have come to be levelled at the Commission for the variable quality of its impact

27 See SEC(2005)791final, p. 9

assessments. Anecdotal evidence from the early months of operation would certainly seem to suggest that the introduction of the IAB has concentrated the minds of Commission departments on the need to be rigorous in preparing their impact assessments.

Cost-Benefit Analysis in EC Impact Assessments

It is often the case that what lies at the heart of critical comments about an EC impact assessment having failed to analyse impacts to a sufficient degree is a belief that attempts at quantifying impacts have been inadequate. Indeed, there tends to be confusion, even within the Commission itself, about the relationship between impact assessment and Cost-Benefit Analysis (CBA), with some people assuming that the two terms are synonymous. This, however, is not the case. Under the current guidelines for impact assessment in the Commission (and indeed from the time of the 2002 founding Communication), it is perfectly possible for an impact assessment to be considered acceptable even if the analysis is purely qualitative in nature. Although there is strong encouragement to make efforts to quantify and/or monetise impacts, there is no blanket requirement to do so. As the 2002 Communication states:
To show the different impacts, make comparisons easier and identify trade-offs and win-win situations in a transparent way, it is desirable to quantify the impacts in physical and, where appropriate, monetary terms (in addition to a qualitative appraisal). Impacts that cannot be expressed in quantitative or monetary terms should not, however, be seen as less important as they may contain aspects that are significant for the policy decision.

Just to underline the point that alternative methods of analysis can be equally acceptable in a Commission impact assessment, it also goes on to say:

> Nor can final results always be expressed in one single figure reflecting the net benefit or cost of the option under consideration … When assessing impacts, strict cost-benefit analysis may not always supply the most relevant information.

Lying behind this caution on applying a strict requirement for CBA in Commission impact assessments are some very practical concerns and some, more principled, considerations.

As the quote from the 2002 Communication given above makes clear, the Commission approach to impact assessment is based on the view that political decisions are taken in a highly complex decision-making environment. Leaving aside for one moment the practical difficulties of reaching a "net" outcome in the *ex ante* analysis of a policy choice which could have potential impacts on a wide range of business sectors, social groups, regions, countries, etc, attempts to reduce this complexity to a single net figure of cost or benefit may be reducing the responsibility of the political decision-maker by presenting an oversimplified answer. Essentially it may mean that those responsible for preparing the impact assessment come to exercise a far greater influence on the final political choice than was originally intended.

The more practical concerns associated with applying a strict CBA approach to Commission impact assessments relate mainly to the highly complex environment within which the Commission operates. Unlike many approaches to impact assessment in place in a national context, the Commission applies the practice to legislative proposals and broad, policy-defining initiatives. The problem with the latter is that they are often setting out policy measures at a highly abstract level, making it extremely difficult to determine the quantitative, let alone the monetary, impacts. The difficulty with the former is that the legislative action being proposed may be in the form of a Directive, meaning that the EU Member States have considerable leeway in how the measure is eventually implemented. This, of course, makes it very difficult for the Commission to be able to fully predict the likely impacts once the Directive is fully applied across all 27 Member States.

Trying to assess the potential impacts in a very wide range of domains, across an EU of 27 Member States, and where the Commission does not control the final outcome on the ground, is often further complicated by a lack of reliable or complete data sets. Notwithstanding the efforts of Eurostat[28] and other specialised areas of the Commission (including the Joint Research Centres) to gather and present data and statistics of direct relevance to many of the impact assessments prepared by the European Commission, many Commission officials tasked with the responsibility of elaborating an impact assessment, still lament the lack of hard data which would allow them to move further towards quantification of impacts. It is interesting to note that there is a potential paradox between two key demands of those who push for a stepping-up of the EU's activities on Better Regulation. On the one hand, there is the demand that greater efforts are made to assess the potential costs of new (or renewed) EU regulation on

28 The statistical office of the European Communities.

businesses, government, etc; and on the other is the demand that information obligations, including the provision of data to Eurostat, be substantially cut back as part of the attempt to reduce administrative burden, meaning that the data sources on which the impact assessments could rely are likely to become less certain or reliable.

Conclusions

The image of the EC bureaucrat, sitting in his or her comfortable Brussels office, cut off from the realities of everyday life for Europe's citizens and businesses, blithely thinking up new and onerous rules under which those citizens and businesses will be obliged to live, is a caricature which has been regularly employed by Euro-sceptics for many years. It is, however, a false image. It would be dishonest to suggest that every legislative proposal made by the Commission over the years has been properly thought through, or that the political decision to proceed with making a proposal has always been taken in the light of a careful and considered analysis of the direct and indirect impacts it is likely to have. However, for the most part, Commission officials have recognised the need to think before proposing action. In order to systematise the practice and ensure that the analysis is as complete as possible, the EC has introduced a system of integrated Impact Assessment. The identified shortcomings of the approach, particularly those relating to a perceived lack of quantification/monetisation and absence of strict Cost-Benefit Analysis, should not blind the observer to the overall success of an approach which has enhanced transparency and offered opportunities for structured input into the policy development process.

Nevertheless, the Commission itself has recognised that there are areas which require improvement - the launch of an external evaluation of the impact assessment system in August 2006 is clear evidence of that. The findings of the external evaluation have not been made available at the time of writing this chapter, but it is probable that one area which is likely to feature in terms of aspects where improvements are needed is in providing greater clarity on the principle of proportionate analysis. As the above has shown, there are mechanisms in place to assist Commission departments in determining the appropriate degree of analysis in any single case. Claims of inconsistency in how these are applied may lead to more specific guidance being provided as to when a more detailed analysis needs to be undertaken. For those cases where an advanced degree of analysis is thought to be required, there will be an increased expectation that this will involve muc greater attempts at quantification of impacts, and the wider use of CBA.

It is unlikely, however, that the Commission will move towards a blanket application of CBA for all advanced-analysis impact assessments, let alone insist that the analysis comes to a "net" finding. The main reasons for not doing so have been briefly explored above. It would be helpful, nevertheless, if the impact assessment clearly explained why it was not possible or appropriate to quantify impacts, or to set out where the data being used is less than complete or totally reliable. If the expectation is that impact assessment is used by the political decision-maker to better inform their decisions, then it is imperative that they are aware of possible doubts about, or gaps in, the evidence being presented to them. Those for whom the decisions are made would expect nothing less.

Part Two

Assessing costs and benefits: concepts, methods and tools

The demand for cost and Benefit Assessment in the EU Water Framework Directive: What has been done in France?

Patrick Chegrani

Abstract

The Water Framework Directive (WFD) establishes a framework for Community action in the field of water policy. One of the greatest innovations brought by this directive is the introduction of economic analysis in water management.

Economics also provides tools to support decisions: identifying the economical context (for instance the turnover and the number of jobs of the activities linked to water), identifying the least cost option for meeting the environmental objective (cost-effectiveness analysis – CEA) and defining realistic objectives, that is to say objectives whose costs are not disproportionate (cost-benefit analysis – CBA).

Economic analysis can generate tools which can be used to meet environmental objectives: this is the case for water-pricing and for the analysis of the cost-recovery of water services. The WFD requires the assessment of environmental costs and benefits. To that purpose, the French Ministry of Ecology issued a "circular" (instruction) to underline the importance of economic analysis and to specify the type of work which should be done in order to update cost-recovery information and the assessment of environmental and resource costs.

Regarding cost-benefit analysis – which can be used to justify exemptions – a three-step process can be developed to assess non-market benefits: first a qualitative assessment, then a phase based on reference-values, and finally, a local study if needed. To that end, the Ministry of Ecology has provided tables of reference-values and guidebooks presenting good practice regarding the implementation of valuation methods. A website (www.economie.eaufrance.fr) has been set up to gather information on the WFD economic analysis.

The directive 2000/60/CE (Water Framework Directive – WFD) of the European Parliament and of the Council establishes a framework for the Community action in the field of water policy. The introduction of economic analysis in water management is one of the greatest innovations brought by this directive, especially concerning the assessment of environmental costs and benefits.

The objective of this article is to present:

- The demand for cost and benefit assessment in the Water Framework Directive: when and how can economics provide tools to support decisions or tools for action?
- How documents developed by the French Ministry of Ecology and Sustainable Development (methodological support and data synthesis) aim to make economic analysis more understandable and environmental costs and benefits easier to assess.

This article first briefly describes the Water Framework Directive (general context and timetable), then considers the economic analysis in such a context (which tools? which aims?), and finally depicts the production of the French Ministry of Ecology for the assessment of environmental costs and benefits (especially a recent circular and the website www.economie.eaufrance.fr).

The Water Framework Directive (WFD)

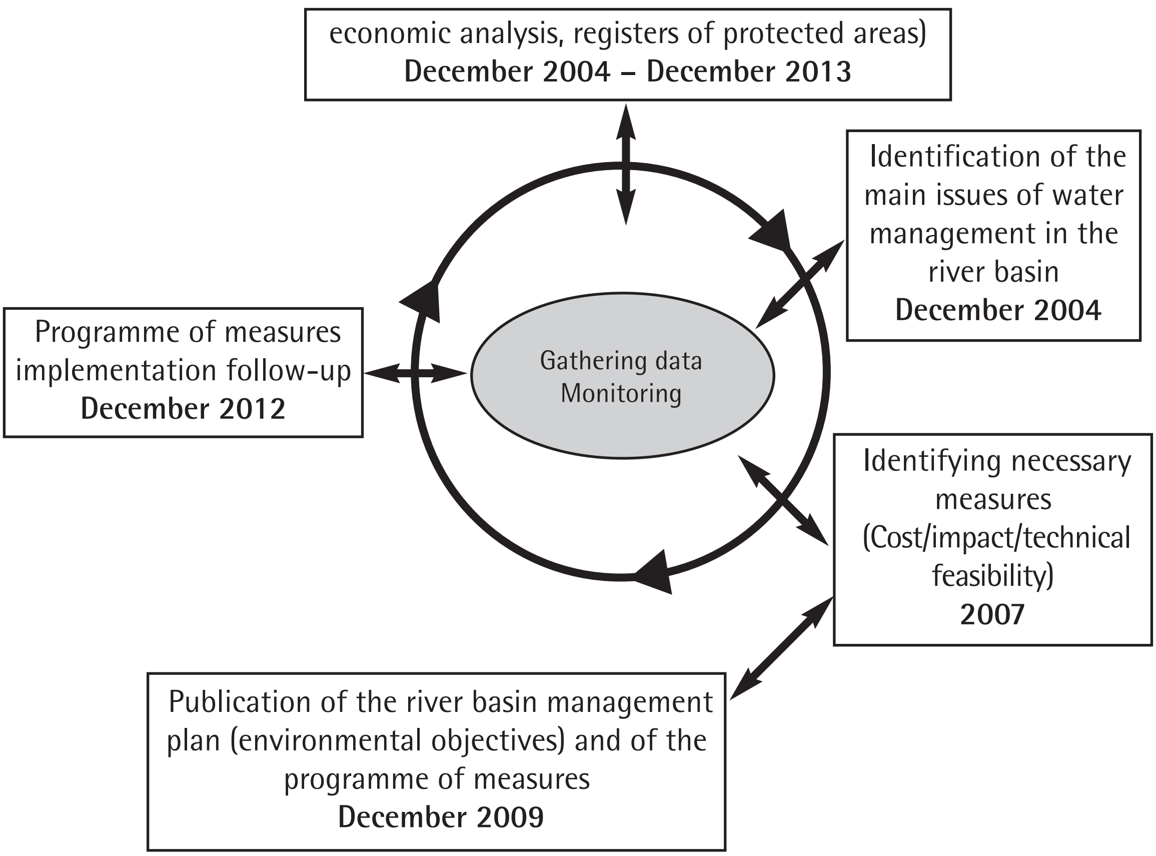
The WFD is the most substantial piece of European water legislation: it will provide a major driver for achieving sustainable management of water in the Member States for many years to come. This directive requires all inland and coastal waters to reach "good status" by 2015, which would be achieved by establishing a river basin district structure within which demanding environmental objectives would be set, including ecological targets for surface waters.

The WFD sets out a timetable for both initial transposition into national laws and for implementation of its requirements.

- For each river basin district, it requires carrying out analysis of characteristics of the surface and groundwater, reviewing the environmental impacts of human activity (industry, farming, etc), preparing economic analysis of water use (article 5) and establishing register(s) of protected areas (articles 6 and 7) (December 2004).
- Defining the necessary measures which may respond to these issues and may allow the environmental objectives specified by the directive to be achieved. The environmental objectives have to be set in accordance with the cost of these

measures, their technical feasibility or their natural obstacles. All exemption requests that would delay the achievement of good water status must be justified.

- Establishing programmes of measures in each river basin district to deliver environmental objectives (article 11) and publishing first river basin management plan for each river basin district, including environmental objectives for each body of surface or groundwater and summaries of programmes of measures (article 13) (December 2009).
- Ensuring follow-up of the implementation of the programme of measures, establishing a report halfway through the programme, thus allowing to identification of possible remedial action.
- Reviewing and updating plans, so that a new cycle of action is beginning (December 2015 and every six years thereafter).



Economic analysis in the Water Framework Directive

Economics is one of the central pillars of the WFD. Economic assessments are widely carried out in other areas, such as air quality, the greenhouse effect and transport. WFD aims to apply this type of assessment in the water sector, where it has not been strongly implemented so far in France.

On the one hand, the aim of an economic analysis is to assess the social, economic, financial and environmental impact of the current situation regarding the water environment and resource. The aim is also to assess actions which could be carried out in order to improve the water environment.

On the other hand, economics aims to optimise the use of financial resources by making appropriate and clear decisions that support the idea of reducing collective costs. The goal is to bring balance to the water supply growth policy through actions on water demand.

Carrying out an economic analysis within the context of the WFD helps the decision-making process. This type of analysis is also a tool that may be used to achieve environmental objectives. The following paragraphs provide a description of these tools by:

- quoting the requests of WFD;
- providing economical data that must be taken into consideration; and
- indicating its relevance within the water management framework.

A. Tools to support decisions

1 - Identifying the economic context

Economic analysis mainly contributes to acquiring a better knowledge of the context by identifying activities linked to water status and its use. It can also provide information regarding economic data: "Each Member State shall ensure that for each river basin district or for the portion of an international river basin district falling within its territory: an analysis of its characteristics, a review of the impact of human activity on the status of surface waters and on groundwater, and an economic analysis of water use is undertaken" (article 5-1).

Data collected during the analysis of the river basin districts in France performed in December 2004 corresponds to the turnover and the number of jobs.

When all parties reflect on the creation of the river basin management plan and on developing a programme of measures, the relationship between technical data (sampling, rejection) and economic data (turnover, number of jobs generated) allows stakeholders to assess:

- the economic impact of water on human activity;
- the future impact of territorial development (by creating a pattern of evolution); and
- the costs supported by various sectors (concerning pricing and cost-recovery, including environmental and water resource costs).

2 - Identifying the least cost option for meeting the environmental objective: Cost - effectiveness analysis (CEA)

Cost-effectiveness analysis (CEA) consists in choosing the least cost option for meeting the environmental objective: "The economic analysis shall contain enough information in sufficient detail (taking account of the costs associated with collection of the relevant data) in order to make judgements about the most cost-effective combination of measures in respect of water uses to be included in the programme of measures under Article 11 based on estimates of the potential costs of such measures" (annex III).

The analysis of the river basin districts carried out in 2004 identified whether the measures known to be compliant with the existing regulations would achieve WFD environmental objectives. If some water bodies are not likely to meet a good ecological water status in 2015 based on the criteria established by the existing legislation, new measures must be identified and considered in order to succeed. Several strategies may therefore be developed and the most efficient one must be selected.

If environmental action costs are assessed in a fairly simple manner – mainly through feedback – it would lead to lack of knowledge regarding their impact upon the natural environment, particularly regarding farming environmental measures as well as other measures linked to restoring the morphology of the water environment.

In order to comply with the principles of CEA despite such difficulties, a pragmatic and progressive approach has been recommended. It consists of gathering available feedback and additional information by carrying out research studies in specific sites.

3 - Defining realistic objectives: Cost-benefit analysis (CBA)

Economic analysis can justify, where necessary, realistic environmental objectives. These objectives might not reasonably be achieved within the timescale set out in article 4 (article 4-4) or Member States may aim to achieve less stringent environmental objectives (article 4-5).

There are three possibilities to justify a longer timescale (of six or 12 extra years):

- Technical feasibility: "The scale of improvements required can only be achieved in phases exceeding the timescale, for reasons of technical feasibility" (article 4-4-a-i);
- Disproportionate costs: "Completing the improvements within the timescale would be disproportionately expensive" (article 4-4-a-ii);
- Natural conditions: "Natural conditions do not allow timely improvement in the status of the body of water" (article 4-4-a-iii).

Such decisions need to be justified. They must be included in the river basin management plan and they must be presented to the general public during consultation.

Less stringent environmental objectives will rather concern Heavily Modified Water Bodies (HMWB) and natural bodies which are affected by human activity.

"Member States may designate a body of surface water as artificial or heavily modified, when:

(a) the changes to the hydromorphological characteristics of that body which would be necessary for achieving good ecological status would have significant adverse effects on : (i) the wider environment ; (ii) navigation, including port facilities, or recreation ; (iii) activities for the purposes of which water is stored, such as drinking-water supply, power generation or irrigation ; (iv) water regulation, flood protection, land drainage, or (v) other equally important sustainable human development activities ;

(b) the beneficial objectives served by the artificial or modified characteristics of the water body cannot, for reasons of technical feasibility or disproportionate costs, reasonably be achieved by other means, which are a significantly better environmental option."

A water body can also be designated as heavily modified:

- if meeting the good status requires hydromorphological changes that would affect economical activities;
- if this activity can not be replaced by an option which is significantly better from an environmental point of view and in which cost is not disproportionate.

How to define a disproportionate cost?
Carrying out a cost-benefit analysis is necessary to answer this question. It is therefore a matter of estimating not only the cost of the actions to be carried out, but also the environmental benefits of meeting the environmental objectives.

The next section provides a more detailed explanation of how to assess the benefits expected from a better water status.

B. Tools for action

Economic analysis should not be considered exclusively as a decision-making tool within the WFD implementation framework. It must also be used to reach environmental objectives. Relevant tools are mainly water-pricing and cost-recovery of water services.

1 – Water-pricing

Water-pricing is an economic tool that may be used to achieve environmental goals of WFD. The descriptive analysis of river basin districts provides information on the average price of water and on water pricing structures.

Consequently, basins must account for public water supply pricing policies and for water treatment practices as well as for the implementation of environmental taxes: “Member States shall ensure by 2010 that water-pricing policies provide adequate incentives for users to use water resources efficiently, and thereby contribute to the environmental objectives of this Directive” (article 9-1).

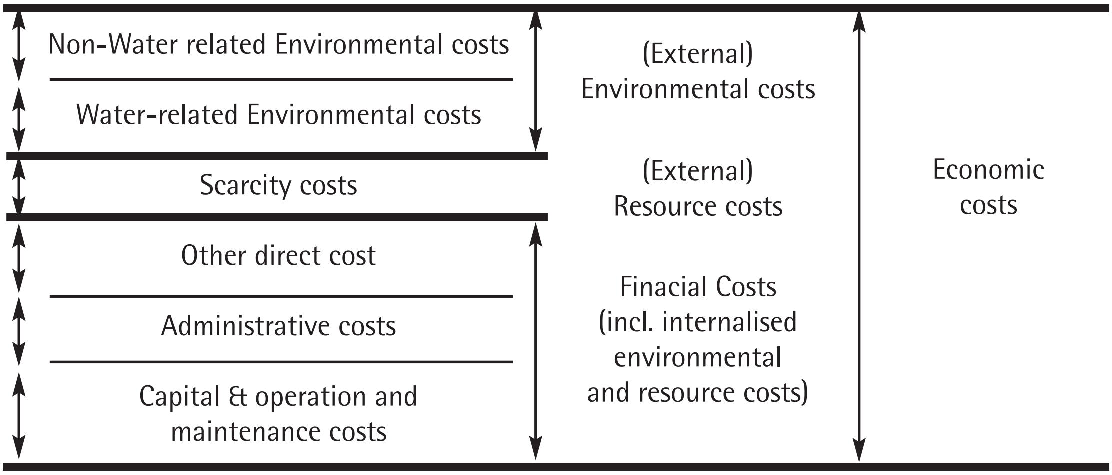
2 – Cost-recovery of water services

Regardless of the fact that the directive does not require recovering the service’s complete cost (including capital and operation costs, environmental costs ...), it does request from Member States that they account for cost-recovery levels, including environmental and resource costs generated by all river basins that form "the hydrographic district" or by their national territory : “Member States shall take account of the principle of recovery of the costs of water services, including environmental and resource costs, having regard to the economic analysis conducted according to Annex III, and in accordance in particular with the polluter pays principle” (article 9-1).

In order to perform this analysis, the directive implicitly requests the calculation of the complete cost. To clarify the WFD request, Wateco guidance represented the full cost of water services on the graph below that sums up two components:

- financial costs, that are the water services' current costs (operation and capital costs), covered by clearly identified economic agents;
- costs for the environment and for the resource.

Box 1 - What are the different types of costs mentioned in the Directive?



Source: Rogers et al. (1997)

When calculating cost-recovery rates, the first objective is to specify the financing of all water service investments and operation costs within the river basin area. The aim is to assess the economic sustainability of service management, mainly by verifying if costs are covered by the receipts (prices and subsidies) as well as the relevance of renewal expenses. At the time of defining the programme of measures, analysing the predictable cost-recovery ratio evolution for each human activity sector will allow assessing how cost-recovery will evolve as a result of the programme of measures.

A financial flow analysis may provide information on costs and expenses for each large activity sector (industry, households and farming) as well as cost-recovery rates for these same sectors. Analysing each economic activity sector may also allow identifying cross subsidies between sectors. Information on environmental and resource costs and on compensation or mitigation costs incurred by various sectors will be available too.

One must notice that there is also a request for performing an externalities assessment in water service environments ("environmental and resource costs"). Section-A of this paper provides a detailed description of the assessment of environmental and resource costs in France: implemented method and available data.

3 – The production of the assessment of Environmental Costs and Benefits by the French Ministry of Ecology

As mentioned above, WFD requirements in terms of economic analysis apply not only to economically-linked data collection (such as turnover, employment generated, costs of measures implemented to restore surface and ground waters), but also the assessment of environmental costs and benefits.

This second requirement is a new concept for two reasons: stakeholders usually do not take these environmental costs and benefits into consideration and the work performed on this subject is mainly research work that may only generate local results.

The role of the Ministry of Ecology and Sustainable Development, through the Water Directorates and the Directorates of Economic Studies and Environmental Assessment, was therefore to make necessary tools available for local stakeholders so that they might be able to perform these assessments. Two of the difficulties encountered were: making these assessments as functional as possible and complying with theoretical micro-economic principles. These goals were achieved through the use of research studies performed in France and through the support and expertise of economy researchers.

This part of the paper presents the documents developed by the French Ministry of Ecology for assessing environmental costs and benefits in France. It contains objectives and content of these documents and also the tools to meet their adequate outreach.

C. Environmental and resource costs (included in the cost-recovery of water services)

1 – Why assess environmental and resource costs?

Considering environmental and resource costs in the cost-recovery of water services could be seen as a luxury among the huge amount of information to collect and to analyse. However, it is important to be aware of their significance in the inventory process.

Firstly, considering environmental costs in the cost-recovery process means that environment is seriously considered as one field, among other traditional economic sectors (households, industry, agriculture). The cost-recovery process will therefore allow the sectors damaging the environment to be identified. By assessing the financial participation of each sector to the environmental damages it causes (through environmental taxes, environmental restoration expenses), it will be possible to assess the extent to which the polluter-pays principle is implemented. In other words, it will be possible to identify the sectors "subsidised" by the environment and the proportion of the subsidy.

Secondly, it seems insufficient to compare a single water service over different water districts on only the restricted basis of its financial costs. For example, for waste water treatment, a low financial cost may be explained by an unjustified low treatment level, which could entail large environmental costs. On the other hand, a high financial cost of sewage treatment could mean a high-level treatment standard and therefore low environmental costs. In this example, both sewage treatment services considered may have very different financial costs, but quite similar total costs.

2 – Definition

The European working group Eco2, made up of economists from European countries, proposed in June 2004 a definition of environmental and resource costs:

- Environmental costs are defined as the economic damage costs to the water environment and other water use(r)s caused by alternative competing water use (e.g. water abstraction or wastewater discharge).
- Resource costs are defined as the opportunity costs of using water as a scarce resource in a particular way (e.g. through abstraction or wastewater discharge) in time and space. They only arise, however, as a result of an inefficient allocation (in economic terms) of water and/or pollution over time and across different water users, i.e. if alternative water use generates a higher net economic value.

3 – How to assess environmental and resource costs

- There are two main, mutually non-exclusive approaches to the estimation of environmental costs : a “cost based” approach and a “benefit based” approach:
  The benefit approach is based on the estimation of the loss of welfare due to environmental damage or the increase in welfare if the environment is improved (through Willingness to Pay – WTP). A list of reference values of benefits and

methodological guidances on the valuation methods have been carried out in France. However, there is currently not enough information available to implement this benefit approach in a consistent manner on all river basins (missing data i terms of unitary benefits and the number of users).

- The cost approach is based on the calculation of the costs of measures, which aim to protect the water environment against environmental damage. In France, a methodological note was issued in 2004 by the Ministry of Ecology. Water Agencies provide summaries presenting the costs of the implementation of the measures. These environmental protection costs are used under certain circumstances as a proxy for the environmental damage costs, even if such a method does not square with the micro-economic theory.

The benefit based approach is the only existing method for cost-benefit analysis. In the case of the environmental costs, this approach cannot be easily applied. Referring to the results of the working group Eco2, the Ministry of Ecology chose the cost based approach for the cost-recovery assessment.
It is however recommended to keep collecting data (number of users, more unitary benefit values) to allow future environmental costs assessment using the benefit based approach.
The following table shows a summary of the comparison between the cost based and the benefit based approaches – making the link with cost-benefit analysis:

Approach / Demand of the WFD	<u>Benefit based approach</u> (loss of welfare)	<u>Cost based approach</u> (costs of measures)
Cost-Benefit Analysis (on a few water bodies)	**Method to apply**	
Environmental costs (on river basins)	Non selected method, implementation is difficult	Estimation is possible with this method (available data)

Table 1: Comparison of the cost based and the benefit based approaches for the assessment of environmental and resource costs

The assessment of resource costs is more complex. Resource costs are opportunity costs. That means it is necessary to identify the situation with optimal allocation of water and/or pollution (in economic terms), and then to assess the net benefits which are generated in that situation (compared to the present situation). It is worth noting that the situation identified then is not necessarily the one where water will reach good status.

The Ministry of Ecology considers that the assessment of resource costs must not be carried out at the moment, because of its methodological uncertainties and scale problems (the assessment would also be performed on a river basin as a whole).
We can note that health costs are excluded in this assessment.

4 – The "circular" of the Ministry of Ecology (January 2007)

The Ministry of Ecology issued a "circular" (instruction) in January 2007 on the definition and assessment of environmental and resource costs[29]. This "circular" is in line with the will of updating the calculation concerning cost-recovery in water services which must be carried out at the moment of publishing the river basin management plans[30].

The text is intended for government services and water agencies. It is meant to specify how economic analysis must be implemented as well as the types of results which are expected.
In a more general manner, this instruction:

- underlines the necessary reinforcement of the importance of economic analysis, by inscribing the tools in the SDAGE specifications (river basin management plan);
- Specifies what type of work should be done to update cost-recovery information and the assessment of environmental and resource costs, while providing the necessary guidelines that will help in achieving WFD compliant results.

Appendix 1 of this circular is dedicated to the cost-recovery of water services: it lists all the data that should be collected in order to perform the calculation correctly (by updating or completing the data that were collected at the time of performing the river basin district descriptive analysis carried out in December 2004).

This appendix requests an assessment of the "mitigation costs", which are costs users must pay as good ecological status has not been reached. Mitigation costs are costs

29 Available on this website: http://texteau.ecologie.gouv.fr/texteau/ServletUtilisateurAffichageTexte?idTexte=815
30 In France these are the « SDAGE » (« Schéma Directeur d'Aménagement et de Gestion des Eaux »), that is the establishment of River Basin Management Plans based on French legislation and according to WFD requirements.

generated by water treatment, bottled water expenses motivated by fear that water resources are of bad quality, as well as all other expenses generated because good status has not been reached (for example new drillings for water).

Appendix 2 is devoted to the assessment of environmental and resource costs. It gives the definitions of these notions and finally proposes (as previously described in this paper) the use of a cost abatement method in order to assess environmental and resource costs.

Environmental and resource costs are also those expenses that have not yet been accrued at the date of the assessment. It will be necessary to calculate them in order to achieve good ecological status for all water bodies in 2015. It is a global indicator of the effort that must still be made to achieve the general objectives of good status and, in a certain way, it may also be considered as an indicator showing the remaining environmental expenses that must be accrued.

D. Cost-Benefit Analysis (CBA)

As seen earlier, cost-benefit analysis can be used to justify exemptions when achieving the environmental objectives may be disproportionately expensive. To that purpose, it is necessary to identify the costs generated by the programme of measures (costs of measures and negative impacts) and to compare them with socio-economic impacts and environmental benefits of these measures.
Socio-economic impact assessments are for example based on the share of cost of water use and treatment in household income, on the best available technologies, on the value added obtained from those activities, etc.

Benefits linked to achieving environmental objectives are also assessed: types of profits, assessment methods, the steps which are proposed for the assessment and the documents produced by the Ministry of Ecology to carry out CBA – as presented on the website *www.economie.eaufrance.fr.*

<u>1 – What kinds of benefits can occur?</u>

The following table summarises the benefits generated by when achieving a better water status.

Table 2: The different types of benefits

<table>
<tr><td colspan="2">Kinds of benefits</td><td>Present users</td><td>Extra users
(because the good status is met)</td></tr>
<tr><td colspan="2">Market benefits</td><td>Avoided costs of water treatment</td><td>Increase in the value added of some activities (to display out of the CBA)</td></tr>
<tr><td rowspan="4">Non-market benefits</td><td rowspan="2">Concerning users</td><td>Increase in welfare of users</td><td>Increase in welfare of users (which alredy have another activity but can practicw another one thanks to the good status)</td></tr>
<tr><td>[Willingness to Pay of users] * [Number of present users]</td><td>[Willingness to Pay for a new activity] * [Number of extra users]</td></tr>
<tr><td rowspan="2">Concerning non-users</td><td colspan="2">Increase of welfare of non-users (for example for biodiversity)</td></tr>
<tr><td colspan="2">[Willingness to Pay of non-users] * [Number of non-users]</td></tr>
</table>

Market benefits may be estimated by observing the existing economic flows.

These are avoided costs in water treatment: a change in resource status may induce the total or partial disappearance of water treatment activity (concerning households and industries). A synthesis of the average cost of water treatment per cubic meter, according to each water treatment type, has been established by the Adour-Garonne Water Agency[31].

The increase turnover of some activities can also be considered, but using these data for such purposes remains delicate. These benefits are subject to substitution or transposition effects for example: the arrival of additional people at a site may generate a decrease in attendance in other sites thus decreasing the turnover in other recreational activities. It seems logical to take increased or reduced economic activity into consideration based on a local point of view. However, this economic impact is generally excluded from CBA, as it does not generate any net benefit for the country's total population.

31 See http://www.economie.eaufrance.fr/IMG/pdf/AEAG_surcouts-trait_pesticides_nitrates.pdf

Besides, one must note that:
- There is no guaranteed method for evaluating the increase in site attendance or for specifically measuring the flow of people from one site to another.
- Turnover is not synonymous with net benefits, as turnover also includes expenses made for intermediary goods. A more sound benefit indicator would be the value added.

Consequently, it is recommended that local discussions consider increase in value added and the number of jobs generated.

Non-market benefits are advantages that may not be observed directly through existing economical flows. These benefits are particularly important in the environmental area. They are based on the degree of importance that the population grants to improving their lifestyle by making environmental quality changes.

Three population categories must be taken into account:
- Current users (general): the satisfaction obtained by the leisure users from changes in water status is measured by the Willingness to Pay (WTP).
- Non-users: the benefit is the interest shown by the population regarding the improvement of the community's natural heritage (also measured by the WTP). This value is often considered to be less accurate, but it may nevertheless represent an important part of the total benefits. The Ministry of Ecology therefore recommends including the natural heritage value when calculating benefits. This is achieved by clearly posting its exact amount in order to perform two calculation operations – with and without this value.
- Additional users (specific): in this case, benefit derives from the satisfaction obtained by observing people who come to practise a new recreational activity (in comparison with past activities) after a change in the water status. This assessment only applies to cases where the increase in the number of users is expected to be high and significant.

<u>2 – A proportionate process with three progressive steps</u>

Progressive analysis must be carried out to assess non-market benefits. The means of assessment must be proportional to the accuracy needed for the decision-making process.

The progression could be divided into three steps as follows:

- Simplified phase: The analysis of obvious cases which confirm the disproportionate aspect of certain costs is based on technical non-monetary indicators (for example: numerous and costly measures applied to ecologically weak issues: thus resulting in a disproportionate cost). This step requires describing the benefits generated by a change in water status (positive impacts), whether it is seen as a natural heritage asset or as normal use and whether it is for the short or long term.
- A phase based on reference-values: A more detailed study is performed based on available benefit values. It is an intermediary approach between a general qualitative approach and an in situ study approach. It has the advantage of being fast and simple, but it is also relatively uncertain. Assessment results must mostly be considered as indicators rather than as fixed theoretical values.
- Detailed phase: More in-depth economic analysis based on specific and local assessments (particularly if certain benefits or costs cannot be monetary because of reference-values and are likely to influence the outcome).

<u>3 – The production of the French Ministry of Ecology for the assessment of non-market benefits</u>

In order to assist a first quantitative analysis, the French Ministry of Ecology offers an overall view of the existing values regarding non-market benefits. This overview is organised into a convenient set of tables: classifying each reference value by type of use, providing a description of the context and the environmental quality change for each. A detailed document for every value (research area, method, results) has also been created, providing background data on environmental benefits in water management. Tables are constantly being updated and are intended for direct use. They provide simple access to benefit information. They allow users to acquire knowledge on the types of benefits which cannot be quickly assessed, which will for instance provide enough information to decide whether more research would be necessary. An Excel file has been created in order to implement these values more easily. It offers a general view of available values, divided by types of benefits, and allows stakeholders to select desired values easily. These are then designated as “likely”, “uncertain” or “impossible to estimate” according to the indications entered by the end user of the Excel file. This data processing tool also calculates the sum of costs and benefits – applying a discount rate. It is indeed possible to include data regarding measure costs, whether it is expressed as an investment, as an average life span or as an operation cost – annual non-market costs can also be included in the calculation.

It is necessary to specify that the reference-values overview is based on several value research studies made in France on water quality and management. The list of all the concerned studies has been performed by researchers (Amigues and Bonnieux, 2003).

A group of university researchers collaborated with the French Ministry of Ecology on providing the opportunity to evaluate the situation and on enabling the elaboration of a programme that aims to develop a more functional implementation of environmental benefit assessments. A workshop was held on September 29th 2006 which discussed opportunity costs, value transfers, aggregation, non-use value and the valuation method better known as "choice experiment". The minutes of this meeting are available (French only) on the French Ministry of Ecology website: *http://www.ecologie.gouv.fr/article.php3?id_article=6916*

The issue of aggregation of reference-values for each unit (per person, per household, etc.) was also discussed. This aggregation consists in transposing unit benefits to a total amount of benefits (expressed in Euros per annum) for a specific area, meaning that the number of users and non-users needs to be estimated. Implementing this type of action remains difficult. Methods that may be used are as follows:

- Using local data, such as expert opinions (fishing guards, sports clubs) and site attendance research studies. This information tends to be the most reliable but is also the most difficult to get access to.
- On the other hand, in order to be able to estimate benefits, ratios of users in a population could be used (for example, the percentage of fishermen living among the population of the river basin). Such data are obtained from existing studies. Using these ratios to measure other site numbers of users and non-users might however unconvincing.
- Whenever important issues must be resolved (if available estimates are not considered to be reliable enough and may strongly influence the decision-making process), performing a telephone urvey may be recommended.

British methods featured in the "Benefits Assessment Guidance" (Environment Agency, 2003) are yet to be tested and validated in France. They could be used as intermediary methods, being less costly than site attendance research and also being more reliable than users' ratios transpositions. These measures have not yet been validated and the French Ministry of Ecology maintains an impartial position regarding the use of such methods.

If reference-values are not sufficient in evaluating whether the costs are disproportionate or not – whether in cases in which a large number of costs and benefits may not be given a monetary value, or in cases in which there is no agreement regarding values – a local research study should be performed.

The French Ministry of Ecology has therefore published four guidebooks regarding valuation research studies:

- A general document that helps to elaborate relevant specifications.
- Three guidebooks presenting good practice regarding the implementation of three methods: contingent valuation, hedonic pricing and travel cost.

These documents have been created based on feedback provided by French and foreign research studies: Which are the relevant questions that should be asked? What information should answers to these questions be based on? Which ways to answer such questions? What are their advantages and drawbacks? How may common errors be avoided?

The main objective is to produce quality results that may be compared and transferred to other uses. These guidebooks include non-technical information, intended for non-economists, to help them follow the guidelines easily and therefore be able to pilot a valuation study.

The guidebooks also feature technical information for people who carry out the study itself, including several econometric data processing guidelines.

The progressive benefit assessment approach tends to target its efforts toward the issues at stake. Collecting all available data (mainly research studies), using this approach in determining reference-values, and publishing valuation method implementation guidebooks are all actions that respond to this objective.

4 –The website *(www.economie.eaufrance.fr)*

The French Ministry of Ecology has created the website *www.economie.eaufrance.fr* which shares all the available information regarding the WFD economic analysis.

The site features information on the following subjects:

- Identifying the economic context (see the “water activities” tab): estimates of turnover and number of available jobs in several sectors are given (industry, agriculture, navigation, fishing, tourism, recreation), as well as a large number of documents about “water services” (service expenses, rainwater management expenses, servic management, etc.) and self-managed water treatment (equipment rates, management expenditures). These data are also essential for the calculation of cost-recovery results. The "water services financing" tab provides a large amount of information regarding this point (water service investment and performance, industrial water supply and water treatment, farming water treatment expenses).
- Water-pricing: the objective is to collect all existing data and research study information concerning households, industry, farming (public or private irrigation).
- Environmental costs and benefits: this includes all of the documents mentioned in the above paragraph:
- Guidebooks presenting good practices regarding implementation of valuation studies.
- A general document providing a description of environmental benefits observed within the WFD framework (list including all types of benefits, progressive approach, value transfer guidelines) and the available methods and data to assess them. The document also includes reference-value tables (in the appendix) and the CBA Excel file.
- Documents describing each value: a database including all French research studies which assess environmental benefits concerning water.

  Several assessment studies regarding mitigation costs: bottled water expenses, additional water treatment costs.

The French Ministry of Ecology website is a powerful tool that gathers all of the relevant information concerning the economic analysis that must be performed within the WFD framework.

In conclusion, we note the importance of economic analysis in the Water Framework Directive (WFD).

Economics does not only provide tools to support decisions (identifying the economical context, cost-effectiveness analysis and cost-benefit analysis), but also provides tools which can be used to reach environmental objectives (water-pricing and cost-recovery of water services). In both cases, the assessment of externalities is central and is also an innovation in the field of water management.

This article described documents the French Ministry of Ecology produced for the assessment of environmental costs and benefits: a "circular" (instruction) to update cost-recovery information and the assessment of environmental and resource costs, tables of reference-values and guidebooks presenting good practices regarding the implementation of valuation methods for a progressive process to assess non-market benefits.

The French experience shows the importance of sharing the available information on environment economics (so as to carry out a synthesis of values and guidebooks, for example) in order to be operational and to get quality operational results. The website is also a good way to gather information regarding the WFD economic analysis. We can add in conclusion that it is fundamental to making economics understandable and to showing that it is helpful so that stakeholders can use it. The gap between practice and theory should then be progressively filled in.

Economic Valuation: Use and Non Use Values, Methods and Case Study

Salvador del Saz-Salazar

Abstract

This paper shows what role non-market valuation techniques can play in assisting decision-making processes, as in the case of the European Water Framework Directive. In particular, the paper first introduces the concept of economic valuation of non-market goods and justifies its necessity in a world of scarce resources. Then the concept of total economic values is explained in the context of water resources. The different economic methodologies, both direct and indirect, available for estimating use and non-use values are also presented. Nevertheless, greatest attention is given to the Contingent Valuation technique through the presentation of the main results obtained from a study aimed to get the recreational use value of a wetland in the Júcar river basin in Spain. Finally, some conclusions and policy implications follow.

1. – Introduction

Economic valuation refers to the assignment of money values to non-marketed assets, goods and services. Non-marketed goods and services refer to those which may not be directly bought and sold in the market place. Examples of non-marketed services and goods would be clean air, natural areas, wetlands, endangered species, etc. However, the absence of a market does not imply the absence of value, as these non-marketed goods allegedly have a high social value contributing to the improvement of the wellbeing of individuals. For example, some people experience an increase in their wellbeing after visiting a natural area, or from the simple fact of knowing that the tropical rain forests in Brazil are protected.

Why is economic valuation necessary? Economics is the science of how society manages its scarce resources. Making choices about the environment is more complex than making choices in the context of pure private goods and services. We therefore need to impute a value to non-marketed goods. Although the valuation methodologies, developed by the economists, have been contributing to a better understanding of the social benefits of environmental improvements for fifty years, the process of environmental valuation still remains controversial, and in many ways divides the environmental economics profession (Alberini and Khan, 2006).

Economic valuation is mainly focused on discovering the demand curve for environmental goods and services, i.e. the value humans beings hold for the environment. Other values, such as the intrinsic values of environmental assets, are ignored. These other values are those that exist not just because human beings have preferences for them, but for themselves intrinsically. The use of money as the measuring rod is merely a convenience that allows comparison of costs and benefits of the different alternatives available when taking decisions. For example, if the costs of protecting a wetland are measured in monetary units, it is necessary to measure the benefits in the same units including the non-market benefits. If the services provided by non-marketed goods are overlooked or ignored in cost-benefit analyses and other empirical economic studies, the accuracy and relevance of the results will be severely undermined (Carson et al., 2001). When there are alternative options, it is generally the case that the option with the highest ratio of benefits to costs will be preferred (provided it has benefits greater than costs).

Once the concept of total economic value is introduced, the paper presents a brief description of the different economic methodologies available for estimating these values, emphasising their advantages and drawbacks. Then, a contingent valuation study conducted a decade ago is presented as an example of the usefulness of these methodologies in the process of informing policy-making under the European Water Framework Directive. This study was aimed at estimating recreational use value of a protected wetland located at the Júcar river basin in Valencia (Spain). Finally, the paper closes with a summary of the main conclusions reached and some policy implications.

2. – The total economic value

A comprehensive assessment of the benefits of a change in the level of a public or environmental good should include all the benefits which will legitimately accrue from a specified change in the provision of a given good. This concept is known as the "total economic value" approach (Mitchell & Carson, 1989). Therefore, the economic value of an environmental asset can be broken down into a set of component parts. For example, let us imagine a public policy that implies an increase in freshwater quality. Firstly, we have to distinguish between use values and non-use values. Use values may be direct (e.g. recreational activities that occur on the water; the use of the water for agricultural irrigation or for drinking) or indirect (e.g. the water body's characteristics enhance nearby activities such as bird watching, or higher water quality provides an aesthetically pleasing setting for recreational activities such as picnicking). Direct use values are fairly straightforward in concept while indirect use values correspond closely to the ecologist's concept of "ecological functions". For example, it is well known that tropical forests store carbon dioxide, so if they are burned for agricultural purposes, the stored carbon dioxide is released contributing to global warming.

In addition to current use values, individuals may be willing to pay (WTP) to conserve the option of future use. That is, through no use is made of it now, use may be made of it in the future. So we could say that option value is like an insurance premium to ensure the continued supply of some environmental good, the availability of which would be otherwise uncertain. In this case, the uncertainty refers to the individual himself. However, if the uncertainty is being experienced by the person who has to take a decision regarding an activity that can have irreversible consequences on the environment, we are then referring to the quasi-option value. This value is regarded as the risk premium people will be willing to pay to delay an activity which, if undertaken, might have undesirable consequences. So this delay provides an opportunity to take a better-informed decision at a later time.

Secondly, non-use values arise in contexts where an individual is willing to pay for a good even though he makes no direct use of it, may not benefit (even indirectly) from it, and may not plan any future use for him or others. This is also referred to as existence value. A good portion of the millions of euros in fees and voluntary contributions paid by members of environmental groups can be cited as evidence for the reality of existence values for wilderness amenities. For example, some individuals can obtain utility knowing that environmental activists are trying to stop the massive killing of whales by the Japanese fishing fleet under the pretext of scientific research purposes. These individuals will never make a direct use of the whales, nevertheless the simple fact of knowing that they are protected make them feel much better.

The bequest value is also referred as a non-use value. This value exists when somebody experiences an increase in his own utility knowing that natural resources are used in a responsible manner that guarantees their use and enjoyment by future generations or by his own descendents.

While, from an analytical point of view, the different non-use values can be easily distinguished, they are likely to be very difficult to separate and to estimate independently even using the Contingent Valuation method unless a careful wording and design of the questionnaire is guaranteed.

3. – Methods for estimating values

The various methods that can be used to measure the benefits of environmental goods differ both in their data needs and in the category of benefits they are able to measure. The main distinction here is between indirect or revealed preferences and direct or stated preferences methods.

Indirect methods rely on data from situations where consumers make actual market choices as, for example, they do in deciding on a trip to a natural park or in buying a new house. The value of the non-market benefits can be inferred from market data from other goods with which they have a known linkage. The most important methods are the Travel Cost Method and the Hedonic Pricing Method. A drawback of these methods is that only use values can be estimated with them.

The Travel Cost method seeks to place a value on an environmental asset, which can be a natural park, by the behaviour observed in related markets. The costs of consuming the services provided by the environmental asset are used as a proxy variable for price (Freeman, 1993). Therefore, this model assumes weak complementarity (Mäler, 1974) between the environmental asset and the marketed goods required to travel there.

As Rosen (1974) formally demonstrated, the hedonic technique allows the researcher to estimate the implicit price of different attributes that compound a heterogeneous good such as a house. This method relies on the idea that the sets of characteristics that compound a composite good have an effect on their market prices. It is therefore assumed that the price of the composite good can be broken down into its different attributes, making it possible, therefore, to assign an implicit price to each attribute once the hedonic equation has been estimated. Among these attributes, the environmental quality of the surrounding area can have a strong impact on the price of a house. These environmental attributes can be the quality of the air, the noise, the distance to a park, etc., so it is possible to estimate, for example, how much people are willing to pay to be a certain distance closer to a park or to be exposed to a lower noise level.

Direct methods are classified into Contingent Valuation (Mitchell & Carson, 1989) and Choice Modelling Techniques (Louviere et al., 2000). Both are survey techniques, but the former implies asking individuals to state their maximum willingness to pay (WTP) or their minimum willingness to accept compensation (WTA) for an increase or decrease in the level of environmental quality. This method presents consumers with hypothetical opportunities to buy public goods, thus circumventing the absence of a real market for them. The attraction of contingent valuation is that it facilitates the construction of a market in which the researcher can observe an economic decision directly related to the good in question (Carson, 1991). The resulting information is more useful than a simple referendum poll, because the CVM records both the direction and the strength of a respondent's preferences (Lockwood et al., 1996).

Choice modelling techniques, also called "Conjoint Analysis", define the environmental goods in terms of their different attributes and the individual must choose, rank or rate different combinations of these attributes that are presented to him in order to obtain the WTP. For example, a wetland can be defined in terms of species diversity, extension of the area protected, recreational facilities provided to visitors, etc. Combining the different levels of each attribute is possible to construct different options to be presented to the respondent to choose, rank or rate. Obviously, in order to derive a measure of the respondent's WTP, it is necessary to include among the different attributes a price in such a way that the higher the level of other attributes, the higher will be the correspondent price or cost attached. Choice modelling differs from contingent valuation in that it asks for rankings or ratings rather than for values, so what is being valued are the different attributes of the good rather than the good itself as a whole.

4. – Case study: Estimating the recreational use value of a wetland in Spain

4.1 Description of the environmental good under valuation

La Albufera Natural Park is located on the east coast of Spain in the Júcar river basin. The park covers an area of 21,000 hectares and is very close to the city of Valencia which has a population of 780,000. Its proximity to Valencia implies that it is an important place for outdoor recreation, bearing a large pressure from visitors. In some ways, it could be said that it is an urban park as 87% of visitors travel less than 25 kilometres to access it.

The park is characterised by four types of habitats (the spit or sand bar; the marsh or wetland; the lake and the woodland) which harbour a great variety of fauna and flora. The fauna is distinguished for its abundance and diversity of species, especially two types of fish -the samaruc and fartet- that are in danger of extinction. However, it is in the context of European wetlands that the park plays a decisive role as a place of resort for waterfowls which migrate to Africa looking for a warmer climate in winter. Due to their major role as a habitat for migratory birds, these wetlands have been designated as a Ramsar site and a European Important Bird Area. In particular, more than 250 bird species regularly use this ecosystem and 90 breed in it.

The main environmental problems stem from land use conflicts such as the trade-offs between preservation versus growth of farmland and preservation versus urban and tourism development. For example, the most representative element of the wetlands is that a lake which at present occupies an area of 2,837 hectares used to be ten times bigger (30,000 hectares) during the Roman Empire. Others problem affecting this wetland are water pollution and the decrease of the flora and fauna.

4.2 Methodology and survey process

The aim of the research conducted was to estimate the recreational use benefits of this natural area. Non-use benefits were not estimated as a consequence of lack of funding. The methodology applied was the Contingent Valuation Method. The survey process began in June 1995 with a pilot study of the questionnaire. Questionnaires must be pre-tested and discussed by focus groups beforehand in order to fine-tune wording and remedy any failings encountered. Schumann (1996) pointed out that surveys which ignore the importance of pre-testing questionnaires are of little use. The final version was then used in 501 interviews conducted between July and November 1995 at three different places in the park (Devesa, the Jetty and El Racó de l'Olla).

The implementation of a hypothetical market through a questionnaire requires three main elements. Firstly, the respondent must receive information about the environmental asset to be valued to familiarise him/her sufficiently with the problem in hand. Poorly defined goods are difficult to value under any circumstances whether the goods are traded in private markets or not (Carson, 1998). Secondly, we must choose the payment vehicle (entrance fee, increased taxes, voluntary contribution to a nature conservation fund, etc.) and elicitation method to obtain willingness to pay. Regarding question format or elicitation method, several options have been developed in CVM, from open-ended formats ("What is your maximum willingness to pay for good X?") to dichotomous-choice ("Answer yes or no to a specific amount offered") or a combination of both (often a dichotomous question is followed by an open-ended question) There is no unanimity regarding which question format elicits real willingness to pay. Hanemann (1994), states that the dichotomous format can eliminate many of the biases detected in open-ended formats. However, other authors argue that the open-ended format provides more reliable estimates (Freeman, 1993; Schulze, 1993) and that the dichotomous-choice may produce an upward bias since the starting bid given to the respondent provides him/her with information about the good under study (Schulze et al., 1996). And thirdly, information is obtained about the socio-economic characteristics of the respondents in order to estimate a value function in which the declared willingness to pay is explained by these characteristics and other pertinent variables.

The payment vehicle chosen was an entry fee per visitor. This seems to be the most neutral vehicle used in evaluating this type of environmental asset in Spain. In fact, previous studies - Riera et al. (1994), Rebolledo and Pérez y Pérez (1994) and Pérez y Pérez et al. (1996) - also applied this payment method. In any case, the vehicle chosen must be the one most suited to the study in question and avoid protest responses. Bennett et al. (1995), for example, used three different vehicles of payment and observed that the percentage of protest zeros was higher when the vehicle chosen was an increment to their annual Council tax as opposed to the other two alternatives (payment of an entrance fee per person and voluntary donation to a fund[32].

The elicitation method chosen was the discrete choice model, first introduced in contingent valuation analysis by Bishop and Heberlein (1979) and ratified by the NOAA (National Ocean and Atmospheric Administration) Blue Ribbon Panel (Arrow et al., 1993) given its popularity. However, with the aim of obtaining the respondent's maximum willingness to pay, a follow-up open-ended question was added. Four different bids were used: 200, 400, 600 and 800 pesetas[33] (which equivalents in Euros are, respectively, 1.2, 2.4, 3.6 and 4.8) based on the results obtained in the pre-test and in the pilot study where an open-ended question was used. As Clinch and Murphy (2001) pointed out, a larger number of bids would have allowed for greater accuracy in the estimation of the

32 Bateman et al. (1996) recorded a similar result.

33 The bids were in Pesetas because the survey was carried out before the Euro came into force in Spain. The exchange rate between the Peseta and the Euro is 166.386.

bid curve but each sub-sample would have been smaller, thereby leading to greater sampling error.

4.3 Theoretical model

The discrete choice model has become the most used approach for determining whether people are willing to pay for a non-market good. Since the CV responses are binary variables, we need a statistical model appropriate for a discrete dependent variable, such as that detailed in Hanemann and Kanninen (1996). In fact, when a household is confronted with a question to accept or reject a project that implies an environmental improvement from Q0 to Q1, we need to ask people about their willingness to pay to obtain the proposed change. However, the "yes" or "no" responses obtained provide only qualitative information about willingness to pay. Therefore, in order to obtain a measure of the willingness to pay we need a statistical model that relates the responses of the respondents to the monetary amount asked for. Hence, consider the following indirect utility function for a representative individual:

$$V = U(Y, S, Q) \qquad (1)$$

where Y is his or her income, S a vector of the socioeconomic characteristics of the individual (age, education, etc.) and Q the current state of the environment. Consider now a local policy that improves the environment such as that mentioned above. In this case, the welfare measure involved is given by the following equation:

$$V(Y-WTP, S, Q1) = V(Y, X, Q0) \qquad (2)$$

where WTP is the amount a respondent would be willing to pay to secure a welfare gain as a result of improving the environment (the change from Q0 to Q1.) This amount corresponds to the Hicksian compensation variation for the proposed change.
Now, following the seminal article by Hanemann (1984), if we assume that the utility function has some components which are unobservable to the researcher and are treated as stochastic, then the individual's utility function can be written as:

$$V(Y, S, Q) = U(Y, S, Q) + e \qquad (3)$$

where e is a random disturbance term with an expected value of zero. When offered to pay an amount of money A for a change in Q ($Q0 \neq Q1$), the individual will accept the offer if:

$U(Y\text{-}A, S, Q1) + e1 \geq U(Y, S, Q0) + e0$ (4)

where e0 and e1 are identically and independently distributed (i.i.d.) random variables with zero means. For the researcher, the individual's response is a random variable that will have some cumulative distribution (c.d.f) GWTP (A). Therefore, the probability that an individual will accept the suggested cost A is given by the equation below (Kriström, 1990a):

Prob {"yes" to A}= Prob (A £ WTP) = 1- GWTP (A) (5)

when GWTP (A) is the standard normal cdf, one has a probit model and when it is the standard logistic model the logit model is obtained.

4.4 Results

Based on the 419 positive observations (protest zeros were ignored[34]), an average willingness to pay of 590 pesetas (3.54 Euros) was obtained from the follow-up open-ended question (see Table 1). However, the highest values were obtained in Racó de l'Olla where the mean of the WTP was respectively 23% and 42% higher than the values obtained in Devesa and Jetty. Users that declined to pay (17%) argued that they should not be charged for access to a public good, that there are substitute natural areas near and that they pay enough taxes[35].

Table 1. WTP by site of survey

	Total sample	Devesa	Jetty	R. de l'Olla
Mean WTP in pesetas (equivalent in Euros) - confidence interval	590 (3.54) 552-629	515 (3.10) 465-565	565 (3.40) 491-639	735 (4.41) 657-813
Median	500	400	500	800
Mode	1000	200	1000	1000
Observations	419	212	82	125

Tables 2 and 3 show the analyses of the dichotomous question. There are different estimation procedures -both parametric and non-parametric[36]- to convert data on "yes" or "no" responses to a referendum question into a monetary measure. First, we have used the parametric model of choice of Hanemann (1984) which is consistent with utility theory and, essentially, referendum votes can be interpreted as a utility

34 Protest zeros were ignored because by definition protest bidders have a negative attitude toward paying for the environmetal good, but a positive attitude toward the good itself (Hoevenagel, 1994).

35 Follow-up questions receive special attention in order to discover the reasons for the answers (Portney, 1994).

36 For non-parametric procedures see Kriström (1990b).

maximising process. Second, we analyse the socio-economic variables which affect the likelihood of someone accepting the proposed bid. Since the values of a and b are 1.8673 and - 0.0024595 respectively in the logit model, the estimated mean WTP is 759 pesetas (4.56 Euros). Note that this outcome is considerably higher (28.6%) than the 590 pesetas (3.54 Euros) obtained using the open-ended question. In fact, an upward bias may arise when using the dichotomous elicitation format since the starting bid shown to the respondent provides information on the value of the environmental asset under study[37]. Hence persons who are not sure of their true valuation are highly likely to answer yes - the phenomenon known as "yea saying" (Kannimen, 1995).

Table 2. Probit and logit models of the dichotomous question without socio-economic variables

	Probit model	Logit model
Variable	Coefficient	Coefficient
Constant	1.1505 (7.567)	1.8673 (7.275)
Starting bid	-0.00151147 (-5.645)	-0.0024595 (-5.522)
Log likelihood function	-306.1911	-306.3581
X^2	32.8052	32.4714
Correct predictions (%)	61.7%	61.7%
n	496	496

t-values in parenthesis

Table 3 shows the models with socio-economic variables. The variables considered are:
STARTING BID: first price mentioned when asking the dichotomous valuation question.

VISITS: number of visits made to the park in the preceding year.

SIZE: size of the group of visitors to which the respondent belongs.

UNDER18: number of children under 18 with the respondent.

SITE1: 0-1 variable, whether the interview was conducted at site one (Devesa).

SITE2: 0-1 variable, whether the interview was conducted at site two (Jetty).

EDUCATION: 0-1 variable, whether the individual has university degree.

DATE: 0-1 variable, whether the interview was conducted in summer (July, August and September)

[37] The typical conclusion, it seems, is that the referendum format generates somewhat higher estimates of WTP (see Schulze et al., 1996, Boyle et al., 1996, Ready et al., 1996 Kriström, 1993). However, Kealy et al. (1988) present evidence that suggests no significance difference between the approaches.

DISTANCE: 0-1 variable, whether the interviewee travelled less than 25 km to the park. The negative sign of the offered bid means the higher the offered bid shown to the respondent, the lower the probability of him/her responding positively to the dichotomous question[38]. There is also an inverse ratio between VISITS and the interviewee's willingness to pay: the more visits he/she makes, the less likely he/she is to accept the proposed bid, a logical result since regular visitors to the park are more likely to be aware of their budget limitations than sporadic users.

The UNDER18 variable is highly significant and also negative, i.e. the higher the number of persons under 18 in the group of visitors, the lower the likelihood of the proposed bid being accepted. Indeed because many of the park's visitors are families it is reasonable to assume that children are a considerable financial burden taken into account when accepting or refusing the proposed starting bid.

Table 3. Logit-probit models of the dichotomous question with socioeconomic variables

	Probit model		Logit model	
Variable	Coefficient	t-value	Coefficient	t-value
CONSTANT	2,4253***	5,537	3,6584***	5,354
STARTING BID	-0,002491***	-6,668	-0,004086***	-6,453
VISITS	-0,010426**	-2,563	-0,017029**	-2,559
SIZE	0,044847***	2,611	0,073643***	2,576
UNDER18	-0,13935***	-2,687	-0,22911***	-2,667
SITE1 (DEVESA)	-1,1624***	-4,117	-1,8984***	-3,983
SITE2 (JETTY)	-0,95423***	-3,404	-1,5500***	-3,276
EDUCATION	-0,31516*	-1,884	-0,49684*	-1,753
DATE	0,89035***	3,451	1,4767***	3,411
DISTANCE	-0,47059**	-2,232	-0,77161**	-2,138
Log-Likelihood function	-274.4656		-274.9427	
c2	81.25108		80.29697	
correct predictions	69.45		69.56	
n	483		483	

*** $p < 0.01$, ** $p < 0.05$, * $p < 0.10$

38 See Gonzalez-Caban and Loomis (1997) and Bullock and Key (1997) for a similar result.

The negative sign of the dummy variables SITE1 and SITE2 shows that persons interviewed at Devesa and the Jetty are less willing to pay the proposed bid than those visiting Racó de l'Olla. Likewise the EDUCATION variable means that the higher a person's education, the greater the probability he/she will reject the starting bid. This atypical outcome[39] suggests that such visitors are more critical when determining their valuation and therefore give more thought to their reply than those without higher education. People living nearer the park show higher visit frequencies, therefore according with marginal utility theory the negative sign on DISTANCE confirms that they are less willing to accept the proposed bid.

Variables with a positive relation to willingness to pay on the other hand are the SIZE of the group and the DATE the survey was conducted. Hence the likelihood of the respondent accepting the proposed bid is greater when he/she belongs to a larger group and is interviewed in summer (holiday period).

It is interesting at this point to study which variables explain willingness to pay declared by visitors in the open-ended question, therefore a censored tobit model was applied using willingness to pay as the dependent variable and the respondent's socio-economic characteristics and other pertinent variables as independent variables[40]. In our case, if WTP* = bX + e, where WTP* is a latent variable with e ~ N [0, s2] , the observed variable WTP is censored with respect to WTP* such that:

WTP = WTP* if WTP* > 0
and (2)
WTP = 0 if WTP* < 0

The following independent variables were considered:
STARTING BID: variable as described in previous model.

INCOME: discrete variable on a scale of 0-12 indicating the net monthly income of the interviewees in intervals of 50,000 pesetas.

SIZE: variable as described in previous model.

AGE: continuous variable indicating interviewee's age.

SATISFACTION: continuous variable rated 1-10 according to degree of satisfaction (pleasant / unpleasant) obtained during the visit.

39 More highly educated persons are usually more aware of environmental problems and therefore have a higher WTP. There are however cases in literature showing an inverse relationship between level of education and WTP, see Danielson et al. (1995).

40 This relationship enables us to prove the theoretical validity of the contingent valuation method since the sign of estimated coefficients should be those expected by economic theory. There should be for example a significant and positive relationship between a person's income and his/her declared WTP otherwise the theoretical validity of the outcome would be dubious. (Bishop et al., 1995).

DISTANCE: variable as described in previous model.

DATE: variable as described in previous model.

SITE1: variable as described in previous model.

SITE2: variable as described in previous model.

The maximum likelihood parameter estimations appear in Table 4, showing that in this specific case, the offered bid is not significant. Therefore, it may be argued that interviewees do not perceive the given bid as information on the valuation of the environmental assets under study. This result must however be interpreted cautiously since the mode of each sub-sample coincides with the respective offered bid, which may suggest the existence of a starting point bias.

The INCOME variable has the sign expected by economic theory, indicating that the higher a person's income, the higher his/her willingness to pay. The SIZE of the group of visitors and the DATE of the survey also have a positive sign, hence the first variable means the larger the group, the higher the declared WTP; and the second, that persons interviewed in summer (July, August and September) give higher valuations than those interviewed in autumn. This phenomenon is possibly due to the visit being linked to other "special" summertime leisure activities. Likewise, the positive sign of the SATISFACTION variable shows that the people most satisfied with their visit give the highest valuations.

Table 4. Determinants of WTP

Variable	Coefficient	t-value
CONSTANT	372.66***	2.407
STARTING BID	0.147	1.241
INCOME	24.481 ***	2.347
SIZE	9.271***	2.8245
AGE	-5.220***	-2.827
SATISFACTION	30.232**	2.496
DISTANCE	-130.130**	-2.036
DATE	163.250*	1.945
SITE1 (DEVESA)	-270.080***	-3.209
SITE2 (JETTY)	-212.3630**	-2.363
Log likelihood function = -3133.006 n = 479		

*** $p < 0.01$, ** $p < 0.05$, * $p < 0.10$

The AGE variable on the other hand shows a negative sign, consequently the older the person, the lower the value assigned to the environmental asset. In fact younger people tend to appreciate environmental assets more, possibly due to their different education more in keeping with current conservationist attitudes, whilst having, logically, greater expectations of future use. The DISTANCE variable again shows that people living closer to the park declare lower willingness to pay due to their higher rate of visits.

The negative sign of SITE1 and SITE2 shows that people interviewed in Racó de l'Olla are willing to pay more than those interviewed in Devesa and the Jetty. We believe this stems from two reasons. Firstly, someone visiting this site is aware that the services available here are more than those of a strictly environmental nature, in comparison with the two alternatives sites. This result coincides with other empirical studies, e.g. Hanley and Ruffell (1993), which note that the existence of other services for visitors considerably increases the willingness to pay. Secondly, it may also be explained by the socio-economic characteristics of the visitors to this site (higher personal income, higher level of education, etc.).

Finally, it must be pointed out that the content validity[41] of a contingent valuation study can be affected by interviewer bias i.e. whether an interviewer influences interviewees' responses. Therefore following Smith and Desvougues (1986), we have included a dummy variable for each interviewer in the regression, to check whether he/she influenced the responses. The outcome was that these dummy variables are not significant so we deduced that in principle the respondents' responses were not affected by the interviewer's personal characteristics.

4.5 Aggregation

Using the findings of a contingent valuation study to obtain an estimate of aggregate individual willingness to pay for a specific quantity of a public good requires making several assumptions which are potentially troublesome. However, considering that there was no visits register when the survey was conducted, it is assumed that the park received 200,000 visitors this year. So, if we multiply this figure by the mean willingness to pay estimated from the dichotomous question, we obtain that the benefits derived from the recreational use of the park are Euros 912,366, whose equivalent in 2007 Euros would be 1,290,085. Likewise, if the mean willingness to pay value chosen is the one obtained from the follow-up open-ended question, then the recreational use value would be Euros 709,194 whose equivalent in 2007 Euros is 1,002, 800.

41 In contingent valuation studies, the content of a survey is valid if designed and conducted such that interviewees are induced to give bias-free responses. The aim is therefore to determine whether the material used in a survey (maps, photographs, plans, etc.) enable us to discover true WTP preferences. A question which encourages strategic behaviour for example would have lower content validity (Bishop et al., 1995).

When comparing costs and benefits of protected areas in a cost-benefit framework, these figures can be considered as modest and probably the costs can exceed the benefits. However, if we had estimated the existence value of the park (the most important value of a wetland) the result would have been the opposite. In fact, it is practically impossible to support a protection policy considering only the use benefits (Dixon & Sherman, 1990).

5. Conclusions and policy implications

When determining the costs and benefits of a specific environmental policy (like the measures proposed by the Water Framework Directive to achieve a good ecological status of surface water and groundwater in Europe), it is easy to overlook the benefits of the services provided by non-marketed goods given the difficulty inherent in their estimation. To overcome this problem, the economists have developed, in the last 50 years, different techniques to estimate these benefits with a high degree of reliability if some major guidelines are followed. However, like any other economic methodology, these techniques, although useful in this context, have their limitations and cannot on their own provide the definitive answer to any major issue (water targets established by the European Water Framework Directive, for example).

Among these techniques, the Contingent Valuation method is the most widely used in measuring the demand for non-market goods given its advantages over the indirect or revealed preferences techniques. However, this technique has at least two limitations in the Water Framework Directive context. Firstly, the willingness to pay estimators obtained strongly depends on the assumptions made about the underlying consumer preferences structure and the empirical model used in the willingness to pay inference process (Bengochea-Morancho et al., 2005). So the results obtained for assisting decision-making can have an excessive variability generating some uncertainty in this process. And secondly, we cannot forget that these studies are very expensive to carry out with a minimum guarantee of reliability. Therefore, if possible, sometimes benefits transfer could be a possibility given that some major characteristics of the river basins considered are very similar.

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The Current Crisis in Water Economics

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Introduction

Economics has been given a central role, notably in the implementation of the Water Framework Directive (WFD) in Europe, just when it is least able to fulfil this role. In part, this is because of the internal contradictions of the WFD, where the triple emphases on, a fixed standard, good ecological quality, economic analysis and stakeholder engagement are mutually incompatible. In particular, in conventional economics, value is given by individual preference and exists prior to each specific choice, and determines which option is chosen. However, by definition, stakeholder engagement is a social process in which the stakeholders argue, debate and negotiate values so values emerge as an outcome of the choice and as the product of a social process.

But there are in addition two other fatal problems for conventional economics when applied to water. First is that water is nothing like the good or resource which is assumed in the standard microeconomic textbooks. The second, which is a wider problem, is that a series of Nobel prizes for economics have been awarded for work which fatally undermines the foundations of neoclassical economics.

Why water is different

Water and water management have a series of characteristics which differ fundamentally from the assumptions about goods and resources made in any standard microeconomic textbook.

1. We manage water in order to make the best use of land; land without access to water is essentially valueless for most purposes. The most important of these uses of land is agriculture which is also the largest consumer of water. Whilst around 80l/d is adequate for all domestic uses for one person, with perhaps a further twice that amount being necessary for commercial and industrial uses, growing the food to feed that person requires between 3 and 6 tonnes of water, all of which is lost by the plants through evapo-transpiration (Falkenmark and Rockstrom 2004). In turn, this requires that water be a low cost input unless food prices are to rise substantially, where the biggest economic change in the last

2. hundred years is the fall in the proportion of household income spend in the developed countries on food from 50-60% (Reeves 1913) to the current 12-16%. When it is argued that water should be reallocated away from agriculture towards urban uses, it is implicitly being argued that too much food is being produced at too low a price. Conversely, the Millennium Ecosystem Assessment (Alcamo et al. 2003) argues that global food production will have to rise by 80% over the next 50 years. Whilst the relationship between yield and water use is not linear (Molder 2007), such an increase in demand will require a substantial additional diversion of water.

3. The textbooks assume that growth in demand is both inevitable and desirable. However, achieving sustainable water management is likely to involve driving some water demands downwards. Thus, the UK Code for Sustainable Homes (DCLG 2006) seeks to cut per capita domestic water consumption levels from the present 140-160 l/p/d to 80l/p/d. Conversely, arguments for water metering, for example, assume both that water demand will continue to grow and that metering will be relatively ineffective; that is, that it will, at best, stabilise demand rather than drive it significantly down from present levels.

4. The problem with water is not how much to produce but how to allocate essentially fixed supplies among competing uses. In addition, the amount of water available is frequently determined by land management decisions, and thus the proportion of rainfall converted to runoff, hundreds of kilometers away.

5. Catchments are systems so that the effect of any abstraction or discharge depends critically upon where it takes place; hence, the externalities associated with any such abstraction or discharge depend upon where within the catchment the abstraction or discharge takes place (Dwyer et al. 2006). Local optimisation can consequently result in sub-optimisation for the performance across the catchment as a whole.

6. As systems, catchments are temporally and spatially dynamic and have to be managed as such: an intervention intended to reduce problems at one point in space or time (e.g. extreme flows, floods) can have negative impacts at other places or times (e.g. by creating a low flow problem in dry periods) (Calder 2004).

In particular, the problem with water is to bring the distributions of supply and demand over time into alignment where the value of water is time varying (i.e. the value for water for irrigation outside of the growing season is zero). Hence, storage, in order to buffer changes in supply or demand, is a necessity in water management systems (Keller et al. 2000).

7. In addition, water is heavy and incompressible so moving water is energy intensive and hence moving it under gravity has been preferable to lifting it. The costs of pumping rapidly make lifting water for use uneconomic and even if the energy costs of desalination fall towards their theoretical limit, the energy costs of pumping water inland and uphill will still limit the extent to which desalination is helpful. Water bought from water vendors is typically more expensive than water supplied through a piped system (Solo 1999) and is to a considerable degree simply a reflection of the costs of using kinetic energy (human, animal or vehicle) as opposed to gravity to move the water.

8. The necessity for storage together with a preferential reliance on gravity to move water means that water management has been and still continues to be capital intensive.

9. Therefore, it is the ability to fund the capital costs that is the crucial problem. Hence, it is the ability of institutions to raise capital, and the cost of that capital, or to reduce that capital cost, which is the primary condition of success in water management.

10. Because water management is capital intensive, water is frequently a Ramsey good (Ramsey 1927); one whose average cost exceeds its marginal cost over the range of supply of interest.

11. In turn, short run marginal costs are frequently constant and may be negative. For example, sewers are designed to be self-cleansing above a specified flow; below that flow they must be cleaned to avoid blockages occurring (Butler and Davies 2004). Thus, the marginal cost of increased flow in sewers falls at some point and then is essentially constant until the limiting capacity of the sewer is reached. Again, once a reservoir is built and an aqueduct is constructed to distribute the stored water by gravity, the marginal cost of increased demand is essentially zero until the capacity of the reservoir and aqueduct is reached.

12. Consequently, marginal cost pricing does not permit the recovery of all costs and Ramsey's own solution to the problem, to load costs onto those consumers whose demand is least price elastic, will typically load those costs upon those with the lowest incomes.
13 The demand for water is to a significant degree technologically rather than behaviourally determined (Green 2003). In turn, prices are relatively ineffective in reducing demand in the short term (Cornish et al. 2004) and price elasticities tend to be low (Cavanagh et al. 2002). The price elasticities are sometimes no more than measures of the degree of market failure (Green 2003); assessments of water use in industry and commerce typically find that water consumption is 15-50% higher than would maximize profitability (Envirowise 2005).
14. There are frequently significant technical economies of scale in water management; physically bigger frequently does mean lower average cost (WRc 1977).
15. Historically, water management has been characterised by cooperative action rather than market or state action. Although Wittfogel (1957) claimed that "hydraulic civilisations" required a dictatorship, the evidence is instead that much water management has taken place through cooperative action. The Waterschappen of the Netherlands (Huisman et al. 1998), who essentially drained the country and then protected the land against flooding, are merely the best known of a widespread practice in Europe (Wagret 1967). Such Water User Associations are widespread across the globe; there are some 12,000 to 18,000 in Germany (Pant 2000); 10,000 in France (Garin and Loubier 2002); 6,200 in Spain (Garcia nd); and some 18,000 water related districts in the USA (US Bureau of Census 2002). If municipalities are taken to be multi-purpose user associations, the history of urban water management is similarly one of cooperative action (Hietala 1987). An important question is why this has occurred rather than market provision being the norm.

The result of these differences is that analyses which uncritically follow the textbooks produce very little that is of practical value to water management.

Competitive markets

Definitions of economics (Robbins 1935; Samuelson 1970) essentially have two components:

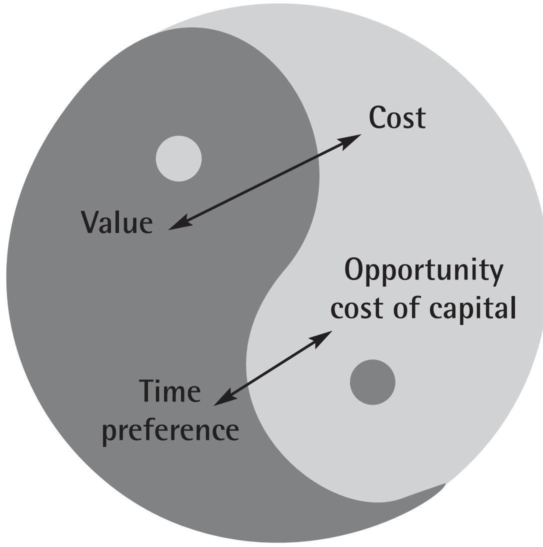
- That economics is the application of reason to choice; and
- In doing so, we should seek to make the best use of available resources.

In this Yin and Yang of economics (Figure 1), neoclassical economics elegantly joined the two through the concept of the perfect competitive market. Through this concept, the idea of value and cost are equated and fall out of the market in the form of a price. But curiously there is only an incidental relationship between the Producer Surplus and fixed costs.

Again, discounting combines two entirely different concepts, preferences for the distribution of consumption over time and the opportunity cost of capital into a single measure through the assumption of a perfectly competitive market. The focus of the intellectual effort in neoclassical economics has, perhaps in consequence, focused upon elaborating the concept of perfect competitive markets and the necessary supporting concept of General Equilibrium.

The negative side of this effort is that because the concept of perfect competitive market solves all the problems, it has not been necessary to think about those problems. For example, although the question what is the nature of value was of concern to the classical economists (O'Brien 1975), it is common now to assume that whatever value is then it is measured by willingness to pay, as measured in conditions as near to those of a perfect competitive market as it is possible to achieve. A simple and rather hollow shell has been left of what is a rich and difficult issue. Thus, in Hicks' (1946) "Value and Capital", the term "value" does not appear in the index at all, and it gets only a glancing mention in chapter one on the way to developing an ordinal theory of preferences.

Figure 1 The Yin and Yang of economics



But, as Boulding and Lundstedt (1988) noted, in everyday speech "value" refers to a guiding moral principle or an endstate. Those values are defined either as social relationships (e.g. justice, democracy) or in relation to others (e.g. honesty, liberty). Our use of the term "value" in economics comes close to fraudulent misrepresentation when used in the public arena because it is being used in a sense which diverges so far from its everyday meaning.

The practical problem with the neoclassical approach of defining value as instrumental is that consequently it is necessarily actions, not things, that have an instrumental value (Green 2003), and hence to a large measure we are trying to explain behaviour by itself. If a hat has a value because of what you can do with it, the behaviour of buying a hat is done in expectation of the value of the actions that can be performed with it. Moreover, in this case, I have many hats because they are different, and rather than there being declining marginal utility of hats, it is that there are fewer and fewer differences between individual hats that makes it less likely that I will buy another one. Only where each unit is a perfect substitute for another does declining marginal utility apply. Fortunately, water is a perfect substitute for itself although there is seldom even an imperfect substitute for water. The second practical problem with the concept of instrumental value is that any action necessarily has as many instrumental values as objectives are engaged by the choice. Only once those objectives are completely ordered can there said to be a single instrumental value (Green 2003).

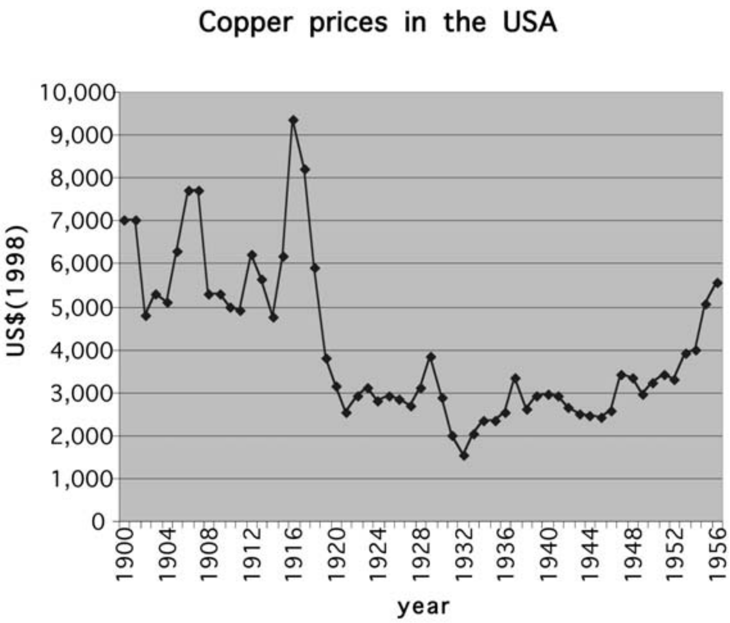
On the other side of the coin, since a perfect competitive market solves the problem of optimising the use of resources, it has not been necessary to think about what is a resource, or rather how resources may differ.

In developing and refining this beautiful intellectual construction of the perfect competitive market, we have had to sacrifice reason for rationality. Arrow (1987) has pointed out that in conventional language, rationality refers to a process of reasoning, akin to Toulmin's (1958) definition of argument, whereas in neoclassical economics, rationality means no more than consistency of outcome. One consequence is that economics is no longer in the position to advise people how they should make a choice, but only as to what choice they should make. Given the focus now upon stakeholder engagement, it is precisely the first task upon which economics should be focused.

Perfect competitive markets also have some very convenient properties: they are not only optimising but homeostatic. They embody a rather nineteenth century view of the world when increasingly it appears that homeostasis and single equilibrium are not natural but atypical. Thus, that very simple linear systems can demonstrate chaotic behaviour (Gleick 1987) and perhaps the majority of systems are complex and exhibit behaviours which are the antithesis of those assumed in the neoclassical model (Ball 2005; Cohen and Stewart 1994; Kauffman 1995). Adam Smith's (1986) time series of wheat prices show very large year to year variations and do not obviously demonstrate a convergence to any form of equilibrium (Green 2003).

Again, the Knightian concept of uncertainty as being completely different from probability (Knight 1921) has had to be sacrificed so that now uncertainty and probability are treated largely as synonyms. This is to assume that it is possible to make meaningful probability statements about the future; a claim which Keynes (1937) denied, using the example of copper prices in twenty years' time. Examination of copper prices in the U.S.A. (Figure 2) suggests that, firstly, he was correct, and secondly, that attempts to understand the future in probabilistic terms are unhelpful. One post-hoc explanation of copper prices in 1956 is that copper has a number of desirable physical properties (malleability, low electrical resistance and high heat transmission) but it also forms some very useful alloys (bronze, brass). This made it useful for armaments (bronze and brass), for electrical and telecommunication equipment (high electrical conductivity), and for the transmission of heat. Demand in 1956 would thus be expected to depend upon the extent of rearmament, together with the output of the construction industry, coupled to the output of the electrical and electronic industries, the degree to which copper was replaced by other materials in those industries, and the degree to which those industries were themselves superceded by others. Trying to explore what would affect copper prices might thus be a more productive exercise than trying to attach probabilities to particular price levels.

Figure 2 Copper prices in the USA
(Source: Kelly and Matos 2005)



This is to follow Davis's (2002) argument, which underlies the scenario approach adopted with great success by Shell (van der Heijden 1996): "We need to do this for a future that is essentially unknowable – but not unthinkable." The mere fact that the future is essentially unknowable does not reduce the necessity to act, the problem in choice is to decide how to choose and what to choose in the certain knowledge that the future is unknowable. To make choices on the basis that the future is merely risky, although comforting, may be dangerously misleading.

The wider crisis in economics

A succession of Nobel prizes for economics have been given for work which each prize winner asserted undercut the basis of neoclassical economics. Thus, in his laureate speech Coase (1988) observed: "If I am right, current economic analysis is incapable of handling many of the problems to which it purports to give answers." Similarly, North (1990) asserted that: "The theory employed, based on the assumption of scarcity and hence competition, is not up to the task. To put it simply, what has been missing is an understanding of the nature of human coordination and cooperation." Consequently, Stiglitz (2001) was able to claim: "Today, there is no respectable intellectual support for the proposition that markets, by themselves, lead to efficient, let alone equitable outcomes."

The problem for economics is that once the perfect competitive market is gone, there is very little left and we are faced with confronting the two separate questions of what is value and what is cost? Something can now have a value without having either a cost or a price, nor is there any necessary equation of the three. Once the perfect competitive market is gone, so too are we confronted with the question of: what is the scope for substitution both in resources and in consumption? At the same time, whilst neoclassical economics with its technical claims to be able to deliver optimal outcomes was ideally suited to the needs of a technocratic bureaucracy, it is ill-equipped to address the needs of a deliberative democracy where the stakeholders want to know not what choice to make but how to make a "better" choice.

Whilst the fundamental question for the stakeholders is what do they mean by "better", implicit in that concept is change. This change may be adaptation to changing conditions, or a desire for invention and innovation as compared to the pre-existing options. It is consequently about learning to do better. One consequence of seeking to do better is that the past is only relevant to the extent to which we can learn from it, often to do something different from what we did before. Conversely, neoclassical economics denies learning and simply assumes all the real problems, through assumptions of consumer rationality and perfect competitive markets. But it is precisely those real problems with which the stakeholders need help in order for them to decide what the best available option is.

On the resource side of the equation, we similarly have problems in part because the conventional financial investment metaphor has been extended beyond its limits. That is, that all decisions have been treated as if they are equivalent to one, say, as to whether or not to buy a new machine tool: an initial upfront capital cost is to be balanced by a long term stream of output from which must be netted the operational and maintenance costs. In this setting, the concept of opportunity cost is another elegant one. But, underlying it is the assumption that all costs are changes in resource costs. In practical cost-benefit analyses, some of the changes are positive changes in the availability of resources (e.g. lower requirements for labour) and others are undesirable changes in the availability of consumption (increased noise, reductions in the recreational value of a river).

The assumption that all good changes will be on the output or consumption side of the equation and all bad things on the input or resource side of the equation does not stand. Whilst it is well-known that the benefit-cost ratio is sensitive to whether changes are treated as "negative benefits" or costs, and "negative costs" or benefits, the argument for using the benefit-cost ratio to choose between options within a capital constraint (Brent 1990) also falls when the cost side of the equation includes reductions in consumption.

The second problem is to define the nature of the resource in question. The Hicksian concept of income (Hicks 1946) necessarily also provides a definition of capital; as that which produces a Hicksian income. The practical problem is that many of what have to be considered in choices are not capital items but better described as a durable. A machine tool will not produce a Hicksian income because it wears out, and it wears out, through physical and chemical processes of decay, irrespective of whether or not it is used: it is not therefore Hicksian capital. Money capital is interesting abstraction, but some of the important details of reality are lost.

Secondly, what we are concerned to evaluate is potential conversions of one resource to another (e.g. trees to paper), substitutions (e.g. timber for brick in construction), production (e.g. the rate at which a forest will produce timber), and exchanges (e.g. the sale of trees). These are quite different actions but are often treated as equivalent. Equally, the extent and rates at which the first three can occur are determined by the physical, biological and chemical properties of the materials in question as expressed through the available technologies. More generally, these possibilities can be quite limiting; nitrogen cannot be substituted for phosphorus, water or labour in agricultural production for example (Loomis and Connor 1992). Similarly, although copper, nickel and zinc are adjacent on the Periodic Table, the scope for substituting nickel for copper is quite limited (Hill and Holman 1995). In construction, timber can be substituted for stone to an extent but this requires radically different constructional forms, requiring different technologies to exist. Our production functions consequently tend to assume a much greater scope for substitution than actually exists. Equally, when we argue that it is simply necessary to get the prices right, this is to assume that this will be sufficient to call the necessary new technology into existence.

Thus, the application of the opportunity cost of capital argument for discounting, whilst logical, requires careful analysis of what is capital and what is income. Moreover, on the time preference side of the argument, we have yet to absorb Modigliani's work (Modigliani and Brumberg 1954) into discounting. This work destroyed the presumption that people are always short-sighted and always prefer consumption now rather than in the future. More widely, the limitation of conventional discounting is that it only calculates some weighted average of the area under the curve of net annual benefits over time. Mathematically, for a given discount rate, there are an infinite number of curves that will yield the same area. Should the individual or society have a preference for the shape or trajectory of net annual benefits over time, this cannot be handled by conventional discounting approaches (Penning-Rowsell et al. 1992; Green 2003).

So what do we do?

For economists, the important question is: what do we do now about economics? For everyone else, the critical question is in the current position where economic theory is essentially bankrupt; what should be done now to improve water management? Fortunately, the two questions are linked: we are trying to make "better" choices, so the obvious and fundamental question is what do we mean by "better" both in the specific case and in general? The starting point is: why do we have to choose, what is choice? Elsewhere, I have argued (Green and Penning-Rowsell 1999; Green 2003) that choice only exists when there are at least two mutually exclusive options, and at least one reason to prefer one option and at least one reason to prefer another. Hence, that the two necessary conditions for the existence of a choice are conflict plus uncertainty. Thus, that choice is a process in which we seek to resolve the conflicts that make the choice necessary and become confident that one option ought to be preferred to all others.

In one respect, the abandonment of the belief in perfect competitive markets to solve all problems automatically liberates us. An increasingly widely expressed view in water economics (Cornish et al. 2004; Green 2003; Molle and Berkoff 2007) is that the three claims as to what prices can do in a perfect competitive market are, in reality, best treated separately. Those claims are:

- As to the allocation of resources between competing users
- The recovery of all costs
- Changing the behaviour of producers and consumers.

For practical problems, where a market does not exist and so prices must be invented, using a single set of prices to solve all these problems simultaneously is an ambitious exercise, and assertions that a single price schedule can solve all three problems is an optimistic assumption.

Taking the three issues individually, using price as the sole allocation privileges one form of power (Lukes 1974; Weber 1968), income, over all other forms of power. It is not self-evident that either reliance on this form of power is superior to all other forms of power, whether it satisfies the principles of procedural equity (Lawrence et al. 1997; Lind and Taylor 1988), or that the resulting distribution, substantive justice (Lloyd 1991), will be fair.

Cost recovery for Ramsey goods poses particular problems. Equally, because water is necessarily a low value good, where agriculture cannot be purely rainfed, transaction costs of charging for water can loom very large. As Coase (1988) observed, the efficient outcome is often determined by the relative magnitude of the transaction costs associated with different solutions. Thus, the first test of water metering is whether any associated reductions in water demand are sufficient to cover the additional costs of metering, costs which are non-trivial (Green 2003). A number of logical ideas for charging for water services, such as for surface water drainage by the area of impermeable surface on a property (Federal Ministry for the Environment), and charging for wastewater by the polluting loads (OFWAT 2000) have been restricted in application because of the very high transaction costs involved. Attempts at approximating charges for water services to demand have a further disadvantage with Ramsey goods: they increase the revenue risk and hence the cost of capital.

The presumption in neoclassical economics is that prices always work and nothing else does in terms of changing behaviour. This leaves the cupboard bare if prices are not very effective, and allows us to avoid the interesting question of under what circumstances prices will be effective and when they will not be. In practice, the processes of consumer choice, as explored in marketing (Schiffman and Kanuk 1994) and applied microeconomics, are much more interesting and it is a problem for economics that marketing has largely discarded economic theory and instead draws upon the experimental and theoretical frameworks of psychology, sociology and anthropology. In the field of water management, the results of the World Bank's NIPR initiative (Wheeler et al. 2000) are fascinating, as are Andersen's (1994) conclusions of the varying degree of success of economic instruments in water pollution abatement in Western Europe.

For economists, perhaps the key question is the one defined by North (1990): we need a theory of cooperation in order to determine when competition is more efficient than cooperation and vice versa. More generally, we need some explanation for the existence of societies and the enormous quantity of resources which individuals have devoted to constructing and maintaining complex social systems. This construction suggests that the benefits of cooperation are best achieved, not through single acts but through a continuing relationship. In turn, the critical willingness to pay question becomes: what amounts are people willing to pay to benefit other people and for what? Arrow's Impossibility Theorem (Arrow 1963) appears to imply that collective choice is impossible, but it is only impossible because it denies the stakeholders the possibility of negotiation and leaves them nothing to negotiate about. If choices can be chained together, and the stakeholders can negotiate across the choices, then it is conceivable that an overall outcome will be achieved that is a Pareto Improvement. In order to chain the choices together so that each stakeholder is prepared to negotiate across the chain of choices, procedural equity may be the key conditionality.

In this regard, it was probably a mistake to start with individual choice and take individual choice as the basis for a general theory of choice. In practice, individual choice is better considered to be the trivial case of choice: individuals spend most of their time in groups of one form or another. As a starting point, Sprey's (1969) model of household choice as a zone of cooperative-conflict looks a much more promising starting point as the basis for a general theory of choice.

The stakeholders have to focus on the question of what is a "better" choice in each individual choice and economists need to explore the wider issues in order to support the stakeholders. For the stakeholders, I would recommend that we focus upon governance (UNDP 1997) issues, both structures in the form of institutions, and processes in the form of stakeholder engagement. The core issues are about what are social relationships or what should they be: all the fundamental human values such as justice, liberty, and democracy are about the nature of social relationships. Concepts of justice or equity in particular have two elements: procedural justice and substantive justice (Lloyd 1991). The shortest definition of equity is "a moral principle consistently applied" (Green 2003), where the moral principle that ought to be applied in a particular case is commonly contested. The concept of economic efficiency is simply another claim as to the moral principle that ought to be applied in collective choices. It is a claim that the objective of such choices ought to be some aggregate of individual preferences. This is a claim which obviously is widely contested in both philosophy and religion.

We not only have to think it, we have to do it: we can seek through stakeholder engagement to resolve the conflicts and to narrow down to one option as being the best available. Defining this as the problem is one thing; resolving it, being successful at stakeholder engagement, is another. This is hardly a new definition: governance is essentially what Aristotle (1932) defined as politics. In part, the rebranding exercise is now necessary because the nature of politics has fallen into disrepute. Because it is social relationships which are central, and we articulate the procedural aspects of justice through the social act of language, we will have to become very good at conversations (Stone et al. 1999).

Framed as social relationships, the parallel with Rittel and Weber's (1974) claim that social science is confronted with "wicked problems" becomes apparent. Rather than resolving a single isolated problem, what has to be managed over the long term is the relationship in which each specific "problem" emerges.

Both aspects imply that in seeking water management we need to focus upon the twin aspects of governance: structures, or institutions, and processes, stakeholder engagement. In doing so, we should recall Keynes' remark that economics is a form of analysis and not a body of dogma.

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Part Three

Confronting the policy demand with the supply: A general discussion

Costs and benefits for supporting regulatory actions: How are costs and benefits integrated in EC impact assessments?

Uffe Nielsen

1. Introduction

The European Commission (EC) has started to undertake impact assessments of all major proposals integrating assessments of environmental, economic and social impacts from 2003. This system was initiated not least with a view to provide a more thorough and balanced input to guide policy makers' views about advantages and drawbacks expected from proposed EU regulation. This is a challenging task. Even with unlimited resources available to analysts, it will always be inherently difficult to get a full and clear picture of the wide range of different types of impacts that prospective regulation may have – not least when the specific scope and type of regulation is not yet clearly defined. This is exacerbated by limited resources for such assessment activity. Thus, trade-offs in the scope and depth of impact assessments are fundamentally unavoidable. This makes it important to ensure that analysis in impact assessment analysis is "proportional" to the problem at hand, as also underlined in the EC impact assessment guidelines (European Commission 2005).[42]

The question of how to ensure such proportionality in practice is not an easy one, but even so, it is not addressed in very detailed form in the 2005 EC impact assessment guidelines. However, one expected outcome of the currently ongoing evaluation of the EC impact assessment system could be further emphasis on how to ensure proportionality in practice in future impact assessments (European Commission 2006).

Fundamental questions that one may ask with respect to proportionality of analysis in practice are:

- What range of EC initiatives to cover under the impact assessment system?
- How many policy options should be analysed in a given impact assessment?
- How many impacts should be analysed?
- How detailed should the coverage othese impacts be?

[42] It is useful to make a distinction between "treaty proportionality" and "impact assessment proportionality" (European Commission 2005). "Treaty proportionality' refers to the policy analysed in the impact assessment: is the policy proportionate to the problem at hand, or are other actions more adequate? "Impact assessment proportionality" is related to methodology: How much effort should be invested in assessing the effects of the policy in question? In the remainder of this chapter, focus will be on "impact assessment proportionality".

A specific challenge with regard to proportionality of impact assessments is how to ensure an integrated perspective – i.e. making sure relevant impacts within environmental, social and economic impact categories are addressed. In principle, the ideal way of ensuring an integrated perspective – at least from a welfare economic point of view – would be to compare impacts on as equal footing as possible, i.e. through establishment of a common measurement unit of different impacts, through e.g. economic valuation of environmental impacts in monetary terms (Pearce 2001). This is the principle underlying a full cost-benefit analysis. Given the need for impact assessment proportionality, such a type of exercise would be deemed overly ambitious in many instances.

The analysis underlying impact assessments will always entail some uncertainty, not least due to the inherently uncertain nature of *ex ante* analysis of expected future outcomes. However, a specific feature of proportional analysis is that – by definition – it will imply further limitations in scope and/or less detailed analysis in some areas. This will necessarily have implications for the certainty with which impact assessments can conclude compared to a more full analysis.

Based on the results from a review of 58 EC impact assessments in 2004 and 2005, this chapter will focus on proportionality, with special emphasis on these two aspects of impact assessments: (1) The integration of analysis of costs and benefits in environmental, social and economic domains, and (2) the treatment of limitations and uncertainty of impact assessments. Although this paper relies heavily on Nielsen et al. (2006), where the overall results of this review are presented and discussed, its focus will be narrower on proportionality, integration of costs and benefits, and on uncertainty.

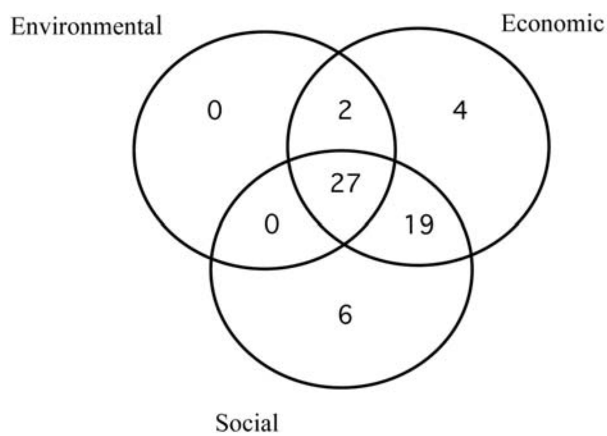
2. Results from a review of 58 EC impact assessments

The review, which is presented in more detail in Nielsen et al. (2006), is based on a checklist mostly focusing on methodological issues. This was applied to 58 impact assessments published by the EC in the period from 2004 and until 1 October 2005. The review constitutes what Harrington & Morgenstern (2004) term a "content test" of the impact assessment system – i.e. a review of what is included in the impact assessments. Thus, the review does not cover the degree to which impact assessments succeed in describing the main impacts relevant in the given impact assessment context. Nor does the review check the quality of the analysis as such. The sample of impact assessments cover a wide range of types of EC initiatives, both with respect to how far in the policy cycle the initiative is, what is the scope of the initiative, what type of regulation is considered, and what main policy area is covered.

2.1 Integration of costs and benefits in EC impact assessments

The results of the review indicate that the range of impacts covered by impact assessments typically is narrow: less than half of impact assessments (27 out of 58) cover environmental, economic as well as social impacts (see Figure 1) and 29 out of 58 do not cover environmental impacts at all.

Figure 1: Impact assessments with different combinations of environmental, economic and social impacts.



(Nielsen et al. 2006)

When only half of the impact assessments address environmental costs and benefits, this hardly qualifies as an "integrated perspective" of all relevant costs and benefits. It could of course be argued that (1) many of the EC initiatives covered may not actually be expected to have any major environmental impacts, and (2) the need for proportional analysis would necessitate lower emphasis on certain main impact areas.

Both arguments are valid, but in the context of the 58 impact assessments covered, very little argumentation – or for that matter documentation – is to be found to explain why certain impacts have not been covered. Therefore, it is difficult to see whether such omissions are deliberate and warranted, and what implications they have.

A transparent scope of impact assessments is not only relevant with respect to relative coverage of main impact areas, but also with respect to the broadness of coverage of specific types of impacts within a main area, and depth of analysis of these specific impacts (e.g. degree of quantification in analysis of impacts). The 58 impact assessments covered in the review generally display a low average number of specific impacts covered per impact assessment (one type of environmental impact, three economic impacts and two social impacts per impact assessment on average). Within the environmental field, these specific types of impacts comprise for example “air quality”, “soil quality”, “climate”, etc. Again, it is only rarely discussed why these specific types of impact have been covered, and others not. Thus, even when impacts have been covered in economic and social as well as environmental domains, the scope of this coverage is often limited.

With respect to the depth of analysis of impacts, the review distinguishes between impacts only briefly mentioned, more detailed qualitative coverage of impacts, quantification of expected impacts, and monetisation of impacts. It is found that most impact assessments do not contain quantitative analysis at all (39 out of 58) – and monetary quantification only takes place in 17 out of 58 impact assessments.

When monetisation is undertaken, it only covers some of the impacts described, since no impact assessments have performed monetary quantification for all impacts. Thus, even though some quantification does take place, it may be difficult to apply the result in terms of direct comparison of impacts – e.g. via cost-benefit analysis – since the monetary quantification supplied in the impact assessment gives an incomplete measure of the overall net benefits or costs.

Furthermore, the low level of quantification is particularly striking in the social and environmental domains, where economic valuation of environmental impacts only have been attempted in two impact assessments (one of which was the impact assessment underlying the Thematic Strategy on Air Pollution and the Directive on Ambient Air Quality and Cleaner Air for Europe, see chapter by Paul Watkiss).

There may be several possible reasons for this: lack of quantification skills, genuine lack of reliable data, lack of resources, or a combination of these factors. Whether this has been made with no deliberate consideration of the significance of the initiative covered, or whether it reflects impact assessment proportionality applied in practice remains an open question, since this question is not generally addressed directly in the impact assessments covered.

As mentioned, it is not surprising that all costs and benefits are not quantified, but unless this is communicated clearly, there will be an implicit danger that policy makers inadvertently focus more on quantified impacts. This is so, both because this may lead to a mistaken perception that these are in fact the main impacts, or because numbers generally may convey a false sense of certainty.

Thus, quantification should not necessarily be seen as an indicator of impact assessment quality. Given the need for proportional analysis and an integrated perspective on impacts, it may be even more important that the most important impacts are identified and potential trade-offs between environmental, economic and social impacts are discussed in impact assessments, than that some of these impacts are quantified. Thus, the fact that 39 impact assessments solely apply qualitative analysis need not be seen as worrying.

However, on average, only four impacts are addressed in the exclusively qualitative impact assessments compared with eight impacts on average in the impact assessments with some quantification. Furthermore, impact assessments without quantification are overrepresented in the group of impact assessments that only assess impacts in one or two of the three main categories of impacts (environmental, economic and social impacts). For example, all 11 impact assessments that only cover impacts in one of three main categories are exclusively qualitative.

When discussing qualitative coverage of impacts, the distinction between a serious qualitative discussion and simply mentioning an impact briefly is useful. For seven impact assessments, the only impacts addressed are briefly mentioned. For example, a formulation like the following: "The impact of the proposed measures will be felt firstly from an environmental or ecological perspective through the improvement in the state of certain important fish stocks" from the impact assessment on "Proposal for a Council Regulation Establishing a Community Fisheries Control Agency" adds at best only marginal value to the impact assessment, since no order of magnitude for the impact is supplied, the importance of it compared to other impacts is not addressed and no references are cited.

Overall, these observations therefore contribute to a picture of a set of exclusively qualitative impact assessments with relatively little direct value in terms of identifying main impacts or identifying trade-offs.

2.2 Coverage of uncertainty and limitations in EC impact assessments

Whenever an impact assessment covers impacts with only minor detail or no detail at all, the analysis – by definition – has certain limitations, and therefore also has implications for the certainty with which the impact assessment can conclude on recommended policy options. At the same time, it is also relevant to assess the certainty of quantitative estimates – since quantification is no guarantee that estimates are more certain, e.g. due to limitations in data availability and the methods used for quantification.

Given limits in scope and limited level of detail of analysis in a significant part of the 58 impact assessments (as discussed above), some deliberations about limitations and uncertainty would be expected, both in general, but most importantly in the conclusions of the impact assessments. However, this is not generally the case. On the contrary, only 9 out of 58 impact assessments mention uncertainty with respect to assumptions or data, and only 2 out of 58 impact assessments address sensitivity issues. Only one of the impact assessments not employing any quantitative analysis refer to limitations of the analysis due to incomplete information or the availability of data.

This very low incidence of a clear statement of limitations is not only surprising because of the prevailing low level of detail of analysis in impact assessment, but also since 22 impact assessments themselves mention that they do not consider the currently available information to be sufficient. This point to a gap between limitations in the results of the analysis undertaken in impact assessments due to data gaps, uncertainties and proportionality and how these limitations are reflected in the conclusions of the impact assessments. Reservations are not communicated fully.

Is this warranted from a proportionality perspective? Is it merely the result of a proportional coverage of uncertainty and limitations? This should be seen as a fallacy, since decisions to limit the depth or scope of analysis due to proportionality considerations has direct implications for uncertainty and limitations of analysis. Thus, the attention devoted to issues of uncertainty ought to be ensured at all levels of proportionality. Given that proportionality may imply a low degree of quantification of impacts, this attention to uncertainty may not be possible in the form of quantitative sensitivity and uncertainty analyses. However, it is important that the impact on the certainty of results of not quantifying results is discussed thoroughly (Mogensen et al. 2007; Nielsen et al. 2006).

3. Conclusions

An integrated perspective on the impacts of regulation necessitates that relevant costs and benefits are covered in a balanced way – i.e. including coverage of relevant economic, social and environmental impacts. Based on a review of 58 impact assessments carried out by the EC in 2004 and 2005, there is reason to question whether the EC impact assessment system ensures a sufficiently balanced and integrated analysis of the impacts of prospective regulations. This conclusion is based on the observations of a generally limited overall scope as well as a limited detail of coverage of social and environmental impacts.

Ideally, decision-makers would like to know the full costs and benefits of all Commission initiatives before they are implemented. In practice, this is a difficult, if not impossible, exercise. Therefore, it is important to at least assess the most important costs and benefits, and to do so at a sufficiently early stage in the policy process to allow this information to influence decision-making.

Often, social or environmental impacts will indeed not be the most important – but there is no reason to assume that this will be a general pattern. If the observed generally low coverage of environmental and social impacts is instead a symptom of a lack of knowledge or data surrounding such impacts, this should rather be used as an argument for further work on narrowing this knowledge gap, when such impacts are indeed important. If there are considerable uncertainties surrounding how important the impacts are expected to be, this could be used as an argument in favour of using further efforts to gather more data. This should of course always be seen in the perspective of where the added value of increased efforts is greatest. Alternatively, full transparency about what potentially important impacts are not covered – or only qualitatively covered – should be seen as a minimum requirement of a truly proportional and balanced analysis.

One concrete way to address this issue would be to require impact assessments to include statements which could invite challenges from stakeholders, for example, "We are not aware of any evidence suggesting impacts on X, so this issue was not investigated further" (Nielsen et al. 2006). Such formulations would make it easier to persuade policy-makers that the impact assessment indeed is a balanced and integrated analysis of the most relevant impacts in the environmental, social and economic fields. In other words, it would increase the value of a given impact assessment if it is more explicit about what proportionality (and other) considerations lie behind the inherent choice of what impacts to include or not.

Clearly, more elaborate analysis, covering many policy options, many impacts, and with detailed quantitative analysis of these impacts should almost by definition lead to more certainty about conclusions from the analysis (unless the area assessed is fundamentally beset with high uncertainties). This means that a limited scope or level of detail of analysis, whether due to conscious proportionality considerations or not, will most often mean less certainty of conclusions and more limitations of analysis.

An important dimension of a balanced and proportional analysis is therefore to ensure full transparency about the limitations and uncertainty surrounding the analysis. This also involves transparency about the degree to which quantitative data covered in the impact assessment are in fact among the most important impacts – in order to avoid misplaced emphasis on these impacts, simply because they are communicated in quantitative form.

Given that full quantification will rarely be possible, the concept of proportionality needs further attention and more concrete guidelines. When no quantification takes place, there should be the same or even higher demands for documenting which impacts are relevant. It is essential that restrictions of scope of analysis due to proportionality in impact assessments are elaborated and explicit. The limitations following from this should also be reflected in the conclusions of the impact assessments. Hopefully, the currently ongoing evaluation of the EC impact assessment system will lead to increased emphasis in this area in the future.

Although there may be weaknesses in the current practice of the EC impact assessment system, it still has the potential to become an important instrument for integration of environmental, economic and social concerns into EU decision-making in a systematic and transparent manner.

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Comparing ex ante and ex post cost assessments of environmental regulations – some lessons
Jakub Koniecki

(The content of this article purely expresses the views of its author and should not in any way be attributed to the European Commission)

In 2006, the Directorate General for Environment at the European Commission undertook a project aiming at assessing the accuracy of *ex ante* estimates of costs of environmental policy. The project, which this paper builds upon, has been prepared by a consortium of think tanks (lead by IVM, full list in the annex) and consultancies, with input from experts sought during an expert workshop[43].

Introduction

A fully fledged policy assessment would normally look at least at two elements: effectiveness and efficiency. Effectiveness shows whether policy objectives have been met, efficiency – shows whether they were met at least costs. In addition, there is normally a comparison of costs and benefits for different levels of ambition. Within this distributional effects or impacts on environmental equity at large can also be considered[44].

The scope of the exercise we undertook at DG Environment was focused on the costs to businesses. This may seem like a rather restrictive approach, but for an assessment of EU environmental policy (entire or parts of it), an estimate of costs to businesses will provide important inputs for any other further steps. Imagine, we introduced a new air emission standards on the power sector. If we can have accurate cost estimates at the plant level, we can go further to see how the whole sector responds, what part of costs are passed on and to whom, and finally, assess effects on the whole economy[45]. But experience shows that this path is not always that simple; nevertheless, without having assessed costs to businesses it would be difficult to go any further.

43 Please see the full project results at *http://ec.europa.eu/environment/enveco/others/index.htm#ex_post*

44 For a detailed discussion on assessing distributional impacts please see The Distributional Effects of Environmental Policy, ed. by Ysé Serret and Nick Johnstone, OECD 2006, *http://www.oecd.org/document/10/0,2340,en_2649_37419_36171914_1_1_1_37419,00.html*

45 A good illustration of employing costs to business as input for further analysis can be found at "Sectoral and macroeconomic impacts of the large combustion plants in Poland: A general equilibrium analysis" by Olga Kiuila and Grzegorz Peszko *http://www.sciencedirect.com/science/article/B6V7G-4JS20CW-1/2/79053b77a29d210051b7281c70e63470*

Analysis of benefits has been deliberately excluded from this project, except for direct to-business ones, such as savings from energy efficient appliances that would lower overall compliance costs. Environmental policies often benefit the public at large, while the costs often are concentrated on specific sectors. Finally, benefits are often visible in far longer term than costs. In short, estimation of benefits requires quite a different methodology than estimating costs and is the subject of analysis elsewhere.

There is a general scarcity of *ex post* studies assessing the efficiency and effectiveness of environmental policy instruments. For instance since 2002, major EU policy proposals are subject to an impact assessment procedure that is to provide an *ex ante* assessment of costs and benefits of a proposed measure. There is no such requirement to always check the *ex post* results. Some evidence shows great disparities between *ex ante* estimates of costs and their *ex post* outturns[46].

Assessing costs *ex post* allows a number of lessons to be learnt for policy-making. Imagine that *ex post* costs turned out to be much higher than estimated at the time of policy design. This may have adverse impacts not only on businesses competitiveness, but also – in extreme cases – lead to lower compliance and, as a consequence, lower environmental improvements. On the other hand, if compliance costs are understood to be higher than actually is the case, then they may affect setting the policy ambition, and environmental benefits associated with the policy change may be delayed, or foregone entirely.

Compliance cost estimates are also crucial in the political process associated with regulatory change. Insofar as compliance costs can never be known with absolute certainty, it becomes a matter of management of uncertainty and understanding the probability of alternative cost outcomes. If there is a significant uncertainty about the numbers, such as costs and benefits, these then become the focus of the debate, rather than the regulatory change itself[47]. The bargaining over the numbers takes over discussion on objectives and means. *Ex post* analysis offers a better understanding of the issues, and of how to manage the potential biases in the cost estimation process offers the prospect of improved efficiency (lower costs, reduced uncertainty) of regulation.

[46] Mind the gap!, Comparing *ex ante* and *ex post* assessments of the costs of complying with environmental regulation, Bailey P., Haq G., Gouldson A., European Environment no. 12, p. 245-256, 2002. See also: "Costs and Strategies presented by Industry during the Negotiation of Environmental Regulations", Stockholm Environment Institute, 1999; and: Eames, M. The Large Combustion Plant Directive (88/609/EEC): An Effective Instrument for SO2 Pollution Abatement?: chapter in "Implementing European Environmental Policy: the impacts of the Directives in Member States", M. Glachant (ed.), Edward Elgar, 2001.

[47] A good illustration of why we should not be too obsessed with exact numbers is betting on a football game. Imagine you put your money in, say, European championship qualifications, and team A (guests) wins 3:0 over team B (hosts). What happens during the game is that in, 89 minutes it is a draw, say 3:3. Then there is a penalty for the guests. Then a crazy football fan makes it to the pitch and hits the referee. The final decision may be to annul the goals scored by the hosts and the official result would be that the guests have won three to nil. You would take money back from the sweepstake. However, if you do not know what really has happened during the game, you may tend to overestimate the team A and underestimate team B. Your future betting is at risk.

The project objectives

The project had two broad objectives. Firstly, to improve policy design (to reduce costs) by learning some lessons from the *ex post* assessment of economic efficiency and distributional consequences of selected legislation. Secondly, to improve future *ex ante* assessments, and notably the methods for assessing *ex ante* costs through a comparison of *ex post* compliance costs assessments with available *ex ante* assessments and focusing on understanding the compliance mechanism and costs drivers rather than aiming at precise compliance costs figures.

The main lessons stemming from case studies and literature reviews can be grouped into those related to costs estimation methods and to processes in which these estimates are carried out.

Lessons on methodology

There seem to be two main methods of estimating compliance costs *ex post*: by building up a counterfactual scenario or by looking at changes in unit costs of specific technologies.

The first method tries to compare effects caused by a given policy measure to a business as it would have been under the usual scenario. In principle the same rules apply as to building scenarios when doing *ex ante* estimates; a number of assumptions needs to be made that can have a bearing on the results. Obviously, the regulatory change must be significant enough to allow back-casting.

The second method is to "simply" compare *ex ante* estimates (e.g. based on unit costs) to *ex post* results. A word of caution needs to be said about the *ex post* data. It is usually only as good as *ex ante* data. Both data sets are just estimates obtained by employing certain methodologies; it is very rare that we have detailed costs data from all individual entities affected.

Main lessons below follow applying the second method and concern various types of costs and costs drivers. Innovation appears to be the main underestimated factor, which leads to incorrect costs analysis.

Definition of costs

Definition of costs is a key issue when assessing impacts of regulation. Often discussions between regulating authority and businesses relate to what costs should be included and what should not[48]. The literature review for this project revealed that there are no uniform costs definition or methodology applied in measuring costs of environmental policies. There are also no uniform discount rates (if there are any at all) and system boundaries vary, with externalities taken account of at random. The biggest difficulty, however, is that it is not always clear in *ex ante* estimates what was the approach applied.

Defining what compliance costs are can be difficult. If a new environmental standard coincides with a natural investment cycle of a related industry, should the costs of upgrading production chains be regarded as compliance costs? Or if a company optimises its use of chemicals in order to protect employees? In both cases the answer should be no. Compliance costs[49] are those that aim at protecting (or improving) the environment and are additional to business as usual practice.

Compliance costs are, though, relatively easier to identify when a regulation imposes a given technological solution. For instance, the large combustion plant directive 88/609/EC was expected to result in power plants investing in flue gas desulphurisation units. Costs of running such an end-of-pipe installation will normally almost always be additional to "normal" company operation.

More difficult to measure are those costs that result in changes in production processes. In recent years, the European chlorine industry has been switching to mercury-free membranes, which create less mercury pollution and at the same time improve the efficiency. In such cases, resulting compliance costs might be close to zero (as benefits of technology improvement and pollution reduction costs outweigh each other)[50].

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[48] For a detailed discussion on an analytical framework for assessing costs of environmental policy to business please refer to Commission Staff Working Document SEC(2004) 769 http://ec.europa.eu/enterprise/enterprise_policy/industry/doc/sec_769_2004.pdf

[49] Environmental protection expenditure is the money spent on all purposeful activities directly aimed at the prevention, reduction and elimination of pollution or any other degradation of the environment. EPE does not include:

- Activities that, while beneficial to the environment, primarily satisfy technical needs or health and safety requirements.
- Expenditure linked to exploitation of natural resources (e.g. water supply).
- Calculated cost items such as depreciation (consumption of fixed capital) or the cost of capital.
- Payments of interest, fines and penalties for non-compliance with environmental regulations or compensations to third parties.

Activities such as energy and material saving are only included to the extent that they mainly aim at environmental protection. One example is recycling which is included to the extent that it constitutes a substitute for waste management http://epp.eurostat.ec.europa.eu/cache/ITY_OFFPUB/KS-NQ-05-010/EN/KS-NQ-05-010-EN.PDF

[50] In this case the well-known Porter discussion appears. If companies were efficient, no such improvement would be possible. If however, companies are floating away from their optimum, then a regulatory change can wake them up and help to wipe out inefficiencies. A useful overview of a history of this discussion was done i.a. by SQW in their project for DEFRA "Exploring the relationship between environmental regulation and competitiveness" in 2006.

The project revealed that the confusion often arises around the following issues:

- direct and indirect costs;
- investment and operational costs;
- unit costs versus total costs;
- discount rates, prices, etc;
- measuring costs of integrated technologies, production process changes, etc.

Direct and indirect costs

Some environmental regulation can result in setting production limits or preventing business from further expansion. A company, by selling less (now or in future) will face costs (materialising as income loss). Should these costs be included in compliance costs? This could be problematic. Firstly, these forgone profits should also be corrected by reduced expenditure on production factors (such as raw materials, labour, operational costs, etc.). Secondly, these negative effects will most likely be offset somewhere else in the economy. If production of “dirty” products is restricted, substitution will occur and companies providing “greener” products will gain. Estimating these effects would normally require applying complex general equilibrium models. A robust estimation of resource costs in almost all cases makes the estimation of indirect effects irrelevant.

Investment and operational costs

Investment costs normally relate to the capital costs of purchasing equipment (or modifying existing equipment) to meet the regulatory requirements; it can generally be regarded as a one-off expenditure, although it can also be annualised (depreciation or discounting). Operational costs will relate to costs of material, energy, labour, and other current expenditures. Additionally, there might also be some administrative costs, such as reporting[51].

The first two categories of costs often interact – installing new equipment may imply hiring additional staff (or making some redundant if the new technology is more efficient) and higher/lower energy bills, etc. Both categories will be reflected in companies’ accounting books as annual costs (investment being depreciated), yet drivers behind them are different, therefore care should be taken when collecting data directly on business level.

[51] Please see proceedings from a conference organised by European Commission Directorate General Environment: "Better Regulation and Outcomes for the Environment" , in March 2007 http://ec.europa.eu/environment/better_regulation/conference_march07/index_en.htm

Unit costs vs. aggregated costs

Equally important are unit costs versus total costs and handling uncertainties about total population size. Total *ex post* costs may appear to be smaller due to incomplete compliance or due to lower than estimated pollution by the companies (e.g. because of a general economic recession). However, the unit costs (e.g. expressed as €/ t of abatement, or costs per company, etc.) might have been estimated correctly; it is just the population size that has changed. The reverse can also happen: population is estimated correctly, but the unit costs are higher than estimated. In both cases, the result would be an over-estimation of *ex ante* costs, but due to completely different factors.

Population size can be particularly difficult to estimate when it comes to highly dynamic businesses, such as SMEs, or new emerging sectors of the economy. It is one of the factors that is important when making any costs extrapolations, say, from one EU member state to all, as business structures can vary significantly.

This is not to say that estimates should be always put into unit costs. When showing overall policy effects aggregated costs will be required to compare them with overall policy benefits. On the other hand, when assessing policy affordability or when searching for vulnerable populations (such as SMEs), costs per company compared to its turnover or profit will be more appropriate.

Innovation

Whereas handling uncertainties about the population size and defining clearly what costs are being measured seem rather straightforward (although in practice not easy), factoring innovation in *ex ante* estimates is more difficult. Firstly some of the technological developments are difficult to anticipate (e.g. development of combined cycle gas turbine (CCGT) that replaced flue gas desulphurisation (FGD) for large combustion plants). A conservative *ex ante* estimate will normally take as an input abatement costs of existing technologies (or the ones well advanced). Break-through innovations will not be taken into account; even if some policy instruments are more likely to prompt innovation the uncertainties of scale and speed of this effect happening can be significant[52]. Secondly, costs for emerging technologies are likely to fall alongside economies of scale, but it is difficult to set the point in time when it will happen. One

52 For a more detailed discussion on effects on innovation of environment policy please see: Innovation dynamics induced by environmental policy, IVM report for DG Environment, November 2006 http://ec.europa.eu/environment/enveco/others/index.htm#innodyn

of the reasons for difficulties in estimating this point is that businesses are rather unlikely to invest time and money in checking likely technological developments until it is certain that new standards will be required. A conservative approach will lead to overestimation of compliance costs. The available evidence suggests that innovation can contribute to annual costs reduction between 4% and 10%; often both to improved efficiencies of technologies, and to lower price.

Process related

The factors presented above relate to methodology. This is by no means an exhaustive list. However, when doing *ex ante* estimates, there are also issues related more to the analytical process that can affect robustness of the results.

Time of measurement and type of regulation

Ex ante estimates are normally carried out for draft policy measures, at the beginning of the legislative process. This is particularly visible in the EU, where there is a significant number of actors empowered to amend an original policy proposal put forward by the European Commission. Both Council of Ministers and the European Parliament can significantly change the original provisions, however, they are not required to carry out assessment of impacts of these changes[53]. Furthermore, instruments allowing some degree of flexibility (the majority of the EU environmental law takes the form of directives) can be differently applied in the Member States and even differently inside the Member States where there is a certain degree of devolution.

On the other hand, involving relevant stakeholders early upstream in the process of forming new policy measures brings a bigger chance to obtain accurate results. Preliminary findings can be tested and fine-tuned. Even if stakeholders are not involved (which would be a bad practice), the time lag between disclosing first policy draft and the final adoption will give them opportunity to prepare better for the regulatory change.

Type of policy measure will also matter. Effects, in particular distributive, of economic instruments may be more complex to assess than those of regulation prescribing specific technologies.

[53] Inter-institutional Agreement between the European Commission, European Parliament and the Council addresses this problem: http://ec.europa.eu/governance/better_regulation/ii_coord_en.htm

Information sources

The main sources of information about costs of compliance are businesses and they are often reluctant to reveal data. This may be because businesses are wary of regulation as information is commercially sensitive. Additionally, businesses do not generally invest time in assessing theoretical compliance options, unless there are incentives to do so. Hence, there is an information asymmetry and the business- to-be-regulated will reveal only as much (little) information as suitable at a given moment.

These difficulties will also have to be faced when doing *ex post* assessment. Time lag between formulation of a policy proposal and its implementation[54] can be quite long, and assessing the full effects of a given policy would require extending this period even further. Business memory tends to be shorter.

An alternative source of information may be suppliers of pollution abatement technologies. The accuracy of information from this source can also vary. On the one hand, as the case study on IPPC shows, the efficiency of some technologies can be overestimated. This is particularly the case when new technologies have only been applied in test plants that can be subject to unique conditions. On the other hand, suppliers of abatement technologies, when placing their products on the market, will rather give a safe margin regarding the guaranteed efficiency, which in reality will often be higher. However, suppliers might have a limited knowledge about application of changes in the production processes other than end-of-pipe installations.

A source that is not normally exploited when searching for costs data is various registers related to public funds, such as subsidies. For instance, when assessing behaviour of agricultural holdings, registers with direct payments and other agricultural subsidies can be helpful.

Laws overlapping

It is often difficult to untangle the impact of a single piece of legislation and define what marginal compliance costs are. As demonstrated in the case study on packaging and packaging waste, the problem is particularly acute when EU legislation is introduced in the area where member states have a great variety of existing systems and requirements. Singling out compliance costs for a given piece of legislation will be also

54 For instance, the Integrated Pollution Prevention and Control Directive, entered into force in 1996, sets requirements for so-called existing plants as late as October 2007.

difficult for business, as it is expected to comply with a whole package of policy measures.

Clearly, “weight” of a policy measure will also matter. Effect of traditional command and control instrument imposing relatively significant costs on particular economic sectors will be easier to assess than impacts of some broader policy guidelines.

Additionally, there are some laws that for particular sectors are closely interlinked - for instance, the Large Combustion Plants Directive and National Emission Ceilings Directive are crucial for power plants. At the same time, fulfilling requirements of one of those directives to a large extent satisfies requirements of the remaining one(s).

Results from the case studies

Literature review revealed that in general compliance costs for business are overestimated, whereas costs for public sector are often underestimated. Overall, costs tend to be overestimated. Case studies confirm the tendency to overestimate compliance costs for business but also point out the difficulty in correcting *ex ante* estimates. The case studies also provide useful insights in to the underlying problems of differences between *ex ante* and *ex post* estimates.

Large Combustion Plants Directive (LCP)

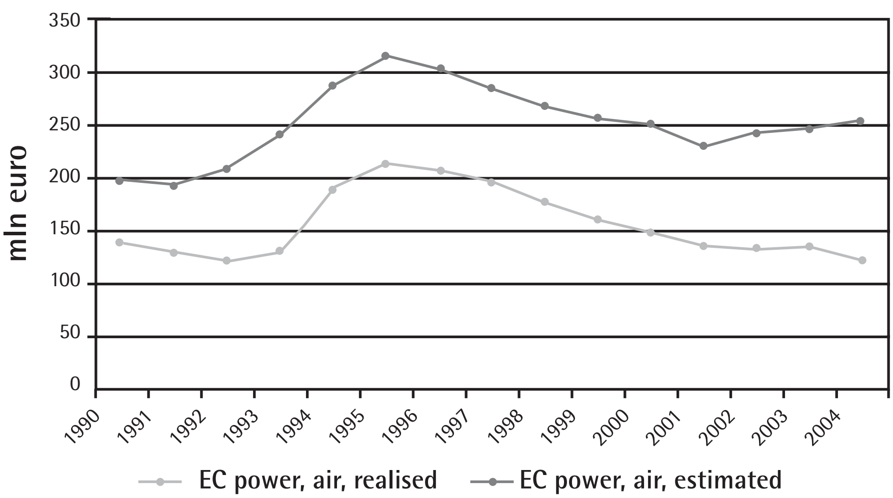
The Large Combustion Plants Directive (LCPD) 88/609/EEC applied to combustion plants with a rated thermal output ≥ 50 MWth. It set national emission ceilings for emissions of SO2 and NOx from existing LCPs, and absolute emission limit values for SO2, NOx and dust for individual new installations based on Best Available Technology (BAT). The directive has been reviewed in 2001, setting compliance dates for 2008 and 2016. That is why the case study concentrated on the old directive. As no data was available for EU-15/25, UK, Germany and Netherlands experiences have been examined.

The case study revealed that a new technology emerged (combined cycle gas turbine) which was cheaper than the one assumed in *ex ante* assessments (flue gas desulphurisation). As a result, only about half of the installed capacity in the UK had to be retrofitted with expensive FDG (flue gas desulphurisation). Whereas total costs for the sector and the economy were lower than *ex ante* estimates, the compliance costs for the units that finally had to apply the FGD was fairly accurate.

It also revealed that careful consideration should be given to choosing a proxy for measuring effects of regulatory change. In the UK, electricity prices at power plants were chosen. They were estimated to increase 25-30%, while an *ex post* check demonstrated increase by only about 2.5% to 5%. However, a number of external factors come into play, such as the above-mentioned fuel switch or liberalization of electricity market. This demonstrates how vulnerable the baseline scenario can be. Flexibility of the LCP (Large Combustion Plants) directive (which allowed adopting either a national reduction programme or standards at plant level) and privatisation of the UK power sector also contributed to lowering the compliance costs.

In the Netherlands, the overestimation of *ex ante* cost estimates of about 1.5 is explained by the fact that these were carried out taking into account existing technologies or small-scale applications of some new technologies, that only later brought cost reductions due to economies of scale. It is worth noting that whereas the costs were overestimated, the breaking point after which costs started decreasing had been estimated correctly.

Environmental costs for air protection in the power sector, the Netherlands, estimated and realised (1990 – 2004) (price level 2004)



Source: TME (2005), estimate of costs of reducing SO2, NOx and dust in the power sector; does not necessarily refer to costs of implementation of the LCP directive

Integrated Pollution Prevention and Control Directive (IPPC)

The IPPC directive concerns industrial plants and agricultural holdings that have a significant impact on the environment. Each installation is delivered an integrated permit, which takes into account "best available technology" (in terms of abatement or resource efficiencies) and which is intended to make it difficult for the operators to switch pollution between different media (for instance causing more water pollution due to air quality limits, etc.). The IPPC directive applies to a wide range of sectors, yet very few costs estimates are available; one of the reasons might be that the provisions of the directive concerning existing installations (cut-off date was 1999) enter into force only in October 2007. At the same time, costs data might be commercially sensitive. Because of the limited availability of data, the case study was limited to the ceramic sector in the Flemish Region of Belgium, and options for SOx reductions. Ex ante data was mainly gathered from the suppliers of abatement technologies, whereas *ex post* data was obtained from the installations that were subject to regulation.

The results reveal that the investment costs (of flue gas treatment) were estimated within a reasonable margin of 20%, however, the operational costs were overestimated. On the other hand, emission reduction efficiency turned out to be lower than expected. Thus, excessive optimism of the technology suppliers corrected over-pessimistic investment and operational costs estimates. The limited scope of this case study does not allow for formulating any general conclusions on the implementation of the IPPC directive, however it once again demonstrates that information provided by stakeholders from one interest group (here technology suppliers) needs to be verified before being employed in policy formulation.

Ozone depleting substances

The ozone depleting substances were gradually phased out under the Montreal Protocol and the EU Regulations (which set shorter deadlines than the Protocol) 3093/94/EC and 2037/2000/EC. The main concern of the industry was that the substitutes to the banned substances would continue to be significantly more expensive. Another concern was that switching to other substances will require significant time lags. Neither of these concerns became real. Substitutes became available in shorter time and at lower prices, also the substitution process was quicker (though not equal in all Member States). The study revealed some evidence for first mover advantage that US industry benefited from, contrary to some of the EU companies.

On the aggregated level, the *ex ante* costs estimates were overshot by a factor of 1.4, while for some individual cases this factor was as high as 40. Costs for downstream users were overestimated by about 1.25. Administrative costs were also substantially lower. The main reason for such a result seems to be that industry, if confronted with a situation where future uncertain technological developments are the main costs driver, prefers to take a conservative line.

Environmental standards in transport

This case study examined a number of environmental regulations concerning the transport sector, such as fuel quality and exhaust standards. It has been limited to Netherlands only, due to data availability. Nevertheless, assuming that there is a similar level of environmental ambition in many other EU Member States and that the car fleet structure is similar, the results can be regarded as fairly representative, at least for the EU-15.

At aggregated level, the compliance costs were overestimated by about factor 2. Yet, the factors behind were different and ranged from a greater innovation to a slower implementation. The main cost elements were unit costs of abatement equipment. The *ex ante* estimates failed to factor in such elements as improvements in technology,

Measure or vehicle concerned	**Fuel**	**Factor of comparison between ex post and ex post and ex ante estimates**	**The main reason for the difference between ex ante**
Passenger and light duty vehicles	LPG and petrol	2	Decrease in unit costs €771 in 1985, €285 in 2000
Passenger and light duty vehicles	Diesel	5	Decrease in unit costs €400 in 1997, €275 in 2001
Heavy duty	Diesel	1.4	Overestimation of costs of particle traps
Unleaded petrol production costs 2000	Petrol	6.3	Decrease in additional €0.02 in 1990, €0.004 in
Low sulphur diesel	Diesel	Possible underestimation	Ex-ante estimates, quantities and additional costs

Source: TME 2006, *http://ec.europa.eu/environment/enveco/ex_post/pdf/transport.pdf* efficiency and economies of scale. At the end, the price of cars supplied by the industry was much lower than originally expected.

Packaging and packaging waste[55]

The Directive 94/62/EC on packaging and packaging waste sets common recycling and recovery targets for the Member States. It is also intended to tackle trade barriers in recovered material. As in many other cases of EU environmental legislation, this directive sets the objectives, but the means of achieving them are left to the Member States.

This case study illustrates the difficulty in assessing effects of an EU policy measure that was introduced on top of existing policies in the Member States. Two broad conclusions can nevertheless be formulated. Firstly, the impacts of the directive, however modest at the time of its introduction, seem to be growing (as the packaging waste problem might well be in some MS). This would suggest that "value-added" is growing and, if the *ex post* estimate is repeated, it would be easier to identify the impacts. Secondly, the impacts of the directive seem to be very unevenly distributed; modest in those MS that have had their own national measures prior to the EU regulations.

Nitrate pollution from agricultural holdings

The objective of the nitrate directive 91/676/EEC is to prevent nitrate run-offs from agricultural holdings. Some animal husbandry (mainly cows, pigs, poultry and sheep) through manure generated can lead to overloading soil with nitrate. It can then cause water pollution, leading among other things to water eutrophication, and negative impacts on human health (*http://ec.europa.eu/environment/water/water-nitrates/pdf/eutrophication.pdf*).

This case study shows significant differences of annual compliance costs (measured either at €/ha or at €/kgN) between the countries (Denmark, Finland, France, The Netherlands, The United Kingdom, Lithuania and Croatia) as well as among industries. The historical rate of fertiliser application seems to play a role. Direct comparison of *ex ante* and *ex post* estimates have been possible only for Denmark, and the Netherlands reveal overestimation at about factor 1.4-2.7. This can be explained by the efficiency gains in fertiliser management that could offset part of the costs. Two other findings follow from the study. Firstly, in managing excessive manure, transport costs play an

[55] For a more comprehensive evaluation of MS policies see the European Environment Agency report: http://waste.eionet.europa.eu/publications/wp2_2005

important role, and these are obviously independent of stringency of nitrate regulation. Transport costs are also easier to assess *ex post* as the actual transport destinations and patterns are revealed. Secondly, the costs per farm vary dramatically depending on its activity. For instance, there are some extensive dairy farms that are net importers of manure, while other farms pay for having it disposed of.

Conclusions

As we can see, the *ex ante* compliance costs to business tend to be overestimated. However, the occurrence and gravity of underlying factors are different and should always be analysed case by case. The most common include unexpected "leap-frog" innovations, underestimation of strength of existing innovation trends, asymmetry of available information between business and government and difference between planned and actually implemented policies.

The results of this project can lead to different conclusions, depending on who will interpret them. The greens will see it as a proof for the regulating authorities being too conservative and setting environmental objectives too low on the basis of overestimated business costs. The business will be partially relieved, but may remain worried, because if the government is constantly overestimating the costs of policies, what is the guarantee the same is not happening with benefits estimates? The policy makers will probably reply that in order to have better estimates they need better input from all stakeholders.

Differences in *ex ante* and *ex post* estimates are due to several factors: unexpected or underestimated technical innovation (which tend to reduce compliance costs); asymmetry of information between industry and government; and overestimates which are linked to the fact that the environmental improvements turn out to be lower than assumed, e.g. because a somewhat different policy was actually implemented than the one for which the *ex ante* estimate was made. A distinction should be made between total costs and costs per unit of improvement.

Broadly speaking these factors can be divided into two groups, depending on their behaviour in *ex ante* estimates. The "misbehaving" would be all significant external factors, influencing policy costs, yet independent. Liberalisation of electricity market and rising supply of gas as a fuel for power plants in the UK provide an example.

Breakthrough innovations would also fit in that hard-to-predict category. Whereas such factors are difficult to forecast, most can be known *ex post* with great accuracy. Therefore a remedy here would be to carry out sensitivity analysis, so as to be completely clear to what extent they influence costs estimates.

A somewhat more "behaving" group would include population size, unit costs and pace of innovation. Most estimation errors related to those factors can be avoided by applying greater care in the *ex ante* analysis. Yet, in some specific cases, such as mapping out SMEs populations, *ex post* analysis will face the same difficulties as *ex ante*.

These policy measures whose impacts are uncertain, yet could be significant, should have built in mechanisms allowing for monitoring the effects in question. Such mechanisms often imply additional costs, but it might be worth investing in.

In short, there are many traps awaiting anyone who will engage in *ex ante* and *ex post* analysis. Some are more manageable than others, some not at all. Perhaps the final one to avoid is focusing too much on very fine technical aspects, as this brings a risk of having the big picture blurred and clouded. After all, it is better to be vaguely right, than precisely wrong[56].

56 http://www.who.int/hac/techguidance/tools/disrupted_sectors/module_02/en/index6.html

Annex 1
Overall Case Study Results

Directive (Sector)	Ex-ante/Ex-post		Shortcoming in Ex ante
	Upstream	Consumers	
LCPD (Power sector)	2 (Germany)	6-10 (UK)	The introduction of CCGT made the high cost of FGD in the UK unnecessary
IPPC (Belgium Ceramics)	>1.2 (OPEX) ~1.1 (CAPEX)	-	Optimistic estimates by suppliers broadly cancelled out other pessimistic components of the *ex ante* estimates
ODS (Ozone Depleting Substances)	2.5 (1.4 -125)	1.25	Resistance and conservative technological assumptions by the chemicals industry and use sectors
Transport	2 (1.4 - 6)	-	Ex ante failures to predict technological advancements
Packaging	-	-	Complexity in the way the Directive was implemented did not make this comparison possible
Nitrates Directive (Agriculture)	~2	-	Possible efficiencies in nitrates use (and some costs savings).

Annex 2
Information about the project and the authors

Title	Authors	Link
The final report on *ex post* estimates of costs to business of EU environmental legislation	Edited by Frans Oosterhuis (IVM) Reviewed by Reyer Gerlagh (IVM)	http://ec.europa.eu/environment/enveco/ex_post/pdf/costs.pdf
Case Study on the Large Combustion Plants Directive	Véronique Monier and Cécile des Abbayes (BIO)	http://ec.europa.eu/environment/enveco/ex_post/pdf/lcpd.pdf
Costs of compliance case study: Packaging & Packaging Waste Directive 94/62/EC	Andrew Jarvis and James Medhurst (GHK)	http://ec.europa.eu/environment/enveco/ex_post/pdf/packaging.pdf
Ex ante and *ex post* costs of implementing the Nitrates Directive	Onno Kuik (IVM)	http://ec.europa.eu/environment/enveco/ex_post/pdf/nitrates.pdf
Ex post estimates of costs to business of EU environmental policies: A case study looking at Ozone Depleting Substances	Robin Vanner and Paul Ekins (PSI)	http://ec.europa.eu/environment/enveco/ex_post/pdf/ozone.pdf
Case study Road Transport	Jochem Jantzen and Henk van der Woerd (TME)	http://ec.europa.eu/environment/enveco/ex_post/pdf/transport.pdf
Ex-post estimates of costs to businesses in the context of BAT and IPPC	Peter Vercaemst, D. Huybrechts and E. Meynaerts (VITO)	http://ec.europa.eu/environment/enveco/ex_post/pdf/ippc.pdf

Part Four

Costs and benefits for supporting regulatory actions- practical illustrations

Lessons from the Hazardous Waste Combustion MACT Standards: A Practitioner's Application of Cost and Benefit Analysis in Evaluating Environmental Regulations

Cynthia J. Manson

Abstract

The paper discusses the methods and results of the cost-benefit assessment undertaken for the U.S. regulation of toxic air emissions from industrial facilities burning hazardous waste. The analysis reveals the challenge of performing economic assessments, including data and resource limitations and attention to the legal and policy context of the analysis. The purpose of this case-study-driven discussion is to outline practical approaches that have proven useful in addressing the various issues encountered. The discussion focuses on identification of effective cost modeling approaches and use of a range of tools to characterise economic impacts and inform a discussion of benefits when the economic and scientific literature do not support monetised estimates.

Economic analysis of environmental regulations and policies is almost always a complex task, and in many cases the resulting analyses can be frustratingly incomplete. Challenges include mundane constraints such as limited project resources, coupled with limited process and cost data from the regulated community, and frequently, incomplete information to support measurement and valuation of the environmental benefits that are the focus of the policies. The central challenge for a practitioner in performing an economic analysis thus becomes one of targeting a limited analysis to assess the specific costs and benefits that are most likely to influence key policy decisions. The goal is to optimise the analytic approach such that analyses and results clarify the economic tradeoffs inherent in environmental policy, and characterise the key uncertainties that could affect the success of the policy.

This paper is intended as a resource for practitioners faced with crafting approaches to assess the costs and benefits of environmental policies. It describes one general strategy for performing regulatory analysis in the face of real-world constraints: an informal "value of information" approach that relies on a series of screening-level analyses to focus more detailed analysis on key variables and to characterize uncertainties. To frame the discussion and highlight common issues and limitations, the paper chronicles the economic analysis of a recent large-scale U.S. rulemaking targeting hazardous air pollution. The Hazardous Waste Combustion MACT Standard encompasses many of the complexities and limitations that are common to large-scale environmental policies. It represents a complex rulemaking with significant costs and uncertain (but probably significant) benefits, and the analysis of the rule illustrates the policy and market complexities that can overshadow benefit-cost analysis.

Introduction

On December 19, 2005, the U.S. Environmental Protection Agency (EPA) completed a rule that implemented more stringent emissions standards on U.S. “hazardous waste combustors” or facilities that burn hazardous waste, either to fuel operations or as a disposal method. The rule is one of several "MACT[57] Standards" implementing "maximum achievable control technologies" to reduce hazardous air pollutant releases under the Clean Air Act.[58] The "Hazardous Waste Combustion MACT Standard" (HWC MACT) governs releases from hazardous waste incinerators, hazardous waste-burning cement kilns and lightweight aggregate kilns (LWAKs); and hazardous waste-burning boilers and industrial furnaces at a range of facilities, including process heaters and hydrochloric acid (HCl) production furnaces. Consistent with MACT standards issued for other industries, the rule addresses releases of hazardous air pollutants, including dioxins/furans, mercury, semi-volatile metals such as lead, low-volatile metals, particulate matter, chlorine gas, carbon monoxide, and hydrocarbons.[59]

57 Maximum Achievable Control Technology

58 Rule citation: NESHAP: National Emission Standards for Hazardous Air Pollutants: Standards for Hazardous Air Pollutants for Hazardous Waste Combustors, Federal Register: December 19, 2005 (Volume 70, Number 242). Rules and Regulations, Page 75042-75047. The MACT standards are promulgated under Section 112 of the Clean Air Act, as amended (CAA). Section 112 of the Clean Air Act requires EPA to promulgate MACT standards for major sources (facilities) emitting hazardous air pollutants, and for other facilities where the Agency finds that these sources present a potential threat to human health and the environment.

59 Hazardous air pollutants addressed by MACT include the following: dioxins/furans; total chlorine (including hydrochloric acid and chlorine gas); mercury; semi-volatile metals (including lead and cadmium); low-volatile metals (including arsenic, beryllium, and chromium); particulate matter (a surrogate for antimony, cobalt, manganese, nickel, and selenium); and carbon monoxide and hydrocarbons (surrogates for non-dioxin, non-furan toxic organic emissions).

The 2005 final rule ended a decade-long effort that included a separate, earlier (1999) rulemaking. The final 2005 rule revised the 1999 standards and included additional facilities (boilers and industrial furnaces).[60] The combined costs and benefits of the rulemakings as reported in the economic analyses are presented in Exhibit 1. The discussion in this paper focuses on the 2005 analysis, but also notes key findings and issues related to the initial 1999 analysis.

One conclusion is clear from these results: based on monetised estimates of benefits, the rule appears to be economically inefficient. In part this is a result of the fact that the ability to monetise - even quantify - benefits associated with reduced hazardous air pollutants is severely constrained. This reduces the utility of a traditional analysis of net benefits (or calculation of a "benefit-cost" ratio), and leaves the analyst to seek other approaches to help policy-makers determine whether benefits could plausibly be commensurate with the costs of the rule.

[60] The initial 1999 rulemaking was vacated by a 2002 court decision and replaced with interim standards similar to the 1999 standards (for complete text of the decision, refer to 255 F3d 855). The 2005 rule replaced the interim standards, and costs and benefits are actually incremental from the interim standards, though the difference is less than $0.2 million. Both the 1999 and 2005 rulemakings included formal economic assessments consistent with Executive Order 12866. See U.S. EPA, Assessment of the Potential Costs, Benefits,and Other Impacts of the Hazardous Waste Combustion MACT Standards: Final Rule, Office of Solid Waste, July 1999; U.S. EPA, Addendum to the Assessment of the Potential Costs, Benefits, and Other Impacts of the Hazardous Waste Combustion MACT Standards: Final Rule, Office of Solid Waste, July 23, 1999; and U.S. EPA, Assessment of the Potential Costs, Benefits, and Other Impacts of the Hazardous Waste Combustion MACT Final Rule Standards, Office of Solid Waste, September 2005.

Exhibit 1

MONETISED COSTS AND BENEFITS OF THE HAZARDOUS WASTE COMBUSTION MACT STANDARDS

(Million 2006US$)

COSTS	**1999 Rule**	**2005 Rule**	**Total**
Cement Kilns/LWAKs	32.5	(0.3)	32.2
Commercial Incinerators	7.2	(16.5)	(9.2)
On-site Incinerators (including federal)	33.8	1.7	35.5
Boilers/Furnaces	None	36.2	36.2
TOTAL	**73.5**	**25.7***	**99.2**
BENEFITS			
Cancer (dioxin)	2.8	0.02	2.8
Non-Cancer (PM)	37.6	7.1	44.7
Non-Monetized	IQ impacts,	IQ impacts, visibility,	IQ impacts, visibility,
	ecological impacts	ecological impacts	ecological impacts
TOTAL	**40.4**	**7.1**	**47.5**

* Total includes government cost estimates and costs to existing generators related to increases in prices of incinerating hazardous waste. Totals may not sum due to rounding.

Sources:

Addendum to the Assessment of the Potential Costs, Benefits, and Other Impacts of the Hazardous Waste Combustion MACT Standards: Final Rule, Office of Solid Waste, July 23, 1999; Assessment of the Potential Costs, Benefits, and Other Impacts of the Hazardous Waste Combustion MACT Final Rule Standards, Office of Solid Waste, September 2005.

Estimates inflated to $2006 using U.S.Department of Commerce Bureau of Economic Analysis Gross Domestic Product deflator at *http://www.bea.gov/bea/dn/nipaweb/TableView.asp#Mid*

In the absence of clear and substantial benefits, other important questions become even more critical in the assessment. Policy-makers and economists may focus less on economic efficiency and more on an accurate accounting of specific costs and impacts, as well as on key distributional issues, such as the economic and environmental impacts on specific sectors, regions, or consumers. In addition, when the net benefits of a policy are not self-evident, the clarity of the analytic process becomes a central support for policy decisions. Ideally, an analysis can provide not only accurate cost and benefit estimates, but can also facilitate the policy process with accessible information that responds to the concerns and perceptions (and misperceptions!) of key stakeholders. A strict "cookie-cutter" approach focused on economic efficiency is often of limited use in supporting a transparent process, which is critical in addressing complex or far-reaching issues such as broad air regulations.

Our approach to the economic assessment of the HWC MACT rule borrows from the value of information literature, though our "process" evolved organically as the analysis proceeded.[61] The approach incorporates a series of bounding, screening, and breakeven analyses to prioritise and target issues for more detailed analysis. The analysis integrated policy concerns and economics as part of the iterative U.S. rulemaking process that includes notices, proposed rules, and several levels of review and public comment (in this case the process included the iterations of a court case and a second rulemaking). The assessment supporting both rules incorporated three phases of analysis, each framed in part by key policy questions:

- Characterisation of high-end compliance costs: this initial engineering-based estimate of direct compliance costs provides important information to frame the size of the rule. This bounds the policy discussion and provides insight into the issues and methods that should drive the more detailed analysis;

Targeted analysis of market dynamics, compliance decisions and market impacts: this set of analyses improves on the initial static cost analysis by incorporating industry structure and market dynamics, and use a series of screening-level economic impact analyses to examine a range of policy concerns such as employment and facility operation; and

- Characterisation of benefits: this analysis centres on a conservative quantitative analysis of key benefits, coupled with a description of other, non-monetised benefits and a focus on the role of uncertainty in benefits estimation.

61 We use the term "value of information" somewhat loosely - for a more formal discussion of value of information approaches, see, for example, Leland B. Deck and Lauraine G. Chestnut, "Benefits Transfer: How Good is Good Enough?" Presented at June 1992 Association of Environmental and Resource Economists Workshop Benefits Transfer: Procedures, Problems, and Research Needs, Snowbird, Utah. Reprinted in United States Environmental Protection Agency Policy, Planning, And Evaluation, April 1993 (EPA 230-R-93-018).

The following chronicle of the analysis outlines the process we developed to address these issues. The discussion highlights our priorities and methods, key issues that emerged during the analysis, and notes on recent methodologies and literature that could potentially improve future analyses.

The role of Economic Assessment in assessing MACT standards

Economic analysis plays a specific and somewhat limited role in the promulgation of a MACT rule. MACT standards must, at a minimum, require all facilities in a target industrial sector to adopt a technology-based standard that is equivalent to the top-performing 12 percent of facilities in that sector.[62] This minimum requirement is the "floor" standard, and is determined by technological and emissions performance. Cost is expressly excluded from consideration in developing the floor standard. However, in cases where EPA determines that the floor standard is not sufficiently protective of human health and the environment, the agency may develop and consider any number of more stringent "beyond-the-floor" options. Consideration of costs and benefits is expressly required in any evaluation of beyond-the-floor standards.

In other words, in the context of the MACT standards, economic analysis is not used to determine whether regulation should occur. Instead, it provides information on which regulatory options might maximise benefits, minimise costs, or both.

Both the 1999 and 2005 HWC MACT rulemakings considered, and ultimately promulgated, beyond-the-floor standards for certain pollutants. Both therefore required economic analysis of the various options considered. Moreover, both rules were potentially large enough to trigger a separate, broader requirement for economic assessment under Executive Order 12866 ("Regulatory Planning and Review"). Executive Order 12866 requires that U.S. agencies perform economic assessments of all "significant" rulemakings, with the aim of establishing that the benefits of a rule justify the costs.[63] Rulemakings are considered significant if they incur more than $100 million dollars a year in total economic impacts, or are expected to have a substantial impact on particular economic sectors or regions.[64]

62 The term "sector" can be a point of discussion as well – the HWC MACT stan dards developed separate standards for specific sub-sectors of "combustors" based on the processes and markets (i.e., commercial incinerators, cement kilns, on-site incinerators, light-weight aggregate kilns, and three different types of boilers and furnaces each were subjectto different standards).

63 Executive Order 12866, "Regulatory Planning and Review," Federal Register, Vol. 58, No. 190, Monday, October 4, 1993. The executive order does not pre-empt the statutory authority to set a floor standard without considering cost, but it does establish another layer of review that focuses on economic analysis.

64 According to the executive order, one definition of "significant regulatory action" is any regulatory action that is likely to result in a rule that may "Have an annual effect on the economy of $100 million or more or adversely affect in a material way the economy, a sector of the economy, productivity, competition, jobs, the environment, public health or safety, or State, local, or tribal governments or communities."

Initial Phase: analysis of direct compliance costs

Our initial analytic question was driven in part by the trigger for Executive Order 12866: was the rule big enough to be "significant" and therefore require a broad analysis of costs and benefits? To address this, we used a simple initial "worst case" cost screen to determine significance and answer a number of additional questions: what does the rule size say about further evaluation of broad economic impacts? What does the potential burden on different sub-sectors and facilities suggest about further analysis of distributional impacts and industry dynamics? And what are the appropriate cost analysis tools and data to address these issues? Finally, knowing the upper cost limit could assist EPA in framing its communication with various stakeholders during the rulemaking.

Our deliberately simplistic "engineering" cost analysis assumed that every facility would upgrade its pollution control equipment to conform to the standard, with no ability to pass costs through to customers, cease operations, consolidate units, or opt for cheaper disposal alternatives. This exercise can be substantial if the number of entities affected by the rule and their options for compliance are uncertain. In our case, however, the facilities were known and the compliance technologies were specified. Moreover, we were fortunate to have access to data on baseline facility technologies and capacity, and on facility-specific upgrade costs. As a result, the screening exercise provided reliable upper-bound information both about total costs of the rule and about the range of potential costs for specific facilities. Exhibit 2 provides detailed results for the 2005 analysis, including per system cost data, and total 1999 analysis costs for comparison.

Exhibit 2

ENGINEERING COSTS OF THE 2005 HAZARDOUS WASTE COMBUSTION MACT STANDARDS

2006$

Costs	Per System	Number of Systems	Number of Facilities	Total Costs	1999 Total Costs*
Cement Kilns	0 – 815,200 average: 129,000	25	13	3.5	39.8
LWAKs	16,100 –144,800 average: 62,900	7	3	0.1	8.4
Commercial Incinerators	0 – 21,100 average: 3,700	15	11	0.5	10.9
On-site Incinerators	0 – 101,600 average: 16,200	92	66	2.3	56.6
Liquid Boilers	0 – 1,820,700 average: 311,300	104	53	35.6	N/A
Coal Boilers	76,800 – 357,700 average: 193,200	12	4	3.0	N/A
HCl Production Furnaces	0 – 148,000 average: 18,800	10	8	0.7	N/A
TOTAL	**N/A**			**46.2**	**115.8**

* 1999 costs reflect different unit costs and different (usually larger) numbers of facilities than 2005 numbers

Sources:

Assessment of the Potential Costs, Benefits, and Other Impacts of the Hazardous Waste Combustion MACT Final Rule Standards, Office of Solid Waste, September 2005.

Estimates inflated to $2006 using U.S.Department of Commerce Bureau of Economic Analysis Gross Domestic Product deflator at *http://www.bea.gov/bea/dn/nipaweb/TableView.asp#Mid*

The screening assessment reveals that "worst case" total costs, though substantial, were less than $100 million in each rulemaking, and also that the range of costs at the facility and category level was considerable. Therefore, while the total impact of the rule might not be "significant," a real possibility existed for significant changes in operations at the sector level as a result of the rule.

The main value of the screening analysis was to inform our selection of an appropriate tool for a refined cost analysis. The $100 million threshold, in addition to providing a policy distinction, is roughly equivalent to the smallest shock that has a measurable equilibrium impact on the U.S. economy. Therefore, economy-wide analysis of economic impacts using tools such as computable general equilibrium models would be fruitless, though these types of analysis are often central to evaluation of broader regulatory programs.

The extent and range of costs identified were, however, great enough to suggest that a simple compliance cost model would also be inadequate. HWC industry could potentially undergo significant consolidation as a result of the rule. We therefore opted to develop an in-house market-based model that would build on available compliance cost data to evaluate specific compliance decisions, resulting changes in operation, and price impacts within the sector.[65]

This phase of the analysis also reflects a methodological and behavioural reality in evaluating regulations in the U.S.: high-end (i.e., overstated) cost estimates are valued by policy-makers for the defensive "cover" they can provide in working with industry, particularly on contentious rulemakings. EPA's record of overstating costs on rulemakings had been well documented at the time of the 2005 rulemaking, but the screening analysis was still preserved as an upper bound in the published analyses.[66]

Second Phase: Market-Based costs and economic impacts

Our high-end estimate of direct compliance costs indicated that evaluation of potential market impacts would be useful. The central question in determining market effects is simple: what will facilities – and their customers – actually do to respond to the

Hazardous Waste Combustion Markets

To model compliance decisions we first characterised industry structure and competition. Hazardous waste combustors fall into three "submarkets," according to whether they are

65 For a more detailed discussion of methods for choosing cost analysis tools, see Industrial Economics, An Overview of Major Economic Modeling Paradigms and their Potential Application in OSWER Analyses, prepared for U.S. Environmental Protection Agency Office of Solid Waste, Economic Methods and Risk Analysis Division, November 2004, and Peter Berck and Sandra Hoffman, "Assessing the Employment Impacts of Environmental and Natural Resource Policy," Environmental and Resource Economics Volume 22: pp. 153-166: 2002.

66 See, for example, Winston Harrington, Richard D. Morgenstern, and Peter Nelson, On the Accuracy of Regulatory Cost Estimates, Resources for the Future Discussion Paper 99-18, January 1999.

commercial facilities (i.e., treating waste from other facilities) and whether they burn waste as fuel. The overall industry is somewhat segmented, with incinerators seeking "high margin" waste and kilns seeking high fuel-value waste. Competition occurs among commercial incinerators and among cement kilns, and, to a more limited extent, between these sub sectors. In brief:

- **Commercial incinerators:** seek "highly contaminated" waste for which they can charge higher prices, but can take all wastes. Many of their customers represent a captive market of generators with limited disposal options and inelastic demand, but the overall quantity of hazardous waste has declined in recent years and cement kilns have been starting to compete for more waste streams. Commercial incinerator compliance options are to upgrade or close the entire facility.

- **Cement kilns:** seek "cleaner" wastes with high fuel value; compete with incinerators for some more contaminated wastes and blend the wastes to fuel cement-making. Their main compliance options are to upgrade or seek alternate fuels.

- **On-site incinerators and boilers:** seek inexpensive disposal for wastes and energy recovery (boilers and furnaces). Their principal compliance options are to upgrade or to seek off-site treatment for wastes, and either close the unit (incinerators) or seek alternate fuel (boilers).

Because kilns and on-site units support other manufacturing operations, closure of an entire facility is unlikely unless the facility's production operations are marginal and compliance costs are high. However, at the time of the initial rulemaking in the late 1990s, the commercial incinerator sector, in particular, was characterized by overcapacity and decline, and policy-makers were very sensitive to the concern that the MACT standard would result in significant facility closures and employment losses.

To determine relative costs of the different options for each facility, we identified facility locations, distances to commercial combustors, unit costs for transportation, alternative fuels, and disposal, and used EPA's Biennial Report database to identify the quantity and type of waste combusted at each facility.

Revised Cost Analysis Reflecting Market Dynamics

For our market-based cost analyses for both rules, we developed a spreadsheet-based model to identify least-cost decisions for each facility. The 1999 model was static, but advances in software allowed the development of a limited partial equilibrium model for the 2005 analysis. Both models examined the options for each facility considering facility-specific capital equipment, permitting, alternate fuel, transport, and disposal costs, as well as consolidation options for on-site facilities with more than one unit. The 2005 model also evaluated commercial facility capacity on a regional basis to identify any constraints, and evaluated the price impacts of passing through compliance costs to hazardous waste generators seeking treatment.

The result was an estimate that was substantially lower than the upper bound engineering costs. The 1999 analysis calculated annual costs of $73.5 million (2006$), a reduction of over 35 percent from the high-end estimate. The 2005 analysis calculated total annual costs of $25.7 (2006$), a reduction of over 40 percent from the $46.2 high-end estimate. In general, the model allowed facilities with high engineering costs to select other options by consolidating units, substituting fuel, or shifting treatment to off-site facilities. It also reveals the positive revenue impacts of the rule for commercial combustors as generators close on-site disposal facilities and seek commercial treatment. Notably, the 2005 analysis estimated that commercial incinerators would actually have negative costs (or positive revenue impacts) of $16.5 million (2006$) annually. For these facilities, compliance costs are more than offset by new revenues, due to the predicted closure of on-site incinerators and the direction of over 42,000 tons of waste to commercial units. The combined impacts of the rulemakings were estimated to be positive $9 million per year for incinerators. The cost analysis also revealed that the final beyond-the-floor standard had a modest three percent impact on costs over the floor standard in 2005, and a potential impact of five to fourteen percent in 1999. Price impacts, if any, on hazardous waste treatment were likely to be modest.[67]

The custom modelling and high quality data of this approach provided reliable estimates that were generally accepted by industry groups during the rulemaking process. It is worth noting that although the market analysis was more optimistic than the upper bound engineering estimate, our model continued to incorporate a number of

67 It is likely that costs can be passed through by raising treatment costs for certain types of waste (typically those with no fuel value and no other disposal options). It is not clear whether all facilities can pass through all costs, but a series of sensitivity analyses using the model determined that the pass-through assumption did not have significant impacts on decisions by on-site facilities to continue or cease operation. We therefore assumed that costs were passed through to waste generators, and included these cost increases separately in the estimates (see Exhibit 1).

conservative (i.e., high-cost) assumptions in areas where information was uncertain.[68] Sensitivity analyses indicated that these assumptions did not significantly change the results, and the conservative emphasis provided policy-makers with a stronger position from which to discuss the more important assumptions in the analysis.

Facility Closures and Employment Impacts

A key issue in the analysis of pollution control policy is its potential to cause facility closures and employment impacts. In the HWC MACT context, concern focused on commercial incinerators and cement kilns, whose operations rely at least in part on the income received/fuel costs avoided from combusting waste. Both the cement and hazardous waste incinerator industries predicted extensive facility closures from the MACT standards, particularly in 1999. The commercial hazardous waste incineration industry was suffering over-capacity, the result of 1980s expansion in anticipation of demand for waste treatment that never emerged, as generators opted instead to reduce generation. The cyclical cement industry was also declining at the time.

Our analysis of this issue highlights both the utility of a breakeven analysis and the importance of establishing a valid baseline for analysis. First, we used information collected by EPA on the baseline fixed and variable costs for waste management at each system in the universe.[69] Coupling this data with information on the current quantities of waste treated in each system, national average prices for treatment of different types of waste, and system-specific compliance costs, we performed a simple analysis to identify baseline profitability, post-rule profitability, and the "breakeven quantity of waste" required for every system at target facilities to meet costs.

The breakeven analysis for both rules revealed that a number of systems were not profitable in the baseline. Most of these systems were on-site incinerators (26 in 1999; another 10 in 2005). This is not surprising, given that on-site incinerators enjoy "profits" only in the form of cost savings, and may be operated for other perceived benefits (such as avoided liability for off-site disposal). But the 1999 analysis also identified three commercial incinerator facilities, and the 2005 analysis identified three commercial incinerator systems that were not profitable in the baseline. In each rulemaking, a number of on-site systems (in addition to baseline systems) were estimated to close as a result of the rule, but only one commercial facility (a cement kiln in 1999) was

68 Specifically, alternate fuel, unless known, was assumed to be natural gas (in many cases cement kilns burn non-hazardous waste and other low-cost fuels); transport was assumed to use trucks and not (less expensive) rail, and consolidation at multi-system facilities was limited to incinerators, because boilers were assumed to be physically integrated into production systems and unable to consolidate. Only transport calculations were optimistic, as they were based on "great arc" distances, but a series of sensitivity analyses confirmed that these costs were not a central driver of most facility decisions. Note that the recent emergence of mapping software that calculates road distances for multiple facilities would improve the transport calculation.

69 It is rare that EPA can make detailed facility-specific cost data available for a regulatory analysis; however, we have also been able to use data from Risk Management Association and other sources to build pro forma profitability analyses for several sectors.

predicted to close as a result of either rule (Exhibit 3). The baseline and regulatory breakeven analysis, coupled with the market-based cost analysis that showed increases in demand for commercial treatment due to the rule, was sufficient to resolve concerns about facility closures, and no more detailed analysis was undertaken.

Finally, the existence of two consecutive rulemakings provides insight into the accuracy of the breakeven analysis: the 1999 analysis predicted the baseline closure of three incinerators, in part due to the severe overcapacity of the industry at the time. By 2005, this prediction was borne out. Several facilities had closed, and remaining plants had higher capacity utilisation and were commanding much higher prices for waste.

A screening analysis of employment impacts based on system closure calculations and engineering estimates of labour requirements likewise concluded that total net employment impacts were small but positive, as a result of production and maintenance requirements for air pollution control devices. Exhibit 3 provides the 2005 analysis results, and provides an interesting illustration of distributional employment effects across sub-sectors.

Exhibit 3

ECONOMIC IMPACT ANALYSIS RESULTS FOR THE 2005 HAZARDOUS WASTE COMBUSTION MACT STANDARDS

Costs	Systems Closing/ Market Exits	Waste Rerouted	Employment Dislocations	Employment Gains
Cement Kilns	0	2,289 ton increase	0	15.3
LWAKs	0		0	0.9
Commercial Incinerators	3*	42,722 ton increase	73.3	2.2
On-site Incinerators	26	45,011 tons transferred to commercial facilities; 13,915 tons consolidated to other on-site units	191.8	9.8
Liquid Boilers	8		36	183.1
Coal Boilers	2		0	13.0
HCl Production Furnaces	0		0	5.6
Pollution control device makers	N/A		0	92.8
TOTAL	39		310.2	322.8

* Note that the 2005 baseline profitability analysis determined concluded that these incinerators may be unprofitable in the baseline, and therefore impacts associated with the closure of these facilities may not be attributable to the MACT standard.

Source: Assessment of the Potential Costs, Benefits, and Other Impacts of the Hazardous Waste Combustion MACT Final Rule Standards, Office of Solid Waste, September 2005.

Overall, the market-based cost analysis provided a reasonable estimate of costs that reflects the key elements in industry decision-making, and also addressed the important "bread and butter" issues of concern to policy-makers, concluding, in contrast to "conventional wisdom," that most commercial facility closures would take place in absence of the MACT rules, and that the rules would have an overall positive impact on commercial incinerators, in particular.

In addition, the analytic process provides several insights into "right-sizing" an analysis to meet demands for economic rigor and provide relevant information. Our custom spreadsheet model allowed us to focus on market dynamics of importance, including pricing dynamics, without significant additional data collection. While a custom model is not appropriate for all analyses, the ability to provide high-resolution analysis to key issues makes it an attractive option in many cases. In addition, a series of quick, nested screening assessments to support both rules determined that facility closures, employment impacts, and price increases for waste disposal were all insignificant.

Recent economic trends and emerging literature suggest two potential improvements to the cost analysis. First, fuel prices and concern about fossil fuel combustion have raised the profile of energy impact analyses, and have increased efforts to by EPA to encourage use of alternative fuels, including hazardous and non-hazardous waste. A more extensive evaluation of the costs of fuel switching at facilities that decide not to upgrade might be valuable. In addition, the cost analyses do not incorporate long-term savings associated with "learning curves" – marginal efficiency improvements over time. Particularly for technology-driven standards, an adjustment to reflect efficiency gains is warranted.[70] In one sense, the impacts of learning are already reflected in the 2005 analysis: the nominal costs of specified air pollution control devices are the same in 2005 as they were in 1999.

Finally, the cost increase associated with the beyond-the-floor option was a modest three percent higher than the cost of the floor option. The remaining question for the analysis, then, is whether total costs are justified by the benefits of the rule.

Final Phase: Benefits Assessment

In the assessment of the human health and ecological benefits of environmental policies, the most important adage may be "timing is everything." At the time of 2005 HWC MACT analysis, new research was underway on quantification and monetisation of three of the pollutants addressed by the MACT standards: mercury, dioxins, and particulate matter. The analysis, however, pre-dated the final publication of all of these efforts, and therefore confirmed another reality of economic assessment in the U.S.: in high-profile rulemakings, policy-makers often emphasise development of conservative and defensible – if partial – estimates of benefits.[71]

70 The U.S. Department of Energy incorporates learning into its energy forecasting models, and EPA has investigated incorporation of learning into rulemakings. For an overview of the issue, see Cynthia J. Manson, Matthew B. Nelson, and James E. Neumann, Assessing the Impact of Progress and Learning Curves on Clean Air Act Compliance Costs, presented at the Air and Waste Management Association Conference, June 24-26, 2002.

71 This approach stems in part from various Agency guidance, such as U.S.Environmental Protection Agency, Guidelines for Performing Economic Analysis, Office of the Administrator, September 2000, EPA-240-R-00-003, and Office of Management and Budget, Circular A-4: Regulatory Analysis, September 17, 2003. While in some cases policy-makers can be aggressive in identifying and characterising benefits, the existing guidance and multi-layered review process for high-profile rulemakings frequently results in an emphasis on "defensible" analysis of monetised benefits.

Much of this caution comes from the nature of uncertainty. Costs, particularly in cases like the HWC MACT standard where facilities and regulatory options are clearly identified, are typically measurable and easy to describe. Benefits analysis, however, is characterised by uncertainty at every step in the analytic process – fate and transport, exposure, dose-response functions, and valuation. This is particularly true in addressing pollutants such as mercury and lead, where adverse impacts are well known, but measuring them and valuing them in specific contexts is more complex.

As with our cost analysis, we employed a staggered value-of-information approach to focus our benefits assessment. We measured the benefits that we could reliably value, and performed a number of screening assessments to determine the usefulness of expanding our efforts to other benefits categories. Attention to the distribution of benefits across facilities and demographics also informed our assessment.

Human Health Benefits

The HWC MACT standard represents a fairly typical large-scale benefits analysis. The initial 1999 rule incorporated a multi-pathway risk assessment of the anticipated changes in the emissions of several pollutants (Exhibit 4). In 2005, the smaller incremental benefits were evaluated using a conservative extrapolation of the previous results. The 2005 extrapolation did not adjust for population increases near facilities, and did not consider regional concentration of facilities due to the inclusion of facilities with boilers and industrial furnaces. We then applied established estimates of cost-of-illness and willingness-to-pay to avoid health effects to the quantified impacts from the risk assessment to develop monetised estimates of human health benefits.

Exhibit 4

EMISSIONS REDUCTIONS PREDICTED UNDER THE 1999 AND 2005 HAZARDOUS WASTE COMBUSTION MACT STANDARDS

	1999	**2005**	**Total**
Particulate Matter (tons/year)	2,449	2,138	4,587
Mercury (tons/year)	3.9	0.2	4.1
Semi-Volatile Metals/Low-Volatile Metal (incl. lead) (tons/year)	97.1	9.4	106.5
Dioxins/Furans (g/year)	28.7	0.2	28.9
Chlorine (tons/year)	5,132	107	5,239

Sources:
Addendum to the Assessment of the Potential Costs, Benefits, and Other Impacts of the Hazardous Waste Combustion MACT Standards: Final Rule, Office of Solid Waste, July 23, 1999; Assessment of the Potential Costs, Benefits, and Other Impacts of the Hazardous Waste Combustion MACT Final Rule Standards, Office of Solid Waste, September 2005.

Ultimately, the benefits of the rule reflected only cancer impacts from reduced dioxin emissions, and non-cancer impacts from particulate matter reductions. The 1999 analysis also estimated the impacts of lead and mercury reductions on exposure in children, but did not attempt to monetise these impacts due to limitations in the valuation literature for assessing lead impacts, and uncertainties related to estimating the number of children potentially exposed to mercury. The limited extrapolation of the risk assessment did not support an estimation of lead and mercury impacts in 2005. Exhibit 5 summarizes the undiscounted annual benefits of the floor and final standards for the 2005 rulemaking.[72]

Other Benefits

Other potential benefits related to reductions in hazardous air pollutants include improvements in visibility (reduced regional haze), crop and forest damage, and ecological impacts including wildlife injury. As with human health benefits, defensible methods exist for measuring and valuing these impacts, and we evaluated the difficulty and usefulness of addressing these benefits formally. With most ecological impacts

72 Discounting presents another set of challenges in benefits estimation that is not discussed here.

(forest and crop damage, wildlife injury), facility and receptor location is so critical to measurement that benefit transfer is difficult, particularly when reductions vary widely across facilities, as was the case with the HWC MACT standards.

In spite of this, we did use some values in existing studies and linear extrapolation to national HWC MACT emissions reductions to establish one outcome with some certainty: the cost of a reliable assessment of the measurable, annual ecological benefits of the rule would exceed those benefits. In the context of MACT standards, where economic analysis is focused on optimising rather than justifying a rulemaking, the value of this information would be minimal.

Regional haze reductions were one potentially significant source of benefits. Our screening assessment was a bounding analysis of potential benefits of increased visibility in recreational areas, based on national estimates from The Benefits and Costs of the Clean Air Act 1990 to 2010. Using two different extrapolation approaches, our estimated potential benefits ranged from $0.2 to $7.5 million dollars per year (2006$).[73] We reported these benefits, but did not include them in final estimates for two reasons. First, the range reflected an unacceptable level of uncertainty, and second, the inclusion of even the higher-bound benefits estimate would not result in total benefits exceeding costs of the rulemakings, or differentially affect the benefits of the beyond-the-floor and floor standards.

Comparison of benefits and costs

The monetised estimate of annual, undiscounted benefits of the HWC MACT standards are clearly less than total annual costs for all regulatory options formally considered under both rulemakings (see Exhibit 1). In fact, the total annual costs of both rules (roughly $99 million) are just over double the monetised annual benefits of $48 million.

Moreover, the beyond-the-floor standards in both rules increase the difference between costs and benefits. In 2005, the beyond-the-floor standard eventually passed as the final rule, increases costs by three percent and benefits by roughly one percent, despite the fact that the standards focus on the most easily valued pollutants: dioxin and particulate matter. The 1999 rule incorporated beyond-the-floor standards for dioxins, lead, and chlorine, and these standards increased "floor" costs by 11 percent and benefits by roughly five percent.

73 U.S. EPA, The Benefits and Costs of the Clean Air Act 1990 to 2010, September 1999. The high-end estimate assumes a linear relationship between PM reductions and total visibility benefits. The low-end estimate assumes a linear relationship between particulate matter-related health benefits and visibility benefits. Estimates were expressed in 1990 dollars in the 1999 report; we convert to 2006 dollars using the GDP deflator.

These general ratios hold true across the range of regulatory options considered during the development of the rule, with one notable exception: during the early stages of the 2005 rule development, EPA examined standards that would incorporate dry scrubber technology at coal-fired industrial boilers. In addition to reductions in other pollutants, use of this technology would have eliminated an estimated 22,000 tons of sulfur dioxide. By extrapolating results from other analyses of Clean Air Act rules (notably the 2004 Interstate Air Quality Rule and the Clear Skies Analyses) we calculated HWC MACT sulphur dioxide benefits ranging from $193 to $350 million dollars annually (1999 dollars), with a lower-bound estimate of $27 million reflecting part time operation of only half the boilers at a single, large facility.[74]

The development of these estimates created a small internal debate in our team about whether "derived benefits" from sulfur dioxide benefits should be important in evaluating the HWC MACT rule. The benefits would undoubtedly have resulted from the HWC MACT rule, and we ultimately agreed that they should be included in benefits estimates. However, the benefits did not accrue from reductions in the hazardous air pollutants, and sulfur dioxide emissions were being targeted by a number of other rules. This raised a concern about "double counting" the benefits of other rules (though we verified that no existing rules had required upgrades). More importantly, however, these derived benefits were driven largely by operations at a single facility. The HWC MACT standard did not have broad ancillary benefits associated with sulfur dioxide across a number of facilities, and we were concerned that an ancillary benefit at a single facility, no matter how large, should not drive the discussion of the total value of a rule. In this case, our debates remained academic; revisions to the floor standard by EPA eliminated the dry scrubber option, and the benefits were eliminated. However, recent revisions to the National Ambient Air Quality Standards for particulate matter may require similar upgrades at the facility.

Ultimately, then, the HWC MACT standards were passed in spite of an overall two-to-one cost/benefit ratio, and in spite of the fact that the stricter beyond-the-floor standards raised costs more than they did monetised benefits. The question is: what other factors could have contributed to the acceptance of a rule that did not meet the benefit-cost priorities outlined under either Executive Order 12866 or the Clean Air Act?

74 Cynthia Manson, Katherine Wallace, and Jason Price, Preliminary Estimates of Benefits from Control of Sulfur Dioxide (SO2) Under the Hazardous Waste Combustion MACT Standards, memorandum submitted to EPA Office of Solid Waste, February 25, 2004. Methodology based on U.S. EPA, 2003, Technical Addendum: Methodologies for the Benefit Analysis of the Clear Skies Act of 2003, U.S. EPA, 2003, Technical Support Document for the Clear Skies Act 2003 Air Quality Modeling Analysis; U.S. EPA, Benefits of the Proposed Inter-State Air Quality Rule.

The answer likely amounts to a characterisation of uncertainty; in this case, resolution of any of several key uncertainties would likely result in increases in benefits. In addition to the benefits completely omitted (e.g., ecological benefits) and the embedded high-cost assumptions in the cost analysis, the estimated benefit range associated with one factor – visibility – could as much as double the total benefits of the rule if refined. Add to this the fact that unquantified benefits include those associated with mercury and lead, which are high-priority pollutants known to have a range of neurological and developmental impacts that disproportionately affect children and therefore have long-term and high-cost impacts related to overall productivity.

Furthermore, as the analysis of this rule unfolded, emerging literature re-examining dose-response functions for both particulate matter and dioxins was underway, and preliminary conclusions from the dioxin research suggested that the impacts of the pollutant might be as much as six times the impacts that were commonly calculated by EPA.[75] Other research published in 2005 clarified the link between mercury and cardio-vascular impacts, opening the possibility of expanded future estimates of mercury-related health effects.[76] Though published since the rule, the research on particulate matter has also concluded that impacts have been underestimated.[77] Thus, the timing of the analysis dictated its benefits and this is a typical limitation in the analysis of environmental policy.

Finally, it would be negligent to overlook the obvious justification for the rule: the Clean Air Act mandates MACT standards without regard to cost, and the total incremental costs – and benefits – of the two beyond-the-floor HWC MACT standards were minimal. Moreover, the beyond-the-floor standards, in general, increased the consistency of the emissions standards across facilities of different types. This has the practical – and market – advantage of avoiding standards that would establish "loopholes" that favor older technologies and allow older facilities to externalise the costs of pollution.[78]

75 For a recent discussion of this issue, see Health Risks from Dioxin and Related Compounds: Evaluation of the EPA Reassessment Committee on EPA's Exposure and Human Health Reassessment of TCDD and Related Compounds, National Research Council, 2006.

76 Glenn Rice and James K. Hammitt. Economic Valuation of Human Health Benefits of Controlling Mercury Emissions from U.S. Coal-Fired Power Plants, prepared for Northeast States for Coordinated Air Use Management, February 2005.

77 See Chapter 5 of the Regulatory Impact Analysis for 2006 National Ambient Air Quality Standards for Particle Pollution, U.S. Environmental Protection Agency Office of Air and Radiation, September 2006, for a description of the recent expert elicitation process that revised estimates of the dose-response relationship for particulate matter.

78 In cases where an incomplete calculation of net benefits alters the policy dialogue, comparative approaches such as cost-effectiveness and break-even analysis may provide useful insights. Cost-effectiveness analysis can be useful in comparing options when methods and data support a consistent analysis (The HWC MACT analysis was required to include a cost-effectiveness analysis, but the analysis was of limited use as it merely allocated total costs according to change in emission, without regard for issues such as co-control of multiple pollutants with one technology). Another emerging approach to evaluating policies with uncertain benefits is break-even analysis, a technique that is increasingly used in analyses dealing with high cost efforts to avoid large-scale events such as hurricanes and terrorist attacks. Break-even analysis involves identifying a "net cost" that equals the magnitude of benefits required for the total benefits of the policy to equal the total costs. In other words, this value represents the threshold at which benefits would "break-even" with the costs of the policy or regulation. The decision-makers can then determine whether the risk reduction anticipated by the policy justifies the net cost.

Conclusions

The HWC MACT standard economic analyses provide some insights into the general process of regulatory analysis, and illustrate the effectiveness of a tailored analytic approach that uses a combination of initial screening assessments and targeted analyses to focus efficiently on specific issues of concern. The results, in this case, were a combination of robust estimates of costs and (partial) benefits, and a range of other economic results that informed the stakeholder process during the rulemakings.

The central, "traditional" benefit-cost analysis for each rule determined that the measurable net benefits of both rules were negative. However, our layered "value of information" process of assessing costs and benefits allowed us to identify and investigate the most significant economic issues and answer the key policy questions surrounding the rule. The conclusion that the 2005 rule would result in positive net impacts for commercial incinerators, and that commercial facility closures would likely occur in the absence of the rulemaking, assisted policy-makers in addressing stakeholder concerns. Similarly, our partial monetised estimates of benefits were lower than costs (and lower than incremental costs for beyond-the-floor standards) but the analysis also noted significant non-monetised benefits and emerging literature to help frame the discussion of rule impacts among decision-makers. Ultimately, the suite of analyses both confirm the usefulness of benefit-cost analysis in informing regulatory decisions, and also highlights methods for supporting the policy-making process by expanding beyond an examination of net benefits.

Assessing the Costs and Benefits of the European Air Pollution Policy (CAFE): Results and Lessons from Experience

Paul Watkiss[79], Mike Holland, Fintan Hurley, Alistair Hunt and Steve Pye[80].

Abstract

This paper summarises the cost-benefit analysis supporting the CAFE Programme and the Impact Assessment of the Thematic Strategy on Air Pollution (the proposals for future air quality policy in Europe). It outlines the methodology used for the analysis, and presents a summary of the CBA results comparing different ambition levels for future air quality. The paper also reports on the uncertainty analysis undertaken as part of the CBA work. Finally, it summarises the lessons learnt from the CBA and how these might improve future Impact Assessments.

Introduction

Concerns over the impacts of air pollution have led to major policies being introduced in Europe over the past few decades. These have had a focus on reducing impacts associated with natural or semi-natural ecosystems (acidification and eutrophication) and have been implemented as international agreements to reduce emissions, set either through the European Commission or agreed within the United Nations Economic Commission for Europe. The most recent are the National Emissions Ceilings Directive (2001/81/EC) and the United Nations Economic Commission for Europe (UN/ECE), Gothenburg Protocol. However, there has also been a more recent recognition of the health impacts of air pollution (at ambient air quality concentrations). This led to the introduction of legally binding air quality standards in Europe (known as limit values), set to protect human health, and introduced through the Air Quality Framework Directive (1996/62/EC) and Daughter Directives.

More recently, the EC's Sixth Environmental Action Programme[81] set out the objective to develop long-term, strategic and integrated policy advice for 'achieving levels of air quality that do not give rise to significant negative impacts on and risks to human

79 Paul Watkiss (paul_watkiss@btinternet.com), Paul Watkiss Associates, *http://www.paulwatkiss.co.uk*

80 The CAFE CBA work summarised here was undertaken by a team including Mike Holland (EMRC), Alistair Hunt (Metroeconomica), Fintan Hurley (Institute of Occupational medicine) and Steve Pye (AEA Environment and Energy).

81 The Sixth Environment Action Programme (EAP) of the European Community 2002-2012. Adopted 2002. *http://ec.europa.eu/environment/newprg/intro.htm*

health and the environment', including 'no exceedance of critical loads and levels for acidification or eutrophication'. It also set out that this should lead to a Thematic Strategy on Air Pollution considering the economic, social and environmental dimensions (of policy) towards these objectives. In response, the European Commission launched the Clean Air for Europe (CAFE) Programme in 2001[82] – a knowledge based approach for technical/scientific analyses and policy development, which led to the proposal and adoption of the Thematic Strategy on Air Pollution in September 2005[83].

Consistent with all European regulatory policy proposals, the Strategy was subject to an Impact Assessment (IA). These IA's consider the likely economic, social and environmental impacts of different options[84]. This paper discusses the cost-benefit analysis undertaken as part of the CAFE programme, undertaken to support the impact assessment of the Thematic Strategy.

The Clean Air for Europe (CAFE) Programme and Ambition Levels

The aim of the EC's CAFE programme was to compile a set of multi-pollutant, multi-effect scenarios to investigate the effects of different objectives (ambition levels) on the future emissions, air quality, and of health and environmental impacts up to the year 2020. The pollutants covered were SO_2, NO_x, VOC, NH_3, $PM_{2.5}$ and the effects considered were human health, acidification, eutrophication, and critical ozone exceedance, as shown below.

Table 1 Pollutants and Effects Considered in CAFE and the Thematic Strategy

	Primary PM	SO2	NOx	VOC	NH3
Health effects:					
- Particulate matter	√	√	√	√	√
- Ground-level ozone			√	√	
Vegetation effects:					
- Ground-level ozone			√	√	
- Acidification		√	√		√
- Eutrophication			√		√

As part of CAFE, five working groups were set up to provide assistance and advice and the Programme also sought external advice from a range of international organisations. It reported to a Steering Group with representatives of the Member States, several

82 COM(2001)245)). *http://ec.europa.eu/environment/air/cafe/index.htm*
83 Thematic Strategy on Air Pollution (COM(2005) 446). Directive on Ambient Air Quality and Cleaner Air for
84 Europe (the "CAFE" Directive) (COM(2005) 447), version http://europa.eu.int/comm/environment/air/cafe/index.htm Impact assessment guidelines*. 15 June 2005. SEC (2005) 791. European Commission.

industry sectors, environmental NGOs, and other European organisations. Two major projects were also commissioned to provide information on the cost-effectiveness and cost-optimisation of air pollution policies (run by IIASA[85]), and on the costs and benefits of proposals (CAFE CBA[86]).

The method used in the Programme was first to establish a baseline assessing air pollution levels and impacts up to 2020 under a business as usual scenario (i.e. including all current and agreed legislation, but with no extra measures or additional legislation)[87],[88]. This baseline was compared against the long-term objectives of the 6th EAP (see above) to determine the "policy gap". This analysis showed a significant gap between the predicted baseline and the objectives and that further action was required.

The CAFE programme then considered various policy scenarios towards the long-term objective. To help decide on the costs and benefits of different levels of action, various options were considered, with reference to a scenario with all technical emissions abatement measures included irrespective of cost, known as the "Maximum Technically Feasible Reduction" (MTFR) scenario. It was found that even under this scenario (which includes the most expensive measures available), there would still be significant negative impacts on health and the environment in 2020. As applying all available technical measures irrespective of cost did not achieve the long-term objectives of the 6th EAP, the CAFE programme set out to establish interim environment objectives that delivered progress in a balanced and cost-effective way.

Various options (levels of ambition) were considered to close the gap between the baseline scenario and the MTFR scenario. The levels of ambition were considered for four areas: loss of life expectancy from exposure to particulates, premature deaths attributable to ozone, exceedence of critical loads for acidification, and exceedence of critical loads for eutrophication. Various interim objectives were explored, using the IIASA RAINS model in an iterative way, and the cost and benefits of closing the gap impact between the baseline emissions in 2020 and the MTFR scenario. The objective was to find a balance between cost-effective measures that would give optimum environmental and health benefits for Member States and the EU as a whole, and accounting for aspects of equity so that no population group or Member State would experience disproportionately high risks or costs.

85 The International Institute for Applied Systems Analysis (IIASA) and the RAINS model *http://ec.europa.eu/environment/air/cafe/activities/basescenario.htm*

86 *http://ec.europa.eu/environment/air/cafe/activities/cba.htm*

87 "The Current Legislation" and the "Maximum Technically Feasible Reduction" cases for the CAFE baseline emission projections. Background paper for the meeting of the CAFE Working Group on Target Setting and Policy Advice, November 10, 2004. Markus Amann, Rafal Cabala, Janusz Cofala, Chris Heyes, Zbigniew Klimont, Wolfgang Schöpp. International Institute for Applied Systems Analysis (IIASA) Leonor Tarrason, David Simpson, Peter Wind, Jan-Eiof Jonson. Norwegian Meteorological Institute (MET.NO), Oslo, Norway. Version 2 (including tables of impact estimates). November 2004

88 Baseline Scenarios for Service Contract for carrying out cost-benefit analysis of air quality related issues, in particular in the clean air for Europe (CAFE) programme. Paul Watkiss, Steve Pye and Mike Holland. April 2005. www.cafe-cba.org

Through several rounds of analysis, it was found that control costs started to increase significantly at about 75% between the baseline and MTFR in 2020. Therefore the assessment focused on the range between 50% and 100% of MTFR. Three possible ambition levels (options) were identified and agreed by the CAFE WG and SG, and these were proposed for a more detailed analysis (including a cost-benefit analysis). These ambition levels combine the health-related PM2.5 and ozone objectives with those of environmental protection for acidification, eutrophication and ozone damage to vegetation, as shown in the table below. The Ambition Levels reflect progressively more ambitious levels towards the maximum technical feasible reduction (MTFR) and were labelled A, B, and C.

Table 2 Health and environmental targets (Ambition Levels) in CAFE for 2020.

			Ambition level (% gap closure towards MTFR)			
	2000	Baseline 2020	Scenario A	Scenario B	Scenario C	MTFR
EU-wide cumulative years of life years lost (YOLL, million)	203	137 (0%)	110 (65%)	104 (80%)	101 (87%)	96 (100%)
Acidification (country gap closure on excess deposition)	120	30 (0%)	15 (55%)	11 (75%)	10 (85%)	2 (100%)
Eutrophication (country gap closure on excess deposition)	422	266 (0%)	173 (55%)	138 (75%)	120 (85%)	87 (100%)
Ozone (gap closure on SOMO35)	4081	2435 (0%)	2111 (60%)	2003 (80%)	1949 (90%)	1895 (100%)

The % refers to the difference between Baseline 2020 and Maximum Technically Feasible Reduction.
Key: YOLLs = years of life lost; SOMO35 = sum of mean (ozone concentrations) over 35ppb.
Note: to reduce the concentrations of particulate matter in air it is necessary to reduce both primary PM, but also NOx, SO2 and NH3, as they are precursors for secondary PM2.5 species.

It is important to differentiate the roles of the RAINS and CBA models in the process. The RAINS model identified a cost-effective set of measures for meeting pre-defined health and environmental quality targets. The CBA model added to this analysis by assessing the magnitude of benefits and assesses whether overall benefits are higher or lower than the estimated costs; in other words, whether it is worth carrying out the measures identified in the RAINS model (though note that the CBA does not include valuation of all environmental benefits, notably the omission of benefits to ecosystems).

Benefits Methodology

The analysis of the costs of potential measures was undertaken in the RAINS integrated assessment model. This builds on a well established database of technical options and an established methodology (from standard appraisal) for technical cost assessment. However, for the analysis of benefits, new methodologies had to be developed to assess the environmental costs of environmental improvements, for input to the cost-benefit analysis. The benefits methodology was compiled by a multi-disciplinary team. The development of this CAFE CBA methodology can be traced back to the beginning of the EC DG Research ExternE Programme that started in 1991 and continues to the present day. Further to this, the methodology developed was the subject of intense consultation in 2003 and 2004 with stakeholders from the European Union Member States, academic institutes, environment agencies, industry and non-governmental organisations. It was also subject to formal peer review by senior experts in the U.S.A. and Europe (the peer review report is available at the website). The full CAFE CBA methodology is described in three volumes (Holland et al., 2005a, b; Hurley et al., 2005[89]).

This approach for quantifying the benefits of reducing air pollution followed a logical progression through the following stages:
1. Quantification of emissions (using results from the RAINS model);
2. Description of pollutant dispersion and chemistry across Europe (again, based on outputs from the RAINS model);
3. Quantification of exposure of people, environment and buildings that are affected by air pollution (linking the pollution concentrations with the "stock at risk", e.g. population data);
4. Quantification of the impacts of air pollution, using relationships linking pollution concentrations with physical impacts;
5. Valuation of the impacts where possible; and

89 Mike Holland, Fintan Hurley, Alistair Hunt, Paul Watkiss (2005). Volume 3: Uncertainty in the CAFE CBA: Methods and First Analysis.Service Contract for Carrying out Cost-Benefit Analysis of Air Quality Related Issues, in particular in the Clean Air for Europe (CAFE) Programme.
Available from http://europa.eu.int/comm/environment/air/cafe/activities/cba.htm and http://www.cafe-cba.org/

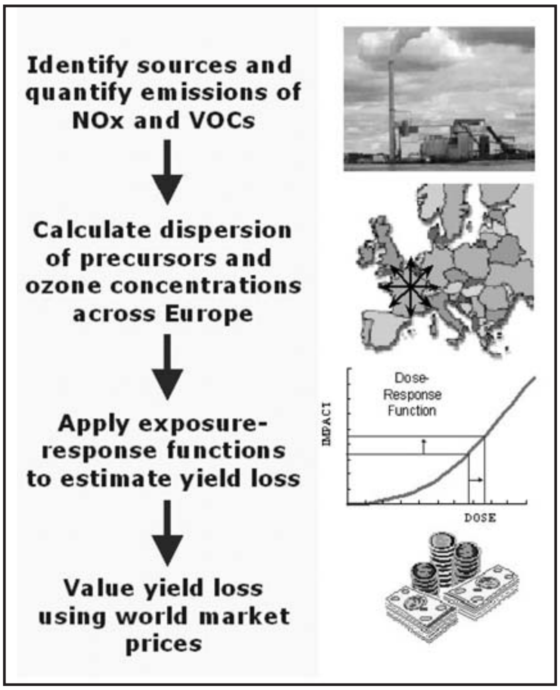
6. Assessment of the potential importance of uncertainty with regard to the balance of the costs of pollution control quantified by the RAINS model and their associated benefits.

This approach is known as the impact pathway approach, shown below. Following from the figure, impacts and damages under any scenario are calculated using the following general relationship:

impact = pollution x stock at risk x response function

economic damage = impact x unit value of impact

Figure 3: Impact Pathway Approach



Although the underlying form of the above equation does not change, the precise form of the equation will vary for different types of impact. For example, the functions that describe materials damage from acidic deposition require consideration of climatic variables (such as relative humidity) and need to account for several pollutants simultaneously. For any type of receptor it is necessary to implement a number of these impact pathways to generate overall benefits. So, for example, in the case of impacts of ozone on crop yield, it is necessary to consider, separately, impacts on a series of different crops, each of which differs in sensitivity. For health assessment it is necessary to quantify across a series of different effects to understand the overall impact of air pollution on the population.

The final stage valuation is generally done from the perspective of "willingness to pay" (WTP). For some effects, such as damage to crops, or to buildings of little or no cultural merit this can be done using appropriate market data. Some elements of the valuation of health impacts can also be quantified from 'market' data (e.g. the cost of medicines and care), though other elements such as willingness to pay to avoid being ill in the first place are clearly not quantifiable from such sources. In such cases alternative methods are necessary for the quantification, such as the use of contingent valuation. The full consideration of different effects in the CAFE CBA is presented below.

Table 3: Effects of the CAFE pollutants, and the extent of assessment

Effect	Impact quantified and valued	Impact only quantified	Qualitative assessment	Comments
Health				
Primary PM, NO3 and SO4 aerosols acute – mortality, morbidity chronic – mortality, morbidity infant mortality	 √ √ √	 √ √ √		 Care taken to avoid double counting with chronic effects
Ozone acute – mortality chronic – mortality acute – morbidity chronic – morbidity	 √ √ 	 √ √ 	 √ √	 Less clear linkage between O3 and mortality than for PM10 No information on possible chronic effects
Direct effects of SO2			√	Limited importance to CAFE
Direct effects of VOCs			√	Lack of data on speciation, etc.
Direct effects of NO2			√	Lack of clear information of effects at ambient levels
Social impacts			√	Limited data availability
Altruistic effects			√	Reliable valuation data unavailable

Agricultural production				
Direct effects of SO2 and NOx			√	Negligible according to past work
Direct effects of O3 on crop yield	√	√		
Indirect effects on livestock			√	
N deposition as crop fertiliser			√	Negligible according to past work
Visible damage to marketed produce		√	√	
Interactions between pollutants, with pests and pathogens, climate…			√	Exposure-response data unavailable
Acidification/liming			√	Negligible according to past work
Materials SO_2/acid effects on utilitarian buildings	√	√		
Effects on cultural assets, steel in reinforced concrete			√	Lack of stock at risk inventory and valuation data
PM and building soiling	√	√		
Effects of O3 on paint, rubbe	√	√		
Ecosystems Effects on biodiversity, forest production, etc., from excess O3 exposure		√	√	Valuation of ecological impacts is currently too uncertain
Effects on biodiversity, etc., from excess N deposition		√	√	Valuation of ecological impacts is currently too uncertain
Effects on biodiversity, etc., from excess acid deposition	√	√		Valuation of ecological impacts is currently too uncertain
Visibility: Change in visual range			√	Impact of little concern in Europe.
Change in greenhouse gas emissions		√	√	Valuation too uncertain
Macroeconomic effects		√	√	Addressed using the GEM-E3 model
Drinking water supply and quality	√		√	Limited data availability

For air pollution, four main categories of receptors were assessed: health, crops and materials, which were all quantified and valued in monetary terms, and ecosystems which were assessed in terms of physical changes but not valued. Data sources are shown below. As highlighted above, whilst the method built on existing work, it was improved through independent advice on key areas, e.g. from the World Health Organisation on health impact assessment. The method was also subject to extensive stakeholder consultation and a formal peer review.

Table 4. Sources of data for the benefits assessment

	Stock at risk	Response functions	Valuation
Health	UN population data, with additional factors for sensitivity of the population from Eurostat	Working group convened by WHO for mortality analysis. Morbidity by CAFE CBA (Hurley) based on ExternE plus input from WHO.	Surveys undertaken in NewExt and other projects and debate under CAFE.
Materials	EC ExternE Project	ICP Materials working under LRTAP Convention	Repair cost data from architectural sources
Crops	Stockholm Environment Institute	ICP Vegetation working under LRTAP Convention	World market prices from FAO
Ecosystems	Coordinating Center for Effects	Coordinating Center for Effects via GAINS, providing outputs in terms of exceedance areas	None

In addition, a detailed uncertainty analysis method using Monte Carlo analysis was developed and applied (see later section), and wider economy effects were also considered using a general equilibrium model (GEME3).

Health benefits

The health impacts from air pollution arise from short-term and long-term exposure. Short-term health impacts include premature mortality (deaths brought forward), respiratory and cardio-vascular hospital admissions, and potentially exacerbation of asthma and other respiratory symptoms. The evidence for these effects is strongest for particles (usually characterized as PM_{10} – particulate matter less than 10 microns in diameter) and for ozone. Long-term (chronic) exposure to air pollution (PM pollution) also damages health and these effects - measured through changes in life expectancy - are substantially greater than the effects of acute exposure. For PM, there is no safe threshold value for effects.

Full details of the methodology and impact functions, plus valuation estimates, for health are presented in the methodology reports. All functions and values are summarised in an Appendix at the end of this paper, and a number of key areas are summarised below:

The analysis of chronic mortality from PM pollution, following WHO (World Health Organisation) guidance[90], used the central estimate of a 6% increase in mortality hazard rates per 10 µg/m3 PM2.5 based on the U.S. Pope et al. study[91], implemented for anthropogenic PM, with no threshold. Consistent with WHO guidance, and a wider emerging consensus in favour of using life table methods, the analysis expressed health impacts in terms of years of life lost from air pollution. In addition, consistent with the recommendations of the external peer review, the analysis also included estimates of the number of deaths per year attributable to long-term exposure to ambient $PM_{2.5}$[92]. The approach used estimated attributable deaths using a "static" approach (without life tables) where the annual death rate is multiplied by the PM risk factor. This method is approximate and is considered to over-estimate the true attributable fraction to some extent. Consequently mortality effects of long-term exposure to PM were expressed both as years of life lost and as attributable cases of premature mortality and both are relevant for monetary valuation.

90 As part of the CAFE process, the WHO was consulted on health issues. WHO is involved in review of health impact data for both CLRTAP (UNECE Convention on Long-Range Transboundary Air Pollution) and the European Commission as part of the CAFE process. As part of the latter, the recommendations of WHO-CLRTAP Task Force on Health (TFH) *(http://www.unece.org/env/documents)* and the WHO "Systematic Review of Health Aspects of Air Quality in Europe" *(http://www.euro.who.int/document/e79097.pdf)* were key to the development of quantification methods for assessing health impacts of air pollution, the WHO-sponsored meta-analyses of the acute effects of PM and ozone based on studies in Europe *(http://www.euro.who.int/document/e82792.pdf)*, and also the process drew on the answers to follow-up questions *(http://www.euro.who.int/document/e82790.pdf)* asked by the CAFE Steering Group.

91 Pope C.A. III, Thun M.J., Namboodiri M.M., Dockery D.W., Evans J.S., Speizer F.E. and Heath C.W. Jr. (1995). Particulate air pollution as predictor of mortality in a prospective study of U.S. adults. Am J Resp Crit Care Med 151: 669-674.
Pope, CA III, Burnett RT, Thun MJ, Calle EE, Krewski D, Ito K, Thurston GD (2002). Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. Journal of the American medical association, 287: 1132 - 1141.

92 Estimates of attributable deaths have their own methodological problems. However, number of premature deaths appear easy to understand, and so are often made in HIAs of air pollution and health.

Table 5. Values for use in CAFE CBA: Effects of chronic exposure on mortality.

	VSL	VOLY	Derived from:
Median (NewExt)	€980,000	€52,000	Median value
Mean (NewExt)	€2,000,000	€120,000	Mean value

For valuation, the analysis was able to take advantage of new research under the EC DG Research NewExt Project[93]. There has been some debate as to whether it is appropriate to take the mean or median values from the NewExt analysis of VSL and VOLY. The most relevant measure of society's willingness to pay (WTP) is the mean, though this can be affected significantly by a few extreme values. In contrast, the median, though less relevant as an indicator of the average societal WTP, is more robust. Consistent with the external peer review guidance, the analysis used both VSL and VOLY approaches, with mean and median values, which gives four alternatives on valuation.

The actual difference in mortality damage quantified using VOLY and VSL-based methods is not as great as the above table might suggest. Much of the difference between VSL and VOLY is cancelled out by the difference between the number of premature deaths quantified compared to the number of life years lost, and there is extensive overlap in the ranges. This issue is addressed in greater depth in Volume 3 of the CAFE-CBA Methodology Report.

For PM morbidity, a set of functions were used based on studies of the effects of acute exposures (from observation of response to day-to-day variations in ambient PM) as well as of long-term (chronic) exposures. A similar approach was also adopted for ozone and morbidity.

For acute mortality from ozone, the analysis quantifies the number of "premature deaths" (deaths brought forward)[94]. Following guidance from WHO, the analysis used a risk estimate of 0.3% increase in daily mortality per 10 µg/m3 O3 – this is the estimate from the WHO-sponsored meta-analysis of time series studies in Europe. Mortality impacts were expressed initially in terms of numbers of cases. Note that the health impact here can best be characterised as a "deaths brought forward" attributed to ozone. This is to signify that people whose deaths are brought forward by higher air pollution almost certainly have serious pre-existing cardio-respiratory disease and so in at least some of

93 NewExt (2004) "New Elements for the Assessment of External Costs from Energy Technologies". Funded under the EC 5th Framework Programme (1998 – 2002), Thematic programme: Energy, Environment and Sustainable Development, Part B: Energy; Generic Activities: 8.1.3. Externalities ENG1-CT2000-00129.

94 This is to signify that people whose deaths are brought forward by higher air pollution almost certainly have serious pre-existing cardio-respiratory disease and so in at least some of these cases, the actual loss of life is likely to be small – the death might have occurred within the same year and, for some, may only be brought forward by a few days.

these cases, the actual loss of life is likely to be small. These cases are valued using a VOLY approach, assuming that on average, each premature death leads to the loss of 12 months of life. The range for the VOLY is therefore applied to these impacts.

Crops

Air pollution is recognised both in Europe and the U.S.A. as having a significant influence on agricultural and horticultural production. The analysis considered the effects of ozone on crop yield. The units of ppm (parts per million) hour refer to total ozone exposure aggregated by hour when concentrations exceed 40 ppb (parts per billion) for a seven hour period each day over a three month growing season. This metric is referred to as AOT40. The valuation of impacts on agricultural production is reasonably straight forward, with estimated yield loss being multiplied by world market prices as published by the UN's Food and Agriculture Organization. World market prices are used as a proxy for shadow price on the grounds that they are less influenced by subsidies than local European prices (in other words, they are closer to the "real" price of production). Functions and Values are shown in the Appendix.

Materials

Air pollution is associated with a number of impacts on materials: acid corrosion of stone, metals and paints in "utilitarian" applications; acid impacts on materials of cultural merit (including stone, fine art, and medieval stained glass, etc.); ozone damage to polymeric materials, particularly natural rubbers; and soiling (PM) of buildings and materials used in other applications. Only the first of these was quantified in CAFE. CAFE uses stock-at-risk data collected in a number of studies, particularly ExternE (1995 and 1999[95]). These studies provide data for individual cities or countries in both eastern and western parts of Europe. Functions have been taken from the ICP Materials (2003) website[96], based on exposures over an eight year period (1987 to 1995). The most important are likely to be those relating to steel, zinc and stone (limestone and sandstone), and application for mortar and rendering.

Calculation Framework

Calculations were made for each cell within a grid system generated by dispersion modelling, using GIS at a 50 by 50 km resolution across Europe.

95 ExternE (1995). European Commission, DGXII, Science, Research and Development, JOULE. Externalities of Energy, 'ExternE' Project. Volume 2. Methodology. (EUR 16521 EN).
ExternE (1999). European Commission Directorate-General XII Science, Research and Development. ExternE Externalities of Energy. Volume 7: Methodology 1998 Update.

96 ICP Materials (2003) Dose-response functions. *http://www.corr-institute.se/ICP-Materials/html/dose_response.html*

Benefit Results

The benefits method above was used to estimate the physical impacts and economic costs of air pollution in 2020 for the baseline and the three ambition levels. The benefits of the different ambition levels, i.e. the change over the baseline, are shown below.

These are also expressed in monetary values below. Values were also generated at Member State level, as well as for the EU25.

Table 6. Total Physical health benefits per year of ambition levels (EU25) over the 2020 baseline.

End point	Pollut-ant	Unit	Scenario A	Scenario B	Scenario C	MTFR
Chronic mortality (years)	PM	thousand	492.5	600.8	654.6	744.6
Chronic mortality (premature deaths)	*PM*	*thousand*	*53.8*	*65.7*	*71.6*	*81.4*
Infant mortality (0-1 years) (premature deaths)	PM		70	80	90	100
Chronic bronchitis (over 27 years)	PM	thousand	25.5	31.1	33.9	38.5
Respiratory hospital admissions (all ages)	PM	thousand	8.5	10.3	11.2	12.8
Cardiac hospital admissions (all ages)	PM	thousand	5.2	6.4	6.9	7.9
Restricted activity days (15-64 years)	PM	million	44.4	54.1	58.9	67.0
Respiratory medication use (children 5-14 years)	PM	million	0.4	0.5	0.5	0.6
Respiratory medication use (adults over 20 years)	PM	million	4.2	5.1	5.5	6.3
Lower respiratory symptom (LRS 5-14 years)	PM	million	17.7	21.7	23.6	27.0
LRS among adults (over 15years) with chronic symptoms	PM	million	41.4	50.5	55.0	62.6
Acute mortality (premature deaths)	O_3	thousand	1.6	2.2	2.5	3.0
Respiratory hospital admissions (over 65years)	O_3	thousand	1.6	2.1	2.5	2.9
Minor restricted activity days (MRADs 15-64 years)	O_3	million	3.2	4.3	4.9	5.9
Respiratory medication use (5-14 years)	O_3	million	1.0	1.3	1.5	1.8
Respiratory medication use (over 20 years)	O_3	million	0.6	0.8	1.0	1.1
Cough and lower respiratory symptom (LRS 0-14 years)	O_3	million	4.9	6.6	7.5	9.1

Note: chronic mortality is quantified in two alternative ways. The estimates are not additive.

The monetary values for crops and materials were added to these values, though they only increased total benefits slightly (i.e. non-health benefits were only measured in millions of Euro, though it is stressed that ecosystems benefits are not included in the valuation).

Table 7. Total Annualised Monetary Benefits of the Ambition Levels (EU25) over the 2020 baseline - Million Euro

Endpoint	Pollut-ant	Scenario A	Scenario B	Scenario C	MTFR
Chronic mortality – VOLY – (median value)	PM	25,750	31,412	34,225	38,927
Chronic mortality – VSL – (median value)	*PM*	*52,726*	*64,313*	*70,122*	*79,680*
Chronic mortality – VOLY – high (mean value)	PM	57,798	70,508	76,822	87,377
Chronic mortality – VSL – high (mean value)	*PM*	*108,479*	*132,319*	*144,271*	*163,935*
Infant mortality (0-1 years) – (median value)	PM	100	121	132	150
Infant mortality (0-1 years) – (mean value)	*PM*	*199*	*242*	*264*	*300*
Chronic bronchitis (over 27 years)	PM	4,786	5,827	6,348	7,219
Respiratory and cardiac hospital admissions	PM	27	34	37	42
Restricted activity days (RADs 15-64 years)	PM	3,703	4,512	4,915	5,589
Respiratory medication use	PM	4	5	6	7
Lower respiratory symptoms	PM	2,272	2,774	3,022	3,440
Acute mortality (VOLY median)	O_3	83	110	127	152
Acute mortality (VOLY mean)	*O_3*	*186*	*248*	*285*	*342*
Respiratory hospital admissions and medication use	O_3	5	6	7	9
Minor restricted activity days (MRADs 15-64 years)	O_3	124	165	190	228
Cough and lower respiratory symptoms (0-14 years)	O_3	189	252	290	349
Total with mortality – VOLY – (median value)		37,043	45,218	49,299	56,112
Total with mortality – VSL – (median value)		64,019	78,119	85,196	96,865
Total with mortality – VOLY –(mean value)		*69,293*	*84,573*	*92,186*	*104,902*
Total with mortality – VSL – (mean value)		*119,974*	*146,384*	*159,635*	*181,460*

The benefits of ambition level A were estimated at 37 to 120 million Euros per year (the range reflecting the low and high values from the four alternative mortality valuation combinations). Benefits increase with more ambitious scenarios up to Ambition level C with benefits of 49 to 160 million Euros per year. These were combined with other information to present the overall effects of each ambition levels, shown below. These benefit values were compared to the cost outputs from the RAINS analysis, also included below.

Table 8: Analysis of the Different Ambition Levels (and final strategy)

		Human health		Natural environment					
Ambition level	Cost of reduction (€bn)	Life Years Lost due to $PM_{2.5}$ (million)	Premature deaths due to $PM_{2.5}$ and ozone (thousands)	Range in monetised health benefits[10] (€bn)	Ecosystem area exceeded acidification (000 km^2)			Ecosystem area exceeded eutrophication (000 km^2)	Forest area exceeded ozone (000 km^2)
					Forests	Semi-natural	Fresh-water		
2000		3.62	370	-	243	24	31	733	827
Baseline 2020		2.47	293	-	119	8	22	590	764
Scenario A	5.9	1.97	237	37 – 120	67	4	19	426	699
Scenario B	10.7	1.87	225	45 – 146	59	3	18	375	671
Scenario C	14.9	1.81	219	49 – 160	55	3	17	347	652
MTFR	39.7	1.72	208	56 – 181	36	1	11	193	381
Strategy	**7.1**	**1.91**	**230**	**42 – 135**	**63**	**3**	**19**	**416**	**699**

Cost-Benefit Analysis Results

The data above were used to undertake a cost-benefit analysis of the different ambition levels. Results are first presented below in terms of total costs and benefits. This shows a high benefit to cost ratio, even with the lower end of benefits for Ambition Level A, with annualised benefits more than six times annualised costs (with the high estimate of benefits, benefits exceed costs by a factor of 20). The ratio of benefits to costs falls with more ambitious levels (as expected) due to the relatively small increases in additional benefits, but the sharp rise in costs reflecting progressively more expensive options from the cost curves. The analysis also looked at the incremental changes in benefits and costs (the marginal changes from moving to progressively more ambitious levels), shown in the second table below.

Table 9: Total Cost Benefit Analysis Results – Comparison of total Annualised Costs (Billion Euro [billion = 1000 million]) for EU25 for Different Ambition Levels

	A	B	C	MTFR
EU Annualised benefits (health, materials and crops) change over base				
Low estimate	38	46	50	57
High estimate	120	147	160	182
EU-25 Annualised Costs - change over base line				
Total	5.9	10.7	14.9	39.7
NET benefits				
Low estimate	32	35	35	17
High estimate	115	136	145	142
Benefit to Cost Ratio				
Low estimate	6.3	4.3	3.4	1.4
High estimate	20	14	11	4.6

Table 10: Marginal Cost Benefit Analysis Results – Comparison of total Annualised Costs (Billion Euro [billion = 1000 million]) for EU25 for Different Ambition Levels

	from CLE to A	from A to B	from B to C	from C to MTFR
EU incremental annualised benefits (health and crops)				
Total with Mortality – VOLY – low (median)	38	8.3	4.1	6.9
Total with Mortality – VSL – high (mean)	120	27	13	22
EU-25 annualised costs in Billion€/year - incremental changes to each scenario				
Total	5.9	4.8	4.2	25
NET incremental benefits				
Total with Mortality – VOLY – low (median)	32	3.5	-0.03	-18
Total with Mortality – VSL – high (mean)	115	22	9.1	-3.0
Benefit to cost ratio				
Total with Mortality – VOLY – low (median)	6.3	1.7	0.98	0.3
Total with Mortality – VSL – high (mean)	20	5.6	3.2	0.9

The analysis of incremental changes (marginal changes) provides information on the likely optimal policy, i.e. the point where net benefits are maximised (this occurs when the marginal abatement costs are equal to the marginal abatement benefits). As can be seen from the table above, this occurs in moving from Ambition Level B to C with the low estimate of benefits (highlighted in the square, where the benefits and costs are approximately equal).

The analysis also considered the benefit to cost ratio in different member states. This was important to ensure that the overall policy is equitable, i.e. trying to avoid cases where large benefit to cost ratios for some countries occur at the expense of very negative benefit to cost ratios for others.

A number of other metrics were used to assess the policies, in addition to the headline CBA figures above. These were useful in trying to test how different presentations were received by stakeholders (examples calculated were the cost per life saved and the benefit per person). Overall, it was found that the simple cost-benefit analysis (the net benefits, and especially the ratio of benefits to costs) provided the clearest and more easily understandable metric to stakeholders.

Uncertainty Analysis

As part of the CAFE methodology, the team produced a separate methodological report on uncertainty (Holland et al.). This considered a number of different uncertainties using a number of techniques including:
Sensitivity analysis using an additional set of health impact functions;
Scenario analysis considering the effect of meteorological year;
A bias analysis, assessing the potential omissions and their likely bias;
- An "extended" CBA where impacts that were monetised were presented alongside omitted effects (such as ecosystem valuation) but with a numerical analysis and commentary of how potentially important the omitted effects were.

The analysis also undertook Monte Carlo analysis. Monte Carlo analysis is a risk modelling technique that presents both the range, as well as the expected value, of the collective effects of various risks. It is very useful when there are many variables with significant uncertainties, or when the implications of uncertainty (and the effect on decision) cannot be adequately captured through sensitivity analysis. At a detailed level, the approach works by undertaking a probabilistic simulation: simulating a probability distribution of outcomes by randomly selecting from the probability distributions of input parameters and repeating the analysis numerous (thousands of) times. The input parameter probability distributions may be derived empirically (e.g. directly from population data or indirectly from regression or other statistical models) or by assumption. In contrast to the other methods, this method has the advantage of weighing explicitly the likelihood of alternative outcomes, permitting evaluation of their relative importance.

The analysis undertook Monte Carlo analysis using the @Risk model. Distributions were included for all of the input parameters associated with impact functions and valuation, and then the distributions of benefits for the different ambition levels were assessed.

The Monte Carlo analysis was then extended to look at the probability that benefits exceeded costs. In order to assess additional uncertainty with respect to cost estimates, the analysis built a distribution based around the recent literature on *ex ante* and *ex post* costs (Watkiss et al., 2005[97]; IVM, 2006[98]). These show that in many cases, *ex post* cost estimates (the costs of the legislation as estimated after the legislation is implemented) differ significantly from *ex ante* estimates (estimates of the costs of legislation in appraisal, i.e. before implementation). In many cases, the *ex post* costs are a factor of two lower (though there are notable exceptions). A distribution of costs was therefore used that built in a cost range from 50% to 120% of the RAINS *ex ante* estimates to take account of this. The analysis then compared the probability distribution of benefits against the range of cost estimates, and investigated the probability that benefits exceeded costs. This metric was found to be the most easily understandable and transparent way of presenting the overall conclusions of the Monte Carlo analysis. An example output is presented below, showing the probability that benefits exceed costs for Ambition Level B, plotted against the range of costs (from 50 to 120%) for each of the four runs on chronic mortality (median and mean VSL and VOLY).

97 Watkiss, P, Baggot, S, Bush T, Cross, S, Goodwin, J, Hunt, A, Hurley, F, Holland M, Stedman, J (2005). Evaluation of Air Quality Strategy (EPES 0203/1). Published by DEFRA, January 2005. AEA Technology Environment, Metroeconomica, and the Institute of Occupational Medicine. http://www.defra.gov.uk/environment/airquality/strategy/evaluation/index.htm

98 EX-post estimates of costs to business of EU environmental legislation, IVM (2006)

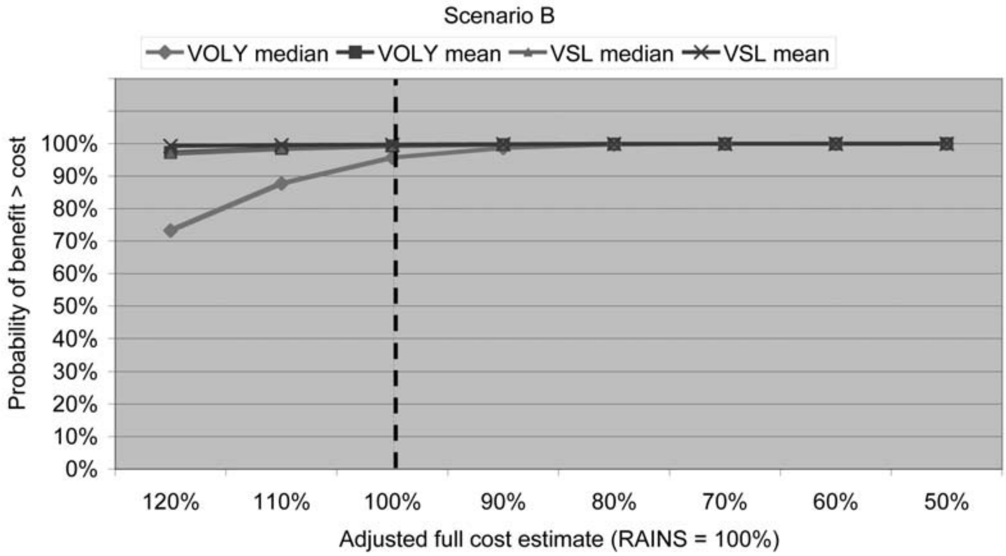


Figure 2: Monte Carlo Results: Probability of benefits exceeding costs for alternative VOLY/VSL choices: Ambition Level B

The overall analysis showed that in all cases, there was an extremely high chance that benefits would exceed costs for Ambition Level A (in excess of 98%). There was also a very high chance that benefits would exceed costs for Ambition Level B - >90% in nearly all cases, except where costs are much higher and when the lower median VOLY run is used (and even then the probability is over 70%).

Macro-economic analysis

The final part of the overall analysis was to undertake a macro-economic assessment of the overall policy, to investigate the impact of the additional costs of the policy on the economy (i.e. on employment, competitiveness and overall GDP). To do this, the study used a general equilibrium model, the GEM-E3 model. Overall the model found that the macroeconomic costs of air pollution reduction were limited when compared to the benefits obtained in terms of air quality, health and ecosystem improvement, and overall effects were estimated to be significantly less than 0.1% GDP.

CBA as an input to Policy Decision Making

The analysis above showed that the cost-benefit analysis results, even with the low estimate of benefits and even with omitted benefits from ecosystem effects, could justify a high level of Ambition. The total benefits of Ambition Level C exceeded the costs, and the marginal analysis showed that the optimal policy point (for the low estimate of benefits) occurred in moving to Ambition Level C. The Monte Carlo analysis tested the uncertainty associated with different ambition levels: with a central estimate of costs, the probability that benefits exceeded costs was greater than 90% with the Ambition Level B (and still above 70% for this scenario even if costs were underestimated by 20%). The probabilities did drop for Ambition Level C, but were still high (i.e. it was still "very likely" that benefits would exceed costs).

The information from the cost-benefit analysis was used as an input to the EC considerations for the proposed Thematic Strategy. This is consistent with the use of CBA as an aid to decision making[99]. The final proposed ambition level, as adopted in the Thematic Strategy, has an Ambition A level for acidification, eutrophication and ozone, and 75% ambition level for health (i.e. between the A and B ambition levels). This achieves estimated benefits (low estimate) of at least €42 billion per annum, and is estimated to cost approximately €7.1 billion per annum (representing about 0.05% of the EU-25 GDP in 2020), so the ratio of benefits to costs of the overall policy is high, and the confidence that benefits would exceed costs (from an additional Monte Carlo run) showed almost a 99% probability.

Conclusions and Lessons Learnt

Overall, the feedback to the cost-benefit analysis was positive. Stakeholders found the presentation of headline figures from the CBA were a simple and clear way to compare the benefits and costs of different policy options. The detailed CBA also formed a significant input to the EC Impact Assessment (the evaluation of which was viewed very favourably in a review of EC Impact Assessments presented by another speaker at the workshop). We conclude that cost-benefit analysis is an extremely useful input to the policy decision, though we highlight that it is only one strand of information that should go into the overall policy analysis and decision. In looking back at the positive and negative aspects of the CAFE CBA, a number of observations are raised with a review of lessons learnt below.

99 Project appraisal is intended to produce an indication of the degree to which a proposed project or scheme is justified. It can also be used to rank or prioritise alternative schemes or options. However, appraisal is an input into decision-making, not a substitute for it. Policy appraisal is one strand of information that informs whether to proceed with a particular course of action. As with any approach, it will inevitably entail some judgements in areas such as distribution, risks and uncertainties.

In order for a cost-benefit analysis to be accepted by different stakeholders, there is a need for the analysis to be scientific and evidence based. In CAFE, the process benefited enormously from independent inputs on key areas (e.g. independent advice from the World Health Organisation on health effects).

- Similarly, the processes of external peer review and stakeholder consultation were both extremely useful in providing input to, and re-enforcing the independent nature of the analysis.
- It was found that the simplest results work best in communicating key findings to varied groups of stakeholders and policy makers. For example, in the case of the CBA itself, simple headline values for the ratio of benefits to costs (and marginal benefits to costs) worked well, rather than more complex but potentially informative detail. Whilst the simple message is good in communicating the overall results, it must be complemented by detailed analysis and documentation to ensure that those that are interested can delve into the analysis.
- Moreover, the headline CBA results were found to work best when complemented by a presentation of physical effects alongside the monetary values. This has the advantage of providing data to stakeholders that are unsure of CBA, and also providing a much richer detail to allow stakeholders to conceptualise what the benefits really mean in practice (e.g. in terms of numbers of deaths per year avoided).
- However, whilst CBA simplifies the story for stakeholders – and has the very distinct advantage of being able to aggregate different impacts together - it remains extremely difficult to present non-monetised data alongside the headline CBA values. These omitted benefits are therefore often forgotten. Some progress was made through an extended CBA in CAFE, which tried to present quantified or semi-quantified analysis of omitted benefits alongside the CBA (rather than trying to extend to a more complex multi-criteria analysis), but this remains a priority for future analysis and presentation.
- The uncertainty analysis did prove extremely useful in helping to convey the confidence in potential decisions. The consideration of how to communicate this information re-enforced the finding above, as it was found that a simple presentation was preferable. This is particularly important especially as Monte Carlo analysis is a very technically detailed tool. In the study here, the output that was most useful was the simple headline of the "probability that benefits exceeded costs".
- The success of a CBA is significantly affected by the person commissioning the work: in the case of CAFE, this was a senior economist and there is no doubt that the CBA benefited from a two way dialogue on how best to assess and present the CBA.

- It is also clear that, however much a CBA seeks to ensure independent input, to clearly communicate the analysis, etc. there will be different stakeholder groups who will look to criticise or strongly back the findings. In CAFE there was a strong division between industry and NGO support for the CBA. Interestingly, the positions of the stakeholders to CBA changed during the CAFE process (i.e. industry became more concerned, whilst the environmental NGOs became more interested and even cited the CBA results). The positions of different groups may have reflected a switch in perception of the method and process, but may also have changed (opportunistically) as results emerged and showed that higher ambition levels could be supported.
- Finally, it is highlighted that addressing all the above areas has important resource implications. To do a CBA that has stakeholder consultation and peer review, seeks external independent input, undertakes detailed analysis and uncertainty analysis, etc. means a significant resource and time implication. This can also prove extremely challenging within a policy timetable. Whilst these resource issues must be recognised, we strongly believe that the CAFE CBA provides a strong case that such analysis is worthwhile, and can significantly improve and enhance the inputs to allow balanced policy decisions.

Acknowledgements

This paper summarises the findings of the CAFE CBA analysis for the CAFE programme which fed into the Impact Assessment of the Thematic Strategy on Air Pollution. The underlying work of the CAFE CBA project was greatly enhanced by the input from IIASA and from the project officers (particularly Matti Vainio) at the European Commission DG Environment. However, the views expressed here (and interpretation of lessons learnt) are solely those of the author.

Appendix: Health Response Functions used in CAFE CBA

AM under pollutant metric = annual mean concentration. ERF = exposure response function. Conversion from PM10 concentration to PM2.5 concentration may be made using a factor of 0.65 in developed countries, and 0.5 in emerging economies.

Effect	Pollutant metric	Popn factor 1	Popn factor 2	Incidence rate	ERF
PM analysis					
Chronic mortality (deaths, VSL valuation)	AM PM2.5	0.628	1	1.61%	0.6%
Chronic mortality (life years lost, VOLY valuation)	AM PM2.5	1	1.00E-05	1	65.1
Infant mortality (infants aged 1 month – 1 year)	AM PM10	0.01	1	0.19%	0.4%
Chronic bronchitis	AM PM10	0.70	1	0.378%	0.7%
Respiratory hospital admissions	AM PM10	1	1.00E-05	617	0.114%
Cardiac hospital admissions	AM PM10	1	1.00E-05	723	0.060%
Restricted activity days (RADs)	AM PM2.5	0.672	1	19	0.475%
Respiratory medication use by adults	AM PM10	0.817	0.001	4.50%	90.8
Respiratory medication use by children	AM PM10	0.112	20%	36.50	0.050%
LRS, including cough, among adults with chronic symptoms	AM PM10	0.817	1	0.3	0.13
LRS (including cough) among children	AM PM10	0.112	1	1	0.185
Ozone analysis					
Acute mortality (deaths, VSL valuation)	8 hrmean	1	1	1.10%	0.03%
Acute mortality (life years lost, VOLY valuation)	8 hrmean	1	1	1.10%	0.03%
Respiratory hospital admissions	8 hrmean	1	1.00E-05	617	0.03%
Minor restricted activity days	8 hrmean	0.64	1	7.8	0.148%
Respiratory medication use by adults	8 hrmean	0.817	0.001	4.50%	71.3
Respiratory medication use by children	8 hrmean	0.13	0.001	1	21.7
Cough and LRS (excluding cough) among children	8 hrmean	0.13	1	1	0.11

*Population factor 1 is the adjustment to the total population weighted number that we apply for each function. This accounts for the age, and the number of people to whom the exposure response functions refer.

For population factor 2 the following are given:

0.001	Function expressed per thousand people
1.00E-05	Function expressed per hundred thousand people
20%	Fraction of children susceptible to asthma
1	Function expressed per person

Summary of morbidity unit values

Health end-point	Recommended central unit values, € price year 2000
Hospital admissions	2,000/admission
GP visits (event): Asthma Lower respiratory symptoms	 53/consultation 75/consultation
Respiratory symptoms in asthmatics (event): Adults Children	 130/event 280/event
Respiratory medication use – adults and children (day)	1/day
Restricted activity day (adjusted average for working adult) Restricted activity day (adjusted average for age >65) Restricted activity day (stay in bed) Restricted activity day (work loss day) Minor restricted activity day	83/day 68/day 130/day 126/day 38/day
Cough day	38/day
Symptom day	38/day
Work loss day	82/day
Minor restricted activity day	38/day
Chronic bronchitis	190,000/case

Crop Exposure-response and valuation data.

	Unit change in yield/ppm.hour	Euro per tonne
Wheat	0.011	120
Barley	Not sensitive	120
Rye	Not sensitive	80
Oats	Not sensitive	110
Millet	0.0039	90
Maize	0.0036	100
Rice	0.0039	280
Soya	0.012	230
Pulses	0.017	320
Rape	0.0056	240
Sugar beet	0.0058	60
Potatoes	0.0056	250
Tobacco	0.0055	4000
Sunflower	0.012	240
Cotton	0.016	1350
Olives	Not sensitive	530
Hops	0.0092	4100
Grape	0.0030	360
Fruit	Not sensitive	680
Carrots	0.0092	340
Tomato	0.014	800
Water melon	0.031	140
Fresh vegetables	0.0095	340

Part Five

Tools for considering costs and benefits in support to policy decision

Costs and Benefits in an Integrated Modelling Framework for Water Management[100]

Frank Ward[101]

Abstract

Implementing water sector policies at the basin scale requires integrated analysis that accurately sorts out conflicting impacts of proposed policies on the environment and society's economic welfare. Cost-benefit analysis (CBA) has considerable potential to support water decisions by consistently appraising proposals in terms of society's total environmental and economic impact. However, the difficulty of correctly applying CBA to environmental programs with complex paths of influence weakens policymakers' confidence in comprehensive economic assessments at the basin scale. This paper describes and illustrates a method by which costs and benefits can be systematically integrated into an integrated physical, institutional and economic simulation model for the water sector. A simple hydroeconomic model is presented. Its size is small enough to build and run with paper and pencil. But its structure is sufficiently flexible to permit expansion for comprehensive policy design that rests on a foundation of a basin's hydrology, institutional constraints and economic relations. The use of cost-benefit analysis to support environmental policy will always be limited by ethical questions on the distribution of benefits and costs among sectors, income groups, locations and generations. Nevertheless, hydroeconomic models offer a potential resource to efficiently and consistently integrate a basin's hydrology, institutions and economics to support the performance of basin scale cost-benefit environmental assessments.

1. Background

The European Water Framework Directive (WFD), adopted in 2000, introduced an integrated approach to water management in Europe. It establishes a common approach to protecting the water environment and setting environmental objectives for all waters of the European Union (EU), and provides a framework for designing and evaluating future EU water legislation. The main objective of the Directive is for member states to

100 The author is grateful for financial support for this work by the Rio Grande Basin Inititiative and by the New Mexico Agricultural Experiment Station

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achieve good water status for both surface and ground water and to prevent degradation of existing quality where good water status has been achieved. Economics operates at the heart of the WFD and will play a uniquely central role in characterising how and to what extent the WFD is implemented across Europe. Basin scale economic analysis in the WFD will provide essential justification for major changes in the management of Europe's waters. Economic analysis required under the WFD includes the following:

- Explicit requirement for economic analysis of water use in the development of integrated river basin management plans;
- Required to support implementation of the "polluter pays" principle;
- Measures for achieving good water status must be subject to cost effectiveness analysis;
- It requires assessments of cost recovery levels for water services provided to the main economic sectors.

Economic analysis of water use in the formulation of integrated river basin management plans for WFD has four parts:

- Identifying linkages between economic activity and water uses;
- Understanding economic benefits of competing water uses;
- Identifying the extent to which economic benefits are reflected in current pricing policies and regulatory standards;
- Identifying implementation options, such as standards, pricing, economic instruments and costs of each.

WFD's central focus on economic analysis in general and on cost-benefit analysis in particular raises questions that have challenged governments since ancient times: designing and carrying out public programs that improve the welfare of a community of private individuals. It is a daunting task to correctly discriminate among community actions that improve a society's welfare from those that reduce it. The task typically becomes more difficult with the size of the community.

Systematic thinking about quantitative evaluation of public policy began in 1844 with the publication of a classic paper on the benefits of public works by the French engineer and economist Jules Dupuit. What later became known as benefit-cost analysis first came into widespread use in the United States, where it owed its origin to the politics and economics of financing public water projects during the early 1930s. Commonly referred to as cost-benefit analysis (CBA) in the EU countries, the EU has seen growing use of CBA to analyse its policies in recent years. Examples include environmental policy issues like air pollution from municipal waste incinerators and various EU proposals for dealing with acid rain (Hanley, 2001). However, as CBA applications have become more widespread in Europe in recent years, so did criticism of their reliability and validity (Turner, 2007). Debate and controversy continue to surface over issues like discounting, measures of economic and social welfare, distribution of benefits and costs and nonuse values of environmental assets.

Despite continued debates over the moral acceptability and utility of CBA, governments are still major players in regulating, managing, and pricing environmental and natural resources in most parts of the world. Local, regional, and federal governments own, regulate, or influence much of what happens to society's environmental and natural resources. Many conflicting demands are placed on these resources by various groups such as farmers, tribal herders, industry, hunters, environmentalists, and outdoor recreationists. Environmental assets often require a considerable amount of management or protection, so democratic governments overseeing these resources must answer to the public, in whose interest the resources are managed.

Most public environmental policy[102] decisions have effects that cause some people to gain while others lose (Barron, 1998). Policymakers and voters want to know if the proposed decision benefits more than it hurts. Because economic impacts are a major factor in influencing public policy outcomes, taxpayers expect economic analysis to be done objectively and fairly. This is especially true when outcomes of the analysis fail to suit interests of politically powerful people or groups who stand to lose something.

This paper's objective is to describe and illustrate a method by which costs and benefits can be systematically integrated into an integrated physical, institutional and economic simulation model for the water sector. A simple hydroeconomic model is presented. Its size is small enough to build and run with paper and pencil. But its structure is sufficiently flexible to permit expansion for comprehensive environmental water policy design that rests on a foundation of a basin's hydrology, institutional constraints, and economic relations. A brief review of the methods, uses, and principles underlying CBA is presented. Then a simple hydroeconomic model is described, critiqued, and interpreted as it could support design of environmental policies such as the WFD. Model results are shown. Finally, the paper concludes.

[102] The term "policy" is used broadly to include any public action to build, design, implement, or evaluate a project, programme, regulation, or legislative action.

2. Overview of CBA

CBA compares the costs and benefits of a government activity, project or regulation over a relevant time period. It is one practical way to find out the economic (not moral) desirability of government decisions. In finding out how desirable a government decision is, it is important to take a long view by looking at impacts in both the near and far term, and a wide view by accounting for many kinds of impacts on many people, economic interests, and regions. CBA involves the enumeration (listing) and evaluation (measuring) of all relevant costs and benefits.

CBA can be expensive, so many ask the question why various government organisations might be motivated to spend all the resources and money to conduct it. One answer is that what counts as a benefit or a loss to to one or more persons or groups does not necessarily count as a benefit or loss to the whole economy. With CBA, we care most about the economy as a whole and not some small part of it. In many respects, CBA is the government's equivalent of the private businesses profit-and-loss statement. The question of "net return" is asked about a wider group, namely society as a whole. Instead of asking whether the owners of a private enterprise will be made better off by a proposal, CBA asks whether the whole society affected by a proposed programme will be made better off by taking it on. Even where a government can make a profit (e.g., polluter fines), that profit may be the wrong yardstick to use for public environmental or natural resource plans.

A CBA compares the desirable and undesirable impacts of proposed policies. It permits analysts to rank the economic performance of environmental and natural resource projects, policies, and programmes in which impacts are measured in nontechnical terms and estimated by scientific methods. It lets us compare all the gains and losses in a common denominator resulting from some public policy action. A CBA organises information in a way which promotes the conduct of rational policy analysis.[103] A complete CBA compares alternative actions to determine which one provides society with the most economically beneficial use of its resources.

CBA can be used to provide information needed for three kinds of public environmental policy decisions: 1) a simple ranking of the comparative benefits of several possible actions, 2) the optimal size or scale of a project produced by a decision, and 3) the optimal timing or sequencing of several elements of a decision. While CBA got its start

[103] Rational policy analysis considers all the relevant alternatives, identifies and evaluates all the consequences that would follow from the adoption of each alternative, and selects that alternative and its associated consequences that would be preferable in terms of society's most-valued ends.

in the 1930s for U.S. water projects, recent years have seen CBA applied to a huge range of environmental and natural resource policy questions in many countries and cultures. A short list of recent examples includes biological control of invasive plants (Van Wilgen et al., 2004), managed health care (Bloom, 2004), sea lamprey control (Marsden et al., 2003), fuel system technologies (MacLean and Lave, 2003), noise reduction (Becker and Lavee, 2003), power plant emissions (Levy et al., 2002), sulfur restrictions (Lee, 2002), common pool management institutions (Kumar, 2002), clean air emissions control (Krupnick and Morgenstern, 2002), mortality risk reductions (Krupnick et al., 2002), forest conservation (Kniivila et al., 2002), windbreak establishment (Jones and Sudmeyer, 2002), environmental programs (Freeman, 2002 a,b), predator control (Engeman et al., 2002), wild nature conservation (Balmford et al.,2002), ozone pollution control (Yoo and Chae, 2001), climate change policies (Tol, 2001), feedlot ammonia emissions (Shi et al., 2001), inland recreational fisheries in Norway (Navrud, 2001), agricultural research programs (Marshall and Brennan, 2001), and drinking water contaminant standards (Gurian et al., 2001). A few other examples include analyses of automobile fuel propulsion technologies (MacLean and Lave, 2003), wheat irrigation (Al-Karaki, 1998), environmental improvement (Parry and Oates, 2000), surface water treatment regulations (Regli et al., 1999), drip irrigation (Tiwari et al., 1998), environmental risk management (Hofstetter et al., 2002), water quality improvements (Thompson, 1999), sustainable forestry (Jagger and Pender, 2003), control of invasive plants (Culliney, 2005), agroforestry interventions (Neupane and Thapa, 2001), groundwater quality improvements (Yadav and Wall, 1998), public housing policy (Johnson and Hurter, 2000), human health risks from unsafe drinking water (Odom et al., 1999), agricultural water pollution control (Qiu, 2003), improvements of sewer systems (Schultz et al., 2004), groundwater recharge (Botzan et al., 1999), rainwater harvesting (Ngigi et al., 2005), salinity control (Hajkowicz and Young, 2002), river health (Bennett, 2002), and water re-allocations (Messner et al, 2006).

CBA uses a simple decision rule. If for some proposed action, the sum of its benefits exceeds the sum of the costs by a larger amount than any other action with the same aim, where economic efficiency is the objective, the proposed action should be adopted. Otherwise it should not. This decision rule assumes that a dollar is worth the same to everyone, whether it is a small or large loss or gain. That is, an additional dollar of benefit produced by a public program for a rich person is worth the same as a dollar of cost paid by a poor taxpayer who finances that benefit. One advantage of CBA is that the monetary unit is easily understood by everyone and does not require technical specialists to interpret. Properly carried out, information provided by a CBA permits government actions to be taken for which the value of resources produced by the public programme is larger than the value of resources used up in financing it. This assures that the programme contributes positively to economic efficiency.

3. Uses of CBA

3.1 Ex Ante versus Ex Post Evaluation

Many applications of CBA deal with *ex ante* (planned) policy questions. However, policy analysis that only conducted *ex ante* analysis might never learn from past mistakes. By contrast, *ex post* analysis looks backward and asks how well an existing project, programme, or regulation performed after it was established. *Ex post* information can be used for three purposes: (1) to look at the stream of actual benefits and costs produced by actual projects built or policies enacted and to see if the previous *ex ante* CBA was accurate, and if not, what kinds of errors were made; (2) to revise methods, forecasts, and assumptions where mistakes were made; (3) to gain informa-tion on the existing economic impact and value relationships on which future CBAs ultimately rest.[104]

3.2 Programme Design, Implementation, and Review

In principle, a CBA can be used to provide the information to help managers design, implement, and review policies. The use of CBA can sharpen environmental programme aims by more comprehensively accounting for economic benefits and costs of various ways of designing policies and setting priorities. CBA can be used to inform decision makers by providing estimates of overall benefits and costs of a proposed policy, as well as identifying people who stand to gain or lose from the policy.

3.3 High Stakes Programmes

A CBA increases in importance as a policy has higher stakes. The utility of a CBA depends on the stakes involved and the likelihood that the information resulting will influence ultimate decisions. The information provided by a CBA justifies devoting more resources to analysing policies as the stakes are higher. Increasing the scale of a CBA increases the cost and these added costs are good investments only if they inform and influence policy decisions (Arrow et al., 1996). Two examples of high stakes programmes are those

[104] For example, if *ex post* analysis showed that reducing each ton of emissions in a water body connected to a drinking water source reduced hospital admissions by X, that fact tells us something about the future value of controlling particulate emissions by Y.

involving sizeable taxpayer cost and proposed regulations for which the cost of compliance is high.

3.4 Information Source

For programmes whose anticipated costs far exceed their expected benefits, the measured negative net benefits can provide valuable information to decision makers on the costs they incur by deciding to carrying the programme anyway in spite of its weak economic performance. Decision makers often wish to weigh factors other than economic benefits and costs into their policy decisions, such as environmental justice or irreversible environmental damages. Yet, even in these cases, CBA provides valuable information by measuring the economic efficiency benefits lost (opportunity cost) from implementing an economically weak program or the economic efficiency benefits that were never realised from failing to carry out an economically strong programme.

3.5 Politically Attractive Programme Design

The fact that CBA focuses on a project's economic feasibility helps it contribute to the political process. CBA can be used to measure both the overall relationship between benefits and costs and the distribution of those benefits and costs among major interest groups in various times and places. Designing a policy that produces a positive economic efficiency payoff for a wide range of major interest groups may make it more politically attractive than concentrating benefits or costs on a small number of people. That is efficiency improvements may get votes.

4. Economic Principles

4.1 Goals

An ancient challenge surrounding the design of government policy is the question of what ends are served by government activity and what programmes can be established to best meet those ends. CBA is one method that provides information for policymakers to better serve the people through government action. Therefore, establishing the goals of a CBA is of special importance.

A CBA is based on the government policy objective of economic efficiency. This principle states that economic efficiency is the important standard for evaluating government regulations or programmes that are proposed for adoption, maintenance or change. For a government programme to meet the efficiency goal, benefits must exceed costs. Efficiency proponents argue that government programmes cannot be justified unless their benefits outweigh their costs. In this view, targeting the efficiency objective increases the likelihood that government actions will only place burdens on businesses and consumers that are in proportion to improvements in health, safety, or the environment, which also accrue to businesses and consumers. Opponents of economic efficiency argue that other goals are also important.

Environmental and natural resource policies typically redistribute economic benefits as well as costs. Critics of the use of CBA state that it ignores the distribution of benefits and costs, and that a simple summing of costs and benefits across all affected individuals leaves out important considerations of equity. CBA might be more politically acceptable and become more widely used if environmental and natural resource programmes are designed to account for equity. One way to accomplish equity is to design programmes for which benefits exceed costs to most groups (i.e., people in most tax brackets, time periods, geographic locations, ethnic backgrounds, and economic sectors). For example, a proposed environmental policy that tightened health and safety standards for toxic chemical manufacture may produce less overall benefits than costs by reducing certain chemicals' effectiveness. However, if it redistributes economic opportunity between those who make and use chemical compounds and those who bear the real costs of their use, people will see it as fair, so it may receive political support. What this means is that economically inefficient proposals (those for which the costs exceed the benefits) can still receive considerable political support if people believe that the most deserving people secure the benefits and the least deserving ones pay the costs.

4.2 Scope of Impacts

Economic values of a proposed programme are defined by benefits and costs of individuals who would be affected by it. For example, if a proposed tightening of water quality standards under the EWFD produced X dollars of health benefits, then there are people for whom reduced doctor visits, reduced work days lost, and the like are worth X dollars to them. CBA is based on principles of democracy. Cost and benefit values that analysts assign to a programme's effects are those of affected people, not theoretical values held by politicians, agency analysts, newspaper editors, or radio talk show hosts. For programmes that are national or international in scope, use of the national accounting stance means that we should try to measure social opportunity costs and values for all inputs and outputs and for all people in the nation affected by the programme, whether those values are correctly or incorrectly priced in the marketplace. CBAs also can be performed for more limited accounting stances (e.g., regional and local; high and low income groups; current and future generations). Environmental proposals are more politically attractive if they can be designed to produce a favorable CBA outcome for a wide range of accounting stances.

4.3 Incremental Analysis

More efficient policies may be designed by using CBA to identify added benefits and added costs resulting from a range of various incremental changes rather than a single all-or-nothing proposal. An excellent example is provided in a study by Montgomery and Brown (1992) of the endangered spotted owl in the U.S. Pacific Northwest. These authors found that the typical all-or-nothing proposal presents policymakers with a single plan for species preservation and a single estimate of its economic costs. But an all-or-nothing proposal presents a problem: by choosing only two extreme policies, one that guarantees species survival and one that guarantees species extinction, policymakers are boxed in one of two corners: they have little opportunity to compare various incremental plans with each other by assessing costs of moving gradually toward more species protection. Analysis of incremental costs associated with increasing the probability of species' survival provides important information to policymakers who may wish to know the added cost of increasing the probability of species survival by a certain amount. The information on added cost can be compared with other possible uses of the same dollars and, hence, can be used to make more informed decisions.

4.3.1 Equimarginal Principle

The incremental (marginal) benefit represents the contribution of one more unit to economic efficiency. It is measured by the change in total benefits from one more policy unit (one higher ambition level). This marginal benefit concept is important in environmental and natural resource policy analysis for achieving efficiency.

Many government programs are intended to develop a natural resource or expand the scale or scope of an existing regulation. A larger scale occurs if an environmental regulation on air emissions is tightened. A larger scope could occur if it applies to a wider geographic region, more economic sectors. Where this occurs, economic efficiency requires that the development or expansion be undertaken until the marginal benefit of the expansion equals the marginal cost. Some proposals would reallocate a resource, such as scarce water among competing users or scarce taxpayer resources among competing human health improvements. For the realloc-ation decision, economic efficiency occurs when marginal benefits per unit of the resource are equal for all uses. This principle could be called the equimargi-nal principle for reallocation. A good example of a policy that reallocated scarce water among competing users is the Central Valley Project Improvement Act of 1992, that reallocated 800,000 acre feet of water from agriculture to improvement of fish and wildlife habitat. (See U.S. Fish and Wildlife Service, 2004).

4.3.2 With and Without Principle

The economic impact of a proposed programme is measured by the benefits and costs with the programme in place minus benefits and costs without the programme in place. Use of this principle assures that measured benefits (or costs) are solely due to the program or project, rather than changes that would have occurred anyway even without it. What this means is that benefits and costs should be measured with versus without a policy, not before and after. Accomplishing this aim requires defining a clear baseline policy to measure the incremental benefits and costs of a proposed policy. Defining that clear baseline helps avoid the double counting problem. For example, many environmental and natural resource policy analyses have counted as benefits those changes that would have occurred even without the policy.

4.3.3 Timing

In the face of growing demands for an environmental regulation (e.g., one that produced better health) or for a project's output (e.g., endangered species habitat), net present value of benefits may actually be increased by postponing a project or regulation even though it might be found to produce a positive net present value if carried out right away. The benefits of waiting will exceed the costs if net benefits are growing at a percentage rate larger than the discount rate.

4.3.4 Sequencing

Programme or policy elements are not always independent. Implementing one programme may influence the benefits or costs produced by a related programme. Introducing environmental regulations that improve human health (e.g., drinking water quality regulations) before introducing a programme whose output is mostly aesthetic (e.g., visibility in a scenic protected area) may produce greater economic efficiency than introducing the programmes in reverse order. If regulatory resources are scarce then where one programme's outputs influence other programmes' benefits or costs, it may be most efficient to consider many different possible time sequences for introducing the programmes.

4.3.5 Environmental Policy Framework

Figure 1 (p. 230) shows the steps required to conduct the kind of economic analysis required for the WFD. Beginning in the upper left, a base policy is defined as the one that would occur into a specified future under status quo conditions (no new policy). For a river basin, that policy produces a series of physical flows into the future, such as a base streamflow and a base level of pollution emissions into several water bodies. These flows give rise to conditions around the basin, including reservoir levels and pollution concentrations in water bodies. These conditions produce short term and long term effects, such as a base level of human health, drinking water quality, commercial fish harvests, and the like. Finally on the far right are economic values associated with these effects, including base costs of ill health, base costs of poor drinking water and the like. The bottom half of the diagram shows the same five steps going from a proposed policy at the left to its economic consequences at the far right. Basin scale water policy analysis such as required by the WFD would require a baseline series of five steps as well as the same five steps for each policy being considered.

5. Simple Hydroeconomic Model

One approach for implementing river basin scale analysis that accounts for hydrology, institutions, and economic at the basin scale, such as required by the WFD, is to build, verify, test, and run a hydroeconomic model. Models of various levels of complexity have been developed, but many of these models remain packed away on a hard disk, and are usually unavailable for open and transparent critique and evaluation. So this section describes a very simple model that can be formulated and solved with pencil and paper using ordinary algebra and calculus.

5.1 River Basin

Figure 2 (p. 232) shows an (expandable) prototype river basin containing the following basic elements at each of the major supply or demand locations (nodes).

- watershed inflows measured in water volume per unit time
- agricultural water use
- urban water use
- gauges to measure streamflows
- minimum delivery requirement to downstream location (e.g. an international agreement)

This simple basin diagram could be expanded to include things like minimum stream flow requirement for endangered species, water pollution emissions and their downstream tracking of concentrations, and various water chemical indicators for water quality. Several farming areas, cities, or other water use nodes could be added using this kind of diagram.

5.2 Objective

Following the spirit of the WFD, this model is based on the simple idea of maximising total basin-wide economic efficiency, i.e. total benefits minus total cost of water used. In the spirit of the WFD approach, the goal of this small model is to allocate water to the two use nodes, irrigation and urban uses, in such a way that total economic benefits are maximised consistent with available water supplies, while meeting downstream water volume delivery requirements. While water volume is not always the central issue throughout the EU countries, it can become an important policy issue in the more water stressed regions of southern and central parts of the EU. The mathematical appendix shows the parameters, variables, and equations used to specify and solve the model.

5.3 Water Uses and Economic Benefits

For this paper and pencil model, the only two uses requiring water diverted from the stream are irrigated agriculture and urban water supply. It is assumed that previous analyses have been conducted from which total and marginal economic benefit functions can be algebraically specified for each of these uses. Economic values for urban uses are often found by analysing urban demands as a function of water's price and price structure, size and structure of households, climate, nature of landscape, and income and population levels of the urban areas. One common method used to estimate economic values for agricultural irrigation is to develop and use representative farm budgets to identify costs and returns of various agricultural enterprises. For known prices of crop and livestock activities and for known costs of production, surface and/or groundwater supplies are varied from wet to dry conditions to see how farm income is likely to vary in the face of water supply changes.

Based on these or similar methods, water-related total and marginal benefit functions are derived and specified for the basin model. Based on those benefit functions, the model's goal could be to identify what combination of diversions at each use node maximised the basin's total economic benefits, subject to whatever relevant constraints limited those diversions. The appendix mathematical documentation shows linear equations in which total benefits are quadratic functions of water diverted and marginal benefit functions are linear functions of diversions. Naturally, this assumption need not be required for more complicated models.

5.4 Hydrologic Constraints

This model uses a simple hydrologic constraint defined by a set volume of water runoff available from the various watershed inflow points. These inflow points are typically defined as being natural or near-natural runoff points above all major water uses. One classic way to implement the hydrologic constraints is to use historical gauged inflows for whatever period of record is available. There are no reservoirs, reservoir evaporation, return flows, groundwater pumping, aquifer levels, or fate and transport of pollutants in this simple model, but many of those can be added if there is sufficient time and money for analysis. An example of a basin scale model with more extensive hydrologic detail is presented in Figure 3 (p.233) descriptively and is documented mathematically in Ward, Booker, and Michelsen (2006).

5.5 Institutional Constraints

One can imagine many kinds of institutional constraints, i.e. rules or conventions that define things like water rights, historical use patterns, international delivery obligations, instream flow requirements to support endangered species habitat, transboundary water sharing agreements, promises to limit pollution emissions, and the like. For the simple model presented, we specify a single downstream delivery requirement, a volume of water, which must be met in every period.

6. Results

Table 1 (p234) shows results of the simple linear model documented in the appendix, whose parameters are also summarized in Table 1. The model is solved using standard LaGrangian multiplier methods, with results of the constrained optimisation presented in the table. Shown is the constrained efficient solution for water diverted by use, gross benefits by use, gross costs by use, and the shadow price for water and the shadow cost for downstream delivery requirements.

Parameters are varied up or down by 10% for headwater flows, downstream delivery requirement, number of acres of irrigated land, number of households served by the urban water supply source, marginal cost of supply for both agriculture and urban use, and price elasticity of demand for irrigation and urban use, both of which are related to their respective economic benefits functions.
All results accord with economic theory and prior expectations. Use by agriculture increases with more acres served while quantity of water supplied to urban areas increases with population. Water supplied to either agriculture or cities falls with an increase in marginal cost of either user. Finally, the shadow price of water increases with increased water scarcity or with an increased downstream delivery requirement.

7. Conclusions

The CBA framework is a way to organise and catalogue hunches about impacts of environmental policy proposals. Its main strength as appraisal methods is that CBA tests for economic efficiency in resource allocation and for efficiency improvements in policies that would change that allocation. Whenever resources are scarce, methods that can help government improve economic efficiency are desirable because they help us get the most bang for our scarce buck. Numerous actual or proposed policy changes impose large costs on society, so decisionmakers often would like a sense of whether they produce similar benefits. For some environmental policies, costs may be found to be too high compared to expected benefits.

A CBA requires analysts, decisionmakers, or the public to identify all benefits and costs of a proposed policy. Such an identification may fail to occur if a CBA is not used. A CBA process produces a considerable amount of information about policy impacts, which can be presented to decisionmakers and to interest groups. Furthermore, by presenting this information in a structured way, CBA raises relevant questions and presents answers in a consistent way. When a policy is selected for which costs exceed benefits, the CBA is a source of information for efficiency losses incurred by that policy.

CBA is attractive because it accounts for social values systematically in environmental decisionmaking. The CBA process incorporates values of the public and not those of experts or leaders, so CBA is a form of economic democracy. CBA also has the advantage that it predicts impacts of policy proposals based on rational human behaviour. For example, if a proposed climate change policy would tax carbon, CBA could be used to predict benefits and costs of the proposal based on buyers' responses to increased prices of electric power and motor vehicle fuel. In addition CBA allows for environmental and natural resource impacts to be directly compared with financial impacts using a common denominator. Otherwise the value of the environment might inadvertently be counted as zero. So it is not just the transparency of the exercise but the fact that the environment is being valued somehow that supporters believe is important.

The greatest strength of CBA may be as an organising tool to force consistency and rigor into environmental policy debates. The process of quantifying and monetising forces discussion over assumptions and makes opposing sides confront the substance of each others' arguments, rather than just attacking the straw men common to many policy debates. Benefit-cost analysis also organises data that influence policy decisions and does so in a way that informs decisionmakers about the important elements of a problem and permits them to test the sensitivity of the decisions to changes in those elements. What this means is that the CBA framework for organising information is arguably more important to the policy process and outcome than any single net benefit number it produces.

Despite its strengths, CBA has several limitations. As described by Turner (2007) critics of CBA applied to environmental policy analysis point to its many measurement problems. Measurement issues are especially true of what may be the most common method for valuing the environment, the contingent valuation method (CVM). Many express considerable doubt about the idea of attaching monetary values to various environmental improvements. Supporters of cost benefit analysis defend it by saying it is better to value the environment out in the open, rather than hide implied values of the environment behind political agendas.

Another measurement problem with CBA is that measured benefits depend on the existing distribution of income. CBA is democratic because it treats all benefits gained and lost equally, regardless of who receives or pays them. However each person's “money vote” depends on his income. The willingness-to-pay for most environmental improvements increase as people’s incomes increase, so any given environmental programme produces smaller benefits when poorer people are the targeted beneficiaries. This shortcoming of CBA becomes more compelling when we see that measured benefits exceed costs for most programmes that would export pollution or other environmental insults from rich to poor countries. Yet another measurement problem occurs with the discount rate. Nobody to date has put forth a convincing argument on what the correct discount rate is, let alone shown how that rate varies with culture, income, and other human conditions important to environmental policy.

Many critics of CBA believe it is morally wrong to value in monetary terms programmes such as environmental programmes that save or prolong human lives through increased safety. When a decision maker sees money values placed on such a project, it may confuse or anger rather than enlighten.

The discount rate raises important ethical issues. CBA critics argue that any discount rate greater than zero is morally questionable because it assigns a lower value to unborn future generations than to current generations and therefore is unfair to the future. CBA also uses values of the current generation as the centre of the exercise, so discounting which leads to lower valuation for future generations is morally objectionable. While there is much to be debated over what the correct discount rate is, its supporters believe it should be larger than zero: as long as the current generation shows positive time preferences, it is difficult to mount a strong argument that no discounting should occur, especially when future generations are expected to be richer than people living today. A more productive debate would seem to be on how discounting methods should be implemented.

Another fairness issue deals with use of willingness-to-pay as a criterion for valuing programme outputs. Use of this criterion for valuing impacts is seen by some as unfair because it depends on project recipients’ ability to pay. For example an environmental regulation that improved safety in the workplace for low income workers may fail a CBA simply because people who get the benefits cannot afford to pay much for them. So an environmental or natural resource management plan may make the rich richer and the poor poorer and still perform well on a CBA.

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9. Appendix: Prototype Hydroeconomic Model

Parameters

$X_{h,t}^{0}$	(Headwater runoff, pd t, ac-ft)
X_{g3t}^{0}	(Delivery requirement at gauge 3, pd t, ac-ft)
B_{0a}, B_{1a}, B_{2a}	(Intercept, linear, and quadratic term ag economic benefits fn)
$B_{0u1}, B_{1u1}, B_{2u1}$	(Intercept, linear, and quadratic term urban economic benefits fn)
Y_a	(Cost per ac-ft ag supply, $ US)
Y_u	(Cost per ac-ft urban supply, $ US)
S_a	(Ag land served, acres)
S_a	(Urban customers served, households)

Variables

Hydrologic

$X_{h,t}$	(Headwater runoff, pd t)
$X_{da,t}$	(Total Ag use, pd t)
$X_{du,t}$	(Total Urban use, pd t)
$Y_{da,t}$	(Per acre ag use, pd t)
$Y_{du,t}$	(Per household urban use, pd t)
$X_{g1,t}$	(Gauge #1 Streamflow, pd t)
$X_{g2,t}$	(Gauge #2 Streamflow, pd t)
$X_{g3,t}$	(Gauge #3 Streamflow, pd t)

Economic

$Benefit_{u,t}$	(Per Household Urban Benefits, pd t)
$Benefit_{a,t}$	(Per Acre Ag Benefits, pd t)
$T\_Benefit_{a,t}$	(Total Ag Benefits, pd t)
$T\_Benefit_{u,t}$	(Total Urban Benefits, pd t)
$Cost_{a,t}$	(Per Acre Ag Costs, pd t)
$Cost_{u,t}$	(Per Household Urban Costs, pd t)
$T\_Cost_{a,t}$	(Total Ag Costs, pd t)
$T\_Cost_{u,t}$	(Total Urban Costs, pd t)
$PVNB$	(Discounted Net Benefits over uses and pds)

Equations

Hydrologic

$X_{h,t} = X_{h,t}^{0}$	(Headwater flows)
$X_{gl,t} = X_{h,t}$	(#1 gauge flows, pd t)
$X_{g2,t} = X_{g1,t} - X_{da,t}$	(#2 gauge flows, pd t)
$X_{g3,t} = X_{g2,t} - X_{du,t}$	(#3 gauge flows, pd t)
$Y_{da,t} = X_{da,t} / S_a$	(Ag use per acre, pd t)
$Y_{du,t} = X_{du,t} / S_u$	(Urban use per household, pd t)

Institutional

$X_{g3,t} > X_{g3t}^{0}$	(delivery requirement, pd t)

Economic Benefit

$Benefit_{a,t} = B_{0a} + B_{1a}Y_{da,t} + B_{2a}Y_{da,t}^2$ (Ag benefits per acre, pd t)

$Benefit_{u,t} = B_{0u1} + B_{1u1}Y_{du,t} + B_{2u}Y_{du,t}^2$ (Urban benefits per household, pd t)

$T\_Benefit_{a,t} = S_a\ Benefit_{a,t}$ (Total ag benefits, pd t)

$T\_Benefit_{u,t} = S_a\ Benefit_{u,t}$ (Total urban benefits, pd t)

Economic Cost

$Cost_{a,t} = Y_a\ Y_{da,t}$ (Ag water cost per acre, pd t)

$Cost_{u,t} = Y_u\ Y_{du,t}$ (Urban water cost per household, pd t)

$T\_Cost_{a,t} = S_a\ Cost_{a,t}$ (Total Ag water cost, pd t)

$T\_Cost_{u,t} = S_u\ Cost_{u,t}$ (Total Urban water cost, pd t)

Economic Discounted Net Benefits

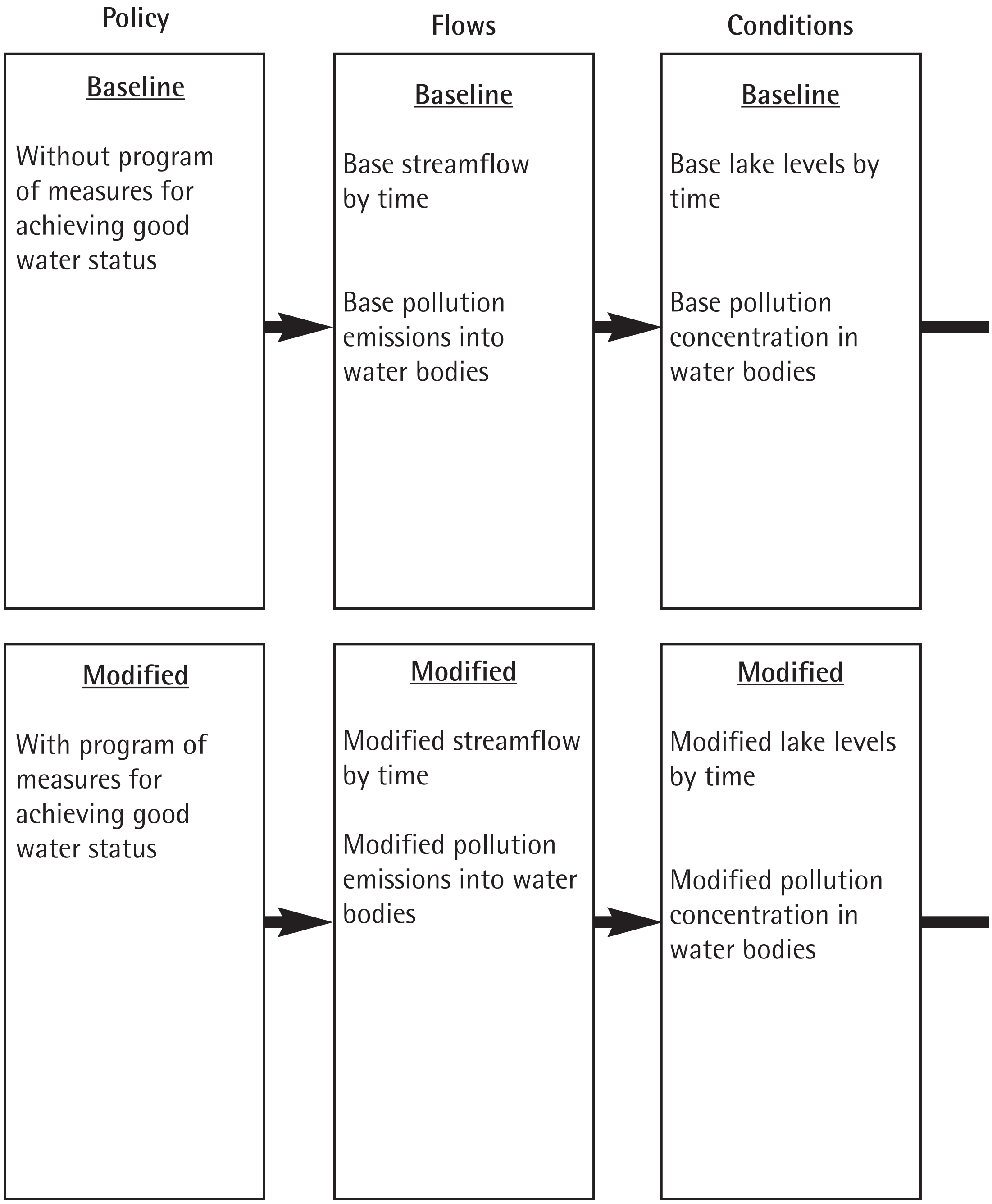
$$PVNB = \sum_t \frac{T_{Benefit_{a,t}} - T - Cost_{a,t}}{(l+r)^t} + \sum_t \frac{T_{Benefit_{u,t}} - T - Cost_{u,t}}{(l+r)^t}$$

Langragian Expression: Constrained Maximization

$$Max\ \phi = PVNB + \sum_t \lambda_i \left(\left(X_{h,t}{}^0 - X_{g3,t}{}^0 \right) - \left(X_{da,t} + X_{du,t} \right) \right)$$

Part Five: Tools for considering costs and benefits in support to policy decision

Figure 1: Environmental Policy Analysis Steps



Effects	Economic Values
Baseline	**Baseline**
Base human health	Base cost of ill health
Base ecosystem health	Base cost of ill ecosystems
Base drinking water quality	Base cost of poor water
Base commercial fish harvests	Base value of fish harvests
Base tourism activity	Base value of tourism activity
Base agricultural activity	Base value of farm output
Base flooding	Base cost of flooding
Base water shortages	Base cost of water shortages
Modified	**Modified**
Modified human health	Modified cost of ill health
Modified ecosystem health	Modified cost of ill ecosystems
Modified drinking water quality	Modified cost of poor water
Modified commercial fish harvests	Modified value of fish harvests
Modified tourism activity	Modified value of tourism activity
Modified agricultural activity	Modified value of farm output
Modified flooding	Modified cost of flooding
Modified water shortages	Modified cost of water shortages

Figure 2: Prototype River Basin

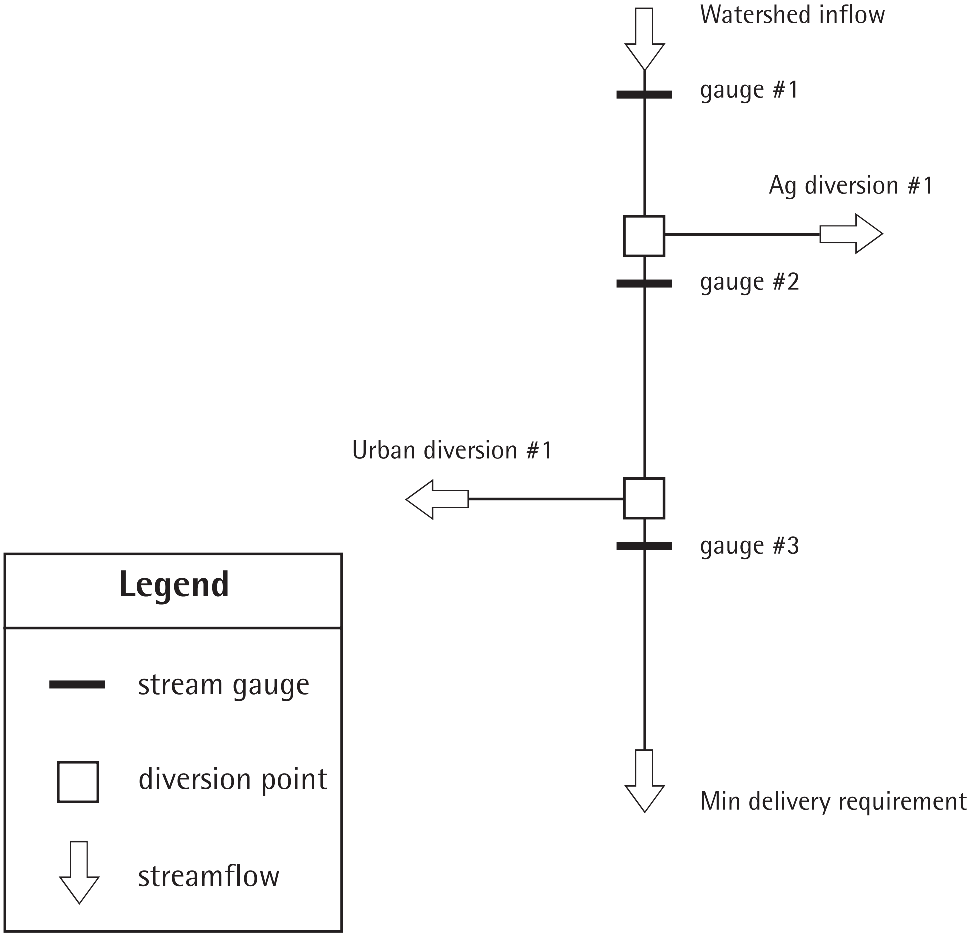


Figure 3: Hydroeconomic Model Structure

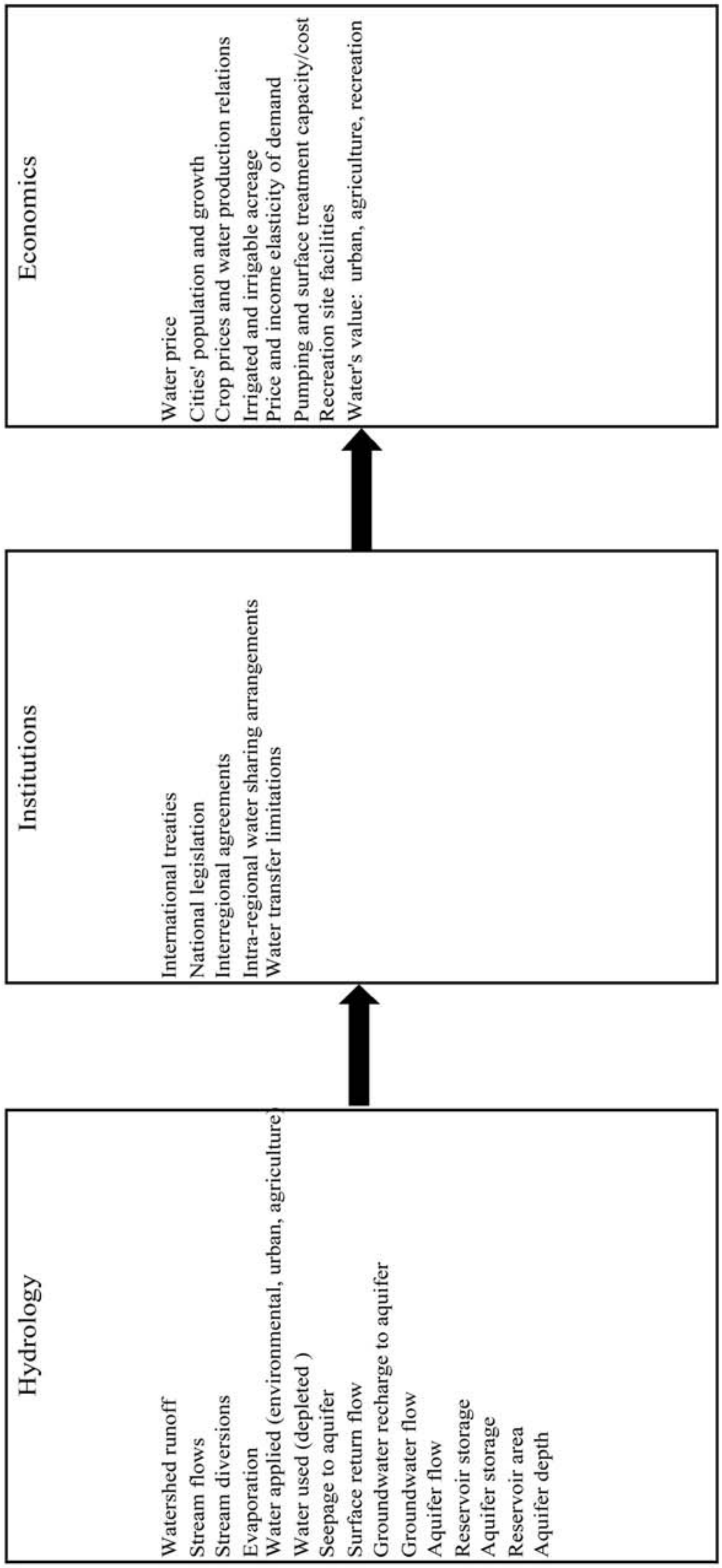


Table 1: Basin Scale Cost Benefit Analysis

		Water Diverted				Gross Benefit			
		Ag		Urban		Ag		Urban	
	Parameter Value	Per Acre (af/yr)	Total (ka-f/yr)	Per HH (af/yr)	Total (ka-f/yr)	Per Acre ($US)	Total ($1000 US)	Per HH ($US)	Total ($1000 US)
Hydrology Headwater flows (a-f/yr)									
Base	600	3.2	142	0.5	58	153	6864	3049	326198
+10%	660	4.5	202	0.5	58	181	8166	3049	326234
-10%	540	1.8	82	0.5	58	103	4617	3048	326160
Institutions									
Minimum outflow reqts (a-f/yr)									
Base	400	3.2	142	0.5	58	153	6864	3049	326198
+10%	440	2.3	102	0.5	58	122	5471	3048	326173
-10%	360	4.0	182	0.5	58	174	7837	3049	326222
Economics									
Ag Scale (1000 acres)									
Base	45	3.2	142	0.5	58	153	6864	3049	326198
+10%	49.5	2.9	142	0.5	58	144	7106	3049	326190
-10%	40.5	3.5	142	0.5	58	162	6568	3049	326207
Urban Scale (1000 households)									
Base	107	3.2	142	0.5	58	153	6864	3049	326198
+10%	117.7	3.0	136	0.5	64	149	6688	3049	358813
-10%	96.3	3.3	148	0.5	52	156	7030	3049	293581
Ag Marginal Cost ($US/a-f)									
Base	10	3.2	142	0.5	58	153	6864	3049	326198
+10%	11	3.2	142	0.5	58	153	6863	3049	326200
-10%	9	3.2	142	0.5	58	153	6864	3049	326195
Urban Marginal Cost ($US/a-f)									
Base	400	3.2	142	0.5	58	153	6864	3049	326198
+10%	440	3.2	142	0.5	58	153	6870	3048	326100
-10%	360	3.2	142	0.5	58	152	6857	3049	326286
Ag Demand Elasticity (unitless)									
Base	-0.79	3.2	142	0.5	58	153	6864	3049	326198
+10%	-0.88	3.2	142	0.5	58	163	7340	3049	326190
-10%	-0.72	3.2	142	0.5	58	144	6482	3049	326204
Urban Demand Elasticity (unitless)									
Base	-0.04	3.2	142	0.5	58	153	6864	3049	326198
+10%	-0.04	3.2	142	0.5	58	153	6871	2743	293475
-10%	-0.04	3.2	142	0.5	58	152	6857	3354	358911

Gross Cost				Net Benefit				Shadow Price	
Ag		Urban		Ag		Urban			
Per Acre ($US)	Total ($1000 US)	Per HH ($US)	Total ($1000 US)	Per Acre ($US)	Total ($1000 US)	Per HH ($US)	Total ($1000 US)	Supply ($US/a-f)	Basin Outflow ($US/a-f)
32	1421	217	23170	121	5443	2832	303027	20	-20
45	2020	217	23205	137	6146	2832	303028	4	-4
18	822	216	23135	84	3795	2832	303025	35	-35
32	1421	217	23170	121	5443	2832	303027	20	-20
23	1021	216	23147	99	4449	2832	303026	30	-30
40	1820	217	23194	134	6016	2832	303028	9	-9
32	1421	217	23170	121	5443	2832	303027	20	-20
29	1421	216	23163	115	5685	2832	303027	23	-23
35	1421	217	23180	127	5147	2832	303028	15	-15
32	1421	217	23170	121	5443	2832	303027	20	-20
30	1363	217	25484	118	5325	2832	333330	21	-21
33	1479	217	20856	123	5552	2832	272725	18	-18
32	1421	217	23170	121	5443	2832	303027	20	-20
35	1563	217	23173	118	5301	2832	303027	19	-19
28	1279	217	23168	124	5585	2832	303027	21	-21
32	1421	217	23170	121	5443	2832	303027	20	-20
32	1423	237	25390	121	5447	2810	300710	20	-20
32	1419	196	20933	121	5438	2854	305353	20	-20
32	1421	217	23170	121	5443	2832	303027	20	-20
32	1421	216	23163	132	5919	2832	303027	23	-23
32	1421	217	23176	112	5062	2832	303028	17	-17
32	1421	217	23170	121	5443	2832	303027	20	-20
32	1423	216	23067	121	5448	2527	270408	20	-20
32	1419	217	23255	121	5439	3137	335656	20	-20

Established in 1996, the Scottish Environment Protection Agency (SEPA) is Scotland's environmental regulator and adviser. Its duty is to protect air, land and water quality, which together form Scotland's environment and contribute to the Government's goal of sustainable development.

In September 2006 SEPA hosted a conference on the Costs and Benefits of supporting regulatory actions. The conference in Edinburgh saw economic experts from around the world presenting their work and approaches to these issues. The papers in this book are based on the presentation and discussions from that event.

ISBN 978-0-11-497342-1

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