Environmental Valuation in Developed Countries
Environmental Valuation in Developed Countries

Case Studies

Edited by

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DWP, Saffron Walden
David Pearce (1941–2005): a tribute

Those working in and around the field of environmental valuation, including the contributors to this volume, owe David Pearce a massive debt of gratitude. This academic field is now burgeoning and is arguably the single most active area of environmental economics. Obviously, this was not always so. In its formative years the subject needed its champions and in David we had a champion of the highest order. He combined numerous qualities that many of us would be content to have in isolation: intellectual agility and clarity, a legendary ability to communicate often complex academic ideas and an undiminished willingness to engage, persuade and not merely be content to preach to the converted. Whether arguing the case for environmental valuation at the ‘frontier’ or more latterly for its extension to the last bastions where economic thinking had previously left little mark, David always carried the conviction that good and broadly accepted environmental policy-making could not proceed without a better understanding of the value that the public placed on environmental improvements.

Sadly, we have now lost our champion. Of course, there are others but as is the way with true innovators each is unique because, to coin a phrase, to make each one ‘they have to break the mould’. David’s sudden passing was a huge loss for all of those in the environmental research and policy community. He is greatly missed already. But our particular condolences and thoughts must go to David’s family who have lost a husband and father far too early. This volume was not intended as a tribute to David Pearce, completed for all intents and purposes only a matter of weeks before his death. We are sure, however, that we echo the sentiments of all of its contributors in acknowledging our gratitude to David for his generosity and encouragement over the years and for what he achieved on our behalves. We hope that the chapters within provide at least a small testament to this achievement.

Giles Atkinson
Brett Day
Dominic Moran
Susana Mourato
October 2005
1. Introduction: valuing environments in the rich world

David Pearce

BACKGROUND

This is the second of two volumes of case studies aimed at showing how environmental economists place value on environmental assets and on the flows of goods and services generated by those assets. The first volume looked at valuation exercises in the developing world (see Pearce et al. 2002). The current volume focuses on the developed, or rich world. This is where the first exercises in economic valuation began. The aim has been to assemble studies that show the very broad areas of application of economic valuation, from amenity and pollution through to water and health risks, from forestry to green urban space. As with the first volume, the chapters come from authors who, at one time or the other, formed teams of environmental economists at University College London. But other UK and continental European economists have been invited to contribute in order to capture the wide array of applications.

All the chapters in this volume take it as read that economic valuation makes sense. Since that is not a view shared by many non-economists, it is as well to devote a short space to explaining what economic valuation is and why it is a rational procedure.

ECONOMIC VALUE AND ITS CRITICS

The notion of economic value has a very precise meaning. It relates to human well-being. Changes in well-being are revealed by people’s preferences. Basically, if someone prefers A to B, their well-being from A is said to be higher than their well-being from B. In common with utilitarian philosophy, the notions of costs and benefits and asset values relate to the sum of these individual economic values. It is assumed therefore that it makes sense to add up what individuals prefer to get some aggregate, which defines the
Environmental valuation in developed countries

Social value of the asset. Also, an individual's well-being is assumed to be measurable through the notion of willingness to pay. Essentially, if someone prefers A to B they will be willing to pay more for A than for B. The extra (marginal) willingness to pay is what is measured by a demand curve. The problem in environmental economics is that many environmental goods and services have no markets, or at least no obvious market. Hence demand curves have to be 'uncovered' by looking for surrogate markets or by creating a market. The former procedure is 'revealed preference' and is typified by something like hedonic property pricing whereby the value of, say, noise nuisance or air pollution, is measured by the price depreciation suffered by properties exposed to these damaging influences. The latter procedure is 'stated preference'. A hypothetical market is established through the use of questionnaires and respondents are either directly asked for their willingness to pay, or it is inferred from their stated choices.

One reason that some critics dislike this notion of economic value is that it looks altogether too much like buying and selling goods in the supermarket. And, of course, it is. Indeed, the whole aim is to put environmental goods and services into the same framework as buying and selling marketed goods and services. The objection of the critic is that this somehow 'debases' the environment to being just another good, like a car journey or a hamburger. Surely, the environment is somehow 'above' this comparison? It is possible to have some sympathy with this view. If people do not like (prefer) the environment, then economic analysis suggests it should be dispensed with. But there are problems with any view that tries to make the environment 'different' or 'special' and outside the scope of economic value.

First, saying that economic value should not count is the same as saying people's preferences should not count. That is undemocratic and, indeed, the main defence of economic valuation is that it embodies a 'democratic presumption'. What the critics are really saying is that some elite (usually themselves!) possesses the right values and only they should decide what happens to environmental assets. Any environmental economist with any practical experience of government will be familiar with that view, which is usually expressed by unelected members of special interest organizations.

Second, economic values derive from trade-offs, from the fact that we cannot have everything and hence have to choose. To say economic values are inapplicable to environmental assets is therefore formally equivalent to saying we can never trade off those assets against other goods and services. One observation is that such trade-offs are made every day and no society anywhere in the world operates as if such trade-offs cannot be made. It is easier to be sympathetic to the view that this trading off has gone too far, that, in some sense or other, we now have 'too little' environment and that no further degradation should be permitted. But it is not very clear what
this constraint on trade-offs means. It cannot mean never cutting down
one more tree, nor can it mean holding the physical stock of environmental
assets constant since we have no metric to measure that stock in physical
terms, and hence no way of knowing if the constraint is being observed
or not. The constraint could be expressed in monetary terms, that is, the
economic value of the stock should be held constant. But this is where
we came in and such a rule offends the value critic because we have now
ended up measuring the environment in economic value terms. The critic
of economic valuation therefore has many more problems to address before
his or her criticisms become robust.

The critics also argue that economic value, if it is permissible at all, is
only one value among many. There are also aesthetic values, spiritual values,
social values (sense of identity and space), historical values, symbolic values
and intrinsic values (and probably others, too). The problem with this line
of criticism is that it is not very clear how these values are to be defined.
More importantly, how do they enter the policy context? If we always have
to make choices (trade-offs) how does, say, spiritual value trade off against
economic value? Does it always trump economic value? If there is a trade-
off, how does one trade off a non-measurable notion of value (spiritual
value) against a measurable one (economic value)? Part of the problem
is that the critics occupy a world in which these choices do not have to be
made by them. As the guardians of their own notions of value, their own
interpretation of that value is always ‘right’. Moreover, unlike those arguing
for economic value, the critics need to have no recourse to what the ordinary
citizen wants. The democratic presumption issue arises again.

Some of the critics argue that the environment is rather like human life:
it is ‘priceless’. Several recent critiques of economics adopt this view – for a
recent well written, but wholly misconceived, critique along these lines see
Ackerman and Heinzerling (2004). In the ‘priceless’ argument something
like environment cannot be brought into the same commensurate units as
willingness to pay. But there are several problems with this view.

First, if one cannot compare costs and benefits in commensurate terms,
it is hard to know how much to spend on the activities that generate them.
If benefits are in some non-monetary units and costs are in money terms,
then the only techniques available for rational decision-making are 1. leaving
it to political judgement and 2. some sort of cost-effectiveness analysis. But
the reason policy analysts develop procedures like cost–benefit analysis is
to ‘check’ on political judgement. Assuming that political judgement is
always ‘right’, without the need for any such checks, makes policy analysis
redundant. No doubt many politicians would prefer to occupy such a
world, but most people would acknowledge that we need the checks and
balances that come from policy analysis. The political judgement view is
also ‘Panglossian’ – everything is for the best in the best of all worlds, and political judgements cannot be bettered.

As to the second apparent escape from cost–benefit analysis, cost-effectiveness analysis is certainly valuable. Indeed, economists would assume that all decisions subsume cost-effectiveness analysis, otherwise one would be wasting resources to achieve a given end. But cost-effectiveness cannot tell us how much conservation to do. The reason is simple. Costs are not in the same units as benefits. Hence one can never say that benefits ‘exceed’ costs, we can never know with cost-effectiveness analysis whether anything is worth doing at all.

Second, no nation functions by assuming that assets, of whatever kind, are ‘priceless’. Nations go to war, which means that they judge the ends (benefits) to be worth the loss of life that war entails. No nation spends resources to bring risks to life down to zero, so costs are being traded against life risks. No nation conserves each and every environmental asset. Logically that means that conservation is traded against cost. The impossibility of not trading environmental, cultural and life assets against money was pointed out 40 years ago by Thomas (1963). All decisions have costs and hence all decisions to incur that cost imply that benefits exceed costs. All decisions not to incur the costs imply that costs exceed benefits. Economic valuation is always implicit or explicit, it cannot fail to happen at all.

A further critique of the notion of economic value argues that the values relevant to environmental assets are those of the citizen, not the individual. The difference is that when voting as a citizen the individual is supposed to think about the social good in general and not about individual self-interest. Citizen preferences are not therefore revealed in the market place, nor even in a hypothetical market place such as that embodied in stated preference techniques. Rather one has to look to the political process for evidence on these preferences. This view is perhaps best associated with the writings of Mark Sagoff (for example, Sagoff, 2004). Here again there are problems. The problems with relying on the political process to reveal citizen preferences were noted above. The additional difficulty in this case is that political decisions are invariably the result of ‘political welfare functions’, which reflect partly what people want, but more what special interest groups and lobbies want. One cannot assume that citizens’ preferences determine political outcomes at the level of detail that would be required.

Finally, consider the notion of ‘intrinsic value’ as a value residing in environmental assets: the assets are ‘worth’ something in themselves and independently of the human being who may value them. If the ‘right’ way of thinking about environmental assets is in terms of their intrinsic value, then economic value is at best subservient and at worst irrelevant. Philosophers have debated notions of intrinsic value for centuries. Whether
Introduction

It exists or not as a meaningful concept, that is, whether there can be ‘value’ independently of a ‘valuer’, is an issue we do not dwell on here. But it is relevant to ask what the policy implications of such notions are, assuming they are meaningful. As we saw, the attraction of the economist’s notion of value is that it is commensurate with costs and costs are unavoidable in the decisions to conserve, enhance, modify and manage environmental assets. Non-monetary notions of value can still fall within the domain of cost-effectiveness analysis and this is an essential requirement for rational policy-making.

The practical problem with ‘intrinsic value’ is that it cannot be brought within the domain of a valuer and hence cannot be measured at all. The implication is that intrinsic value fits neither the economic value concept (deliberately so) nor the cost-effectiveness paradigm. It is also unclear exactly what role it could play in decision-making. Decision-making is about priorities and unless one can say that X has ‘more’ or ‘less’ intrinsic value than B, no one can say whether X should have priority over B, or vice versa. The practical issue, therefore, is not whether intrinsic value exists, but what use can be made of it if it does exist. If it exists and cannot be measured, then perhaps the only way in which intrinsic value can influence decisions is through the decision-making discourse. That is, it would be up to those who wish to argue for more resource allocation to particular assets that they have ‘high’ intrinsic value, without there being any obligation to measure this notion of value. That takes us back to the problem of political welfare functions and special interests. Intrinsic values are therefore extremely problematic.

The ‘standard’ economic approach would reject notions of value that are different from economic value. However, it is important to understand why. The standard approach does not argue that other values are irrelevant. What it would argue is that all of these other values are determinants of economic value, rather than values in themselves. In the language of economic valuation, they are motives for value.

A further argument for treating environmental assets differently from other goods and services is that they may be irreplaceable in the sense that, once lost, the original cannot be recreated. The degree of irreplaceability is perhaps more severe for built heritage than it is for environmental assets. Apart from the extinction of species and, say, primary forests, many environmental assets can be recreated with barely discernible differences from the original. The question is whether irreplaceability makes some assets special in the sense of making them a challenge for economic valuation (or even incapable of economic valuation).

But even if an asset is irreplaceable, it does not mean that it has no substitutes. Substitutes may not be perfect, but economic analysis makes
no assumption about perfect substitutability between money and conventional goods. Rather, perfect substitution is itself a special case. There are two senses to substitution here. First, one environmental asset may be a partial substitute for another environmental asset. That is, there may be ‘within-asset’ substitution. Second, any given environmental asset may be substitutable, partly at least, by non-environmental assets. In the sustainable development literature, the second notion of substitutability is known as ‘weak sustainability’. The former, in which the loss of one environmental asset could be compensated by another environmental asset is known as a ‘strong sustainability constraint’. However, since one cannot recreate irreplaceable assets, compensation via the creation of assets is not possible. Strong sustainability would therefore require that each and every irrereplaceable environmental asset be conserved. Provided a strong sustainability stance is adopted, the relevance of irreplaceability for valuation is that far stronger pressure exists to conserve everything there is that could be called irreplaceable than in the weak sustainability context. That pressure translates into higher implied values for the marginal environmental asset. In contrast, the weak sustainability paradigm is consistent with the cost–benefit approach in which all kinds of assets are substitutable to some degree. On this paradigm, the values elicited by economic valuation techniques are the relevant ones, that is, the various willingnesses to pay to conserve the assets.

There is a potential implication of irreplaceability for the way in which decisions about environmental assets are made. Suppose some environmental assets are at risk because conservation budgets are limited. Suppose further that the assets at risk are not fully documented, in the sense that further information about them could be generated by more research etc. There are opportunities for ‘learning’, that is, for generating more information. Then, economic analysis tells us that the combination of uncertainty, the opportunity for learning and irreplaceability, should dictate a more cautious approach to the conservation decision. The value of the information generated by waiting before allowing the asset to decay or be replaced is known as ‘quasi option value’. What the quasi option value literature is telling us is that one cannot simply compare costs and benefits of conservation in the conventional manner (deriving a discounted present value based on expectations about future economic values) if the context is one of irreplaceability and uncertainty, and if there is the chance of learning. If everything is known about the asset, then quasi option value does not effectively arise – being cautious about losing the asset will not generate new information.

Overall, then, while the notion of economic value is not without its own problems, it does have a long history, and it would be surprising if those who developed it had not thought about the criticisms made against it. The
argument here is that those criticisms are far more problematic than those
who advance them might think.

COSTS AND BENEFITS

The case studies in this volume illustrate the wide array of environmental
issues that have been addressed using the notion of economic value. By and
large they focus on measuring the benefits of an environmental
improvement, or the damage done by environmental degradation. From
a policy standpoint, one of the weaknesses of the past few decades of
economic valuation studies is that they have focused heavily on benefit or
damage measurement, rather than on comparisons of these measures with
the costs of environmental improvement, or the benefits of the damaging
activity. In short, full-blown cost–benefit analysis is less common than
benefit measurement.

One reason for being concerned about this asymmetry is that it fails to
provide the environmental economist with the full array of weapons in the
policy context. All too often, what the Minister of Finance wants to be
persuaded of is that investing in the environment achieves as high a social
rate of return as investment in other forms of national wealth. In other
words, most policy contexts are defined by a general suspicion that the
environment is not a national priority and that other things matter more.
When it comes to public infrastructure, roads and airports and the like, the
presumption is that the environmental impacts are less important than the
benefits in terms of the national economy. Those concerned with raising
the profile of the environment therefore need to remember that it is both
costs and benefits that matter.

VALUE TRANSFER

Nonetheless, it is easy to understand why environmental economists have
focused on benefit and damage estimation. First, the ‘value critics’ discussed
in the previous section place the onus on the economist to show that
economic value exists, is relevant to policy-making, and can be measured.
Many valuation studies are therefore about showing that economic value
is measurable. Second, as the essays in this volume show, the science of
valuation has developed rapidly in the past few decades. Many case studies
are designed to test the valuation theory and to advance it. The policy
implications have been a lower order priority. Where case studies have been
designed to answer policy questions they tend to be a mix of rough-and-
ready analysis and sophisticated analysis. Many of the rough-and-ready studies adopt ‘benefit transfer’ as the mode of analysis, that is, assets, goods and services are valued in terms of values already demonstrated for similar assets, goods and services in previous studies.

This process of benefit (or value) transfer is potentially dangerous. It is only recently that environmental economists have begun to test for the size of the error that is likely to be involved in ‘borrowing’ numbers. The search for large ‘value databases’ – in which studies and their results are stored and accessed – adds to the risks, since it encourages the idea that values can be borrowed and transferred with a reasonable degree of accuracy, without anyone being sure that this is the case. Some of the chapters in this volume (notably those by Clinch (Irish forestry, Chapter 2), Crabtree (UK forestry, Chapter 3), Söderholm and Sundqvist (energy externalities, Chapter 8), Horton and Fisher (water in the UK, Chapter 12), and Newcombe, Özdemiroğlu and Atkinson (agricultural accounts, Chapter 18)) use or comment on transferred values. Chapter 13, by Willis and Scarpa, casts doubt on the large-scale value transfer exercise discussed by Horton and Fisher (Chapter 12). Nonetheless, it is hard to see how value transfer can be improved until original studies are designed with value transfer in mind. At the moment, the ‘jury is out’ on the likely size of error in value transfer. Much more work is needed.

Value transfer can be advanced through ‘meta-analysis’, an analysis of the various studies that have analysed economic value of a specific environmental asset, forests or wetlands, say. Meta-analysis usually involves statistical techniques that attempt to explain why the various studies come up with different values. It is not only the characteristics of the assets in the original studies that matter, the characteristics of the studies themselves also matter. Thus, it may matter if the studies have been carried out using particular valuation methods. For example, in a stated preference study it may matter if a telephone survey has been used compared with a face-to-face questionnaire. It may even matter who has conducted the studies. The study by Hall, Moran and Allcroft (Chapter 6) illustrates this kind of meta-analysis in the context of perceived risks to consumers from genetically modified food. A form of meta-analysis, but this time with less focus on statistical assessment, is also provided by Brouwer in his overview of valuation studies in the Netherlands relevant to the European Water Framework Directive (Chapter 7).

THE PRIMARY STUDIES

The remaining chapters show how ‘primary’ valuation is conducted, that is, they show how the various techniques that have been developed are
applied to specific environmental assets. Groom, Hepburn, Koundouri and Pearce (Chapter 5) look at an interesting development in environmental economics, namely the idea that the discount rate is not a constant with respect to time, but actually declines with time. Time-varying discount rates overcome those objections to economic valuation based on its apparent discrimination against future generations. This ‘tyranny of discounting’ arises simply because people prefer the present to the future, with the result that investment in long-term environmental protection – as with global warming control – appears to have a low economic value.

The other chapters on primary valuation procedures adopt variants of stated preference approaches to valuation, usually contingent valuation but occasionally choice modelling. The former asks respondents directly for their willingness to pay, the latter infers willingness to pay from choices that individuals make across composite goods that are described by ‘bundles’ of characteristics. Christie and his colleagues tackle the complex issue of valuing biodiversity (Chapter 4) where biodiversity is construed as the diversity of life forms, rather than just a stock of biological resources. Ek (Chapter 9) analyses Swedish preferences for windpower, finding, perhaps unsurprisingly, that there are strong preferences for it as long as it is offshore!

Atkinson, Day and Mourato (Chapter 10) focus on another neglected issue – visual disamenity from overhead transmission lines. It is an interesting feature of private sector decision-making that corporations show little interest in finding out what the public actually wants by way of avoiding the disamenity that corporate activity generates. This stands in contrast to their espousal of market research when selling their products. Yet market research and stated preference valuation have very close similarities, the main distinction being that market research focuses on private goods, so that buyers can choose to buy or not once the good is available, and stated preference focuses on public goods that, once available, are available to all without further choice. The transmission line study shows that it is perfectly possible to find out people’s preferences for ‘undergrounding’ (burying the cables) and, given that ‘overgrounding’ is much cheaper, what people’s preferences are for various styles of transmission tower.

Bullock (Chapter 11) adopts a choice modelling approach to valuing urban green space in Dublin. Willis and Scarpa (Chapter 12) summarize what is probably one of the most sophisticated choice modelling studies in Europe. This was conducted for a major private water company in England (Yorkshire Water), which showed the imagination to commission the work in order to find out how its customers valued the various services provided by the company. As noted above, this visionary approach contrasts starkly with how most private corporations approach their wider social and environmental impacts. Chapter 14 by Mourato and colleagues describes another English
water company initiative in finding public preferences for reducing the risks of sewage overflows into the River Thames in London, a legacy of long-past sewer infrastructure decisions. They also use a choice modelling approach, as do Söderholm and Sundqvist in their own evaluation of energy externalities (Chapter 8). Bateman and colleagues (Chapter 15) use contingent valuation (with a twist!) to value the benefits of reduced eutrophication in water bodies in Eastern England. Chapter 16, by Mourato and colleagues, raises some questions about the suggested strengthening of the European Union Directive on bathing waters. It is interesting that, despite the clear requirement for the European Commission to carry out cost–benefit studies of each Directive, comparatively few are actually conducted or funded by the Commission itself. These essays all show the power of stated preference approaches to elicit economic values that, a few decades ago, would have been regarded as being incapable of economic valuation.

Day and colleagues’ chapter on noise nuisance shows the power of another widely used valuation technique, hedonic property price analysis (Chapter 17). The basic idea is simple, property prices reflect the many characteristics of houses, so that the overall price is itself an amalgam of a set of individual (hedonic) prices for these characteristics. Day’s study employs powerful econometric techniques to explore these values. In theory, hedonic prices are not quite the same as the economic value of the change in the asset in question (in this case, peace and quiet), but many empirical studies have adopted hedonic prices as the best guess at the relevant economic values.

The last two chapters explore the use of valuation in wider accounting exercises. It is well known that conventional measures of economic output, such as GNP, are potentially poor indicators of well-being. This explains the drive for better accounting systems that try to measure the externalities involved. In their study, Newcombe and colleagues (Chapter 18) report on an exercise that provides the first set of reasonably rigorous economic accounts for UK agriculture. In their study, Brouwer and colleagues (Chapter 19) illustrate a further application of economic accounting, this time focusing on the information needed to understand economic and environmental interactions at the river basin level.

Overall, this volume brings together leading experts in the field of economic valuation, showing just how sophisticated techniques have become and how powerful their application can be.

REFERENCES

Introduction

PART I

Natural Resources: Forests, Biodiversity, Water and Energy
INTRODUCTION

The literature on the non-market costs and benefits of forestry varies in its quality. Pearce (1999) states that the results are a ‘mixture of legitimate and illegitimate valuation procedures’. A large number of studies of the total economic value of forestry have been undertaken in developing countries. According to Pearce, most of these have been concerned with the destruction of indigenous forests and the associated loss of non-timber benefits rather than by any concern regarding the supply of timber. Thus, a large body of research has focused upon tropical forests and their provision of such external benefits as bush meat, firewood, nuts and berries, and medicinal plants. Despite much of the work on forestry valuation being concerned with tropical forests, there is an extensive literature focused on woodland in developed countries. Summaries of this research can be found in Wibe (1995) and Prins et al. (1990) and a collection of papers on the non-market benefits of forestry is contained in Roper and Park (1999). Most of the research has been carried out in North America although there is a considerable body of literature from the Nordic countries and the United Kingdom including, for the latter, the comprehensive study by Willis et al. (2003).

In general, while there is a substantial literature on the economic value of forestry, much of the research is related to deforestation and the associated loss of non-timber benefits. Temperate commercial afforestation brings with it quite different externalities and the consideration of external costs is most important. This chapter presents the results of an ex ante cost–benefit analysis of the Irish government’s Forestry Plan. In doing so, a range of externalities is examined using, inter alia, contingent valuation and production function approaches. The study provides insights into the methodological difficulties and solutions for assessing the social efficiency of large-scale environmental projects.
IRISH FORESTRY AND FOREST POLICY

Forestry as a land use in Ireland involves a paradox: the rate of tree growth is among the fastest in Europe\(^2\) and neighbouring Great Britain is a significant importer of wood, yet forests comprise less than 10 per cent of the state's land area, the lowest proportion in the European Union. The reasons for this are as follows: once the indigenous forests were cleared towards the end of the seventeenth century, those who controlled the cleared land rarely felt that the financial and other returns from tree planting justified the cost. Only for a relatively short time towards the end of the eighteenth century and the first half of the nineteenth century were the conditions met for making the long-term investments required to expand the forest estate, and then only on the part of a small number of landowners who developed some estate. However, most of these forest estates did not survive the transfer of the estates from landlord to tenant. Indeed, forestry was sometimes identified as an avocation of the 'landlord class', a group regarded by many as symbolic of a suppressive past rather than as models to be emulated!

In the twentieth century, the new Irish state took the lead in restoring forests as a land use. There was a steady increase in forest area, concentrated mainly on relatively high elevation and poor nutrient sites, which were not perceived as being of value for farming, with much of the expansion concentrated in the western counties. In the years immediately following membership of the European Community in 1973, the boom in farming resulted in an escalation in the real price of land, and forestry could not compete with highly-subsidized farming alternatives. However, the 1990s saw very striking changes in forestry in Ireland and they were driven by the country's membership of the EU. Reform of the Common Agricultural Policy made forestry an obvious target for structural fund support under two Forestry Operational Programmes. Generous planting grants, tax-free timber revenues and annual payments for farmers combined with growth rates of timber among the fastest in Europe to make investments in forestry seem most attractive. Today's grants are worth between €2000 and €5000 per hectare with a tax-free annual premium of between €200 and €500 per hectare for 15 to 20 years.

Current Irish Forest Policy at a macro level is set out in the Strategic Plan for the Development of the Forestry Sector in Ireland (Department of Agriculture, Food and Forestry, 1996). This envisages a doubling of the Irish forest estate by 2030 at a cost of over €3.9 billion. This involves increasing the forest area from 570,000 hectares (in 1996, the year of publication of the Strategic Plan) to 1.29 million hectares by 2030. The €3.9 billion would finance the grant and premium schemes necessary to encourage such a large increase in afforestation, 75 per cent of the funds...
coming from the EU. It is clear that this plan would involve a major land-use change, which would have significant effects on the environment. It is generally thought that planting trees is good for the environment in that they provide recreational areas, sequester and store carbon, improve the landscape scenery and provide wildlife habitats. However, in Ireland, where planting consists of predominantly commercial forests, 80 per cent of which are non-indigenous conifers, there is considerable apprehension regarding the external costs of afforestation. Of particular concern are the potential negative effects of forestry on landscape, water and wildlife. The objective of the research presented here is to assess the social efficiency of the Strategic Plan for forestry.

**METHODOLOGY**

An *ex ante* assessment of the social efficiency of the Irish government’s Forestry Plan was carried out using cost–benefit analysis (CBA). The Kaldor–Hicks (potential Pareto optimality) efficiency criterion was used. The total economic value of the plan was measured using both market and non-market valuation methods, the former involving shadow pricing where appropriate, and the latter comprising production function approaches and contingent valuation. The components of the economic value of the forestry plan and the method of valuation are presented in Table 2.1.

*Table 2.1 Components of the total economic value of the Forestry Plan and valuation method*

<table>
<thead>
<tr>
<th>Function</th>
<th>Category</th>
<th>Valuation Method</th>
</tr>
</thead>
<tbody>
<tr>
<td>Timber production</td>
<td>Benefit</td>
<td>Shadow pricing</td>
</tr>
<tr>
<td>Inputs (land, labour, capital)</td>
<td>Cost</td>
<td>Shadow pricing</td>
</tr>
<tr>
<td>Carbon sequestration</td>
<td>Benefit</td>
<td>Damage costs/Opportunity cost</td>
</tr>
<tr>
<td>Water impacts</td>
<td>Cost</td>
<td>Replacement cost</td>
</tr>
<tr>
<td>Recreation</td>
<td>Benefit or cost</td>
<td>Contingent valuation</td>
</tr>
<tr>
<td>Habitat</td>
<td>Benefit or cost</td>
<td>Contingent valuation</td>
</tr>
<tr>
<td>Landscape impacts</td>
<td>Benefit or cost</td>
<td>Contingent valuation</td>
</tr>
<tr>
<td>Community stability</td>
<td>Benefit or cost</td>
<td>Not valued</td>
</tr>
<tr>
<td>Archaeological impact</td>
<td>Cost</td>
<td>Not valued</td>
</tr>
</tbody>
</table>

Discount rates were used to enable the comparison of costs and benefits that arise in different time periods. The choice of discount rate
Natural resources

is of particular importance in forestry as it is assumed that the land is reforested after felling and so the rotation is considered to be perpetual. There is a huge literature on discounting and space does not permit an in-depth discussion of appropriate measures. Suffice to say that there is no agreement on which particular number is appropriate. In practice, the discount rate used to evaluate public projects is chosen via the political system. The Irish government recommends a rate of 5 per cent be used to reflect the opportunity cost of capital (Department of Finance, 1994). For the purposes of this research, a range of discount rates varying between 0 per cent and 10 per cent were used with 5 per cent being considered the key rate for the purposes of drawing implications for government policy.

COMPONENTS OF THE TOTAL ECONOMIC VALUE OF THE FORESTRY PLAN

A brief outline of the procedure used for calculating each of the components of the economic value of the Forestry Plan and the results obtained are presented below. A comprehensive sensitivity analysis was carried out for each component but space does not permit a full outline of the analyses and results. The results are rounded to the nearest million euro (€).

Inputs

The inputs required for afforestation consist of land, labour and other inputs such as plants, fertilizer and pesticides, and materials for roads.

Land

Under the EU Common Agricultural Policy, farming in Ireland has been heavily subsidized. Similarly, afforestation has been in receipt of major subsidies under the EU Operational Programmes for forestry. Since land is in relatively fixed supply, this has resulted in the distortion of the price of land such that it no longer provides an adequate measure of the opportunity cost of its use in the absence of subsidies. The opportunity cost of using land for forestry can be measured by the agricultural output forgone valued at world prices. However, due to the existence of subsidies and trade restrictions, when calculating the value of the agricultural output lost, shadow pricing must be used.

Fitzgerald and Johnson (1996) estimate the social value of agricultural output on an average forestry site to be approximately €100 per hectare. This figure can then be capitalized at the test discount rates to give estimates of the opportunity cost of keeping land under forestry. At the 5 per cent
discount rate, the shadow price of land is €2007 per hectare and the cost of the land required for the targets of the Forestry Plan to be reached is calculated to be €701 million in present value terms.

**Labour**

If labour markets clear, the shadow price of labour will equal the market wage. Any increase in employment in one sector of the economy will merely reduce the availability of labour to another sector. However, in a country with a high unemployment rate, it could be argued that an increase in the demand for labour in one sector of the economy will not necessarily displace a job in another sector, that is, there may be employment additionality whereby if the new job is filled by a person who was previously unemployed, no cost in terms of output forgone is imposed upon society.\(^3\) In this case, the shadow price of labour would equal zero. It has been the practice of Irish government cost–benefit analyses to assume a shadow price of labour of zero. However, this is against the international practice of setting the shadow price of labour at most a fraction below the market wage. For example, in Canada the shadow wage is 95 per cent of the market wage and in the UK the shadow wage is set equal to the market wage (Honohan, 1998).

Honohan argues that it is hard to justify a shadow wage far below the going market wage in Ireland. He bases his conclusions on the recognition that migration responds so readily to job creation such that when the economy is in recession, there is net emigration as people travel abroad (mostly to the UK) in search of employment. The present boom in the economy has resulted in net immigration resulting in a rate of growth of the labour force much more rapid than the rate of decline of unemployment. At present it would seem that it is not justified to use a shadow wage of less than the market wage given that the economy is close to full employment and it is unlikely that the emergence of forests on the landscape will reduce the stock of long-term unemployed. Therefore, the shadow price of labour is considered to be equal to the market price of labour (net of taxes and social insurance). At a 5 per cent discount rate, the present value of the labour required under the Forestry Plan amounts to €173 million.

**Other inputs**

Input costs other than labour and land include costs associated with cultivation and drainage, plants and planting, fertilization and weeding, fencing, brashing, roads and roads repair, and marking and measuring. The total cost of these inputs amounts to €215 million at a 5 per cent discount rate. The total cost of inputs into the Forestry Plan therefore amounts to €1089 million – Table 2.2.
Natural resources

Table 2.2  Cost of inputs into the Forestry Plan (€ million)

<table>
<thead>
<tr>
<th>Discount Rate (%)</th>
<th>Land</th>
<th>Labour</th>
<th>Other Inputs</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>21822</td>
<td>3943</td>
<td>3416</td>
<td>29181</td>
</tr>
<tr>
<td>3</td>
<td>1515</td>
<td>328</td>
<td>414</td>
<td>2257</td>
</tr>
<tr>
<td>5</td>
<td>701</td>
<td>173</td>
<td>215</td>
<td>1089</td>
</tr>
<tr>
<td>8</td>
<td>318</td>
<td>94</td>
<td>173</td>
<td>585</td>
</tr>
<tr>
<td>10</td>
<td>212</td>
<td>70</td>
<td>123</td>
<td>405</td>
</tr>
</tbody>
</table>

Timber

The principal marketed output of the Forestry Plan will be timber. The plan involves afforesting 725,000 hectares of trees. Prior to calculating the value of the timber, appropriate assumptions were made regarding timber prices based on a survey of timber markets and prices. The state forestry company, Coillte, produces 97 per cent of total roundwood in the state and 87 per cent of total production on the island of Ireland (Competition Authority, 1998). The importation of untreated timber is prohibited by law for pest control reasons and, unlike many state forestry companies, Coillte has a commercial brief. For these reasons, the potential exists for it to act as a monopoly supplier and thereby inflate prices. However, there is also market power on the demand-side, where ten wood processing firms consume approximately 80 per cent of the roundwood (Competition Authority, 1998). It is unclear at this point whether, and in what direction, prices are being distorted by this monopoly power in the wood market. However, given that price distortion may occur, Irish prices are an unreliable indicator of the social value of Irish timber. Border prices are normally considered to be the most appropriate (Pearce, 1998) but, because of the import restrictions, they are not suitable in this case. Therefore, British prices were used, as they provide a good indicator of the likely value of Irish timber in a competitive market due to Britain's reliance upon imports and the similarity of British and Irish wood.

A perpetual rotation was assumed when calculating the timber benefits of the Forestry Plan. The present value of timber output over one rotation at the five test discount rate was first calculated. A spreadsheet model was then created to simulate a perpetual rotation and to calculate its net present value. The timber value under varying price and discount rate assumptions is presented in Table 2.3.

If 5 per cent is taken to be the social rate of discount and average historical prices are expected to prevail, it is reasonable to conclude that the (discounted) timber value of the Forestry Plan is €1126 million. As expected, the rate of discount used has a major effect on the result due to
the bulk of the timber benefits arising at the end of the forest rotation. A 10 per cent rate gives a value of just €107 million compared with a value of over €3.8 billion at a 3 per cent discount rate. The expansion of the EU from 12 to 15 member states and the consequent shrinking of the timber deficit rendered the economic security value (the strategic value of growing timber in Ireland rather than importing it from abroad) close to zero.

Table 2.3  Timber value of the Forestry Plan under various price assumptions (€ million)

<table>
<thead>
<tr>
<th>Discount Rate (%)</th>
<th>Low Prices</th>
<th>Average Prices</th>
<th>High Prices</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>102 686</td>
<td>114 096</td>
<td>125 505</td>
</tr>
<tr>
<td>3</td>
<td>3 504</td>
<td>3 893</td>
<td>4 282</td>
</tr>
<tr>
<td>5</td>
<td>1 013</td>
<td>1 126</td>
<td>1 240</td>
</tr>
<tr>
<td>8</td>
<td>234</td>
<td>259</td>
<td>284</td>
</tr>
<tr>
<td>10</td>
<td>97</td>
<td>107</td>
<td>117</td>
</tr>
</tbody>
</table>

**Carbon Sequestration**

Trees absorb (sequester) carbon dioxide (CO₂) and store it in the wood. The carbon is released when the wood, or the products that have been made from the wood, decay. In this way forests delay the release of CO₂ to the atmosphere. There are three approaches to placing a monetary value on this reduction in CO₂ emissions. The damage-avoided approach values a tonne of carbon sequestered by the cost of the damage that would have been done by global warming in the event that it had been emitted. The offset approach measures the value of not emitting a tonne of carbon using one method, by the next cheapest alternative method; the tonne of carbon sequestered is valued by the cost of substituting a non-carbon fuel for a fossil fuel at the margin. The avoided-cost-of-compliance approach measures the tonne of sequestered carbon by the avoided cost of compliance with a global or regional CO₂ emissions’ reduction agreement (a form of ‘offset’).

There are three possible policy scenarios. Under the scenario of a fixed emissions quota, the measurement is, essentially, the same as the offset approach. Under the scenario of a carbon tax, the value of a tonne of carbon not emitted is measured by the tax that would have been paid had the carbon been emitted. With an emissions trading system, the value of not emitting carbon is equal to the cost of the permit(s) that would have been purchased if the carbon had been emitted, or, by the income received from the selling of those permits that are not required. This study uses the damage-avoided approach with a mid-range value for the marginal benefit of a tonne of carbon sequestered of €19 (from Fankhauser, 1995).
The simplest approach to converting a carbon price into a forest value is to approximate the cycle of carbon sequestration by a simple curve, rising asymptotically to a mean level of carbon fixed. However, this implies that the uptake rate is fastest at the beginning of the rotation. Price (1997) points out that, while this formulation gives a reasonable estimate of the carbon sequestered, on average, by a forest and its products, with discounting it is likely to be inaccurate, for example, using a 6 per cent discount rate, the overestimation of benefits at the beginning of the rotation leads to a three-fold overvaluation of discounted benefits. This study used the CARBMOD model developed by Colin Price and Rob Willis of the University of Wales at Bangor. This is a computer simulation that models the temporal sequestration of carbon for different yield classes of a range of species of tree based on Forestry Commission yield tables and carbon sequestration figures. For each species type, the model includes assumptions regarding the end uses of the timber to give a complete profile of the carbon dynamics of afforestation.

A further consideration is that peat soils emit methane and CO₂ when disturbed but the net sequestration effect of forests on peat soils in Ireland is still unclear. A conservative estimate of total sequestration benefits can be calculated by assuming peat oxidation rates are fast, such that there are no net sequestration benefits on peat soils. Results for a variety of discount rates, assumptions regarding the annual growth in the marginal benefit of sequestering carbon (ΔP) and the rate of oxidation of peat upon which forests are planted are provided in Table 2.4. Using a 5 per cent discount rate, a zero growth rate in the marginal benefit of sequestration, and fast peat oxidation gives a total value of carbon sequestration benefits of the Forestry Plan of €58 million.

**Table 2.4 Carbon sequestration value of the Forestry Plan (€ million)**

<table>
<thead>
<tr>
<th>Discount Rate (%)</th>
<th>0</th>
<th>3</th>
<th>5</th>
<th>8</th>
<th>10</th>
</tr>
</thead>
<tbody>
<tr>
<td>ΔP (%)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>–3</td>
<td>145</td>
<td>46</td>
<td>23</td>
<td>10</td>
<td>6</td>
</tr>
<tr>
<td>0</td>
<td>375</td>
<td>145</td>
<td>66</td>
<td>23</td>
<td>13</td>
</tr>
<tr>
<td>3</td>
<td>–</td>
<td>375</td>
<td>220</td>
<td>66</td>
<td>32</td>
</tr>
<tr>
<td>ΔP (%)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>–3</td>
<td>128</td>
<td>41</td>
<td>20</td>
<td>9</td>
<td>6</td>
</tr>
<tr>
<td>0</td>
<td>331</td>
<td>128</td>
<td>58</td>
<td>20</td>
<td>11</td>
</tr>
<tr>
<td>3</td>
<td>–</td>
<td>331</td>
<td>193</td>
<td>58</td>
<td>28</td>
</tr>
</tbody>
</table>
Cost–benefit analysis of Irish forest policy

Water

Plantation forestry can have a number of effects on the water supply. Acidification may occur in poorly buffered areas as a result of ‘scavenging’, whereby trees ‘trap’ pollutants and release them into the local water supply. In areas prone to acidification, this can increase the acid content of rivers and lakes, which may result in damage to fisheries. Pollution from fertilizers and/or pesticides can also have damaging effects on the water supply. Finally, forests tend to reduce stream flow.

The effects of forestry on water are some of the most difficult externalities to value as they tend to be very localized. There have been relatively few attempts to value the impact of forestry on water quality and quantity. Price (1997) quotes a study by Collet (1970), which established a technique for evaluating runoff loss and the effects of sedimentation on reservoir life in the UK based on the cost of replacement. It is thought that HM Treasury’s (1972) study, which suggested a cost of £5 per acre afforested (1972 prices) in North Wales, was based upon Collet’s method. Barrow et al. (1986) surmise from studies in Wales and Scotland that afforestation in upland sites is unlikely to be justifiable given the effect of water loss on hydroelectric power. Stretton (1984) suggested that cultivation of a 100-hectare plantation in South Wales imposed extra water treatment costs of £400 000 due to increased sediment while Milner and Varallo (1990) estimate that forestry-induced acidification of waters had a potential cost to Welsh Fisheries of £25 million (Price, 1997). Whitman (1991) estimates the cost of replacement of the water restricted by forests in East England to be £0.5 million per year. In addition, he estimates the discounted cost of forgoing the use of fertilizers in afforestation to be between £50 and £80 per hectare. However, he suggests that the cost of environmental damage is much lower than the cost of abatement and suggests an average figure of £20 per hectare.

In terms of stream-flow reduction, in general, Ireland has a very generous supply of water as a result of a low population density and high rainfall levels. However, the lowest rainfall levels and the highest population levels occur on the east coast, specifically in the Greater Dublin Area. Thus, while increased afforestation in other parts of Ireland is unlikely to impose a significant cost in terms of reduced water supply, the Greater Dublin Area is an exception. Using Whitman’s (1991) method, it is possible to calculate the cost of reduced water supply from the existing forest estate. Most of the water supply comes from sources in Wicklow, Kildare and Dublin. These counties combined have an average forest cover of 8.33 per cent such that the non-afforested area is 91.67 per cent. Total water supply in the Greater Dublin Area is 442 mega litres per day. The actual water supply will equal
the potential supply in the non-afforested area plus the potential supply in forested area reduced by the percentage rate by which evapotranspiration is above normal.

Table 2.5 gives the potential water supply and the probable loss due to afforestation for five possible rates by which evapotranspiration is above normal. Taking a study on interception loss due to forestry by Johnson (1990), which suggests the loss is 28 per cent, this indicates that the probable loss in the Greater Dublin Area is approximately 11 mega litres per day. If we assume that the forest estate of the three counties from which water is supplied doubles in line with the Forestry Plan, this would suggest a further reduction in the water supply of 11 mega litres per day when the forest matures. The cheapest way of replacing this water would be to repair leaks at a cost of €7 million (M.C. O’Sullivan Ltd, 1996). At a 5 per cent discount rate, the cost amounts to approximately €3 million.

Table 2.5  Potential water supply in the Greater Dublin area and the probable loss due to the Forestry Plan

<table>
<thead>
<tr>
<th>Evapotranspiration Above Normal (%)</th>
<th>Potential Water Supply (mega litres per day)</th>
<th>Probable Loss due to Afforestation (mega litres per day)</th>
</tr>
</thead>
<tbody>
<tr>
<td>10</td>
<td>446</td>
<td>4</td>
</tr>
<tr>
<td>20</td>
<td>450</td>
<td>8</td>
</tr>
<tr>
<td>30</td>
<td>453</td>
<td>11</td>
</tr>
<tr>
<td>40</td>
<td>457</td>
<td>15</td>
</tr>
<tr>
<td>50</td>
<td>461</td>
<td>19</td>
</tr>
</tbody>
</table>

Due to the site-specific nature of the effects of forestry on water it is extremely difficult to predict the likely costs of future afforestation. This is particularly the case in relation to water quality. The magnitude of the cost will depend in a large part on the effectiveness of the Irish Forest Service guidelines with regard to fisheries. In relation to acidification, Allott and Brennan (1993) suggest that the guidelines will be unable to prevent acidification in poorly buffered catchments exposed to an atmosphere charged with acidifying acids (pollutant or otherwise) nor will they protect the soil from organic acids resulting from the drainage and afforestation of peats. However, the Forestry Plan restricts grant aid to areas of yield class 14 and above. In addition, it is expected that conventional afforestation will not be permitted in EU-designated Natural Heritage Areas (NHAs). There is a high correlation between areas designated by the Environmental Protection Agency as sensitive to acidification, areas of below yield class 14 and the
NHAs. It seems reasonable to assume, therefore, that the cost imposed by
acidification from forests planted under the Forestry Plan will be minimal.
The extent of eutrophication from fertilizers and pollution from biocides
will depend upon the extent to which forestry contractors comply with the
guidelines and the extent of consultation with the fisheries boards.

It is most difficult to value the likely cost of pollution from an increase
in the forest estate. The actual cost can depend very much on particular
events, for example, there may be only one or two pollution incidents in a
year involving forestry but the costs would be very high if they happened
to occur, for example, in areas of great value to angling. The magnitude
of the cost will also depend upon the former land use. Given that most
of the land that is likely to be planted is currently being used for grazing,
it is predicted that there will be a net increase in the use of fertilizers,
particularly at establishment phase. While it is not possible to predict the
cost of pollution very accurately, Whiteman's (1991) figure of an average
net (of the cost imposed by the former land use) discounted cost of £20
per hectare gives an indication of the likely order of magnitude of this
cost. At a 5 per cent discount rate, total water pollution costs (assuming an
acidification cost of zero) amount to €10 million. The total costs imposed by
the effect of the Forestry Plan on the water supply at the five test discount
rates are given in Table 2.6. At a 5 per cent discount rate, the total cost
amounts to €13 million.

Table 2.6 Cost of water supply effects of the Forestry Plan

<table>
<thead>
<tr>
<th>Discount Rate (%)</th>
<th>Present Value (€ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>121</td>
</tr>
<tr>
<td>3</td>
<td>19</td>
</tr>
<tr>
<td>5</td>
<td>13</td>
</tr>
<tr>
<td>8</td>
<td>8</td>
</tr>
<tr>
<td>10</td>
<td>6</td>
</tr>
</tbody>
</table>

Recreation, Biodiversity and Landscape

Forests often are of value for recreation even if they are not specifically
designed for such a purpose. In addition, afforestation alters the character
of the landscape and has implications for biodiversity. The visual impact of
afforestation is probably the most controversial external effect of afforestation
in Ireland. This is primarily the result of the predominance of coniferous
forestry and the non-native Sitka spruce species in particular (this species
is favoured because of its rapid growth in Irish damp weather conditions).
In addition, afforestation has the potential to destroy valuable habitats if planting takes place in ‘sensitive’ areas. However, if planting is kept out of NHAs and sensitive areas as designated by local authorities, the threat to biodiversity is low, such that one rather limited diversity will be replaced by another limited diversity. The valuation of biodiversity is extremely difficult when existence values are thought to be significant. This is because there is often a lack of knowledge of the precise effects a proposed development will have on an ecosystem and there is also the difficulty of describing these effects in a contingent valuation questionnaire. Nevertheless, contingent valuation was considered to be the only appropriate way to calculate the landscape, wildlife and recreational benefits of the trees to be planted under the Forestry Plan.

A feature of many private goods is that only a subset of all consumers buy them even at a zero price and there is no reason why this should not also be the case for public goods (Kriström, 1997). If one is indifferent about consuming a good and not consuming it, whether it is marketed or otherwise, one’s willingness to pay for the good will be zero. Moreover, depending on one’s tastes etc., the consumption of a good may increase or decrease one’s utility, for example, some individuals enjoy a trip on a rollercoaster while others would find such a trip thoroughly unpleasant. The same can be said for public goods, for example, the reintroduction of wolves in an area would increase the utility of those who like wolves but decrease the utility of farmers whose sheep fall prey to the wolves (all else being equal). However, while an individual can choose not to consume a private good they cannot choose to avoid consuming a public bad.

It had been ascertained in a survey of public attitudes to afforestation that the views of the Irish public differ regarding the environmental impact of forestry in Ireland. When valuing an increase in the quality of a public good, the appropriate measure is Hicksian compensating variation, that is, the respondent’s willingness to pay (WTP) for the improvement is elicited. However, if the public good also exhibits features of a public bad, an appropriate consumer’s surplus measure must be chosen that can measure the loss of welfare as a result of an increase in its provision. The most appropriate measure would be willingness to accept compensation (WTA). However, since the study of Hammack and Brown (1974), which showed that WTA amounts were over four times greater than WTP amounts for the same amenity, there has been some concern about the practical difficulties of eliciting willingness to accept in contingent valuation surveys. Given these difficulties, the question arises as to whether WTP to avoid a public bad is an appropriate proxy for WTA.

From the work of Hanemann (1991), we can say that WTP will be a valid proxy for WTA when the public good/bad is not unique and irreplaceable.
and when WTP is unlikely to be a large proportion of income. This is likely the case in relation to the Forestry Plan. The agricultural land that would be replaced is not unique and the damage is not irreversible such that the elasticity of substitution is unlikely to be very low. WTP is not likely to be a large proportion of income. Therefore, a questionnaire was designed that ‘filtered’ respondents into two separate contingent markets based on their responses to questions regarding their preferences for forestry. Those who ‘liked’ afforestation were asked to ‘bid’ for a specific increase in the forest estate (the Forestry Plan) while those who ‘disliked’ forestry were asked to ‘bid’ to preserve the present land use and thereby avoid forestry (an environmental protection scheme).

A dichotomous choice elicitation format was used and the questionnaire had the usual checks as recommended by the best practice guidelines of Arrow et al. (1993) and others. The survey was mixed mode (personal interview and telephone) and was carried out in two parts to allow for consistency checks. A 78 per cent response rate was achieved resulting in a total effective sample of 2895 households. The results show that mean willingness to pay to avoid forests is significantly higher than mean willingness to pay for forests, suggesting that, as a result of an expansion in the forest estate, those who dislike forestry will, on average, endure a greater welfare loss than the average increase in welfare experienced by those who like forestry. Nevertheless, a majority were willing to pay for more forestry. At a 5 per cent discount rate, the net present value to the Irish population of the landscape, wildlife and recreational benefits of the Forestry Plan amounts to €164 million.

Other Considerations

Community stability

A potential benefit of an expansion in forestry is what is known as ‘community stability’ or ‘community integrity’. Community stability relates to the value that society puts on the conservation of rural communities (Pearce, 1998). The contribution of forestry to community stability is measured by the increase in the welfare of society that results from the conservation of rural communities as a direct consequence of an expansion in the forest estate. One aspect of community stability is captured by the willingness to pay on the part of individuals, in terms of reduced salaries, to return from a job in a foreign location to their home town. Theoretically, this could be measured using contingent valuation by eliciting the maximum salary reduction that individuals would be willing to accept in order to take a forestry job in their home town. Those who, it is adjudged, would be forced to migrate in the absence of employment created by forestry could be asked...
for the minimum salary increase they would be willing to accept in order to move away from their home town. However, it would also be necessary to measure the extra utility that relatives and friends gain by the presence of these individuals in their hometown. It is likely that the assessment of these values using contingent valuation would prove rather difficult. To complete the valuation of community stability it would be necessary to measure the increase in the welfare of the rest of society from the contribution that forestry makes to conserving rural communities. This would also prove to be difficult in practice.

If the Forestry Plan does not create more rural jobs than it displaces, the community stability value will be zero. If it displaces more than it creates it will have a negative effect. While there have been studies on the contribution of the US Forest Service to the stability of rural communities (Boyd and Hyde, 1989), there have been no such studies in Ireland and, therefore, the community stability value of the Forestry Plan has not been included in this CBA.

Archaeology
Ireland is fortunate to be endowed with a rich archaeological resource that is of value in terms of its direct use for tourism, both domestic and overseas, and for education and research. In addition it is likely to have both option and existence value. However, none of these values has been assessed to date. The difficulty with most archaeological sites is that their recognition requires specialized archaeological techniques and skills. Thus, two potential problems arise. First, an individual may recognize a feature on their land to be of archaeological importance but may destroy it to avoid interference with his or her desired activity (an individual is not compensated for the external benefit of preservation) and second, the individual may not recognize it as of archaeological value and destroy it by accident. Therefore, a change in land use as a result of increased afforestation might result in damage to the archaeological heritage. However, given the gaps in knowledge regarding the location of archaeological artefacts and the absence of any valuation of such artefacts, it is not possible to estimate the likely cost of increased afforestation in Ireland in terms of its impact on archaeology.

Cost of public funds
Due recognition must be made of the fact that the provision of subsidies to forestry will create distortions in the economy. These funds have to be raised through taxation or, if money is allocated to this project, there is an opportunity cost since funds for other projects must be paid from taxation. When distortions are created through revenue raising, the cost of raising such funds will be higher than the size of the funds raised. In 1997 the
marginal cost of public funds was estimated to be 1.5 (Honohan, 1996) such that the marginal excess burden equals 0.5, that is, the total cost of raising €1 via the tax system equals €1.50. Multiplying the subsidy by the marginal excess burden gives the excess burden of the forestry subsidies. This equals €808 million at a 5 per cent discount rate. However, this assumes that, were the 75 per cent funding from the EU not to go to forestry, the funds would be available for an alternative use in Ireland.

OVERALL RESULTS

Bringing together the various components of the total economic value of the Irish government’s Forestry Plan shows that it passes a cost–benefit test if the discount rate is 4 per cent or below. At rates of 5 per cent and above, net social benefit is negative (Table 2.7). If the government’s test rate of 5 per cent is considered appropriate, the results would suggest that the Forestry Plan should not proceed as it is not socially efficient. Timber is the dominant benefit at €1,126 million with net external benefits (amounting to €209 million) being equivalent to just 19 per cent of the timber value.

Table 2.7 Costs and benefits of the Forestry Plan (€ million)

<table>
<thead>
<tr>
<th>Discount Rate (%)</th>
<th>0</th>
<th>3</th>
<th>5</th>
<th>8</th>
<th>10</th>
</tr>
</thead>
<tbody>
<tr>
<td>Timber</td>
<td>114096</td>
<td>3893</td>
<td>1126</td>
<td>259</td>
<td>107</td>
</tr>
<tr>
<td>Land</td>
<td>-21822</td>
<td>-1515</td>
<td>-701</td>
<td>-318</td>
<td>-212</td>
</tr>
<tr>
<td>Labour</td>
<td>-3943</td>
<td>-328</td>
<td>-173</td>
<td>-94</td>
<td>-70</td>
</tr>
<tr>
<td>Other inputs</td>
<td>-3416</td>
<td>-414</td>
<td>-215</td>
<td>-173</td>
<td>-123</td>
</tr>
<tr>
<td>Carbon</td>
<td>331</td>
<td>128</td>
<td>58</td>
<td>20</td>
<td>11</td>
</tr>
<tr>
<td>Water</td>
<td>-121</td>
<td>-19</td>
<td>-13</td>
<td>-8</td>
<td>-6</td>
</tr>
<tr>
<td>Landscape, wildlife, recreation</td>
<td>213</td>
<td>182</td>
<td>164</td>
<td>142</td>
<td>131</td>
</tr>
<tr>
<td>Excess burden</td>
<td>-1979</td>
<td>-1119</td>
<td>-808</td>
<td>-532</td>
<td>-420</td>
</tr>
<tr>
<td>Net social benefit</td>
<td>83359</td>
<td>808</td>
<td>-560</td>
<td>-702</td>
<td>-583</td>
</tr>
</tbody>
</table>

AGGREGATION ISSUES

Randall (1991) has pointed out the risks in independent valuation summation (IVS) particularly when using different valuation methods. However, it is important to use the most appropriate valuation method. Therefore, there is a trade-off between the possibility of inaccuracy from double counting
and the possibility of inaccuracy from using an inappropriate valuation method. Bergland (1998) recommends that simpler methods be used where possible and that revealed preference methods are widely accepted valuation methods, which provide valid, and often reliable, estimates of use values. However, in this study, it was necessary to use a stated preference method in order to assess the value of future planting in relation to, for example, landscape impacts. Therefore, if just one valuation method were to be used, it would have to be a stated preference method. Describing all the possible costs and benefits of the Forestry Plan in a contingent valuation questionnaire would be hard enough but, even if this could be achieved, it seems unreasonable to expect a respondent to place a value on, for example, the reduction in streamflow that would result in the Greater Dublin area.

In this study, it was considered that the implications of double counting were likely to be less of a concern than the inaccuracies that would result from using inappropriate valuation methods. Survey research into the preferences of the Irish public regarding forestry has shown that there is little awareness of the benefits of forestry in terms of carbon sequestration and virtually no awareness of the impact of forestry on water amongst the general public (Clinch et al., 2000). There was no mention of these impacts in the contingent valuation questionnaire and the preference surveys suggest that the public was unlikely to have these impacts in mind when answering. Therefore, it is hoped that double counting has been minimized. In any case, if double counting has occurred, it, most likely, has resulted in an overestimation of the benefits of the Forestry Plan. Therefore, the overall result of the CBA should be adjusted down and it would still return a negative net social benefit so the policy implications remain the same.

GOVERNMENT SUBSIDIES

If the Forestry Plan is implemented, there will be a massive transfer of public funds to afforestation in order to fund the forest subsidy schemes, namely €3.9 billion over 35 years. The CBA suggests that, even before subsidies are considered, the Strategic Plan for the Forestry Sector in Ireland should not be implemented as it will not improve the welfare of society as it returns a negative net social benefit. However, if, for illustrative purposes, a rate of return of 4 per cent is considered to be acceptable, then the Forestry Plan would be socially efficient and consideration would have to be given to whether and to what extent the plan should be subsidized from public funds. Conventionally, since transfers are omitted from cost–benefit analyses, the results of such studies do not give any indication of
the appropriate size of the subsidy. Further discussion of this topic can be found in Clinch (2002).

ASSESSING THE ECONOMIC VALUE OF MARGINAL CHANGES IN THE STRATEGIC PLAN

Environmental valuation, in many cases, is limited to providing point estimates of the value of externalities of a project or policy. It is often difficult to assess the overall impact on net present value of a change in project or policy specification. For example, the study presented above merely tells us whether the Irish government’s Strategic Plan for forestry passes a cost–benefit test; it does not tell us the consequences of modifying the plan itself. As discussed previously, one of the most controversial aspects of Irish forest policy is the planting of non-native coniferous trees. Using a contingent valuation methodology as part of a face-to-face survey of 1202 Irish adults, Clinch et al. (2000) examine the environmental value of increasing the percentage of broadleaf planting from 20 per cent to 50 per cent of the total in the Strategic Plan. The results suggest that the environmental value of the plan would increase by €65 million. However, one should not infer that this change in species mix would necessarily increase the NSB of the plan. Given the dominance of the ‘market values’ of commercial afforestation shown above, reducing the percentage of fast-growing coniferous trees would likely reduce the NSB of the plan under all but the lowest discount rates.

DISCUSSION

The research summarized in this chapter has endeavoured to build on existing research on forest valuation by providing a template for undertaking comprehensive cost–benefit analyses of large-scale temperate plantation forestry programmes. Given the comprehensive range of externalities evaluated, this study provides some insights into the nature of the economic value of temperate plantation forestry and its assessment. The results of the study show that the private benefits of commercial forests dominate. Net external benefits, while significant, are only equivalent to 19 per cent of the timber value. This can be explained by the type of forestry under consideration. The primary purpose of plantation forestry is timber production. Therefore, the external benefits are likely to be lower than estimates from old forests that are likely to have greater landscape and
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amenity value. This study devoted considerable effort to considering external costs as well as benefits. Temperate plantation forestry can produce negative externalities such as water restriction and negative landscape impacts. These reduce the overall net external benefit figure to below what might be expected given the figures from studies of non-commercial or old forests. This highlights the fact that figures from valuation studies of such forests may not be applicable to plantation forestry and this should be taken into account when using benefits transfer. It is also interesting to note that this study suggests that EU subvention of afforestation in Ireland constitutes an intervention failure.

The limitations of the study are typical of CBAs of large-scale projects. Ideally, a number of projects would have been evaluated to see which project would give the greatest return, whereas the results of this CBA merely answer a binary question, that is, whether or not the project provides a net benefit to society. Therefore, it does not tell us whether the money would be better spent elsewhere and so, in practice, one often relies on government test discount rates. In addition, despite a large range of externalities being evaluated, a number of impacts remain unvalued. The physical impact of plantation forestry on community stability is difficult to assess; it is even harder to value. An interesting area for research is the assessment of the willingness to pay of individuals to avoid emigrating from their home town. However, to capture the full value of community stability it is necessary to find ways of evaluating the benefits to others from that individual staying at home. This will prove more difficult. The inability of this study to value the impact of increased afforestation on the archaeological heritage points to the problem of information gaps and uncertainty. Many environmental impacts that we value now were ignored in the past because of lack of knowledge. There are likely to be benefits and costs of which we are, as yet, unaware.

Ex ante assessments make it necessary to use techniques such as contingent valuation in order to ask people what benefit they think a proposed project will bring. However, the respondents must work with what limited information can be provided to them in a questionnaire and there may be considerable uncertainty about the actual impact a project will have. In addition, if respondents do not have much experience of the good in question (in this case, Ireland is the least forested country in the EU), their ex post valuation of that good may be quite different from their ex ante bid. Some Irish foresters suggest that people will grow to like plantation forests, that is, they merely need to experience them. A related and equally important issue is that preferences may change with time. This contingent valuation study showed that older people were less favourably disposed to forestry, which could mean either that, by the time the forests develop, more
of the population will be favourably disposed to forestry, or, by that time, the formerly young people, having aged, will have changed their minds!

A further difficulty in trying to assess the total economic value of a project that has multiple inputs and outputs is that it usually requires a number of valuation methods. Thus, it is necessary to engage in independent valuation summation (IVS). As pointed out earlier, there is a trade-off between the risk of double counting from IVS and the risk of inaccurate results from using inappropriate valuation techniques.

CONCLUSION

In conclusion, this chapter has presented techniques for valuing a range of externalities, both positive and negative, from plantation forestry programmes. However, the study shows that problems still remain in assessing some costs and benefits and there is considerable uncertainty regarding how costs and benefits change over time. Large-scale environmental projects that necessitate a range of externalities to be valued will generally require more than one valuation technique and therefore run the risk of double counting. CBAs such as this merely say whether or not the project being considered is socially beneficial, that is, it is a binary choice and the difficulty of calculating the marginal benefits and costs of projects makes calculation of the optimal subsidy difficult. This highlights the need for more research on the use of environmental valuation in setting targets for environmental policy.

NOTES

1. This chapter is a modified and extended version of a paper that first appeared in Forest Policy and Economics, 1 (3–4), 2000.
2. Average growth rates for Sitka spruce (Picea sitchensis) are nearly four times the European average.
3. For this to be the case, involuntary leisure time must have no value.
4. In practice, any benefits accruing after 300 years can be ignored. In the case of a zero discount rate, timber values would be infinite so a 300-year life is also assumed for illustrative purposes.
5. See Byrne (1999) for the most comprehensive work.
6. Pearce (1998) also notes that this should not be confused with the benefits of creating rural employment, which, if they are adjudged to exist, are adjusted for using a shadow price.

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Cost–benefit analysis of Irish forest policy


3. Costs and benefits of UK forestry policy

Bob Crabtree

INTRODUCTION

Forestry has been the subject of periodic cost–benefit analysis (CBA) in the UK. The first study was in 1972 (HM Treasury, 1972), a second in 1986 (National Audit Office, 1986), and a third in 1991 (Pearce, 1991). These appraisals and others have all demonstrated that timber produced in isolation from other public benefits gives a very low social return (Price, 1997). Even when market failure adjustments (land values, strategic issues, recreation and environment) have been included, it has proved very difficult to demonstrate a return approaching the test discount rate in any context.

Despite the apparently weak economic case for intervention in forestry the Forestry Commission (2004a, 2004b) has continued to manage a public estate of 1.05 million hectares (which accounts for 29 per cent of all GB woodland) and a net expenditure of around £150 million per year on grant aid, regulation and research. The area of woodlands and forests is still expanding, with 12 456 hectares of new planting in GB in 2003–04.

This chapter assesses more recent evidence on the benefits from forestry. It concentrates on Scotland and England, which together accounted for 96 per cent of new grant-aided planting in GB in 2004.

POLICY

Forestry policy was progressively devolved in the late 1990s, culminating in the Forestry Devolution Review (Forestry Commission, 2002). The Forestry Commission is the forestry department of three administrations – the UK government, the Scottish Executive and the National Assembly for Wales. Its aim is ‘the sustainable management of existing woods and forests, and a steady expansion of tree cover to increase the many, diverse benefits that forests provide to meet the needs of present and future generations’. Different
strategies and implementing mechanisms now exist for each country. For example, the England Forestry Strategy (EFS, Forestry Commission, 1998a) is based on four key programmes:

• rural development;
• economic regeneration;
• recreation, access and tourism; and
• the environment and conservation

whereas the Scottish Forestry Strategy (Scottish Executive, 2000) details the strategic directions for Scotland’s forests as:

• maximizing value;
• creating a diverse forest resource;
• making a positive contribution to the environment;
• enjoying trees, woods and forests; and
• helping communities to benefit.

These strategies reflect country priorities – hence the stronger support for commercial forestry in Scotland where the public and private commercial estates are substantial, and a stronger focus on public use of forests in England. Even so, these national strategies are all designed to provide the legitimacy for a wide range of intervention in forestry with little clarity on the directions that provide greatest public benefit. A further policy twist is that since 1992 planting on farmland has been eligible for additional grant aid under the CAP. Farm forestry has thus become an element in agricultural policy. Forestry policy also has to deliver both on the government’s sustainability indicators and the spending review agreements with HM Treasury.

From this we can deduce that in broad terms, forestry policy is to expand the (private) forest area, manage forests and woodlands sustainably, and deliver a wide range of public good outputs.

THE RATIONALE FOR INTERVENTION

The usual argument to justify intervention is that of market failure, that is, where the market alone does not supply the socially optimal quantities of forest outputs. This can then be taken further to assess whether any proposed intervention provides net returns at least equal to the public sector discount rate. The reality in the forestry sector is slightly more convoluted since there is already substantial intervention both in terms of state forests,
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built up by land purchase since 1919, and through private sector grant aid. The issue is whether continued intervention is justified, taking into account the implications of disposals of the public forest and closing grant aid schemes.

The context for intervention is typically a change in the area under forestry or a change in the management of existing forests to deliver an increased output of public goods (e.g. recreation and biodiversity). Whilst intervention has typically been through the creation of state forests or subsidy to the private sector, Pearce makes the point that intervention could consist of stimulating mechanisms whereby forest owners capture part of the non-market value of their outputs, thus converting them into marketable goods (CJC Consulting, 2003). A case in point is carbon sequestration. In that context, intervention may be more efficiently used to assist producers to trade carbon rather than to subsidize producers to increase the rate of carbon sequestration.

Grounds for considering intervention in forestry exist whenever there is a non-market benefit or disbenefit. There is evidence for environmental benefits from forests in the form of biodiversity conservation, landscape and amenity, recreation, human health, carbon sequestration, hydrology and air pollution reduction (Willis et al., 2003). Environmental disbenefits are less well documented but much of the historic concern about disbenefits has been addressed in the UK Forestry Standard (Forestry Commission, 1998b), which defines the sustainable management standards for UK forests.

A number of other justifications for intervention are made on occasion. These include contributions to regional priorities for economic regeneration and rural development, employment creation, compensation for distortions in land prices from a supported agriculture, and action to displaceexternality-creating activities such as imports of timber grown under non-sustainable conditions.

TIMBER

Although timber is normally a part of the social income from forests there is no market failure case for intervention to secure timber supplies. The original argument for state forestry, that of providing a strategic timber supply, has long ceased to be a justification for policy. The return from timber is now extremely low on all but the most productive sites. Nominal GB prices for standing timber peaked in September 1995 at £17.74 per cubic metre overbark, and have since fallen to £6.19 in September 2004, 35 per cent of the peak value (Forestry Commission, 2005). This collapse has been brought about mainly by increased supplies from the Baltic States and
CIS states and exchange rate movements. UNECE (2005) considers that the economic viability of European forests will remain threatened as they face competition from lower-cost producers.

The effects of low timber prices have been severe for both public and private forestry. A recent review of the public estate in Scotland (CJC Consulting, 2004b) showed that the estate had an income from timber of £39.4 million but costs (excluding specific costs associated with recreation, conservation and heritage) of £71.6 million. At least 15 per cent of the estate, and probably much more, is unprofitable to harvest at these prices. Appraisals of the return from restocking showed that only 9 per cent of the current estate could achieve a 3.5 per cent real return (the current Treasury test discount rate) from timber outputs alone (these appraisals attribute no price to the land and value labour at market rates). The private sector situation parallels this, and private owners have commonly taken the decision to delay felling both in the hope of improved prices and to avoid the costs of obligatory restocking.

A case has been made for shadow pricing land and labour. The CAP increases farmland prices and rents, and Harvey (1991) estimated the extent of public subsidy required to compensate for the distortion at that time. Grant aid for forestry under the Farm Woodland Premium Scheme reflects the need to compensate for this loss of income from farming if planting on farmland is to occur. However, land purchase is no longer a relevant issue for the public estate, because expansion of the estate has long ceased to form part of forestry policy. The employment argument continues to be used to support intervention in forestry in remoter areas where alternative employment may be limited. Whilst Pearce (1994) found some evidence to support shadow pricing of labour this is difficult to justify given the current low unemployment in most rural areas (e.g. CJC Consulting, 2005) and the mobility of forestry contractors. In addition, net employment in forestry on farmland is usually small when the displacement of agricultural employment is taken into account, and most of the impact does not occur until harvest (see Eiser and Roberts, 2002).

Neither shadow pricing of land nor labour has much impact on the return from trees grown for timber because the return is dominated by a low income stream. However, the Forestry Commission continues to provide support for commercial forestry through grant aid for new planting and restocking. The public choice explanation would be that forest owners are a major client group for the Commission and timber is their main output. Subsidy is a sine qua non for their support.

In a recent Scottish scheme designed to encourage farmers to plant trees mainly for timber on agricultural land the income stream from timber and shooting had a present value (PV) of £1204 per hectare at 3.5 per cent
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(CJC Consulting, 2004a). Timber was valued at the mean price in the five years prior to the scheme commencing (£13.5 per cubic metre). The PV of establishment and management costs based on typical contractor charges was £3586 per hectare. This demonstrates the extent of the negative return to society (£–2382 per hectare) from forests that produce only timber even where silvicultural conditions were above average for UK woodlands at yield class 16. However, the woodlands did produce some additional social gain in the form of environmental and recreational benefits but these were estimated to have a PV of only around £800 per hectare – insufficient to allow the investment to pass the cost–benefit test.

RECREATIONAL BENEFITS

The recreational benefits from the public forestry estate were estimated in the late 1980s through travel cost models on a representative sample of forests (Willis, 1991). The consumer surplus averaged around £2 per person per visit (in 1988 prices) but varied depending on the design and location of the site. In a more recent study Scarpa (2003) used both open-ended and discrete modes to estimate visitors’ willingness to pay (WTP) on seven sites in England and Wales. The maximum WTP varied between £1.66 and £2.75 per visit depending on the model used.

Such studies are widely used for developing an aggregate recreational value of the public forest estate. In Scotland, these benefits were estimated at £5.4–8.8 million per year (CJC Consulting, 2004b). Whilst such evidence does contribute to the overall economic evaluation of forest policy, specific policy decisions usually relate to investment in additional recreational stock by extending the forest area. An incremental benefit measure is therefore required. New woodlands are likely to be less attractive to users than the major sites used in most recreational benefit studies. There may also be some displacement of existing use. However, there are exceptions. The Commission has developed forests to offer specialized facilities for mountain biking and horse riding where benefits per visit are probably much higher, although there is lack of good evidence on this aspect. Adaptation of one forest for mountain biking was estimated to generate a net benefit of almost £1 million per year (CJC Consulting, 2004b).

For several years the Forestry Commission provided discretionary grant aid to GB woodlands owners prepared to allow public access to new woodlands in areas where there were no other accessible woods. In an evaluation of 35 woodlands Crabtree et al. (2001) estimated local residents’ WTP for continued procurement of access. The mean WTP for future use was £0.35 per household per year and this did not vary significantly with
the size of the wood. WTP declined to zero for residents more than four miles from the wood. Comparisons of benefits, as measured by mean WTP per site and the increment in grant aid showed that in aggregate the scheme delivered benefits in excess of exchequer costs. However, 20 to 30 per cent of woodlands failed to deliver a net benefit. The issue for policy is whether the efficiency of such schemes can be improved by more effective discrimination against poor value applications without excessive administrative cost. This may be very difficult without detailed analysis of expected local use in relation to population size, accessibility and the extent of substitute opportunities for walking.

One problem with estimates of the consumer surplus from recreation studies is that the recreation benefit estimates may well contain benefits other than those strictly related to access and recreation. Any aggregation to estimate total benefit can then lead to double counting. Crabtree et al. (2001) found that many respondents were willing to pay for continued access to a local woodland but had not actually visited them. The WTP may then include an element of existence value rather than solely an option value related to expected future use.

LANDSCAPE AND AMENITY

Afforestation has a profound effect on landscape and a variety of valuation methods have been applied to estimate the benefits from both existing and new woodlands. Willis and Garrod (1992) used a hedonic pricing model of property values to estimate the impact of the density of woodlands near housing on property prices. They found small positive or negative impacts on property prices depending on the woodland species present. In other studies the public were willing to pay sizeable sums (£24 per household per year) for the restoration of native Caledonian pine forest in Scotland (Macmillan and Duff, 1998). In northern Scotland, Hanley and Craig (1991) also found a sizeable WTP (£300 per hectare), in this case to prevent afforestation of bogs, which were valued for their biodiversity. Neither of the latter studies separates out the pure landscape effects from those on biodiversity.

The most detailed study on the visual attraction that woodlands can provide is that of Garrod (2003) who presented the public with computer generated images of coniferous and broadleaved forest types. Choice experiments were used to explore public preferences for views from homes that were located in a range of different landscapes from urban fringe to upland and mountain. The clearest preference that emerged from the study was for views of broadleaves in an urban fringe setting. Here residents’ WTP for a woodland view varied from £200–£500 per household per
year, depending on the model used and the forest configuration. Negative preferences for woodland were also expressed most notably for woodlands in atypical contexts (e.g. broadleaves in a mountain setting). The Garrod (2003) estimates are useful for providing aggregate estimates of the landscape value from existing forests or from new forests that alter the nature of the landscape view. The annual benefits from GB forests were estimated to be £150.2 million per year. The stated preference research suggests that the main landscape benefits from an expansion in forestry will occur principally on the urban fringe where new woodland views are created.

CARBON SEQUESTRATION

Dewar and Cannell (1992) have calculated carbon retention estimates for different species and yield class, taking account of the temporal pattern of sequestration and the release of carbon from timber, litter and soil. These estimates provide the basis for much of the estimation of carbon sequestration in GB forests (Cannell and Milne, 1995). Bateman and Lovett (2000) and Brainard et al. (2003) further developed models to enable estimates to be made for a wide range of species and sites with alternative assumptions regarding the release of carbon after thinning and harvest. These provided the basis for the recent estimates of net carbon sequestration in GB forests (Willis et al., 2003).

In order to include this benefit in forest cost–benefit analysis the flows of net carbon sequestered need to be priced. Pearce (2003) has reviewed the literature on the global social cost of anthropogenic carbon dioxide emissions and concludes that an unweighted ‘price’ of $4–9 per tC, or roughly, £3–6 per tC, without equity weighting and using a constant discount rate, is probably about right. This is slightly lower than estimates made by Fankhauser (1994) and much lower than the Clarkson and Deyes (2002) estimate of £70 per tC. Much depends on the particular assumptions used in relation to discount rates and equity weighting. Willis et al. (2003) used a price of £6.67 per tC to derive an estimate of the net present value of carbon sequestration benefits from GB forests of £2.68 billion (at 3.5 per cent).

Government policy on the use of forests to sequester carbon has to date not been specific. The UK Climate Change Programme established in 2000 failed to give a clear role for sequestration although it is stated to be part of the plan for meeting the 2010 targets. This programme is now under review, and DEFRA (2004) indicates that government would like to see a rise in biomass production, which presumably includes forestry. However, planting forests to produce timber and fix carbon without other benefits is very unlikely to pass the cost–benefit test unless sequestered carbon is
valued at much higher prices than the £3–7 per tonne of carbon typically used in cost–benefit analyses. Allowances to emit CO₂ are substantially higher within the EU Emissions Trading Scheme, which began operation in January 2005. Since inception, prices have varied between €7 and €34 per tonne of CO₂. Even so, this market price would only be relevant to afforestation decisions were sequestration to be an allowable mechanism for offsetting emissions within such trading schemes. However, if £70 per tonne were to be adopted consistently as the price for carbon, the case for increased afforestation would become much stronger.

**BIODIVERSITY CONSERVATION**

It has proved extremely challenging to quantify the benefits from conserving and creating forestry habitats and related species. The complexity of the topic and the potential for information bias have added to the problems normally encountered in estimating non-use values. The use of choice experiments to value more detailed elements of biodiversity policy (forest design, creation of specific habitats) is also problematic because the public lack sufficient comprehension to make informed trade-offs (Price, 2000). The best that can be attained is some indicative estimates of the public’s preferences for different species.

The most comprehensive studies on the value of forestry-related biodiversity are those of Garrod and Willis (1997) and Hanley et al. (2002). Garrod and Willis concentrated on the value of remote Sitka spruce forests and the Hanley study attempted to generalize from this to cover a much wider range of forest types. Hanley used focus groups to assess the values of marginal expansion of re-structuring of different forest types including native woodlands, conifers and broadleaves. The approach was principally aimed at estimating the relative values of different forest types. It sought to isolate as far as possible the non-use biodiversity value from the landscape and recreational benefits. Even so the limited size of the focus groups and their lack of representation of the full socio-economic diversity in the UK suggest that the estimates should be used with some caution. The respondents expressed strong preferences for native and new broadleaved woodlands as compared with coniferous forests. The results can be used to assess the benefits from restructuring the forest estate on restocking, and the benefits over time that may accrue as relatively young woodlands take on the properties of ancient semi-natural woodlands. Willis et al. (2003) used the estimates to produce an aggregate benefit from GB forest biodiversity of £386 million per year, almost all of which was located in England in the form of ancient, semi-natural and native woodlands.
AGGREGATE BENEFITS

Willis et al. (2003) estimated the aggregate environmental and social benefits from the current GB forest estate (Table 3.1). The biodiversity non-use values are aggregated to the national populations and an estimate for the benefit of air pollution reduction is included. Eighty seven per cent of the total valuation is derived from England’s forests, a reflection of the larger population and the much stronger investment in environmental woodlands.

Table 3.1  Annual aggregate value of social and environmental benefits of British forestry

<table>
<thead>
<tr>
<th>Benefit</th>
<th>Valuation (annual, £m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Recreation</td>
<td>392.65</td>
</tr>
<tr>
<td>Landscape</td>
<td>150.22</td>
</tr>
<tr>
<td>Biodiversity</td>
<td>386.00</td>
</tr>
<tr>
<td>Carbon sequestration (at £6.67 per tC)</td>
<td>93.66</td>
</tr>
<tr>
<td>Air pollution absorption</td>
<td>0.39</td>
</tr>
<tr>
<td>Total</td>
<td>1022.92</td>
</tr>
</tbody>
</table>

Not all the costs and benefits of forestry are included in Table 3.1 although these are likely to be the predominant items. It has proved difficult to fully quantify the effects of afforestation on water quantity and quality. It is possible to estimate the additional costs of water supply to compensate for losses from afforestation but water quality effects are more complex. Forestry can exacerbate effects of acidification but mitigate agricultural pollution by displacing farm production. In some contexts there may be a beneficial effect in reducing soil erosion and flood hazards. But the magnitude of these effects depends on the specific location, and Willis et al. (2003) conclude that the information set is too limited to provide a satisfactory statement of the overall impacts on benefits derived from the water resource.

Health benefits are a second example of an area where information is lacking. Benefits from recreation are generally assumed to be captured under the consumer surplus from recreation but this may be mistaken. It is unlikely that survey respondents are well informed on the long-term health benefits from physical activity. They may therefore not fully take them into account in their stated preferences. Swales (2001) has estimated the social benefit from increased physical activity in Northern Ireland in terms of reduced mortality and the value of preventable fatalities (DETR, 1997). He concluded that reducing the sedentary element in the population
by 1 per cent (from 20 per cent to 19 per cent) would save 24 lives with an economic benefit valued at £26.2 million. The break-even investment in an exercise programme would be around £2000 per person moved from a sedentary to active lifestyle. However, a comparison with typical costs for health interventions that reduce avoidable lost life years suggests that greenspace-related health programmes will have to operate at much lower costs than this if they are to be cost-effective. Neither analysis includes the psychological and quality of life benefits from greenspace (Pretty et al., 2005) and benefits may therefore be underestimated. This is an area where more research is required to establish exercise programmes that can be reliably evaluated in terms of the behavioural change of participants and the improvement in health outcomes. This will better inform government policies that encourage healthier and less sedentary lifestyles (e.g. Welsh Assembly Government, 2005).

COST–BENEFIT ANALYSIS OF FORESTRY POLICY

Aggregate estimates of forest benefits as in Table 3.1 are useful to the Forestry Commission in the political and economic debate over public subsidy for forestry. It has been less obvious that benefit valuation research has been used to inform more specific aspects of public expenditure on forestry and the management of the public estate. However, for external economic analysis the benefit transfer models have proved very informative. For example, CJC Consulting (2004a, 2004b) has shown that much of the more commercial afforestation has proved uneconomic, and continues to be so, whereas forests delivering high levels of public goods have often proved valuable social investments.

A good example of this dichotomy is the public forestry estate, which is largely located in Scotland. Here cheap land provided the basis for widespread ‘commercial’ planting when the public estate was expanding. The value of the main environmental and recreation benefits from the estate, based on Willis et al. (2003), is £40–43 million per year (CJC Consulting, 2004b). Adjusting the Commission’s accounts to take account of these benefits indicated a return to investment of around the 3.5 per cent benchmark. More detailed analysis reveals that much of the estate is now uneconomic to restock following harvest. Typically, such uneconomic woodlands are in poor locations for tree growth, remote from people and with little biodiversity interest. Hence they deliver little in the way of additional benefit to the public. Macmillan (1993) identified the scale of this problem some years ago in a spatial cost–benefit analysis on restocking policy for existing forests. He showed that uneconomic sites were mainly
located in the north, west and southwest of Scotland where yield classes are relatively low, windthrow risk high and other public benefits limited. The situation has now been aggravated by the collapse in timber prices. The Commission is heavily constrained in its options because its sustainability rules under the UKFS do not permit large-scale abandonment or inclusion of open space. Just as benefit research has supported the development of public good forestry so now it is needed to identify the losses or gains from abandonment or conversion of existing low-yield forests. There have been no valuation studies on this aspect to inform policy. The alternative is the application of arbitrary sustainability rules that entail substantial costs to the public exchequer.

With forestry policy committed to forestry expansion, CBA has informed decisions on the extent, type and location of new planting. As predictive models for forestry benefits become better developed and benefit transfer more secure it should be feasible to generate GIS spatial models of the type explored by Bateman et al. (1999) for recreation in Wales. Analysis of different types of new planting in England emphasized the wide variation in social return between different types of woodlands in different locations (CJC Consulting, 2003). Those providing visible and recreational amenity close to urban centres have high rates of return. There are likely to be positive returns from woodland investment as part of urban regeneration programmes but more information is required to quantify benefits because the opportunity cost of land can be high. In contrast, woodlands in rural locations that produce timber but few social or environmental benefits offer limited social returns. For these reasons it is not easy to support forestry as a mechanism for rural development. In addition any benefits to local economies from forestry activity mainly occur at the time of harvest which is possibly 50 years into the future.

FORESTRY IN THE WIDER POLICY FRAME

With no direct case for intervention in timber production the central core of forest policy has now disappeared. Most of the content of forestry policy is in fact a contribution by forestry to other policy agendas within government. The institutional manifestation of this is the gradual absorption of forestry within the devolved administrations in Wales and Scotland and stronger links between the Forestry Commission and DEFRA in England. Forestry then becomes a mechanism for delivering on priorities in agriculture, environment, economic development and other policy domains. These agendas may be related to internal departmental targets and indicators or those derived from EU legislation. For example, the EU Water Framework
Directive places major demands on government to improve the quality of surface and ground waters in the UK. Pearce (2004) has questioned the merit of the Directive as it becomes more apparent that its implementation costs will exceed the social benefits derived. Nevertheless, forestry can reduce nitrogen, phosphorus and pesticide pollution by substituting for agriculture. It may also provide other water quality benefits (see above). What then becomes important is its cost-effectiveness as a mechanism as compared with the cost of modifying agricultural practices in catchments at risk.

The danger is that if forestry is assessed as a delivery mechanism for single-objective policies, it will prove costly compared with other instruments because the value of its other outputs will be ignored. It will be important to make sure that other benefits are fully accounted for in these cost-effectiveness assessments since they are likely to determine forestry’s role in the future.

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INTRODUCTION

The aims of this chapter are to identify problems surrounding the economic valuation of ‘biodiversity’, and then to present results from a recent stated preference study on changes in biodiversity on UK farmland, which attempts to get around one major problem, namely the information deficit that typifies the knowledge level of most members of the general public regarding biodiversity. We also provide a first choice experiment estimation of the attributes of biodiversity, an approach that may prove useful in developing policy on biodiversity protection and enhancement; obtain contingent valuation estimates for different policies, which would increase biodiversity on farmland; and compare values obtained using standard survey procedures with those obtained using the ‘valuation workshop’ technique (Macmillan et al., 2003). Finally, we test for benefits transfer in both values and valuation functions across geographic areas.

In what follows, the first section discusses motivations for estimating biodiversity values and problems encountered, whilst the second presents a brief review of the literature. Our study design is explained in the third section, with results presented in the fourth. A discussion concludes the chapter.

WHY DO WE WANT TO ESTIMATE THE ECONOMIC VALUE OF BIODIVERSITY?

Society needs to make difficult decisions regarding its use of biological resources, for example in terms of habitat conservation, or changing how we
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manage farmland through agri-environmental policy (Hanley and Shogren, 2002). Environmental valuation techniques can provide useful evidence to support such policies by quantifying the economic value associated with the protection of biological resources. Pearce (2001) argues that the measurement of the economic value of biodiversity is a fundamental step in conserving this resource since ‘the pressures to reduce biodiversity are so large that the chances that we will introduce incentives [for the protection of biodiversity] without demonstrating the economic value of biodiversity are much less than if we do engage in valuation’. Assigning monetary values to biodiversity is thus important since it allows the benefits associated with biodiversity to be directly compared with the economic value of alternative resource use options (see also Nunes and van den Bergh, 2001). OECD (2001) also recognizes the importance of measuring the economic value of biodiversity and identifies a wide range of uses for such values, including demonstrating the value of biodiversity, in targeting biodiversity protection within scarce budgets, and in determining damages for loss of biodiversity in liability regimes.

More generally, the role of environmental valuation methodologies in policy formulation is increasingly being recognized by policy-makers. For example, the Convention on Biological Diversity’s Conference of the Parties decision IV/10 acknowledges that ‘economic valuation of biodiversity and biological resources is an important tool for well-targeted and calibrated economic incentive measures’ and encourages parties, governments and relevant organizations to ‘take into account economic, social, cultural and ethical valuation in the development of relevant incentive measures’. The EC Environmental Integration Manual (2000) provides guidance on the theory and application of environmental economic valuation for measuring impacts to the environment for decision-making purposes. Within the UK, HM Treasury’s Green Book provides guidance for public sector bodies on how to incorporate non-market costs and benefits into policy evaluations.

The idea of placing economic values on the environment has been challenged by many authors on a variety of grounds, from ethical objections to participatory perspectives. However, what concerns us here is not whether one should attempt to place economic values on changes in biodiversity, but rather what the particular difficulties are in doing so. These include incommensurate values or lexicographic preference issues (Spash and Hanley, 1995; Rekola, 2003) and – the issue we focus on here – people’s understanding of complex goods (Hanley et al., 1996; Christie, 2001; Limburg et al., 2002).

Stated preference valuation methods ideally require survey respondents to make informed value judgements on the environmental good under investigation. This requires information on these goods to be presented
to respondents in a meaningful and understandable format, which in turn will enable them to express their preferences consistently and rationally. Herein lies the problem: many studies have found that members of the general public have a low awareness and poor understanding of the term ‘biodiversity’. If one is unaware of the characteristics of a good, then it is unlikely that one has well-developed preferences for it that can be uncovered in a stated preference survey.

Various surveys have examined the public’s understanding of the term ‘biodiversity’. A recent UK survey found that only 26 per cent of respondents had heard of the term ‘biodiversity’ (DEFRA, 2002). Similar findings are also reported in Spash and Hanley (1995). The lack of public understanding of the term ‘biodiversity’ will make the valuation exercise difficult; however, people can learn during a survey, and may have preferences for what biodiversity actually means, even if they are unaware of the term itself. The DEFRA (2002) survey also found that 52 per cent considered the protection of wildlife to be ‘very important’, even though they did not know what biodiversity itself meant.

An additional complication is that biodiversity itself is not uniquely defined by conservation biologists. Scientists are in general agreement that the number of species per unit of area provides a useful starting point (Harper and Hawksworth, 1995; Whittaker, 1977). Although such a measure appears to be relatively straightforward, issues such as what constitutes a species and what size of area to count species overcomplicate this measure. Even if these questions were resolved, ecologists recognize that some species, such as keystone species, may be more important and/or make a greater contribution to biodiversity than others. A further complicating factor relates to the extent to which the public are capable of understanding ecologists’ concepts. The issues highlighted above indicate that research that attempts to value changes in biodiversity will be challenging, since it requires us to identify appropriate language in which biodiversity concepts can be meaningfully conveyed to members of the public in ways that are consistent with underlying ecological ideas on what biodiversity is.

PREVIOUS LITERATURE

A general comment on much of the existing biodiversity valuation literature is that it mostly does not value diversity itself, but focuses rather on individual species and habitats (Pearce, 2001). In this section we review a number of key studies that have attempted to measure the economic value of different elements of biodiversity. In particular, we distinguish between studies that have valued biological resources (e.g. a particular
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species, habitat area, or ecosystem function) and those that have valued the biological diversity of those resources (e.g., components of biodiversity such as rarity or charismatic species).

Studies that Value Biological Resources

A summary of the range of value estimates for three categories of biological resources can be found in Table 4.1. The first category of biological resources includes both genetic and species diversity. Studies that have quantified genetic diversity have predominantly measured direct use benefits of biological resources in terms of inputs to the production of market goods such as new pharmaceutical and agricultural products. The majority of studies have based valuations on market contracts and agreements for bioprospecting by pharmaceutical industries. Ten Kate and Laird (1999) provide an extensive review of such bioprospecting agreements. Franks (1999) provides a useful contribution on the value of plant genetic resources for food and agriculture in the UK and also the contribution of the UK’s agri-environmental schemes to the conservation of these genetic resources.

There have been a large number of studies that have valued species. Most of these studies have been undertaken in the US and utilize stated preference techniques, thus enabling both use and passive use values to be assessed. Nunes and van den Bergh (2001) provide an extensive review of valuation studies that have addressed both single and multiple species. Valuations for single species range from $5 to $126, and for multiple species range from $18 to $194 (Table 4.1). In the UK, there have been a limited number of studies that have valued both single and multiple species. For example, Macmillan et al. (2003) estimated the value of wild geese conservation in Scotland, while White et al. (1997 and 2001) examine the value associated with the conservation of UK mammals including otters, water voles, red squirrels and brown hare. Macmillan et al. (2001) also take a slightly different perspective by valuing the reintroduction of two species (the beaver and wolf) into native forests in Scotland.

Biological resources may also be described in terms of the diversity within natural habitats. Studies have addressed the valuation of habitats from two perspectives. One approach is to link the value of biodiversity to the value of protecting natural areas that have high levels of outdoor recreation or tourist demand. A second approach to the valuation of natural areas involves the use of stated preference methods. Table 4.1 summarizes the range of passive use values elicited for terrestrial, coastal and wetland habitats. UK examples of contingent valuation (CV) studies that have valued habitats include: Garrod and Willis, (1994) who examined the willingness to pay...
Natural resources

Table 4.1 Value ranges for biological resources

<table>
<thead>
<tr>
<th>Life Diversity Level</th>
<th>Biodiversity Value Type</th>
<th>Value Ranges</th>
<th>Method(s) Selected</th>
</tr>
</thead>
<tbody>
<tr>
<td>Genetic and species diversity</td>
<td>Bioprospecting Single species</td>
<td>From $175,000 to $3.2 million From $5 to $126</td>
<td>Market contracts Contingent valuation Contingent valuation</td>
</tr>
<tr>
<td></td>
<td>Bioprospecting Multiple species</td>
<td>From $18 to $194</td>
<td></td>
</tr>
<tr>
<td>Ecosystems and natural habitat diversity</td>
<td>Terrestrial habitat (passive use)</td>
<td>From $27 to $101</td>
<td>Contingent valuation</td>
</tr>
<tr>
<td></td>
<td>Coastal habitat (passive use)</td>
<td>From $9 to $51</td>
<td>Contingent valuation</td>
</tr>
<tr>
<td></td>
<td>Wetland habitat (passive use)</td>
<td>From $8 to $96</td>
<td>Contingent valuation</td>
</tr>
<tr>
<td></td>
<td>Natural areas habitat (recreation)</td>
<td>From $23 per trip to $23 million per year</td>
<td>Contingent valuation Travel cost, tourism revenues</td>
</tr>
<tr>
<td>Ecosystems and functional diversity</td>
<td>Wetland life support</td>
<td>From $0.4 to $1.2 million to $454 million per year</td>
<td>Replacement costs Replacement costs, hedonic price, production function</td>
</tr>
<tr>
<td></td>
<td>Soil and wind erosion protection</td>
<td>Up to $454 million per year</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Water quality</td>
<td>From $35 to $661 million per year</td>
<td>Replacement costs, averting expenditure</td>
</tr>
</tbody>
</table>

Source: Adapted from Nunes and van den Bergh (2001).

(WTP) of members of the Northumberland Wildlife Trust for a range of UK habitat types; Hanley and Craig (1991) who valued upland heaths in Scotland’s ‘flow country’; and Macmillan and Duff (1998) who examine the public’s WTP to restore native pinewood forests in Scotland.

Ecosystem functions and services describe a wide range of life support systems including waste assimilation, flood control, soil and wind erosion, and water quality. Many of these functions and services are complex and it is likely that members of the public will possess a poor understanding of these issues. The consequence of this is that attempts to value ecosystem functions and services will be difficult, particularly in methods (such as
the stated preference methods) where respondents are required to make a value judgement based on the description of the good in question. Analysts often use other techniques including averting behaviour, replacement costs and production functions to measure the indirect values of ecosystem functions.

**Studies that Value Biological Diversity Itself**

A number of valuation studies have attempted to value biodiversity by explicitly stating to respondents that the implementation of a conservation policy will result in an increase in the biodiversity of an area. For example, Garrod and Willis (1997) estimated passive use values for biodiversity improvements in remote upland coniferous forests in the UK. The improvements in forest biodiversity were described in relationship to a series of forest management standards that increased the proportion of broadleaved trees planted and the area of open spaces in the forest. The marginal value of increasing biodiversity in these forests was estimated to range between £0.30 to £0.35 per household per year for a 1 per cent increase and between £10 to £11 per household per year for a 30 per cent increase in increased biodiversity forest area. Willis et al. (2003) extend this work to examine public values for biodiversity across a range of UK woodland types. Other studies have assessed public WTP to prevent a decline in biodiversity. For example, Macmillan et al. (1996) measure public WTP to prevent biodiversity loss associated with acid rain; whilst Pouta et al. (2000) estimate the value of increasing biodiversity protection in Finland through implementing the Natura 2000 programme. White et al. (1997 and 2001) examine the influence of species characteristics on WTP. They conclude that charismatic and flagship species such as the otter attracted significantly higher WTP values than less charismatic species such as the brown hare. They further suggest that species with a high charisma status are likely to command higher WTP values than less charismatic species that may be under a relatively greater threat or of more biological significance in the ecosystem. In a meta-analysis of WTP for a range of species, Loomis and White (1996) also find that more charismatic species, such as marine mammals and birds, attract higher WTP values than other species.

The above review has demonstrated that from those studies that have claimed to value biodiversity, only a handful have actually examined biological diversity; most studies have alternatively tended to value biological resources. Furthermore, studies that have valued biological diversity currently only provided limited information on the value of the components of biological diversity. Research effort has yet to provide a comprehensive
assessment of the value attached to the components of biological diversity such as anthropocentric measures (e.g. cuteness, charisma and rarity) and ecological measures (e.g. keystone species and flagship species).

STUDY DESIGN

The policy setting for this research is the development of policy on biodiversity conservation and enhancement on farmland in England. The principal challenges in study design were to identify what aspects of the ecological concept ‘biodiversity’ needed to be communicated to the general public, and thus form the focus of the valuation exercise. For a concept to be relevant in this context, it has to have ecological significance, be capable of being explained to ordinary people, and be something that they might in principle care about. We also needed to design effective ways of conveying information.

In a review of ecological literature (Christie et al., 2004), we identified 21 different concepts that ecologists use to describe and measure biodiversity. Clearly, it would be extremely difficult to attempt to value all of these concepts. In an attempt to simplify this, a conceptual framework was drawn up to provide a simplified and structured framework in which biodiversity could meaningfully be presented to members of the public (Figure 4.1). This framework is split into sections according to which perspective we take on the importance and meaning of biodiversity: ecological or anthropocentric. Within each of these headings, we identify different aspects of biodiversity that need to be considered for inclusion. The final row of the figure shows the biodiversity attributes that were eventually selected for the experimental design. We now explain how these were chosen.

A series of focus groups composed of members of the general public were arranged. The discussions held in the focus groups aimed to identify the level of understanding that the public had for each of the elements of the framework in Figure 4.1, and also to identify their views on the importance of each element. The framework was then amended to reflect this input from the focus groups.

One of the first issues discussed in the focus groups related to an assessment of the level of public understanding of the scientific terms and concepts associated with biodiversity. Discussions indicated that over half of the participants had never knowingly come across the term ‘biodiversity’ before. Furthermore, some of those who had indicated a familiarity with the term ‘biodiversity’ were unable to provide a clear or accurate definition of the concept. Alternative ways of describing biodiversity were discussed and the phrase ‘the variety of different living organisms within a particular area
**Figure 4.1** Conceptual framework – biodiversity concepts

<table>
<thead>
<tr>
<th>Biodiversity Concepts</th>
<th>Ecological concepts</th>
<th>Anthropocentric concepts</th>
</tr>
</thead>
<tbody>
<tr>
<td>Keystonespecies</td>
<td>Keystone species</td>
<td>Keystone species</td>
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<tr>
<td>Umbrellasspecies</td>
<td>Umbrella species</td>
<td>Umbrella species</td>
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<tr>
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<td>Flagship species</td>
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<td>Ecosystemfunction</td>
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<td>Endangered species</td>
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<tr>
<td>Rarespecies</td>
<td>Rare species</td>
<td>Rare species</td>
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<tr>
<td>Charismaticspecies</td>
<td>Charismatic species</td>
<td>Charismatic species</td>
</tr>
<tr>
<td>Cutepecies</td>
<td>Cute species</td>
<td>Cute species</td>
</tr>
<tr>
<td>Familiarspecies</td>
<td>Familiar species</td>
<td>Familiar species</td>
</tr>
<tr>
<td>Locallyimportantspecies</td>
<td>Locally important species</td>
<td>Locally important species</td>
</tr>
<tr>
<td>Habitatquality</td>
<td>Habitat quality</td>
<td>Habitat quality</td>
</tr>
<tr>
<td>Ecosystemprocesses</td>
<td>Ecosystem processes</td>
<td>Ecosystem processes</td>
</tr>
<tr>
<td>Rare, unfamiliar species of wildlife</td>
<td>Rare, unfamiliar species of wildlife</td>
<td>Rare, unfamiliar species of wildlife</td>
</tr>
<tr>
<td>Familiarspecies of wildlife</td>
<td>Familiar species of wildlife</td>
<td>Familiar species of wildlife</td>
</tr>
</tbody>
</table>
or habitat’ was considered to be both useful and meaningful. Participants indicated that they were familiar with some related terms including ‘species’, ‘habitat’ and ‘ecosystem’, but were not familiar with the majority of scientific concepts of biodiversity such as keystone species, flagship species etc. On a more positive note, however, it was also found that most participants of the focus groups appeared to be capable of quickly picking up a basic understanding of most biodiversity concepts if these were explained in layperson's terms. However, some participants indicated that they were often confused with regards to the precise definitions of the more closely related ecological concepts. The conclusion from this is that the survey would need to employ alternative, non-scientific terminology to meaningfully describe biodiversity.

Focus group participants considered ecosystem processes to be important. However, they were not able to clearly recognize the differentiation between the terms ‘ecological functions’ and ‘ecosystem health’. It was therefore concluded that these definitions be made less precise to allow these two concepts to be combined into a single category ‘ecosystem processes’. A further issue raised in the discussions related to the level of impact that ecosystem processes had on humans, and this was considered to be an important attribute of ecosystem processes worthy of further investigation.

The second group of biodiversity concepts identified relate to anthropocentric concerns. Within this group, the concepts of rare and endangered species were both considered to be very important. There was, however, confusion regarding the precise definitions of these terms. For this reason, it was argued that these two concepts should be combined into a single category. Participants were also aware of the alternative levels of threat that a species may be under and they considered this to be very important. The concepts of ‘charismatic’, ‘cute’ and ‘familiar’ species were all considered to have significant overlap and therefore it was considered that there would be no benefit from attempting to differentiate between the concepts. The concept of ‘cute’ species, however, was not considered to be helpful or important and therefore the concept could be dropped from the framework. ‘Locally important’ species were also considered to be important both because people valued the fact that they would be able to see, first hand, the benefits from protecting these local species and because they valued the symbolic nature of local species. In all cases, a common theme was that these species were in some way or another likely to be familiar to the public. Thus, it was concluded that it would be useful to identify a group based on familiar species of wildlife.

Thus two distinct themes emerge from the anthropocentric concepts: familiarity and rarity. Based on evidence from the focus groups and comments from the research steering committee, it was proposed that a
distinction should be made between familiar species and unfamiliar species, and that a level of rarity be considered within each grouping. Based on this focus group evidence, it was decided that the choice experiment part of the study would focus on four biodiversity characteristics:

- **Familiar species of wildlife.** This attribute includes the concepts charismatic, familiar (recognizable) and locally symbolic species, and both common and rare familiar species.
- **Rare, unfamiliar species of wildlife.** This attribute focuses on those species that are currently rare or in decline, which are unlikely to be familiar to members of the public. It was considered that this was an important policy question. Also, it was considered important to incorporate an assessment of the effect that the degree of protection from rarity has on values.
- **Habitat quality.** Habitat quality was used as a proxy for the preservation of ecologically significant species such as keystone and umbrella species. A key feature of the habitat quality attribute is to examine the totality of the habitat in terms of supporting a mix of species, rather than to focus on individual species.
- **Ecosystem processes.** Ecosystem processes focuses on preserving the health of ecosystem functions and services. It was also considered useful to distinguish between ecosystem processes that have a direct impact on humans and those that do not.

However, we also wanted to estimate values for three types of biodiversity changes that were considered to be of particular relevance to policy makers, namely: biodiversity enhancement associated with agri-environmental schemes, biodiversity enhancements associated with the re-creation of wildlife habitats, and biodiversity loss from farmland associated with development activities (e.g. house building). Contingent valuation scenarios could be designed to directly elicit the values of the three proposed policy programmes, and thus seems a neater, more direct approach with regard to this second research objective.

Two case study areas were selected: Cambridgeshire and Northumberland. In each area, in-person interviews were undertaken with a random sample of the population, in people’s homes. Information on biodiversity was conveyed to respondents using a PowerPoint audio-visual presentation (see below). The survey instrument included both a choice experiment and a contingent valuation exercise. As a check on validity, we also ran a series of valuation workshops in Northumberland, which included the same questions as the main survey.
A key factor affecting the validity of stated preference studies relates to the success to which the good under investigation can be meaningfully, accurately and consistently presented to survey respondents. Although this can be a challenge in many valuation studies, the very fact that only a small proportion of the public have knowingly heard of the term biodiversity before presents a significant challenge to this research. In this study, the survey instrument was required to present a lot of information on biodiversity that is likely to be complex and new to respondents. The majority of valuation studies tend to describe the environmental good under investigation using verbal descriptions, perhaps supported by some written script and/or pictorial images. Although such an approach to presenting the good can be successful with goods that are familiar to survey respondents, evidence gathered in the focus groups indicated that such a standard approach was unlikely to be suitable for presenting biodiversity that was found to be unfamiliar and considered complex. Feedback from focus groups also indicated that the large volume of new information required to be presented on biodiversity was found to lead to both confusion and respondent fatigue. The adoption of a more visual and interactive approach was therefore considered to be more suitable.

For these reasons, we used a PowerPoint show to convey information to respondents at the start of the survey. This has a number of advantages in terms of using a range of formats (pictures, audio tracks and text), which helps minimize respondent fatigue and maximize the effectiveness with which information is conveyed. The PowerPoint presentation introduced survey respondents to a simple definition of biodiversity: ‘biodiversity … is the scientific term used to describe the variety of wildlife in the countryside’. The narrative that accompanied this slide provided further elaboration of this definition and provided examples to illustrate various aspects of biodiversity. Slides 3 to 8 then introduced the four attributes of biodiversity that had been identified in the focus groups: familiar species of wildlife, rare (unfamiliar) species of wildlife, habitat quality and ecosystem processes. Each attribute was defined, and the alternative levels of biodiversity enhancements associated with these attributes were introduced. Within these descriptions, named examples of relevant species, habitats and ecosystem processes within the study areas were provided and images presented. These were included to help respondents attain a clearer understanding of the various aspects of biodiversity being discussed. Respondents were also made aware of alternative motivations that people may have for protecting the various aspects of biodiversity. For example, respondents were reminded that they ‘might recognize an individual mammal, reptile, bird or even plant because it possesses impressive features such as being large or colourful, or alternatively that it has a particular significant place in local culture’.
Following the presentation of this information, respondents were provided with an opportunity to discuss and clarify any issues of outstanding confusion. In slides 9–12, the case study area (Cambridge or Northumberland) was then introduced. Details presented included a description of the predominant land uses found within the case study areas, and the current levels of biodiversity that exist in those areas. Respondents were then informed that human activities, such as farming and development, are currently threatening overall levels of biodiversity in the area and the consequences of this on the four biodiversity attributes were outlined. Slides 13–18 informed respondents that the government could introduce policies to help protect and enhance biodiversity in the respective case study areas. Policies described included agri-environmental schemes and habitat re-creation schemes. The slides also outlined how such policies could be introduced to specifically enhance the four aspects of biodiversity identified earlier. In each case, the potential improvements were described in terms of the attribute levels used in the choice experiment. Respondents were then asked to think about which aspects of biodiversity they would like to see being protected and enhanced. Finally, at the end of the presentation respondents were given a further opportunity to clarify any issues of confusion/uncertainty regarding any aspect of the presentation.

The feedback from respondents of a pilot survey indicated that the majority of respondents understood the concepts presented. Respondents also indicated that the presentation of more information (to try to increase understanding) would likely be detrimental to the study as a whole since this would lead to respondent fatigue. Thus, the inclusion of further opportunities for respondents to discuss issues of confusion with the interviewer was seen as a better option to ensure that respondents fully understood the information presented.

Following the PowerPoint presentation, respondents of both the household survey and valuation workshops were asked to complete a choice experiment exercise. The choice experiment was introduced as follows:

In the presentation you were provided with information on different aspects of biodiversity. You were also informed that biodiversity within Cambridgeshire [Northumberland] is under threat. We as a society have some options over how we respond to the threats to biodiversity. We are therefore interested in your opinions on what action you would most like to see taken.

We are now going to show you five alternative sets of policy designs that could be used to enhance Cambridgeshire’s [Northumberland’s] biodiversity. In each set, you will be asked to choose the design which you prefer.

An example of a choice task was then presented to respondents and the choice task was explained. Once the respondents had undertaken all five choice tasks, they were asked to indicate the main reason that they had for...
making the choice that they did. This was to allow protest responses to be identified.

We have already explained how biodiversity attributes were selected for inclusion in the choice experiment (above). Each of these attributes was then defined according to three levels of provision, including the status quo and two levels of improvement/enhancement. Table 4.2 below provides a summary of the four biodiversity attributes used in the choice experiment, along with the three levels of provision of each attribute.

Table 4.2  Summary of biodiversity attributes and levels used in the choice experiment

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Policy Level Option A</th>
<th>Policy Level Option B</th>
<th>Do Nothing (Biodiversity degradation will continue)</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Familiar species of wildlife</em></td>
<td>Protect rare familiar species from further decline</td>
<td>Protect <em>both rare and common</em> familiar species from further decline</td>
<td>Continued decline in the populations of familiar species</td>
</tr>
<tr>
<td><em>Rare, unfamiliar species of wildlife</em></td>
<td>Slow down the rate of decline of rare, unfamiliar species</td>
<td>Stop the decline and ensure the recovery of rare unfamiliar species</td>
<td>Continued decline in the populations of rare, unfamiliar species</td>
</tr>
<tr>
<td><em>Habitat quality</em></td>
<td>Habitat restoration, e.g. by better management of existing habitats</td>
<td>Habitat re-creation, e.g. by creating new habitat areas</td>
<td>Wildlife habitats will continue to be degraded and lost</td>
</tr>
<tr>
<td><em>Ecosystem processes</em></td>
<td>Only ecosystem services that have a direct impact on humans, e.g. flood defence, are restored.</td>
<td>All ecosystem services are restored</td>
<td>Continued decline in the functioning of ecosystem processes</td>
</tr>
<tr>
<td><em>Annual tax increase £</em></td>
<td>10</td>
<td>25</td>
<td>100</td>
</tr>
</tbody>
</table>

In choice experiments it is common practice to include a standard option within all choice tasks. In this study, we choose a ‘Do nothing’ policy option. The ‘Do nothing’ option was designed to reflect the situation where no new policies would be implemented to protect and enhance biodiversity on farmland in the case study areas. The consequence for this option in terms of the four attributes of biodiversity was reported as a continued decline in biodiversity in the study area.
The payment vehicle used in the choice experiment was an increase in general taxation. The reasons for using this payment vehicle include the fact that biodiversity enhancement programmes are generally paid for through taxation and that participants of the focus groups indicated that taxation was their preferred payment option. Six payment levels of taxation were used in the choice experiment, including the £0 level in the status quo option. The actual levels used were identified following a small open-ended pilot contingent valuation study, which identified the likely range of bid levels for biodiversity enhancements. These levels were then tested in a pilot choice experiment. The final tax levels used in the choice experiment were: £10, £25, £100, £260, £520, plus the no tax increase in the ‘Do nothing’ option. Tax rises were annual increases per household for the next five years. An SPSS program was used to generate a \((3^4 \times 5^1)\) fractional factorial experimental design, which created 25 choice options. A blocking procedure was then used to assign the options to ten bundles of five choice sets. Thus each choice experiment respondent was presented with a bundle of five choice tasks, where each choice task comprises two policy options and a status quo. Both the household interviews and valuation workshops used this experimental design.

In the Cambridgeshire survey, respondents were also presented with three contingent valuation policy scenarios:

- WTP for agri-environmental schemes such as conservation headlands, and reduced use of pesticides and fertilizers – funded by higher taxes;
- WTP for habitat creation, including seasonal flood plains, reed beds and more natural river flows – funded by higher taxes;
- WTP to protect farmland currently under agri-environmental schemes from development in the form of new houses – conservation here would be financed by a trust fund.

Each respondent only received one of these three scenarios.

In the Northumberland survey, two scenarios were used in the contingent valuation:

- WTP for habitat creation, focusing on wet grasslands funded by a trust fund;
- WTP to protect farmland currently under agri-environmental schemes from development in the form of new houses – conservation here would be financed by a trust fund.

Again, each respondent only received one of these two scenarios.
In addition to the household survey six valuation workshops were undertaken during this research. All workshops were administered in Northumberland, and the location where workshops were conducted was stratified between rural villages, towns and city. A sampling frame based on gender and age was used, and ten individuals were selected on the day before the actual workshop. A £20 incentive was provided to encourage participation in the workshops. The workshops used the same survey instruments as the main study, but the structure of the workshops allowed much greater time for reflection on the information provided, whilst participants were encouraged to discuss the issues involved with each other. Opportunities for questions to the moderator also existed.

RESULTS

The Choice Experiment

In the main household survey, 741 respondents (343 in Cambridgeshire and 398 in Northumberland) each undertook five choice tasks. In the valuation workshops, 53 respondents undertook five choice tasks before the discussion and five choice tasks after the discussion.

Table 4.3 shows results from the choice experiment data for both Cambridgeshire (a) and Northumberland (b), based on a conditional logit model. The pseudo R^2 value is higher for the latter sample, and is very close to the 20 per cent level suggested by Louviere et al. (2000) as indicating a very good fit in this kind of data. The Cambridgeshire model shows significant estimates for all the attribute parameters. In almost all cases, parameter signs are in accord with a priori expectations. As may be seen, improving familiar species from continued decline to either protecting rare species only or protecting all species increases utility; moving habitat quality from continued decline to habitat restoration or habitat re-creation is positively valued; moving ecosystem processes from continued decline to a recovery of either directly relevant services alone or all services creates higher utility. The only exception is for rare, unfamiliar species of wildlife. Here, although a move from continued decline to stopping decline and ensuring recovery increases well-being, a move to slowing decline is negatively valued. All tax increases reduce utility, as expected.

For Northumberland, the same pattern is repeated, except that the ECOALL and RARESLOW attributes are not significant. This means that any improvement in habitat quality or familiar species is positively and significantly valued, as is an improvement in directly relevant ecosystem processes – although not an improvement in all processes. This implies the
Table 4.3 Logit models for Cambridge and Northumberland CE samples

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Parameter Estimate</th>
<th>t-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>(a) Cambridgeshire</td>
<td></td>
<td></td>
</tr>
<tr>
<td>FAMRARE</td>
<td>0.126</td>
<td>2.1</td>
</tr>
<tr>
<td>FAMBOTH</td>
<td>0.331</td>
<td>5.2</td>
</tr>
<tr>
<td>RARESLO</td>
<td>–0.165</td>
<td>–3.0</td>
</tr>
<tr>
<td>RARECOV</td>
<td>0.408</td>
<td>5.7</td>
</tr>
<tr>
<td>HABREST</td>
<td>0.122</td>
<td>2.3</td>
</tr>
<tr>
<td>HABCREAT</td>
<td>0.217</td>
<td>3.5</td>
</tr>
<tr>
<td>ECOHUMA</td>
<td>0.19</td>
<td>3.2</td>
</tr>
<tr>
<td>ECOALL</td>
<td>0.15</td>
<td>2.2</td>
</tr>
<tr>
<td>PRICE</td>
<td>–0.004</td>
<td>–15.2</td>
</tr>
<tr>
<td>Pseudo R² (%)</td>
<td>14</td>
<td></td>
</tr>
<tr>
<td>n (individuals)</td>
<td>343</td>
<td></td>
</tr>
<tr>
<td>(b) Northumberland</td>
<td></td>
<td></td>
</tr>
<tr>
<td>FAMRARE</td>
<td>0.309</td>
<td>5.1</td>
</tr>
<tr>
<td>FAMBOTH</td>
<td>0.334</td>
<td>5.2</td>
</tr>
<tr>
<td>RARESLO</td>
<td>–0.08</td>
<td>–1.5</td>
</tr>
<tr>
<td>RARECOV</td>
<td>0.645</td>
<td>8.1</td>
</tr>
<tr>
<td>HABREST</td>
<td>0.243</td>
<td>4.7</td>
</tr>
<tr>
<td>HABCREAT</td>
<td>0.253</td>
<td>4.3</td>
</tr>
<tr>
<td>ECOHUMA</td>
<td>0.359</td>
<td>5.9</td>
</tr>
<tr>
<td>ECOALL</td>
<td>0.064</td>
<td>1.0</td>
</tr>
<tr>
<td>PRICE</td>
<td>–0.003</td>
<td>–15.3</td>
</tr>
<tr>
<td>Pseudo R² (%)</td>
<td>19</td>
<td></td>
</tr>
<tr>
<td>n (individuals)</td>
<td>398</td>
<td></td>
</tr>
</tbody>
</table>
Northumberland group only cared about ecosystem processes that seemed to directly impact on their well-being. The Northumberland group also had a negative value for RARESLOW, but since this estimate is insignificant, this is unimportant.

The statistical equivalence of the parameter estimates of the two models can be compared using a likelihood ratio test. The probability value for this test is < 0.01, indicating that the models are different. In other words, the valuation of biodiversity attributes varies significantly between the two samples, so that simple benefits transfer of valuation functions is rejected.

Table 4.4 (a and b) shows the implicit prices estimated from the logit model results in Table 4.3. These implicit prices show the ‘marginal’ WTP on average of moving from one level – the excluded level, which in our case is always the worst case, ‘Do nothing’ level – to a higher level. For example, the value of £35.65 for FAMRARE for Cambridgeshire means that people were on average willing to pay £35.65 extra per year in higher taxes to move from continued decline in familiar species to a situation where rare, familiar species are protected from further decline. These are ‘ceteris paribus’ values, so should be treated with care in a cost–benefit context. We can see from Table 4.4 that a scale effect is present in almost all cases for Cambridgeshire, meaning that higher levels of protection are valued more highly for each attribute, with the exception of the odd result on RARESLOW, and in the case of ECOALL, where the value of protecting only directly relevant ecosystem processes is higher than that of protecting all. The highest benefits in per person terms come from ensuring the recovery of rare, unfamiliar species. For Northumberland, the implicit prices for RARESLOW and ECOALL are omitted, since the parameter estimates were not significantly different from zero. Furthermore, there was little evidence that the Northumberland sample considered the scale effects between the levels of the familiar species and for habitat quality attribute. Highest WTP is associated with ensuring the recovery of rare, unfamiliar species – the same result as for Cambridgeshire.

Comparison of Main CE Study and Valuation Workshop

In Table 4.5 models are presented for the choice exercises during the valuation workshops. Participants made two sets of choices, one near the outset, after receiving the same information as the main survey participants (Discuss1), and one near the end, having had a chance to discuss the issues further (Discuss2). Neither model fits very well due to the small sample size, but we can note that the number of significant variables increases from three to seven between the two treatments, whilst the overall fit also improves. In
other words, a learning effect seems to be present. Looking at the model for Discuss2, we see that it compares quite well with the main survey CE results for Northumberland, with only RARESLOW having a negative sign, and with ECOALL still being insignificant. The workshop choices also show habitat restoration to have an insignificant effect on utility. Implicit prices are also very similar, with a complete recovery of rare, unfamiliar species having the highest welfare gain. Finally, we note that a formal LR test shows that the parameters of the main survey CE model for Northumberland are not significantly different than either the Discuss1 or Discuss2 models for
the valuation workshops. In this sense, the valuation workshops provide
support for the main survey choice experiment results.

Table 4.5  Choice experiment results: workshops versus main survey,
Northumberland

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Main Survey Parameter</th>
<th>Workshop Discuss1 Parameter</th>
<th>Workshop Discuss2 Parameter</th>
<th>t-statistic</th>
<th>t-statistic</th>
<th>t-statistic</th>
</tr>
</thead>
<tbody>
<tr>
<td>FAMRARE</td>
<td>0.309</td>
<td>0.172</td>
<td>0.327</td>
<td>5.1</td>
<td>1.1</td>
<td>2.0</td>
</tr>
<tr>
<td>FAMBOTH</td>
<td>0.334</td>
<td>0.257</td>
<td>0.343</td>
<td>5.2</td>
<td>1.6</td>
<td>2.0</td>
</tr>
<tr>
<td>RARESLO</td>
<td>–0.080</td>
<td>–0.028</td>
<td>–0.316</td>
<td>–1.5</td>
<td>–0.2</td>
<td>–2.1</td>
</tr>
<tr>
<td>RARECOV</td>
<td>0.645</td>
<td>0.166</td>
<td>0.654</td>
<td>8.1</td>
<td>0.8</td>
<td>3.0</td>
</tr>
<tr>
<td>HABREST</td>
<td>0.243</td>
<td>0.093</td>
<td>0.149</td>
<td>4.7</td>
<td>0.7</td>
<td>1.1</td>
</tr>
<tr>
<td>HABCREAT</td>
<td>0.253</td>
<td>0.323</td>
<td>0.332</td>
<td>4.3</td>
<td>2.0</td>
<td>2.0</td>
</tr>
<tr>
<td>ECOHUMA</td>
<td>0.359</td>
<td>0.386</td>
<td>0.319</td>
<td>5.9</td>
<td>2.4</td>
<td>2.0</td>
</tr>
<tr>
<td>ECOALL</td>
<td>0.064</td>
<td>0.116</td>
<td>0.211</td>
<td>1.0</td>
<td>0.6</td>
<td>1.2</td>
</tr>
<tr>
<td>TAX</td>
<td>–0.003</td>
<td>–0.004</td>
<td>–0.004</td>
<td>–15.3</td>
<td>–6.2</td>
<td>–5.8</td>
</tr>
<tr>
<td>A_OPTA</td>
<td>–0.012</td>
<td>0.823</td>
<td>–0.295</td>
<td>–0.1</td>
<td>2.3</td>
<td>–0.8</td>
</tr>
<tr>
<td>A_OPTB</td>
<td>–0.205</td>
<td>0.894</td>
<td>–0.081</td>
<td>–1.5</td>
<td>2.4</td>
<td>–0.2</td>
</tr>
<tr>
<td>–2*lnL</td>
<td>3172.6</td>
<td>417.4</td>
<td>440.7</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>p value</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pseudo R²</td>
<td>19.2%</td>
<td>16.7%</td>
<td>18.7%</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>n (individuals)</td>
<td>398</td>
<td>53</td>
<td>53</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Contingent Valuation

Table 4.6a gives summary measures for the WTP bids, comparing the
Cambridgeshire results with those from Northumberland, and with the
total, pooled sample across both studies. We also consider all conservation
scenarios merged together. Table 4.6a thus shows what people in
Cambridgeshire/Northumberland are WTP for any policy amongst the
options presented for increasing biodiversity. As may be seen, about one-
third of respondents had a WTP of zero, in other words, did not value
these increases in biodiversity (note that protest responses were coded as
zeros; these mean WTP figures are thus conservative estimates). Mean
WTP is higher for Cambridgeshire respondents (£58.87) than for those
from Northumberland (£42.47): this difference is statistically significant at
the 95 per cent level. Median WTP is considerably less than mean WTP in
all cases, illustrating a common finding in CV studies.
Table 4.6a  Summary WTP measures: Cambridgeshire and Northumberland

<table>
<thead>
<tr>
<th>SITE</th>
<th>n</th>
<th>Mean (£)</th>
<th>Standard Error (£)</th>
<th>95% Confidence Interval</th>
<th>95% Trimmed Mean (£)</th>
<th>Median (£)</th>
<th>Percentage with WTP = 0</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cambridgeshire</td>
<td>341</td>
<td>£58.87</td>
<td>£5.84</td>
<td>£47.38→£70.36</td>
<td>£42.84</td>
<td>£20.00</td>
<td>32.3</td>
</tr>
<tr>
<td>Northumberland</td>
<td>395</td>
<td>£42.47</td>
<td>£3.97</td>
<td>£34.67→£50.27</td>
<td>£30.09</td>
<td>£10.00</td>
<td>35.9</td>
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<tr>
<td>Both</td>
<td>736</td>
<td>£50.07</td>
<td>£3.45</td>
<td>£43.29→£56.85</td>
<td>£35.81</td>
<td>£20.00</td>
<td>34.2</td>
</tr>
</tbody>
</table>

Note:  \( t \)-test for difference in means: \( t = 2.3 \) and \( p = 0.02 \).

Table 4.6b  Summary WTP measures by type of programme: Cambridgeshire only

<table>
<thead>
<tr>
<th>SITE</th>
<th>n</th>
<th>Mean (£)</th>
<th>Standard Error (£)</th>
<th>95% Confidence Interval</th>
<th>95% Trimmed Mean (£)</th>
<th>Median (£)</th>
<th>Percentage with WTP = 0</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agri-environmental schemes</td>
<td>124</td>
<td>£74.27</td>
<td>£13.26</td>
<td>£48.03→£100.51</td>
<td>£53.28</td>
<td>£24.00</td>
<td>29.8</td>
</tr>
<tr>
<td>Habitat creation schemes</td>
<td>107</td>
<td>£54.97</td>
<td>£6.56</td>
<td>£41.96→£67.98</td>
<td>£48.42</td>
<td>£24.00</td>
<td>29.9</td>
</tr>
<tr>
<td>Development loss</td>
<td>110</td>
<td>£45.30</td>
<td>£7.82</td>
<td>£29.80→£60.79</td>
<td>£31.26</td>
<td>£16.00</td>
<td>37.3</td>
</tr>
<tr>
<td>ALL</td>
<td>341</td>
<td>£58.87</td>
<td>£5.84</td>
<td>£47.38→£70.36</td>
<td>£42.84</td>
<td>£20.00</td>
<td>32.3</td>
</tr>
</tbody>
</table>

Note:  \( f \)-test for difference in means: \( f = 2.2 \) and \( p = 0.11 \).

Table 4.6c  Summary WTP measures by type of programme: Northumberland only

<table>
<thead>
<tr>
<th>SITE</th>
<th>n</th>
<th>Mean (£)</th>
<th>Standard Error (£)</th>
<th>95% Confidence Interval</th>
<th>95% Trimmed Mean (£)</th>
<th>Median (£)</th>
<th>Percentage with WTP = 0</th>
</tr>
</thead>
<tbody>
<tr>
<td>Habitat creation schemes</td>
<td>209</td>
<td>£47.49</td>
<td>£5.98</td>
<td>£35.70→£59.27</td>
<td>£34.35</td>
<td>£12.00</td>
<td>27.8</td>
</tr>
<tr>
<td>Development loss</td>
<td>186</td>
<td>£36.84</td>
<td>£5.07</td>
<td>£26.82→£46.85</td>
<td>£25.29</td>
<td>£3.00</td>
<td>46.8</td>
</tr>
<tr>
<td>ALL</td>
<td>395</td>
<td>£42.47</td>
<td>£3.97</td>
<td>£34.67→£50.27</td>
<td>£30.09</td>
<td>£10.00</td>
<td>35.9</td>
</tr>
</tbody>
</table>

Note:  \( t \)-test for difference in means: \( t = 1.4 \) and \( p = 0.18 \).
Table 4.6b presents results for Cambridgeshire only, and shows how WTP varies by scheme. WTP is highest for agri-environmental schemes, and lowest for preventing development loss. Habitat re-creation is valued between these two. This is of general interest, since the theory of loss aversion suggests that losses are often valued more than gains. However, these changes are not symmetrical in our case. What is more, these mean values are not statistically different from each other at 95 per cent \( (p = 0.11) \). Note that sample sizes in each of the three treatments are quite small \( (n = 107, 110, 124) \). In Table 4.6c, this analysis is repeated for Northumberland, where WTP across the two scenarios used (habitat re-creation and development loss) is compared. WTP is higher for the former, but again this difference is not significant \( (p = 0.18) \).

The conclusions we can draw from Tables 4.6a to 4.6c are that the value people place on increases in biodiversity is positive, and significantly different from zero. This value is higher in the Cambridgeshire sample than in the Northumberland sample. However, in no case does WTP differ across schemes to a significant degree. It thus appears that people care about increasing biodiversity, but not how this is achieved.

With regard to benefits transfer, a simple transfer of mean WTP values across geographic areas is rejected, since the mean values are significantly different from each other at the 95 per cent level of confidence. We also tested whether the bid curve underlying the Cambridgeshire data is different from that underlying the Northumberland data, using a Chow test. This test shows that the inverse demand curves (bid curves) are statistically different: it would be incorrect to estimate WTP in either region using the bid curve parameters from the other.

Comparing the CV main survey results with the valuation workshops (Table 4.7a) we find that WTP was higher in the valuation workshops than in the main survey (£50.33 versus £42.47, across both schemes); but this difference is not statistically significant \( (p = 0.40) \). Interestingly, the variance in the workshop sample is higher than in the main survey: this implies that the ‘opportunity to discuss and reflect’, and the social context, in the workshops encourage a greater spread of values compared with the survey. Again, however, there is a big difference in sample size. The small sample size for the valuation workshops \( (n = 106) \) reflects the high costs of collecting valuation data in this way. Table 4.7b shows a comparison of values across the two schemes for the valuation workshops. WTP is now statistically different between habitat creation and development loss, with that for the former being more than double the latter. However, the variance of bids for habitat creation is high, as may be seen in the very wide confidence interval. Median WTP is four times higher for the habitat creation scenario than the development loss case.
Table 4.7a  Summary WTP measures: valuation workshops versus main survey

<table>
<thead>
<tr>
<th>Scheme</th>
<th>n</th>
<th>Mean</th>
<th>Standard Error</th>
<th>95% Confidence Interval</th>
<th>95% Trimmed Mean</th>
<th>Median</th>
<th>Percentage with WTP = 0</th>
</tr>
</thead>
<tbody>
<tr>
<td>Valuation workshops</td>
<td>106</td>
<td>£50.33</td>
<td>£9.08</td>
<td>£32.32↔£68.34</td>
<td>£35.22</td>
<td>£10.00</td>
<td>33.0</td>
</tr>
<tr>
<td>Main survey</td>
<td>395</td>
<td>£42.47</td>
<td>£3.97</td>
<td>£34.67↔£50.27</td>
<td>£30.09</td>
<td>£10.00</td>
<td>35.9</td>
</tr>
<tr>
<td>Pooled</td>
<td>501</td>
<td>£44.14</td>
<td>£3.67</td>
<td>£36.92↔£51.34</td>
<td>£30.96</td>
<td>£10.00</td>
<td>35.3</td>
</tr>
</tbody>
</table>

Note:  $t$-test for difference in means between main survey and workshops: $t = 0.8$ and $p = 0.40$.

Table 4.7b  Summary WTP measures: valuation workshop

<table>
<thead>
<tr>
<th>Scheme</th>
<th>n</th>
<th>Mean</th>
<th>Standard Error</th>
<th>95% Confidence Interval</th>
<th>95% Trimmed Mean</th>
<th>Median</th>
<th>Percentage with WTP = 0</th>
</tr>
</thead>
<tbody>
<tr>
<td>Habitat creation</td>
<td>53</td>
<td>£68.72</td>
<td>£15.89</td>
<td>£36.83↔£100.60</td>
<td>£52.95</td>
<td>£20.00</td>
<td>28.3</td>
</tr>
<tr>
<td>Development loss</td>
<td>53</td>
<td>£31.94</td>
<td>£8.23</td>
<td>£15.42↔£48.46</td>
<td>£22.57</td>
<td>£5.00</td>
<td>37.7</td>
</tr>
<tr>
<td>Pooled</td>
<td>106</td>
<td>£50.33</td>
<td>£9.08</td>
<td>£32.22↔£68.341</td>
<td>£35.22</td>
<td>£10.00</td>
<td>33.0</td>
</tr>
</tbody>
</table>

Note:  $t$-test for difference in means between schemes: $t = 2.1$ and $p = 0.04$. 


DISCUSSION

Two key questions that can be asked of this data are: is there evidence that the general public are willing to pay additional taxes to support biodiversity conservation, and if so, then why? Here we are first interested in whether respondents chose a biodiversity enhancement policy option (Options A or B) as opposed to the ‘Do nothing’ option. In fact, only 15 per cent of respondents chose the ‘Do nothing’ option. In other words, these respondents were not willing to pay additional taxes to achieve biodiversity enhancements. Eighty-five per cent of the choices made by CE respondents were for choice options A or B. This demonstrates that the majority of respondents were willing to pay some amount of additional taxation to attain biodiversity enhancements. This finding is backed up by the contingent valuation results, where positive WTP values existed for all three policy options to either increase biodiversity or stop it from declining further. Approximately one-third of respondents in the contingent valuation were unwilling to pay for biodiversity enhancement, compared with 15 per cent in the choice experiment.

In terms of the reasons given by CE survey respondents for making these choices, over half of the respondents (52.6 per cent) stated that they considered that the biodiversity improvements in policy options A or B were ‘good value for my money’. Three per cent of respondents stating a zero bid stated that the biodiversity improvements were not good use of their money, while 5 per cent stated that they already contribute to environmental causes. Protest votes included ‘I do not think that increases in taxation should be used to fund biodiversity improvements’ (6.5 per cent) and ‘The costs of biodiversity improvement should be paid for by those who degrade biodiversity’ (14.2 per cent).

Another question our research enables us to address is: what aspects of biodiversity protection policy do the public value most? Examining the implicit prices in Table 4.4 provides some answers. In the choice model, familiar species attained positive and significant implicit prices. In Cambridgeshire, scale effects were evident in that the implicit price for the protection of both rare and common familiar species (£93.49) was significantly higher than the protection of only the rare familiar species (£35.65). This was not, however, the case in the Northumberland sample, where the two levels of protection had similar implicit prices (£90.59 and £97.71 respectively for the protection of rare only and rare and common familiar species). In conclusion, evidence from the choice experiment suggests that the public do support policies that target rare familiar species of wildlife, but the evidence is less clear for the contribution of policies that target common familiar species.
The second attribute addressed in the choice experiment related to rare unfamiliar species of wildlife. Two levels of provision were addressed. RARESLO, which aimed to ‘slow down the rate of the decline in the populations of rare unfamiliar species. … however, it is likely that some rare unfamiliar species may still become locally and nationally extinct’. The second level RARECOV aimed to ‘stop decline and ensure recovery of rare unfamiliar species’. The findings for the RARESLO attribute level were interesting since it was found to be negative in the Cambridgeshire sample (indicating that negative utility would be gained from a slowdown in the decline of the population of rare unfamiliar species – which was not predicted), while the attribute level was not significant in the Northumberland CE model. The implications of this finding was that it appears that the public are unwilling to support policies that simply delay the time it takes for such species to become extinct. This conclusion was further emphasized by the fact that highest implicit prices were attained from the RARECOV attribute level. Thus, the policy implication of these findings is that the public appear to only support policies that aim to achieve recovery of the populations of rare species, rather than those that simply attempt to slow down decline in population numbers. A further implication of these findings relates to the fact that survey respondents were told that they were unlikely to ever see these rare, unfamiliar species. Thus, these values can be considered to represent passive use values.

The habitat quality attribute was included to assess whether the public valued the restoration of existing habitats (HABREST) or the re-creation of new habitats on farmland (HABRECO). Both attribute levels were found to be positive and significant in the two case study locations. In Cambridgeshire, the value for habitat restoration (£34.40) was just over half that for habitat re-creation (£61.36), while similar values were attained for both levels in Northumberland (£71.15 and £74.01 respectively). The reason for this difference may be similar to those stated above for familiar species. In other words, the Cambridgeshire respondents may have been more able to distinguish between attribute levels and/or the Cambridgeshire sample may have considered that there were very few existing habitats within Cambridgeshire that would benefit from restoration. Again, evidence was not collected to identify which, if any, of these reasons could be verified. However, there was evidence that the public would support policies that aimed to protect and enhance habitats, although the value of the implicit prices were found to be slightly lower than those found for the two species attributes.

Finally, the ecosystem processes attribute was included to assess whether the public valued ecosystems that only had a direct impact on humans (ECOHUMA) and all ecosystem processes include those that did not directly affect humans (ECOALL). The ecosystems processes that had direct
impacts on humans were found to be both positive and significant. However, the ECOALL attribute level was not significant in the Northumberland model and was lower than the ECOHUMA attribute level in the Cambridge sample. It would thus appear that survey respondents 'cared' about ecosystem functions that affect humans, but were less interested in the other ecosystem processes.

Another question that can be posed is how valid the value estimates obtained from the choice experiment are. Two ways of answering this question are to, first, examine the theoretical validity of our results. As already noted, the parameter signs in the CE equations were overwhelmingly in accord with a priori expectations. Second, a convergent validity test can be made by comparing the main survey CE results with those from the valuation workshops. Again, as already noted, we cannot reject the hypothesis that the choice models obtained from both are significantly different from each other, which provides some evidence as to the stability of the results obtained.

Finally, we can ask how transferable the CE results are between the two case study areas. A likelihood ratio test was used to compare the beta values (parameter estimates) between the Cambridgeshire and Northumberland models (Table 4.3). The p value for this test is < 0.01, indicating that the two models were different. Based on this evidence we would reject the transfer of the indirect utility functions between the two areas. Another test for benefits transfer undertaken on the choice experiment data was to test whether the implicit prices for each attribute were significantly different from each other between the Cambridgeshire and Northumberland samples. Evidence from Table 4.4 indicates that the 95 per cent confidence intervals for implicit prices do overlap between the models in two out of six cases – for FAMBOTH and HABCREATE. However, this is largely due to the large standard errors on the implicit prices. So again, there is little evidence in support of benefits transfer in the choice experiment data.

CONCLUSIONS

Policy-makers may benefit from information on the economic value of biodiversity protection, but also on which aspects of biodiversity are most valued by taxpayers. Stated preference methods such as choice experiments and contingent valuation can provide these type of value estimates, but implementing these methods is difficult in this particular case since the general public have a rather low level of understanding of what biodiversity is and why it matters. In this study we make use of a novel way of conveying information to respondents, information that is consistent with ecological
understanding of what aspects of biodiversity might be considered. We then use choice experiments to estimate the relative values people place on these attributes, and contingent valuation to gauge the strength of preferences for specific real policies that protect biodiversity.

How policy-makers might choose to use such information is something we have not addressed here. But economists would argue that, in a world of scarce resources and conflicting demands, some information is better than no information on the relative values society places on biodiversity conservation.

REFERENCES


5. Implications of declining discount rates for UK climate change policy

Ben Groom, Cameron Hepburn, Phoebe Koundouri and David Pearce

INTRODUCTION

Discussions about applied cost–benefit analysis (CBA) are incomplete without the thorny issue of discounting emerging at some point. Indeed, since the calculation of net present value (NPV), and hence the efficiency of a project or policy, hinges so crucially upon the level of the discount rate applied across time, the analysis of time preference and discounting has become an active area of research in its own right. Nowhere is this debate more hotly contended than when CBA is used to evaluate projects with impacts that extend into the far distant future such as biodiversity conservation, nuclear power and, of course, climate change. This chapter aims to review some of the more recent contributions to this debate and in particular, the theory that underpins recent calls for the use of declining discount rates (DDRs). We then discuss how a schedule of DDRs can be estimated and illustrate their impact upon two topical policy questions: climate change and nuclear power.

Economists and others have argued at length over which of several potential discount rates should be used as the Social Discount Rate (SDR) (e.g. Marglin, 1963; Baumol, 1968; Lind, 1982; Portney and Weynant, 1999). Several candidates exist, the most widely recognized of which are 1. the social rate of return on investment and 2. the rate at which society values consumption at different points of time (the social rate of time preference), henceforth $r$ and $\delta$ respectively. The distinction between these discount rates is most important in the second-best world in which distortions to the economy, such as corporate and personal taxes or environmental externalities, prevent these rates from being equalized. The choice of SDR is inherently complicated in such situations.¹ Common practice in CBA has been that, however one chooses the SDR, the relative weights applied
to all adjacent time periods would be invariant across the time horizon considered.

One criticism of discounting is that it militates against solutions to the long-run environmental problems mentioned above. Some policy questions and projects need to be evaluated over a time horizon of several hundred years. With a constant rate, the costs and benefits accruing to generations in the distant future appear relatively unimportant in present value terms. Hence decisions made today on the basis of CBA appear to tyrannize future generations and in extreme cases leave them exposed to potentially catastrophic consequences. Such risks can either result from current actions, where future costs carry no weight, for example, nuclear plant decommissioning, or from current inaction, where the future benefits carry no weight, for example, climate change. Hence the question arises: what is the appropriate procedure for such long time horizons? There is wide agreement that discounting at a constant positive rate in these circumstances is problematic, irrespective of the particular discount rate employed. These intergenerational issues associated with discounting have puzzled generations of economists. Pigou (1932) referred to the apparent myopia of exponential discounting with regard to future welfare as a ‘defective telescopic faculty’. More recently Weitzman (1998) summarizes this puzzle succinctly when he states: ‘to think about the distant future in terms of standard discounting is to have an uneasy intuitive feeling that something is wrong, somewhere’.

Discounting also appears to be contrary to the widely supported goal of ‘sustainability’, which by most definitions implies that policies and investments now must have due regard for the need to secure sustained increases in per capita welfare for future generations (World Commission on Environment and Development, 1987; Atkinson et al., 1997). Also, by attaching little weight to future welfare, conventional discounting appears to ignore any notion of intergenerational equity. So, in short, the correct procedure in these circumstances is not immediately obvious.

A recently proposed solution to this problem is to use a discount rate that declines with time, according to some predetermined trajectory, thus raising the weight attached to the welfare of future generations. It is immediately obvious that using a DDR would make an important contribution towards meeting the goal of sustainable development. So, what formal justifications exist for using a DDR and what is the optimal trajectory of the decline?

As far as the former issue is concerned, there are a number of rationales that effectively assume a deterministic world. For example, Dasgupta (2001) shows that DDRs can arise as a result of known changes in the growth rate or the consumption smoothing/risk aversion parameter. A seminal contribution by Fisher and Krutilla (1975) was the first to suggest that the evolution of willingness to pay for the environment could also be captured...
Implications of declining discount rates for UK climate change policy

by the discount rate, a theme also touched upon by Weitzman (1994) in the presence of environmental externalities. The strengths and weaknesses of these rationales have been well documented (e.g. Arrow et al., 1995; Horowitz, 2002).

Additional motivations emerge once uncertainty is considered. Uncertainty of the discount rate itself provides a simple and intuitive approach in a risk-neutral environment (Weitzman, 1998, 2001). In the presence of uncertain growth Gollier (2002a, 2002b) shows that the shape of the yield curve, that is, the term structure of the interest rate, depends upon preferences for risk and prudence, and higher order moments of the utility function. DDRs also emerge from the specification of a ‘sustainable’ welfare function à la Chichilnisky (1997) and Li and Löfgren (2000). Lastly, there is considerable empirical and experimental evidence to show that individuals are frequently hyperbolic discounters (e.g. Loewenstein and Prelec 1992; Frederick et al., 2002 – see Groom et al. (2005) for a review). Henderson and Bateman (1995) argue that this is sufficient reason for similar discounting schedules to be employed in social decision-making.

Once a rationale for DDR has been subscribed to, implementation requires the practitioner to identify a particular set of parameters, that is, an answer to the second question raised: what trajectory should a DDR follow? The required parameters for determining the time invariant discount rate in the deterministic case have been discussed extensively elsewhere (see, for example, Pearce and Ulph, 1999) and are well understood. In this chapter, we focus upon the application of the more recent contributions. The next section gives the background literature, the second section discusses the implications of declining discount rates by using a case study on the climate change policy in the UK and the third section concludes.

BACKGROUND LITERATURE

Uncertainty and DDRs

In the case of Gollier (2002a, 2002b, 2004b) and Weitzman (1998) it is uncertainty that drives DDRs, with regard to future growth and the discount rate respectively. One thing common to both of these approaches is that the eventual schedule of discount rates is highly dependent upon the characterization of the background uncertainty and hence the question of implementation is one of characterizing the uncertainty of the uncertain variables in some coherent way. However, of these two approaches it is Weitzman (1998) that has proven to be more amenable to implementation mainly because the informational requirements stop at the characterization
of uncertainty, and do not extend to specific attributes of future generations’ risk preferences as would be unavoidable in the case of Gollier.\textsuperscript{2}

Weitzman’s certainty equivalent discount rate (CER) is a summary statistic of the distribution of the discount rate and the level and behaviour over time of this statistic is clearly dependent upon the features (static and dynamic) of the associated probability distribution. The two applications that exist have taken different approaches stemming from different interpretations of uncertainty. Weitzman (2001) defines uncertainty by the current lack of consensus on the appropriate discount rate for the very long term. His survey of professional economists results in a gamma probability distribution for the discount rate, which leads to the so-called ‘gamma discounting’ approach, a version of which can also be found in Sozou (1998).

In particular, Weitzman uses certainty equivalent analysis for risk-neutral agents and defines the certainty equivalent discount factor (CEDF) as the expectation of the discount factor. From this he derives the certainty equivalent discount rate (CER). Supposing that each of \( n \) potential discount rates, \( r_j (j = 1, 2, \ldots, n) \), is realized with probability \( p_j \), such that

\[
\sum_j p_j = 1 \quad \text{and} \quad r_j \in [r_{\min}, r_{\max}].
\]

Defining the discount factor for a particular scenario as:

\[
a_j (t) = \exp \left( -\int_0^t r(s) ds \right), \tag{5.1}
\]

the certainty equivalent discount factor for a risk-neutral agent is defined as:

\[
A(t) = E \left[ \exp \left( -\int_0^t \tilde{r}(s) ds \right) \right] = \sum_j p_j a_j (t) \tag{5.2}
\]

From this it is possible to define both the average and marginal certainty equivalent discount rates at time \( t \), \( r_a^{CE} \) and \( r_m^{CE} \) respectively:

\[
\exp (-r_a^{CE} (t)) = A(t) \Rightarrow r_a^{CE} (t) = -\frac{1}{t} \ln (A(t)) \tag{5.3}
\]

\[
r_m^{CE} (t) = -\frac{\partial}{\partial t} \ln A(t) \tag{5.4}
\]
The former is the rate of discount that if applied in every period from time 0 to time \( t \) would yield the same value as the expected discount factor at time \( t \). The latter is the instantaneous, period-to-period rate. Weitzman (1998) works with equation (5.4) noting that at the limit, as \( t \to \infty \), they are precisely the same. Importantly he shows that \( r^{CE}_m \) declines continuously and monotonically over time and that its limit as \( t \to \infty \) is \( r^{CE}_{min} \). Gollier (2002b) provides a justification for the definition of \( r^{CE}_a \) in terms of arbitrage. He explains that an arbitrage exists if, prior to the realization of \( r \), there is potential for arbitrage if equation (5.3) does not hold. Hence, the certainty equivalent discount rate is the equilibrium socially efficient rate for risk-neutral agents prior to the realization of \( r \).

The mechanics of the result are shown in Appendix 5.1. However, the intuition is as follows. In calculating the weighted average that is the certainty equivalent, each potential realization of the discount rate is weighted by a term that contains the discount factor associated with that scenario. In scenarios with higher discount rates the discount factors decline more rapidly to zero. As such, the weight placed on scenarios with high discount rates itself declines with time, until the only relevant scenario is that with the lowest conceivable interest rate. In effect, the power of exponential discounting reduces the importance of future scenarios with high discount rates to zero, since in the ex ante equilibrium the certainty equivalent rate of discount must equal the socially efficient discount rate in all periods of time, this results in an SDR that declines over time.

**Numerical Example of Weitzman’s CER**

Suppose that there are two potential realizations of the discount rate \( r_1 \) and \( r_2 \) with associated probabilities \( p_1 \) and \( p_2 \). Using the definitions (5.2) and (5.4) we obtain the certainty equivalent discount factor and rate at time \( t \) as weighted averages:

\[
A(t) = p_1 \exp(-r_1 t) + p_2 \exp(-r_2 t) = p_1 a_1(t) + p_2 a_2(t) = \sum_j p_j a_j(t) \quad (5.5)
\]

\[
r^{CE}_m(t) = -\frac{\partial}{\partial t} \frac{A(t)}{A(t)} = \frac{r_1 p_1 a_1(t) + r_2 p_2 a_2(t)}{p_1 a_1(t) + p_2 a_2(t)}
= w_1(t)r_1 + w_2(t)r_2 = \sum_j w_j(t)r_j \quad (5.6)
\]
where the weights are

\[ w_j(t) = p_j a_j / \sum p_j a_j, \text{where} \sum w_j(t) = 1. \]

This formula is used for \( r_m^{CE} \) in Table 5.1 below. This formula for \( r_a^{CE} \) is:

\[ r_a^{CE} = -\frac{1}{t} \ln \left[ p_1 \exp(-r_1 t) + p_2 \exp(-r_2 t) \right] \quad (5.7) \]

Using (5.6) and the fact that:

\[ \dot{w}_j(t) = \frac{p_j a_j(t)}{\sum p_j a_j(t)} \cdot \frac{\sum p_j a_j(t) - r_j p_j a_j(t)}{\sum p_j a_j(t)} = w_j(t) \left( r_m^{CE} - r_j \right) \quad (5.8) \]

the derivative of \( r_m^{CE} \) with respect to time then becomes:

\[ \frac{d}{dt} r_m^{CE} = -\left[ w_1 (r_m^{CE} - r_1) r_1 + w_2 (r_m^{CE} - r_2) r_2 \right] = -\sum \omega_j(t) \left( r_m^{CE} - r_j \right)^2 \quad (5.9) \]

which is clearly negative.

Table 5.1 shows the resulting schedule of marginal and average discount rates over continuous time assuming that \((r_1, r_2) = (5\%, 2\%)\) and \((p_1, p_2) = (0.5, 0.5)\). Table 5.1 reflects the aspects of the certainty equivalent discount rate described above. Both the average and the marginal certainty equivalent rates are declining monotonically through time while approaching the lowest possible realization in the long run: \(r_{min} = 2\%\).

**Table 5.1 Numerical example of Weitzman's declining certainty equivalent discount rate**

<table>
<thead>
<tr>
<th>Interest Rate Scenarios</th>
<th>Discount Factors in Period t</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>10</td>
</tr>
<tr>
<td>2% ((p_1 = 0.5))</td>
<td>0.82</td>
</tr>
<tr>
<td>5% ((p_2 = 0.5))</td>
<td>0.61</td>
</tr>
<tr>
<td>Certainty equivalent discount factor: (A(t))</td>
<td>0.72</td>
</tr>
<tr>
<td>Average CER: (r_a^{CE}(t) %))</td>
<td>3.38</td>
</tr>
<tr>
<td>Marginal CER: (r_m^{CE}(t) %))</td>
<td>3.28</td>
</tr>
</tbody>
</table>
More recently, Newell and Pizer (2003) consider the interest rate as a stochastic process, that is, there is uncertainty about future interest rates. Newell and Pizer characterize this uncertainty using time series econometric modelling of the auto-correlation process of interest rates. The estimated model is used to forecast future rates based upon their behaviour in the past. From these forecasts they derive numerical solutions for the CER. In doing so they are also able to provide a test of another assumption important to the Weitzman (1998) result, namely the presence of persistence of discount rates over time. They compare the discount rates modelled as a mean reversion process to a random walk model, and find support for the latter. The practical implications of implementing the declining discount rates that result are significant.

When applied to global warming damages, the present value of damages from carbon emissions increases by 82 per cent, compared with the same damages evaluated at the constant treasury rate of 4 per cent. In monetary terms this translates into an increase in the benefits of carbon mitigation from $5.7/tonne of carbon, to $10.4/tonne of carbon. However, using UK interest rate data Groom et al. (2006) provide a more thorough econometric analysis of the extent to which uncertainty in the future causes DDRs and find that model specification is crucial to the analysis, not least because of the distributional assumptions contained therein. Indeed, they find little evidence of the persistence noted by Newell and Pizer, suggesting that in the UK context the effect of future uncertainty upon the valuation of global warming damages is minimal.

The rationale for declining discount rates provided by Gollier (2002a, 2002b) is perhaps the most theoretically rigorous of all the contributions. But determination of the trajectory requires very specific information concerning the preferences of current generations at the very least, and, in the long run, the preferences of future generations. These parameters include the aversion to consumption fluctuations over time, the pure time preference rate, and the degree of relative risk aversion. For the case with zero recession, restrictions on the fourth and fifth derivatives of the utility function become necessary. In addition, the probability distribution of growth needs to be characterized in some way. Clearly, the informational requirements of the Gollier approach could be daunting.

**Intergenerational Equity and Sustainability**

Then we have the contributions that take sustainable growth and intergenerational equity as their departure point. The main focus of the discussion is on the important contributions of Chichilnisky (1996, 1997)
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and Li and Löfgren (2000), both of which explicitly introduce the notions of intergenerational equity and sustainability. Each paper models optimal sustainable economic growth and each is concerned with deriving the welfare effects of growth paths that are sustainable in the sense that they satisfy particular axioms with regard to intergenerational equity. The axioms employed imply social preferences that are ‘sustainable’ or ‘intertemporally equitable’. Welfare is measured in terms of the utility of a social planner and, with utility as their numeraire, the discussion of discount rates concerns the utility discount rate, $\rho$, rather than the social rate of time preference, $\delta$ or the social rate of return, $r$. Both contributions show that a declining utility discount rate is consistent with a rule whereby current (future) generations must always take into account the well-being of future (current) generations. That is, there must be no ‘dictatorship’ of one generation over another.

Chichilnisky (1997) introduces two axioms for sustainable development. She also characterizes the preferences that satisfy these axioms. The axioms require that the ranking of alternative consumption paths is sensitive not only to what happens in the present and immediate future, but also to what happens in the very long run. Sensitivity to the present means that there is no date before which events are given zero weight. Sensitivity to the long-run future means that there is no date where changes after that date do not matter, in the sense of affecting the ranking. Chichilnisky’s criterion can be represented in the following objective function:

$$\max_{c,s} \pi \int_0^\infty u(c(t), s(t)) \exp(-\rho t) dt + (1 - \pi) \lim_{t \to \infty} u(c(t), s(t))$$

(5.10)

Instantaneous utility $u(\cdot)$ is a function of consumption $(c)$ and the resource stock $(s)$ at each time period $(t)$, while $\exp(-\rho t)$ is the conventional exponential utility discount factor. $u(\cdot)$ is assumed to be the same for all dates so that generations are assumed to be the same in the way they rank alternatives. Intuitively, the limit term reflects the sustainable utility level attained by a particular policy decision regarding $c(t)$ and $s(t)$. This can be interpreted as the well-being of generations in the far distant future. Chichilnisky’s approach is a mixture of the two approaches: a generalization of the discounted utilitarian approach, mixed with an approach that ranks paths of consumption and natural resource use according to their long-run characteristics, or sustainable utility levels. This criterion can be applied under the two main axioms regarding the ranking of alternative utility paths. Notice that $\pi \in [0,1]$, can be interpreted as the weight that the decision-maker applies to each component of the criterion, with $\pi$ providing the weight given to the present generation, and $(1 - \pi)$ representing the weight placed upon the future generation.
Implications of declining discount rates for UK climate change policy

In contrast to Chichilnisky (1997) who treats present and future generations as separate entities in the objective function of the decision-maker, Li and Lofgren (2000) treat the future differently. Li and Lofgren assume society consists of two individuals, a utilitarian and a conservationist, each of which makes decisions over the intertemporal allocation of resources. The utility functions of these two individuals are identical, and again have consumption and the resource stock as their arguments. The objective function employed by Li and Lofgren is:

\[ \max U = \pi U_1 + \left(1 - \pi\right) U_2 = \int_0^\infty u(c(t), s(t)) D(t) \, dt \quad (5.11) \]

where,

\[ U_1 = \int_0^\infty u(c(t), s(t)) \exp(-\rho_C t) \, dt \quad (5.12) \]

\[ U_2 = \lim_{\rho_C \to 0} \int_0^\infty u(c(t), s(t)) \exp(-\rho_C t) \, dt \quad (5.13) \]

where \( D(t) \) is the discount factor. The important difference between these two decision-makers is that they are assumed to discount future utilities at different rates. The utilitarian, who wants to maximize the present value of her utility \( U_1 \), has a rate of time preference equal to \( \rho_U \). The conservationist, who derives utility from conserving the stock of the natural resource, has a rate of time preference equal to \( \rho_C \) and maximizes her utility. The overall societal objective is to maximize a weighted sum of well-being for both members of the society, given their different respective weights upon future generations. The effective utility discount rate in Li and Lofgren is given by:

\[ \rho(t) = -\frac{1}{t} \ln \left\{ (1 - \pi) \exp(-\rho_C t) + \pi \exp(-\rho_U t) \right\} \quad (5.14) \]

A time profile of discount rates can therefore be found by merely selecting the discount rates for the conservationist and the utilitarian, \( \rho_C \) and \( \rho_U \), respectively. For example, if the conservationist discounts the future at a rate of zero: \( \rho_C = 0 \), the discount factor becomes:

\[ D(t) = (1 - \pi) + \pi \exp(-\rho_U t) \quad (5.15) \]

In the distant future when \( t \) is large it has a minimum value of \( (1 - \pi) \), the weight attaches to the conservationist, or future generations. It is in this way
that the effective discount rate can be thought of as declining over time to zero. Thus, unlike the utilitarian discount function, which tends to zero as time reaches towards infinity, the weighted discount function tends to the weight for the far distant future. Hence Li and Löfgren’s model results in a positive welfare weight for the conservationist and there is no dictatorship of present over future generations. As the utilitarian’s welfare level is explicitly considered, there will also not be any dictatorship of the future over the present. Thus, the model explicitly considers intergenerational equity. Within this framework, the conservationist will dominate the far distant future. Therefore the discount rate will be a declining function of the time horizon.

Implementation of the Li and Löfgren and Chichilnisky approaches requires the identification of several other parameters, including specification of the utility discount rate for the ‘utilitarian’, and perhaps more importantly, the relative weight to be assigned between ‘conservationist’ and ‘utilitarian’ preferences. Although the selection of this weighting might appear to be relatively arbitrary, it makes the trade-off between present and future generations explicit, and could possibly be determined by an appropriate political process.

IMPLICATIONS OF DECLINING DISCOUNT RATES: CLIMATE CHANGE POLICY IN THE UK

In this section we describe a declining discount rate schedule derived from the application of the estimation procedure used by Newell and Pizer (2003) to UK interest rate data. In short, interest rates are forecast over a period of 400 years using the results of an estimated reduced form time series model. First, we present the results associated with the autoregressive model (AR) used by Newell and Pizer. The schedule of certainty equivalent discount rates is derived from the simulation of up to 100,000 interest rate forecasts and use of Weitzman’s definition of the certainty equivalent discount rate (CER). We also present the results of a ‘state-space’ model applied to the UK data, which takes into account the possibility of structural breaks and allows for the auto-correlation process driving interest rates to change over time. These are important determinants of discount rate uncertainty, which represent a more appropriate methodology for forecasting discount rates for the very long term and a departure from Newell and Pizer.

Figure 5.1 compares the schedule of the certainty equivalent discount factors derived from the two forecasted models to the discount factor that is derived from discounting at a flat rate of 3.5 per cent. It is easy to see that the schedule of certainty equivalent discount factors derived from the state-
Figure 5.1  Empirical certainty equivalent versus conventional discount factors (UK)
space model is higher than those derived from the Newell and Pizer method, whilst the latter is fractionally higher than with constant discounting. These results are similar to those of Newell and Pizer for the US: interest rate uncertainty in the UK provides a rationale for DDRs to be employed in project appraisal. However, there are two further practical points that arise from this analysis. Firstly, in applying Newell and Pizer, we fail to establish the existence of persistence, indicating that the mean reverting model is more appropriate than the random walk model. The simulation associated with the mean reverting model is given by AR in Figure 5.1. This is the inverse of Newell and Pizer’s finding for the US. Secondly, model selection is important. The state-space model is represented by:

\[ r_t = c_t + \alpha_t r_{t-1} + e_t \]  

\[ \alpha_t = c_t \alpha_{t-1} + u_t \]

where \( u_t \) and \( e_t \) are vectors of serially independent zero-mean normal disturbances. In other words, we model uncertainty of the interest rate as an AR(1) process with AR(1) coefficients. Details of this and other specifications can be found in Groom et al. (2004). The state-space model is introduced to add greater flexibility in the characterization of the uncertainty surrounding the interest rate. For example, the state-space model provides a means by which structural breaks and other likely changes in the data generating process witnessed in the past, can be modelled parsimoniously to predict the future. Groom et al. (2004) present some evidence to suggest that the state-space model is to be preferred for estimating certainty equivalent discount rates. These econometric modelling issues are not without their policy consequences and it is this issue that we turn to in the next section.

**Social Cost of Carbon**

The social cost of carbon is an estimate of the present monetary value of damage done by anthropogenic carbon dioxide emissions. The UK has an ‘official’ value of this shadow price (Clarkson and Deyes, 2002) at £70 per tC, although the validity of the number is disputed (Pearce, 2003) and the official value is under review at the time of writing. Self-evidently, higher values of the social cost of carbon imply that investment in climate change mitigation is more attractive. The discounting framework employed has a significant impact upon such estimates. It is obvious, for instance, that a lower (constant) discount rate will increase the present value of the marginal damage from emissions. For example, the marginal damage values from
Implications of declining discount rates for UK climate change policy

In order to illustrate the difference between the various discounting frameworks on the social cost of carbon, we start with an approximate profile of the economic damage done by one tonne of carbon emissions in 2000, shown in Figure 5.2. This is the profile of damages generated by the DICE model of Nordhaus and Boyer (2000). Applying the various discounting regimes to this damage profile over the next 400 years results in estimates of the social cost of carbon presented in Figure 5.3. For the 200-year period, the estimates vary from approximately £2.50/tC at a 6 per cent flat discount rate, to about £20.50/tC under a discounting regime based on the Li and Löfgren approach.

Increasing the time horizon from 200 to 400 years makes no difference when constant discount rates are employed, because the discount factor approaches zero well before the 200-year mark. In contrast, marginal damage estimates under declining discount rate regimes are noticeably larger when the time horizon is extended to 400 years.

Furthermore, the application of Newell and Pizer’s methodology to UK data increases the 400-year estimates of marginal damage costs by a mere 4.3 per cent compared with the constant discounting regime. This contrasts with Newell and Pizer’s finding of an 84 per cent increase. This reflects the lower level of persistence found in the UK case compared with the US, resulting in the mean reverting model being more appropriate than the random walk model of Newell and Pizer. The state-space model leads to a 150 per cent increase in the value of marginal damage. This model is well specified and is therefore more credible. The magnitude of the differences reflects once more the practical implications of model selection in determining the schedule of CER.

This illustration suggests that estimates of the social cost of carbon are likely to at least double if declining discount rates are employed. This would have formidable implications for policy in several areas. For example, a higher social cost of carbon would make it more likely that commitments to Kyoto targets would pass a cost–benefit test (Pearce, 2003).

CONCLUSIONS

The realization that actions taken today can have long-term consequences presents a challenge to decision-makers in assessing the desirability of policies and projects. The use of the classical net present value (NPV) rule to assess the economic efficiency of policies with costs and benefits that
Source: Nordhaus and Boyer (2000).

Figure 5.2 The time profile of carbon mitigation benefits
Figure 5.3  Value of a 1 tonne of carbon emissions reduction with alternative declining discount rates

Present Value (200 years)  Present Value (400 years)
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accrue in the long term is felt by many to be particularly problematic. The welfare of future generations barely influences the outcome of such a rule when constant socially efficient discount rates are used for all time. The deleterious effects of exponential discounting ensure that projects that benefit generations in the far distant future at the cost of those in the present are less likely to be seen as efficient, even if the benefits are substantial in future value terms. In this respect it appears that the present yields a dictatorship over the future. The idea of using DDRs has emerged largely in response to these awkward implications and recently DDRs have even been entertained at an official level in the UK (HM Treasury, 2003).

The approaches reviewed here are predominantly theoretical contributions to an inherently practical issue. Ultimately, the practitioner is faced with a potentially confusing array of rationales and a sense that almost any discount rate can be applied. Moreover, it is important that the practitioner is aware that the implications of employing declining discount rates are of considerable moment. As our case studies show, there is the potential to reverse the recommendations of social cost–benefit analysis in the long-term policy arena. This is especially important given the nature of this policy arena and the considerable changes that might be required in order to prevent the impact of global warming.

That social discount rates should be declining is still not clear, despite the sometimes compelling contributions described above. In many cases only the conditions under which DDRs are said to exist have been defined. Whether or not these conditions prevail is another question altogether. Indeed, the use of DDRs may put us in danger of placing more weight upon potentially far richer individuals in the far distant future than we place on present or even near future generations. What is more widely agreed is the limited extent to which discount rates can be manipulated to simultaneously reflect the numerous underlying issues that have motivated their investigation, namely intergenerational equity, sustainability and efficiency. Practitioners would be wise to note this as well as the potentially fundamental limitations of CBA in dealing with long-term investments (Lind, 1995).
APPENDIX 5.1

The mechanics of Weitzman’s results are as follows. From (5.5) and (5.6) it is easy to show that the certainty equivalent marginal rate can be written as a weighted average of the potential realizations of $r$:

$$r_m^{CER} = \sum_j w_j(t) r_j$$ (A5.1)

where the weights in this case are simply: $w_j(t) = p_j a_j(t) \Sigma p_j a_j(t)$ and $\Sigma w_j(t) = 1$. Taking the derivative of this with respect to time we obtain:

$$\frac{d}{dt} r_m^{CER} = \sum_j \dot{w}_j(t) r_j = - \sum_j w_j(t) (r_j - r_m^{CER})^2$$ (A5.2)

which is clearly negative. That the limit of $\lim_{t \to \infty} r_m^{CER} = r_{\min}$ comes from noticing that, where $r_1 = r_{\min}$:

$$\lim_{t \to \infty} w_1(t) = 0$$

which means that as $t \to \infty$ the weights associated with all but the lowest discount rate tend to zero due to the presence of $a_j(t)$, and yet, since $\Sigma w_j(t) = 1$, the weight for the lowest discount rate, $w_1(t)$, must tends towards 1.

NOTES

1. See Lind (1982) for an excellent review of these issues.
2. Weitzman (1998) assumes risk-neutral agents for exposition, but this represents a special case of his general point. For realistic scenarios, determination of DDRs à la Gollier (2002a, 2002b) requires knowledge of the fourth and fifth derivatives of utility functions, something that he admits is very far from being accomplished.
3. However, central to this interpretation is a bequest motive: the infinitely lived agent reflects an immortal extended family containing many finitely lived altruistic families. These families are connected by a series of intergenerational transfers to their children who in turn give to their children etc.
4. A discussion of this model is also found in Heal (1998).
5. The last step is not entirely obvious, so we elaborate. Dropping the $m$ subscript from $r_m^{CER}$, note that:

$$w_j(t) = w_j(t) [\Sigma w_j(t) r - r_j] = w_j(t) (r^{CER} - r_j),$$
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Therefore

\[
\frac{d}{dt} C_{ER} = \sum w_j(t) \left( r_{CER}^2 - r_j^2 \right) = \left( r_{CER}^2 \right) - \sum w_j(t) r_j^2.
\]

This term is equal to that obtained by multiplying out (A5.2). That is, noting that \( \Sigma w_j(t) = 1 \) we get:

\[
- \sum w_j(t) \left( r_j^2 + \left( r_{CER}^2 \right)^2 - 2 r_{CER}^2 \right) = 2 \left( r_{CER}^2 \right) - \left( r_{CER}^2 \right)^2 - \sum w_j(t) r_j^2,
\]

and we are done.

6. Gollier (2002a) provides an elegant proof of the following: \( \lim_{\tau \to \infty} r_{CER} = r_{\text{min}} \), that is, for the average CER, by appeal to Pratt’s Theorem.

REFERENCES


Implications of declining discount rates for UK climate change policy


Natural resources


6. Valuing perceived risk of genetically modified food: a meta-analysis

Clare Hall, Dominic Moran and David Allcroft

INTRODUCTION

The future of genetic modification (GM) technology in European agriculture and food is uncertain. Although the ‘unofficial’ moratorium on commercial planting and importation has been lifted, and new applications are being submitted, there remains a large degree of uncertainty surrounding the willingness of consumers to buy GM products. Despite assurances about the safety of GM foods, consumers still perceive there to be potential risks. Policy-makers, biotechnology companies and food growers need to know the extent of the risk that consumers perceive GM foods to contain, and how this is likely to affect demand. Of particular interest is the reduction or premium required to compensate for negative or positive, perceived or real, product attributes. Stated preference studies (contingent valuation (CV) studies, auction experiments and choice experiments) have been conducted to discover how much consumers would be willing to pay (WTP) to purchase GM foods with traits such as less fat (Buhr et al., 1993), or which require less pesticides in production (Boccaletti and Moro, 2000). However, the majority of stated preference studies have asked how much consumers would be willing to pay to avoid products that contain GM ingredients. Others have asked how cheap GM food would need to be in order to induce consumers to buy (see for example, Burton et al. 2001; Noussair et al. 2001; Chen and Chern, 2002; Mendenhall and Evenson, 2002). In this case the question can be described as willingness to accept compensation (WTA) to forego a benefit. The benefit foregone is the perceived risk-free status of non-GM food. This chapter reports on a meta-analysis of stated preference studies relating to GM foods. Objectives were to derive mean estimates for these measures of consumer preferences, and to determine the explanatory variables that influence these values.

Meta-analysis is now established in environmental economics as a way of summarizing the findings of a growing body of non-market valuation studies. Applications cover a range of topics (see for example, Smith and...
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Huang, 1995; Loomis and White, 1996; Poe et al. 2000; Woodward and Wui, 2001; Bateman and Jones, 2003 and Brouwer et al. 2003). The method helps to develop consensus on value estimates for environmental goods, and to investigate factors that explain variation within and between different studies (Poe et al., 2000).

Given the absence of market information on consumer intentions regarding GM foods in Europe we aimed to provide the best prediction of consumer behaviour derived from non-market evidence. We drew together results obtained through contingent valuation (CV) studies, choice experiments and auction experiments that have produced values representing the risk (or benefit) that consumers perceive there to be in GM food. The remainder of this chapter is structured as follows. In the next section we describe the data used in the meta-analysis. This is followed by a section covering methodological issues. Next we present and discuss the results of our study and finally present conclusions.

DATA

Meta-analysis involves considerable methodological complexity (Matarazzo and Nijkamp, 1997) and subjective interpretation in compiling a dataset. The typical approach of non-market valuation meta-analyses involves the compilation of a set of dependent variable observations extracted from studies that have valued the same item. These can be summarized using univariate or multi-variate analysis. In the latter case, dependent data are regressed against independent variables that characterize each study and respondents (or site) (Woodward and Wui, 2001). Several models may be posited. Ultimately a predictive model is desirable but recent applications have been more concerned with checking the underlying experimental validity.

A range of information sources was used to identify studies for inclusion.1 In all, 22 stated preference studies were used in the meta-analysis, producing a total of 56 WTP values. If a single study provided numerous values, as is common in CV studies, then several data points were obtained. For example, Chern and Rickertsen (2002a) presented WTP figures for four different countries. All of these values were initially treated as separate data points. This, as shown below, influences what statistical analysis is conducted. The time period covered by the surveys is approximately a decade, from 1992 to 2003.

We compiled three data sets corresponding to:

- studies that asked how much respondents were WTP for GM with clear benefits (11 values);
• studies that asked how much respondents were WTP for GM-free (or to avoid GM) (21 values);
• studies that asked how much respondents were WTP for GM without clear benefits (or how much cheaper it would have to be before respondents would buy) (24 values).  

The two WTP for GM data sets were treated separately because of the nature of the questions they asked. The 11 values included in the ‘WTP for GM with clear benefits’ data set are from studies that asked respondents how much they would be WTP for GM food with benefits such as reduced use of pesticides in production or improved taste characteristics. This is in contrast to the 24 values included in the ‘WTP for GM without clear benefits’ data set. In this case, the studies presented GM food as being a new and potentially risky alternative to conventional food. Table 6.1 summarizes the 22 studies. Additional tables are appended (Appendices 6.1 and 6.2).

METHODOLOGY

In the present study our dependent variable took the form of a percentage value (of spend on individual food item or on weekly food bill). In most cases this was a comparable statistic reported in study results. In the studies where information was presented as a cash figure we used the contemporary retail food price of that item to produce a percentage figure. In some cases certain assumptions had to be made. For example, in a study by Buhr et al. (1993) we assumed that the price of a meat sandwich bought in a shop would cost twice the price of a loaf of white bread, which in 1992 was $0.75. Hence we used that figure to convert the result to a percentage. We rejected some studies (for example, Fox et al., 1994), which could not be satisfactorily converted. We assumed that if a study asked ‘How much more would you be WTP for non-GM?’ the resulting value would be a lower WTP than if the question asked ‘How much cheaper would GM have to be before you would be willing to buy?’ This is equivalent to the difference between WTA and WTP.

Independent Variables and Hypotheses

The choice of independent variables was informed by the characteristics of the studies included in the meta-analysis and previous environmental economic meta-analyses. Here we outline all assumptions made for the independent variables (see Table 6.2 for a summary).
<table>
<thead>
<tr>
<th>Study number</th>
<th>Author(s)</th>
<th>WTP Question</th>
<th>Number of WTP values</th>
<th>WTP values</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Buhr et al., 1993</td>
<td>WTP for GM-free</td>
<td>1</td>
<td>23% to avoid GM</td>
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<tr>
<td>2</td>
<td>Wang et al., 1997</td>
<td>WTP for GM-free</td>
<td>1</td>
<td>16% extra for rBST-free milk (50% of respondents)</td>
</tr>
<tr>
<td>3</td>
<td>Kuperis et al., 1999</td>
<td>WTP for GM-free</td>
<td>1</td>
<td>13% for GM-free</td>
</tr>
<tr>
<td>4</td>
<td>Boccaletti and Moro, 2000</td>
<td>WTP for GM-free with clear benefits</td>
<td>4</td>
<td>8% for GM foods that reduce the use of pesticides</td>
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<td></td>
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<td>8% for GM foods that have increased nutritional properties</td>
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<td>5% for GM foods with improved taste</td>
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<td>5% for GM foods with longer shelf life</td>
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<td>5</td>
<td>Noussair et al., 2002</td>
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<td>1</td>
<td>27% reduction in bid with awareness of GM</td>
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<td>6</td>
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<td>WTP for GM-free</td>
<td>3 (different countries)</td>
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<td>11% extra for GM-free (Germany)</td>
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<td>9% extra for GM-free (UK)</td>
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<td>Authors</td>
<td>WTP for</td>
<td>(consumer subgroup)</td>
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<td>7</td>
<td>Burton, Rigby et al., 2001</td>
<td>GM-free</td>
<td>5 (consumer subgroups)</td>
<td>9% extra for GM-free (infrequent purchaser of organic food)</td>
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<td>14% extra for GM-free (occasional purchaser of organic food)</td>
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<td>17% extra for GM-free (committed purchaser of organic food)</td>
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<td>10% extra for GM-free (from CV question only)</td>
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<td>Loureiro and Hine, 2001</td>
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<td>4 (by gender and type of GM)</td>
<td>14% reduction required to purchase GM (female – plants only)</td>
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<td>4% reduction required to purchase GM (male – plants only)</td>
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<td>52% reduction required to purchase GM (female – plants and animals)</td>
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<td>26% reduction required to purchase GM (male – plants and animals)</td>
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<td>78% extra to avoid GM</td>
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<td>Baker and Mazzocco, 2002</td>
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<td>40% less for GM</td>
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<td>Mendenhall and Evenson, 2002</td>
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<td>20% extra for GM-free (50% of respondents)</td>
</tr>
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<td>WTP values</td>
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<td>--------------</td>
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<td>------------</td>
</tr>
<tr>
<td>12</td>
<td>Lusk et al., 2001a</td>
<td>WTP for GM-free</td>
<td>2 (different parts of study)</td>
<td>33% extra for GM-free</td>
</tr>
<tr>
<td>13</td>
<td>Huffman et al., 2001</td>
<td>WTP for GM without benefits</td>
<td>3 (different products)</td>
<td>14% less for GM (vegetable oil) 14% less for GM (tortilla chips) 14% less for GM (russet potatoes) 8% extra for GM-free 38% reduction in bids for product labelled 'contains GM' 21% more for GM rice 30% more for GM rice</td>
</tr>
<tr>
<td>14</td>
<td>Noussair et al., 2001</td>
<td>WTP for GM-free and WTP for GM without clear benefits</td>
<td>2</td>
<td>8% extra for GM-free</td>
</tr>
<tr>
<td>15</td>
<td>Lusk, 2002</td>
<td>WTP for GM with clear benefits</td>
<td>2 (different auction types)</td>
<td>7% less for GM vegetable oil 22% less for GM salmon 15% less for GM cornflakes 6% less for GM sweetcorn 6% extra for GM sweetcorn 8% extra for GM sweetcorn 2% extra for GM sweetcorn</td>
</tr>
<tr>
<td>16</td>
<td>Chen and Chern, 2002</td>
<td>WTP for GM without clear benefits</td>
<td>3 (different food items)</td>
<td>6% extra for GM sweetcorn 8% extra for GM sweetcorn 2% extra for GM sweetcorn</td>
</tr>
<tr>
<td>17</td>
<td>James et al., 2002</td>
<td>WTP for GM</td>
<td>4 (different stores)</td>
<td>60% reduction needed for GM noodles 64% reduction needed for GM tofu</td>
</tr>
<tr>
<td>18</td>
<td>McCluskey et al., 2001</td>
<td>WTP for GM without clear benefits</td>
<td>2 (different food items)</td>
<td>60% reduction needed for GM noodles 64% reduction needed for GM tofu</td>
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<tr>
<td>Reference</td>
<td>Research Question</td>
<td>WTP for GM</td>
<td>Sample Size</td>
<td>Results</td>
</tr>
<tr>
<td>---------------------------------</td>
<td>----------------------------------------</td>
<td>------------</td>
<td>-------------</td>
<td>---------</td>
</tr>
<tr>
<td>Grimsrud et al. 2002</td>
<td>WTP for GM without clear benefits</td>
<td>2 (different food items)</td>
<td></td>
<td>48% reduction needed for GM bread 56% reduction needed for GM salmon</td>
</tr>
<tr>
<td>Chern and Rickertsen, 2002a</td>
<td>WTP for GM-free</td>
<td>4 (different countries)</td>
<td></td>
<td>56% extra for GM-free (US) 37% extra for GM-free (Japan) 62% extra for GM-free (Norway) 19% extra for GM-free (Taiwan)</td>
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<td>Chern and Rickertsen, 2002b</td>
<td>WTP for GM without clear benefits</td>
<td>6 (2 different countries, 3 products)</td>
<td></td>
<td>55% less for GM soya-bean oil (Norway) 54% less for GM-fed salmon (Norway) 67% less for GM salmon (Norway) 84% less for GM soya-bean oil (US) 46% less for GM-fed salmon (US) 71% less for GM salmon (US)</td>
</tr>
<tr>
<td>Bugbee and Loureiro, 2003</td>
<td>WTP for GM with clear benefits</td>
<td>2 (different products)</td>
<td></td>
<td>3% extra for GM tomatoes with higher nutritional value 2% extra for GM beef with higher nutritional content and less calories</td>
</tr>
</tbody>
</table>

*Note:* 56 WTP values (11 WTP for GM with clear benefits; 24 WTP for GM without clear benefits (price reduction); 21 WTP for GM-free).
### Table 6.2 Assumptions made about independent variables

<table>
<thead>
<tr>
<th>Independent Variable</th>
<th>Assumption</th>
<th>Options</th>
</tr>
</thead>
<tbody>
<tr>
<td>Response rate</td>
<td>Lower response rate = greater WTP for GM-free or greater reduction required for GM without clear benefits</td>
<td>Percentage</td>
</tr>
<tr>
<td>Survey year</td>
<td>Pre-1999 smaller WTP for GM-free or smaller reduction required for GM without clear benefits</td>
<td>1998 or earlier 1999 or later</td>
</tr>
<tr>
<td>Survey country</td>
<td>USA = smaller WTP for GM-free or smaller reduction required for GM without clear benefits</td>
<td>USA Rest of World</td>
</tr>
<tr>
<td>Description of food in survey (general or specific)</td>
<td>General basket of goods = smaller WTP for GM-free (no assumption for WTP for GM without clear benefits)</td>
<td>Named food item GM general</td>
</tr>
<tr>
<td>Participant group</td>
<td>Shoppers = smaller WTP for GM-free or smaller reduction required for GM without clear benefits</td>
<td>Students Shoppers General population</td>
</tr>
<tr>
<td>Survey distribution method</td>
<td>Supermarket = smaller WTP for GM-free or smaller reduction required for GM without clear benefits</td>
<td>Mail Telephone In-person Supermarket</td>
</tr>
<tr>
<td></td>
<td>Personal interviews and auctions = smaller WTP for GM-free or smaller reduction required for GM without clear benefits</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Phone = greater WTP for GM-free or greater reduction required for GM without clear benefits</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Mail = greatest WTP for GM-free or greater reduction required for GM without clear benefits</td>
<td></td>
</tr>
<tr>
<td>Survey topic</td>
<td>GM-specific survey = greater WTP for GM-free or greater reduction required for GM without clear benefits</td>
<td>GM-specific survey General food survey</td>
</tr>
<tr>
<td>Independent Variable</td>
<td>Assumption</td>
<td>Options</td>
</tr>
<tr>
<td>----------------------</td>
<td>------------</td>
<td>---------</td>
</tr>
<tr>
<td>Elicitation technique</td>
<td>Dichotomous choice = greatest WTP for GM-free or greater reduction required for GM without clear benefits</td>
<td>Auction</td>
</tr>
<tr>
<td></td>
<td>Iterative bidding (auction) = next greatest WTP for GM-free or greater reduction required for GM without clear benefits</td>
<td>Payment card Choice experiment</td>
</tr>
<tr>
<td></td>
<td>Payment card = smaller value than two above, greater than open-ended</td>
<td>Dichotomous choice CV questions</td>
</tr>
<tr>
<td></td>
<td>Revealed preference = smaller WTP for GM-free or smaller reduction required for GM without clear benefits</td>
<td>Open-ended CV questions</td>
</tr>
<tr>
<td></td>
<td>Open-ended = smallest WTP for GM-free or smaller reduction required for GM without clear benefits</td>
<td>Revealed preference in-store purchase</td>
</tr>
<tr>
<td></td>
<td>No assumption made regarding impact of choice experiments</td>
<td></td>
</tr>
</tbody>
</table>

We assumed that pre-1999 respondents considered GM to be no more risky than conventional food and would not be WTP such a high premium for it. Similarly, GM food would not need to be a lot cheaper than conventional food before consumers would be willing to buy. We chose 1999 as a watershed because in 1999 there was an ‘outburst of media hysteria relating to genetically modified food products’ (Burrell, 2000).

Higher response rates have been found to result in lower average WTP (Brouwer et al., 2003). We expected that people interested in GM and food safety would be most likely to respond and a lower response rate would therefore include a larger percentage of those with concerns about the issue. Assuming that they were opposed to GM this would result in a high WTP to avoid GM. However, in some surveys GM was only one topic among many, hence the response decision of an individual may not have been affected by opinions on GM issues.

The use of open-ended elicitation format has been found to result in lower average WTP amounts than any other format (Brouwer et al., 2003). The dichotomous choice (DC) format has shown the highest average WTP, followed by the iterative bidding format (auction experiments), and then payment card method. We based our assumptions on these earlier
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findings. There appears as yet to be no indication as to how the use of choice experiments might affect values obtained.

Personal interviews have been found to yield a lower average WTP than mail surveys (Brouwer et al., 2003). We assumed that supermarket surveys would yield lower values because the setting was more realistic and would help to avoid unrealistically high bids. We also assumed that personal interviews and telephone surveys would yield lower values as they allow more opportunity for clarification. We assumed that surveys conducted by mail would result in the highest values.

It has been widely reported that US consumers have been less suspicious of GM foods. We therefore assumed that values obtained from studies conducted in the US would yield lower values than studies conducted elsewhere in the world. We assumed that a survey asking respondents about a specific food item would result in higher values than a general basket of food (weekly food expenditure). This assumption is based on the idea that, for example, 25 per cent extra for a single item is not as large in people's minds as 25 per cent extra on their weekly food bill.

We assumed that if the survey topic was specifically about GM food then the value would be greater. We assumed this because a GM-specific survey gives the subject an importance that it might not have in people's minds if it is one subject among others in a general food survey.

Finally, we assumed that values would differ depending on the participant group. These groups were: shoppers (for surveys that were conducted in stores), students (for surveys that were conducted in university agriculture departments) and general population (for all other surveys). Our assumption was that shoppers and students would provide lower, more realistic values because of having a more realistic setting and more knowledge of the subject, respectively.

RESULTS AND DISCUSSION

Initially, descriptive statistical procedures were used to explore the dependent variable of each of the three data sets. Next, bivariate analysis was used to compare mean WTP values with a range of independent variables, including date of study, country of study, response rate, survey distribution method and elicitation technique. Finally, in order to explore the significance of our findings and to account for the fact that some papers provided more than one estimate of WTP, we carried out multi-variate analysis using REML. Table 6.3 presents summary statistics for each of the three data sets.

The mean values show that, on average, respondents were WTP 24 per cent to avoid GM food, but were willing to buy GM food at a 37 per cent
**Table 6.3 Summary statistics**

<table>
<thead>
<tr>
<th></th>
<th>Summary Statistics for WTP for GM Food Without Clear Benefits</th>
<th>Summary Statistics for WTP for GM-free Food</th>
<th>Summary Statistics for WTP for GM Food with Clear Benefits</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number of values</td>
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<td>21</td>
<td>11</td>
</tr>
<tr>
<td>Number of observations</td>
<td>24</td>
<td>21</td>
<td>11</td>
</tr>
<tr>
<td>Mean (%)</td>
<td>37 (27, 47)</td>
<td>24 (15, 33)</td>
<td>9 (3, 15)</td>
</tr>
<tr>
<td>Median (%)</td>
<td>39</td>
<td>17</td>
<td>6</td>
</tr>
<tr>
<td>Minimum (%)</td>
<td>4</td>
<td>5</td>
<td>2</td>
</tr>
<tr>
<td>Maximum (%)</td>
<td>84</td>
<td>78</td>
<td>30</td>
</tr>
<tr>
<td>Range (%)</td>
<td>80</td>
<td>73</td>
<td>28</td>
</tr>
<tr>
<td>Lower quartile (%)</td>
<td>14</td>
<td>11</td>
<td>4</td>
</tr>
<tr>
<td>Upper quartile (%)</td>
<td>56</td>
<td>33</td>
<td>8</td>
</tr>
<tr>
<td>Standard deviation (%)</td>
<td>24</td>
<td>20</td>
<td>9</td>
</tr>
</tbody>
</table>

**Notes:**
The percentage values in the first column refer to how much cheaper GM food has to be than conventional food prices before consumers are willing to buy.
The percentage values in the second and third columns refer to a percentage premium over and above the conventional food price.
The figures in brackets are 95% confidence intervals for the means.
discount. In addition, respondents were on average WTP 9 per cent extra for GM food with clear benefits. The mean ‘WTP for GM without clear benefits’ value is higher than the mean ‘WTP for GM-free’ value. This was as expected as the former represents a discount while the latter represents a premium. Both values are higher than the mean ‘WTP for GM with clear benefits’ value (9 per cent), suggesting that, so far, the perceived risks of GM food outweigh the promised benefits (in the minds of some consumers). The ‘WTP for GM food without clear benefits’ value of 37 per cent could be interpreted as the value of risk associated with GM. However, the ‘WTP for GM-free’ figure is perhaps even more significant as this suggests that people perceive the risks to be so great that they are WTP 24 per cent extra to avoid them. This could also be interpreted as the value of risk that GM food is perceived to present.

The histogram in Figure 6.1 displays the frequency distribution of values for the data set ‘WTP for GM without clear benefits’. The histogram appears bimodal, with noticeable peaks between 10–20 per cent and 50–60 per cent. Figure 6.2 displays the frequency distribution of values for the data set ‘WTP for GM-free’. The distribution is skewed with most values below 20 per cent and only a few values around 60 per cent and above.

Figure 6.1  Histogram showing the distribution of the mean WTP values (WTP for GM without clear benefits) from the 24 studies
Valuing perceived risk of genetically modified food

Figure 6.2  Histogram showing the distribution of the mean WTP values (WTP for GM-free) from the 21 studies

Figure 6.3 displays the frequency distribution of values for the data set ‘WTP for GM with clear benefits’. Again the distribution is skewed, with most values below 10 per cent and only a few above 20 per cent.

The results in Table 6.4 show the number of observations and mean WTP value for each category of each variable. This allows comparison of the different categories and suggests how some variables may be influencing values.

Results for the ‘WTP for GM without clear benefits’ data set show that the mean WTP value from GM-specific surveys was 40 per cent, but only 24 per cent for general surveys. This suggests that GM-specific surveys have a greater tendency to give the subject an apparent importance that it may not have in people’s minds. The highest mean WTP value relating to distribution method was from telephone surveys, at 63 per cent. The next highest was from studies conducted in supermarkets (47 per cent), with mail surveys and in-person studies being approximately the same (23 per cent and 21 per cent respectively). We expected telephone surveys to elicit lower values. However, it may be that this format places pressure on respondents to answer quickly thus preventing them from thinking carefully about their responses. We produced rugplots for this variable (Appendix 6.3).
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The ‘WTP for GM-free’ data set was the only one of the three where we could evaluate the variable ‘year of study’ as it was the only one with results in both categories of the variable. Results suggest that studies carried out in 1998 or earlier produced an average WTP that was lower than the average WTP from studies conducted in 1999 or later (17 per cent as opposed to 25 per cent). Despite the fact that only three of the studies were pre-1999, this might suggest that increased media attention leads to a greater degree of scepticism among the public, in turn leading to an increase in the value placed on the perceived risks of GM food by respondents. Regarding elicitation technique, DC CV studies produced the highest WTP value, as predicted (44 per cent). Auction experiments produced an average value lower than this (22 per cent), again as predicted. However, the lowest mean WTP was not from open-ended questions (mean WTP 31 per cent) but from the payment card method (5 per cent). However, there was only one study that used a payment card method in this data set. Results show that auction experiments yielded a lower mean WTP value than open-ended questions, which is not as expected. Interestingly, the average WTP value from choice experiments was 15 per cent, among the lowest. Results from the WTP for GM without clear benefits data set showed that choice experiments elicited

Figure 6.3  Histogram showing the distribution of the mean WTP values (WTP for GM with clear benefits) from the 11 studies

The ‘WTP for GM-free’ data set was the only one of the three where we could evaluate the variable ‘year of study’ as it was the only one with results in both categories of the variable. Results suggest that studies carried out in 1998 or earlier produced an average WTP that was lower than the average WTP from studies conducted in 1999 or later (17 per cent as opposed to 25 per cent). Despite the fact that only three of the studies were pre-1999, this might suggest that increased media attention leads to a greater degree of scepticism among the public, in turn leading to an increase in the value placed on the perceived risks of GM food by respondents. Regarding elicitation technique, DC CV studies produced the highest WTP value, as predicted (44 per cent). Auction experiments produced an average value lower than this (22 per cent), again as predicted. However, the lowest mean WTP was not from open-ended questions (mean WTP 31 per cent) but from the payment card method (5 per cent). However, there was only one study that used a payment card method in this data set. Results show that auction experiments yielded a lower mean WTP value than open-ended questions, which is not as expected. Interestingly, the average WTP value from choice experiments was 15 per cent, among the lowest. Results from the WTP for GM without clear benefits data set showed that choice experiments elicited
Table 6.4  Bivariate analysis: WTP for GM without benefits

<table>
<thead>
<tr>
<th>Variable Factor</th>
<th>Number of Observations</th>
<th>Minimum Value</th>
<th>Maximum Value</th>
<th>Standard Deviation</th>
<th>Mean WTP (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Country of study</td>
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<td></td>
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<tr>
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<td>4</td>
<td>67</td>
<td>20</td>
<td>43</td>
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<td>84</td>
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<td>In-person</td>
<td>5</td>
<td>14</td>
<td>38</td>
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<td>21</td>
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<td>Mail</td>
<td>8</td>
<td>4</td>
<td>52</td>
<td>16</td>
<td>23</td>
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<tr>
<td>Supermarket</td>
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<td>6</td>
<td>64</td>
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<td>46</td>
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<td>14</td>
<td>63</td>
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<td>Elicitation technique</td>
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<tr>
<td>Auction</td>
<td>5</td>
<td>14</td>
<td>38</td>
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<td>Choice experiment</td>
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<td>24</td>
<td>47</td>
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<td>CV questions</td>
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<tr>
<td>Revealed preference in-store purchasing</td>
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<td>6</td>
<td>6</td>
<td>0</td>
<td>6</td>
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<tr>
<td>Gen or Spec food (Does survey deal with a general 'basket of food' or a specific food item?)*</td>
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<td>4</td>
<td>52</td>
<td>21</td>
<td>24</td>
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<tr>
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<td>84</td>
<td>24</td>
<td>40</td>
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<tr>
<td>Gen or Spec survey (Is survey a general food survey or GM-specific?)</td>
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<td>64</td>
<td>24</td>
<td>47</td>
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<td></td>
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<td>19</td>
<td>27</td>
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<td>7</td>
<td>84</td>
<td>27</td>
<td>47</td>
</tr>
<tr>
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<td>60</td>
<td>64</td>
<td>3</td>
<td>35</td>
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<td>6</td>
<td>38</td>
<td>16</td>
<td>24</td>
</tr>
</tbody>
</table>

* No assumption was made about the impact of this variable on WTP values.

The results in Table 6.5 show the number of observations and mean WTP values for each category of each variable for the ‘WTP for GM-free’ data set.
the highest average values. We suggest that an area where more research would be useful is the effect of choice experiments on WTP values. We have produced boxplots for this variable (Appendix 6.4). The mean WTP value obtained from the general population was higher than that obtained from
shoppers (19 per cent as opposed to 5 per cent). The highest value of all was from students, at 36 per cent. However, we had only one value for the shoppers’ category in this data set. There may be a number of reasons why the student participant group produced a surprisingly high WTP value. Firstly the group is unrepresentative of the general population in terms of socio-economic group and age. In addition, student groups may contain a higher than average percentage of individuals who feel strongly about environmental and social issues.

The results in Table 6.6 show the number of observations and mean WTP value for each category of each variable for the data set ‘WTP for GM food with clear benefits.’ We have not included the survey topic variable in this data set as all of the studies were GM-specific.

<table>
<thead>
<tr>
<th>Variable Factor</th>
<th>Number of Observations</th>
<th>Minimum Values</th>
<th>Maximum Values</th>
<th>Standard Deviation</th>
<th>Mean WTP for GM (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Country of study</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>Revealed preference in-store purchasing</td>
<td>3</td>
<td>2</td>
<td>8</td>
<td>3</td>
<td>5</td>
</tr>
<tr>
<td>Gen or Spec food (Does survey deal with a general ‘basket of food’ or a specific food item?)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>General</td>
<td>4</td>
<td>5</td>
<td>8</td>
<td>2</td>
<td>7</td>
</tr>
<tr>
<td>Specific</td>
<td>7</td>
<td>2</td>
<td>30</td>
<td>11</td>
<td>10</td>
</tr>
<tr>
<td>Participant</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>General population</td>
<td>8</td>
<td>2</td>
<td>30</td>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td>Shoppers</td>
<td>3</td>
<td>2</td>
<td>8</td>
<td>3</td>
<td>5</td>
</tr>
<tr>
<td>Response (%)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>14</td>
<td>2</td>
<td>21</td>
<td>30</td>
<td>6</td>
<td>26</td>
</tr>
<tr>
<td>16</td>
<td>2</td>
<td>2</td>
<td>3</td>
<td>1</td>
<td>3</td>
</tr>
<tr>
<td>52</td>
<td>4</td>
<td>5</td>
<td>8</td>
<td>2</td>
<td>7</td>
</tr>
<tr>
<td>Unknown</td>
<td>3</td>
<td>2</td>
<td>8</td>
<td>3</td>
<td>5</td>
</tr>
</tbody>
</table>
Results from this initial comparison of means show the average WTP for GM food from US studies was 10 per cent, while in the rest of the world it was 7 per cent. This makes sense since, as with the other two data sets, we made the assumption that US respondents would be less worried about the risks of GM food. In this data set we can therefore assume that US consumers are less sceptical about the benefits of GM food, and WTP will have a higher average value for GM with clear benefits. Surveys about GM foods in general produced a lower mean WTP than surveys about a specific food item (7 per cent as opposed to 10 per cent).

Of the three data sets, it is ‘WTP for GM with clear benefits’ that produces bivariate analysis results that most consistently support our predictions. There may be an explanation for this. In the studies included in the other two data sets the question that respondents were asked to consider was problematic. First, the values in the ‘WTP for GM-free’ data set are drawn from studies that asked people how much extra they would be WTP to maintain the right to consume a product they were already able to purchase. In the case of the ‘WTP for GM without clear benefits’ data set, respondents were expected to indicate how much cheaper they wanted a new, unknown and potentially undesirable and risky product to be, before they would be willing to buy. This would involve giving up the ownership of a familiar and acceptable product, conventionally produced food. When we consider these problematic questions we can see that the question in the studies covered by the third data set, ‘WTP for GM with clear benefits’, was much more straightforward for respondents. In this case, respondents were presented with a new product with clear benefits over existing alternatives and asked how much more they would be willing to pay for that product.

There is only one variable for which the bivariate analysis results from all three data sets support our predictions. This was that a GM-specific survey would result in higher WTP values. Our assumption relating to the variable ‘description of food in survey’ was supported by the results from the two data sets to which it applied and suggests that a survey relating to a specific food item may elicit a larger WTP value. Our assumptions about a further two variables, survey country and elicitation technique, were supported by two of the data sets. For the former it appears that a study conducted in the USA may, in some cases, elicit a smaller WTP value (reduction required) for GM food without benefits or a larger WTP value for GM food with benefits, than studies from the rest of the world. For the variable elicitation technique, in some cases predictions hold true, for example, that open-ended questions produce the lowest values, DC CV produce the highest values and iterative bidding techniques provide values somewhere in between.
However, the fact that our data sets included studies that used a number of other techniques such as revealed preference in-store purchasing and payment card method made the results less clear, particularly because of the small numbers of studies using these approaches.

In addition, the use of choice experiments by a large number of studies further complicated results. Results about distribution methods, response rate and participant group are problematic. The effect of different distribution methods is unclear. A number of assumptions about in-person studies and telephone studies for example, were not supported by our results in all cases. Overall, results relating to the effect of distribution method, response rate and participant group on mean values are largely inconclusive, depending on which data set is considered. Results relating to year of study do suggest an increase in the value of risk associated with GM food after 1998 but are drawn only from one data set. Table 6.7 summarizes the extent to which the results of the bivariate analysis support our predictions.

Table 6.7 Summary of bivariate analysis results by variable

<table>
<thead>
<tr>
<th>Variable</th>
<th>Assumption</th>
<th>Dataset 1*</th>
<th>Dataset 2*</th>
<th>Dataset 3*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Response rate</td>
<td>Lower response rate = greater WTP for GM-free or greater reduction required for GM without clear benefits</td>
<td>No</td>
<td>Partially</td>
<td>Yes</td>
</tr>
<tr>
<td>Survey year</td>
<td>Pre-1999 = smaller WTP for GM-free or smaller reduction required for GM without clear benefits</td>
<td>N/A</td>
<td>Yes</td>
<td>N/A</td>
</tr>
<tr>
<td>Survey country</td>
<td>USA = smaller WTP for GM-free or smaller reduction required for GM without clear benefits</td>
<td>Yes</td>
<td>No</td>
<td>Yes</td>
</tr>
<tr>
<td>Description of food in survey</td>
<td>General basket of goods = smaller WTP for GM-free</td>
<td>Yes</td>
<td>Yes</td>
<td>N/A</td>
</tr>
<tr>
<td>Participant group</td>
<td>Shoppers = smaller WTP for GM-free or smaller reduction required for GM without clear benefits</td>
<td>No</td>
<td>Partially</td>
<td>Yes</td>
</tr>
</tbody>
</table>

*Assumption supported by results of bivariate analysis.
<table>
<thead>
<tr>
<th>Variable</th>
<th>Assumption</th>
<th>Is Assumption Supported by Results of Bivariate Analysis?</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Dataset 1</td>
</tr>
<tr>
<td>Survey distribution method</td>
<td>Bulk = smaller WTP for GM-free or smaller reduction required for GM without clear benefits</td>
<td>Partially In-person studies lowest value</td>
</tr>
<tr>
<td>Survey topic</td>
<td>General food survey = smaller WTP for GM-free or smaller reduction required for GM without clear benefits</td>
<td>Yes</td>
</tr>
<tr>
<td>Elicitation technique</td>
<td>Dichotomous choice = greatest WTP for GM-free or greater reduction required for GM without clear benefits</td>
<td>Yes</td>
</tr>
<tr>
<td></td>
<td>Iterative bidding (auction) = next greatest WTP for GM-free or greater reduction required for GM without clear benefits</td>
<td>Yes</td>
</tr>
</tbody>
</table>

**Notes:**

*Dataset 1: WTP for GM without clear benefits.
Dataset 2: WTP for GM-free.
Dataset 3: WTP for GM with clear benefits.*
Bivariate statistics gave us some indication of what was explaining mean values, but we were also interested in developing a more robust predictive model and analysis, and in determining the statistical significance of the variables. Two sets of explanatory variables are of interest. First, we were interested in how various exogenous variables, such as country of study and description of food in survey, influence WTP. These give insights into the political and psychological facets of the GM acceptance question. Second, there is a range of study-specific variables that are artefacts of the design process. These include elicitation technique and the distribution method. Arguably, the existence of a single valuation protocol would obviate the need for including these variables. However, we note that some recent meta-analyses (for example, Bateman and Jones, 2003) have focused more on this finding, of construct validity, than the predictive ability of the technique.

To investigate the significance of the findings from the simple comparison of means we subjected two of our data sets to multi-variate analysis using residual maximum likelihood (REML). Multi-variate analysis is common in meta-analysis, typically using ordinary least squares (OLS) methods such as regression or analysis of variance. However, in our meta-analysis, the use of these straightforward techniques was not appropriate. The fact that some of the studies yielded numerous WTP values gave the data sets a ‘multi-level’ structure. As pointed out by Rasbash et al. (2000) ‘the point of multi-level modelling is that a statistical model explicitly should recognize a hierarchical structure where one is present’. Using ordinary regression to establish relationships between the dependent variables and the independent variables means that it can be unclear how to interpret any relationship found since a level of information is ignored (that is, the clustering or blocking of dependent variable values from within the same study). By focusing attention on the levels of hierarchy in the data set, multi-level modelling enables the researcher to understand where and how effects are occurring. Carrying out an analysis that does not recognize the existence of clustering will cause standard errors of regression coefficients to be wrongly estimated.

Using the notation of GenStat (2000) we can write a general multi-level model in terms of the observed data $y$, $p$ parameters corresponding to fixed effects, and $q$ parameters corresponding to random effects (see below for an explanation of fixed and random effects):

$$ y = X\alpha + Z\beta + \varepsilon $$

Here, $y$ is the vector length $n$ of observations (WTP), $X$ is an $n \times p$ design matrix (containing fixed effect parameters), $Z$ is an $n \times q$ design matrix (containing random effect parameters) and $\varepsilon$ is a vector length $n$ of...
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independent and identically distributed normal random errors with mean zero and variance \( \sigma_e^2 \), that is, \( \varepsilon \sim N(0, \sigma_e^2) \). For this model we can write the expectation and covariance of the data as:

\[
E(y) = X\alpha
\]

\[
\text{Cov}(y) = \mathbf{V} = Z \text{Cov}(\beta) Z' + \sigma_s^2 \mathbf{I}_n
\]

and the maximum likelihood estimates for \( \alpha \) and \( \beta \) can be derived in terms of the observed data and estimated parameters (see GenStat, 2000).

An example of the type of model fitted is:

\[
WTP_{ijk} = m + \text{STUDY}_i + \text{COUNTRY}_j + \text{YEAR}_k + \varepsilon_{ijk}
\]

where \( WTP_{ijk} \) is the mean WTP for study \( i \) in country \( j \) in year \( k \), \( m \) is an overall mean, \( \text{STUDY}_i \) is a term corresponding to study \( i \), \( \text{COUNTRY}_j \) allows for the effect of country \( j \), \( \text{YEAR}_k \) for year \( k \) and \( \varepsilon_{ijk} \) is the error term.

In this example, country and year are fixed effects, that is, effects for which we are interested in the comparison between the levels. Conversely, study is a random effect; we are not particularly interested in making comparisons between studies, we are actually hoping that the collection of studies for which we have the data are a representative sample from the whole set of hypothetical studies that could have been carried out or from which data could have been obtained. For the fixed effects, Wald statistics, with approximate chi-squared distributions, may be used to assess the statistical significance of the terms.

For the analyses here, we wanted to end up with a model that described WTP in terms of a combination of the important variables (fixed effects). The first step was to enter each fixed effect individually into a model to assess their potential importance separately; that is, find out which were significant at the 10 per cent level. Those variables that were significant were retained and entered into multiple regression models. This allowed us to explore any relationships between the variables. For example, given two variables, if both contain similar information, either one on its own may be important, whereas if either one is already included in a model, the addition of the other may no longer add any extra information. Conversely, if each of the two variables contains very different information, it will be desirable to retain both in any final model.

As noted above, the model was first run for each variable singly. The results from this suggested that three variables may be important. These were the country of study variable, the distribution method variable and the
elicitation technique variable ($p$ values 0.019, <0.001 and 0.055 respectively). Wald statistics and $p$ values for all variables are shown in Table 6.8.

Table 6.8 P values – WTP for GM without clear benefits (WTA)

<table>
<thead>
<tr>
<th>Variable</th>
<th>Wald</th>
<th>df</th>
<th>$p$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Country (USA/RoW)</td>
<td>5.52</td>
<td>1</td>
<td>0.019*</td>
</tr>
<tr>
<td>Distribution</td>
<td>26.61</td>
<td>3</td>
<td>&lt;0.001*</td>
</tr>
<tr>
<td>Elicitation</td>
<td>7.61</td>
<td>3</td>
<td>0.055*</td>
</tr>
<tr>
<td>General/spec food</td>
<td>0.28</td>
<td>1</td>
<td>0.596</td>
</tr>
<tr>
<td>General/spec survey</td>
<td>0.28</td>
<td>1</td>
<td>0.596</td>
</tr>
<tr>
<td>Participant</td>
<td>0.06</td>
<td>1</td>
<td>0.801</td>
</tr>
<tr>
<td>Response rate</td>
<td>0.80</td>
<td>1</td>
<td>0.370</td>
</tr>
<tr>
<td>Year (pre/post-1999) missing</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note: * Significant or worthy of further investigation.

Next the model was run with just these three variables. Results showed that, although the country of study variable was significant on its own in a model, it seemed only to explain variation that could be better explained by distribution method and/or elicitation technique. Country of study was therefore dropped from the model. Running the model again, with just the remaining two variables of elicitation technique and distribution method, showed that both variables seemed important even after allowing for the other.

At this stage one category of the elicitation technique variable was dropped as it contained only one value (revealed preference in-store purchasing). Once again the model was run for each of the three more important variables singly (country of study, distribution method and elicitation technique). We also ran the model with all three terms and then with each pair. We concluded that after fitting distribution method, the effect of the country of study was no longer significant on WTP values. The main difference between our conclusions from running the model with the full data set and running it again with one case removed, was that in the full data set elicitation technique was shown to have an effect. However, this was mainly due to the revealed preference in-store purchasing category being lower than the others. There was still evidence that WTP was larger in the RoW than in the USA and still evidence that distribution method had an effect on values – telephone surveys and supermarket surveys had greater WTP values than in-person and mail surveys. There was no real evidence to
suggest that elicitation approach had a significant effect on WTP values in this data set. Fitting multiple models showed that the effect of the country of study variable could largely be explained by the effect of the distribution method. We therefore concluded that the distribution method was most important in affecting WTP values in this data set. Means and standard errors of difference (SEDs) are presented in Table 6.9 (the mean for the variable supermarket is different from the mean in Table 6.4 as one case has been excluded).

Table 6.9  Means and SEDs – distribution method, WTP for GM without clear benefits (WTA)

<table>
<thead>
<tr>
<th>Distribution</th>
<th>In-person</th>
<th>Mail</th>
<th>Supermarket</th>
<th>Telephone</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>21.69</td>
<td>22.46</td>
<td>55.39</td>
<td>62.90</td>
</tr>
</tbody>
</table>

Standard errors of difference between pairs

<table>
<thead>
<tr>
<th></th>
<th>In-person</th>
<th>Mail</th>
<th>Supermarket</th>
<th>Telephone</th>
</tr>
</thead>
<tbody>
<tr>
<td>In-person</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mail</td>
<td>2</td>
<td>7.87</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Supermarket</td>
<td>3</td>
<td>10.06</td>
<td>9.07</td>
<td></td>
</tr>
<tr>
<td>Telephone</td>
<td>4</td>
<td>8.45</td>
<td>7.53</td>
<td>9.44</td>
</tr>
</tbody>
</table>

For two WTP values to be significantly different, the values must differ by more than twice the SED. For example, in Table 6.9 means for in-person studies and telephone surveys are 22 per cent and 63 per cent respectively, hence this difference is significant (more than two times 8 per cent).

For the REML analysis of WTP for GM-free we again ran the model for each variable singly. This revealed that the only variable that was almost significant was elicitation technique ($p$ value 0.06). The next most important variable was participant group, with a $p$ value of 0.198. None of the others were at all significant. Wald statistics and $p$ values for all variables are shown in Table 6.10.

We noted that both the elicitation technique variable and participant group variable contained a category with only one observation (the same observation in both variables). This was the payment card method conducted with the shopper category. This case was omitted and we ran the model again for just the elicitation technique variable and then with just the participant group variable. We ran the model for a third and fourth time with both variables, first with participant group first, and then with elicitation technique first. It seemed that both of these variables might be significant but whichever way round we ran the model they were not significant in themselves after allowing for the effect of the other one. So we looked at
the relationship between these two variables. The categories of these two variables were shown to be closely related, that is, the general population studies were either choice experiments or open-ended CV studies, and the student participant group studies were either auction experiments or DC CV surveys. We therefore only really needed one of these variables in any model. It seemed that the best model was the one with just the elicitation technique. The Wald statistic for this variable had a significance of 0.051 with means and standard errors of difference as shown in Table 6.11.

**Table 6.10**  P values – WTP for GM-free

<table>
<thead>
<tr>
<th>Variable</th>
<th>Wald</th>
<th>df</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Country</td>
<td>0.06</td>
<td>1</td>
<td>0.799</td>
</tr>
<tr>
<td>Distribution</td>
<td>1.22</td>
<td>3</td>
<td>0.748</td>
</tr>
<tr>
<td>Elicitation</td>
<td>9.04</td>
<td>4</td>
<td>0.060*</td>
</tr>
<tr>
<td>General/spec food</td>
<td>0.02</td>
<td>1</td>
<td>0.888</td>
</tr>
<tr>
<td>General/spec survey</td>
<td>0.02</td>
<td>1</td>
<td>0.897</td>
</tr>
<tr>
<td>Participant</td>
<td>3.24</td>
<td>2</td>
<td>0.198*</td>
</tr>
<tr>
<td>Year (pre/post-1999)</td>
<td>0.17</td>
<td>1</td>
<td>0.682</td>
</tr>
</tbody>
</table>

*Note:* * Significant or worthy of further investigation.

**Table 6.11**  Means and SEDs – elicitation technique, WTP for GM-free

<table>
<thead>
<tr>
<th>Means</th>
<th>Auction</th>
<th>Choice experiment</th>
<th>DC CV questions</th>
<th>Open-ended CV questions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Elicitation</td>
<td>22.00</td>
<td>14.88</td>
<td>43.50</td>
<td>31.00</td>
</tr>
<tr>
<td>Standard errors of differences between pairs</td>
<td>1*</td>
<td>2</td>
<td>10.70*</td>
<td></td>
</tr>
<tr>
<td>Auction</td>
<td>2</td>
<td>10.70*</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Choice experiment</td>
<td>3</td>
<td>12.35</td>
<td>10.70*</td>
<td></td>
</tr>
<tr>
<td>DC CV questions</td>
<td>4</td>
<td>12.35</td>
<td>10.70</td>
<td>12.35*</td>
</tr>
<tr>
<td>Open-ended CV questions</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
</tr>
</tbody>
</table>

We identified another potential problem with the data set. This was that it included three values that were part-study values (from 50 per cent or 47 per cent of respondents), and therefore median WTP values, not mean WTP. So we conducted a final analysis, excluding these three cases. Again,
by running the model for each variable singly, results showed that the only almost significant explanatory variable was elicitation technique ($p$ value 0.021). The $p$ value for participant group was 0.162. None of the others was at all significant.

We noted that for the distribution method variable, there was now only one observation for the telephone survey. However, discarding this observation and running the model again did not change the significance of this variable. Once again we ran the model with just elicitation technique and participant group (both together and both ways round). This showed that participant group was not important and therefore the best model was the one with just elicitation technique. In conclusion, the only significant variable for this data set was elicitation technique. There was evidence that WTP values were higher for DC CV questions and open-ended CV questions than for choice experiments. In this case, the difference between DC CV, open-ended CV and auction experiments was not quite significant.

Overall, the results from our REML analysis, which we conducted for only two of our data sets (WTP for GM-free and WTP for GM without clear benefits (WTA)), revealed that there was only one significant variable for each of the data sets. For the ‘WTP for GM without clear benefits’ data set the only variable that was shown to be significant in affecting WTP values was the distribution method. Results revealed that telephone surveys and supermarkets surveys elicited greater WTP values than in-person surveys and mail surveys. This hardly supports our assumptions at all, other than that we predicted that telephone surveys would elicit greater WTP values than in-person surveys. For our WTP for GM-free data set the only variable shown to be significant was elicitation technique. Results showed that WTP values were higher from DC CV and open-ended CV questions than from choice experiments. The result concerning DC CV questions was as predicted, the result regarding open-ended CV questions was surprising as this is usually expected to elicit relatively low WTP values. As previously noted we did not make any assumptions about the effect that choice experiments might have on WTP values.

CONCLUSIONS

Overall, we conclude that the value of risk that consumers perceive GM food to contain is between 24 per cent and 37 per cent of conventional product prices. Key findings from our comparison of means are that:

- mean WTP for GM-free food is 24 per cent above existing conventional food prices;
Valuing perceived risk of genetically modified food

- willingness to buy GM food without clear benefits occurs at a mean price reduction of 37 per cent less than existing conventional food prices;
- WTP for GM with clear benefits is 9 per cent above existing conventional food prices.

The latter suggests that some GM food might be acceptable to some consumers if it has clear benefits. Nevertheless, this is far less than the other two values, which can be said to represent the value of food-borne risk perceptions in the case of GM food.

There is only one variable for which the bivariate analysis results from all three data sets support our predictions. This was that a GM-specific survey would result in higher WTP values. Our assumption relating to the variable ‘description of food in survey’ was supported by the results from the two data sets to which it applied and suggests that a survey relating to a specific food item may elicit a larger WTP value. Our assumptions about a further two variables, survey country and elicitation technique, were supported by two of the data sets. For the former it appears that a study conducted in the USA may, in some cases, elicit a smaller WTP value (reduction required) for GM food without benefits or a larger WTP value for GM food with benefits, than studies from the rest of the world. For the variable elicitation technique, in some cases predictions hold true, for example, that open-ended questions produce the lowest values, DC CV produce the highest values and iterative bidding techniques provide values somewhere in-between. Overall, results relating to the effect of distribution method, response rate and participant group on mean values are largely inconclusive, depending on which data set is considered. Results relating to year of study do suggest an increase in the value of risk associated with GM food after 1998, but are drawn only from one data set.

The multi-variate analysis confirmed a number of our hypotheses about the influence of independent variables on mean WTP. Overall, the results from our multi-variate analysis, revealed that there was only one significant variable for each of the two data sets. For the ‘WTP for GM without clear benefits’ data set the only variable that was shown to be significant in affecting WTP values was the distribution method. Results revealed that telephone surveys and supermarket surveys elicited greater WTP values than in-person surveys and mail surveys. For our WTP for GM-free data set the only variable shown to be significant was elicitation technique. Results showed that WTP values were higher from DC CV and open-ended CV questions than from choice experiments. In light of our findings we suggest that there is a need for additional research into elicitation techniques, particularly the effect of choice experiments on WTP values.
NOTES

1. This included databases such as EconLit and ArticleFirst and online sources such as the University of Minnesota’s web-based Agecon Search – ‘A full text library of agricultural and applied economics scholarly literature’ (http://agecon.lib.umn.edu/). We also referred to the website of Master-point, a research group at the Department of Spatial Economics of the Free University in Amsterdam (http://www.feweb.vu.nl/en/master-point/), for information on constructing a meta-analysis. In addition we searched the EVRI online database of environmental valuation studies (www.evri.ca). A range of other books, journals and conference papers were also used.

2. A number of studies asked different questions relating to more than one of the data sets.

REFERENCES


Second World Congress of Environmental and Resource Economics, Monterey, California, 24–27 June.


Noussair, C., S. Robin and B. Ruffieux (2002), ‘Do consumers not care about biotech foods or do they just not read the labels?’, *Economic Letters*, 75, 47–53.


## APPENDIX 6.1 STUDY DETAILS

<table>
<thead>
<tr>
<th>Study Number</th>
<th>Year of Study</th>
<th>Country</th>
<th>Response Rate</th>
<th>General Food or GM-specific Survey</th>
<th>Food Item</th>
<th>Details of WTP Question</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>1993</td>
<td>USA</td>
<td>Information not available (student survey)</td>
<td>GM-specific</td>
<td>Meat sandwich</td>
<td>Bid to exchange GM meat sandwich for conventional sandwich</td>
</tr>
<tr>
<td>2</td>
<td>1995</td>
<td>USA</td>
<td>Information not available</td>
<td>General food survey</td>
<td>Milk</td>
<td>WTP a premium for GM-free</td>
</tr>
<tr>
<td>3</td>
<td>1995</td>
<td>Canada</td>
<td>73%</td>
<td>General food survey</td>
<td>General</td>
<td>Extra percentage of weekly food bill to restrict use of biotechnologically derived hormones</td>
</tr>
<tr>
<td>4</td>
<td>1999</td>
<td>Italy</td>
<td>52%</td>
<td>GM-specific</td>
<td>General</td>
<td>Extra for GM food with positive attributes</td>
</tr>
<tr>
<td>5</td>
<td>1999</td>
<td>France</td>
<td>Information not available (auction)</td>
<td>GM-specific</td>
<td>Chocolate bar</td>
<td>Extra for GM food without positive attributes</td>
</tr>
<tr>
<td>6</td>
<td>2000</td>
<td>France</td>
<td>12% France 7% Germany 15% UK</td>
<td>GM-specific</td>
<td>Beef steak</td>
<td>WTP for GM-free</td>
</tr>
<tr>
<td>7</td>
<td>2000</td>
<td>UK</td>
<td>Information not available</td>
<td>General food survey</td>
<td>General</td>
<td>WTP for GM-free</td>
</tr>
<tr>
<td>8</td>
<td>2000</td>
<td>USA</td>
<td>40%</td>
<td>General food survey</td>
<td>Potatoes General</td>
<td>WTP for GM-free</td>
</tr>
<tr>
<td>9</td>
<td>2000</td>
<td>Australia</td>
<td>18%</td>
<td>General food survey</td>
<td>General</td>
<td>Extra for GM food without positive attributes and for GM-free</td>
</tr>
<tr>
<td>10</td>
<td>2001</td>
<td>USA</td>
<td>19%</td>
<td>GM-specific</td>
<td>Bananas</td>
<td>WTP for GM-free</td>
</tr>
<tr>
<td>11</td>
<td>2000?</td>
<td>USA</td>
<td>41%</td>
<td>GM-specific</td>
<td>General</td>
<td>WTP for GM-free</td>
</tr>
<tr>
<td>12</td>
<td>2000?</td>
<td>USA</td>
<td>Information not available (student survey)</td>
<td>GM-specific</td>
<td>Corn chips</td>
<td>Bidding for non-GM corn chips</td>
</tr>
<tr>
<td>13</td>
<td>2001?</td>
<td>USA</td>
<td>7% Des Moines 6% Minneapolis (out of total invited to participate)</td>
<td>GM-specific</td>
<td>Vegetable oil Tortilla chips Russet potatoes</td>
<td>Lower price required before willing to buy GM</td>
</tr>
<tr>
<td>Study Number</td>
<td>Year of Study</td>
<td>Country</td>
<td>Response Rate</td>
<td>General Food or GM-specific Survey</td>
<td>Food Item</td>
<td>Details of WTP Question</td>
</tr>
<tr>
<td>--------------</td>
<td>--------------</td>
<td>---------</td>
<td>---------------</td>
<td>-----------------------------------</td>
<td>-----------</td>
<td>------------------------</td>
</tr>
<tr>
<td>14</td>
<td>2000</td>
<td>France</td>
<td>Information not available (auction)</td>
<td>GM-specific</td>
<td>Soya products</td>
<td>WTP more for ‘sans GM’ and reduction in bids for ‘avec GM’</td>
</tr>
<tr>
<td>15</td>
<td>2001</td>
<td>USA</td>
<td>14%</td>
<td>GM-specific</td>
<td>Rice, Salmon, Cornflakes, Sweetcorn</td>
<td>WTP more for GM rice with positive benefit</td>
</tr>
<tr>
<td>16</td>
<td>2001</td>
<td>USA</td>
<td>27%</td>
<td>GM-specific</td>
<td>Vegetable oil, Salmon, Cornflakes, Sweetcorn</td>
<td>Lower price required before willing to buy GM</td>
</tr>
<tr>
<td>17</td>
<td>2001</td>
<td>USA</td>
<td>Information not available (in-store survey)</td>
<td>GM-specific</td>
<td>Sweetcorn</td>
<td>Both GM and non-GM product offered at same price or GM cheaper or GM more expensive</td>
</tr>
<tr>
<td>18</td>
<td>2001</td>
<td>Japan</td>
<td>50% (50% of shoppers approached)</td>
<td>GM-specific</td>
<td>Noodles, Tofu</td>
<td>Willingness to buy GM at lower price than conventional</td>
</tr>
<tr>
<td>19</td>
<td>2002</td>
<td>Norway</td>
<td>95% (5% of shoppers approached)</td>
<td>GM-specific</td>
<td>Bread, Salmon</td>
<td>Price reduction required to induce consumers to buy GM</td>
</tr>
<tr>
<td>20</td>
<td>2001</td>
<td>Japan, Taiwan, USA</td>
<td>Information not available (student survey)</td>
<td>GM-specific</td>
<td>Vegetable oil</td>
<td>WTP for GM-free</td>
</tr>
<tr>
<td>21</td>
<td>2002</td>
<td>USA, Norway</td>
<td>29% US, 20% Norway (80 per cent of calls rejected)</td>
<td>GM-specific</td>
<td>Soybean oil, GM-fed salmon, GM salmon</td>
<td>Lower price required before willing to buy GM</td>
</tr>
<tr>
<td>22</td>
<td>2003</td>
<td>USA</td>
<td>16%</td>
<td>GM-specific</td>
<td>Tomatoes, Beef</td>
<td>WTP for GM with benefits</td>
</tr>
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</table>
### APPENDIX 6.2 FURTHER STUDY DETAILS

<table>
<thead>
<tr>
<th>Study Number</th>
<th>No. of participants</th>
<th>Participant Group</th>
<th>Survey Distribution Method</th>
<th>Elicitation Technique</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>106</td>
<td>Students</td>
<td>In-person (university)</td>
<td>Auction (Vickrey auction)</td>
</tr>
<tr>
<td>2</td>
<td>702</td>
<td>General population</td>
<td>Telephone</td>
<td>Open-ended CV questions</td>
</tr>
<tr>
<td>3</td>
<td>1240</td>
<td>General population</td>
<td>Telephone</td>
<td>Choice experiment</td>
</tr>
<tr>
<td>4</td>
<td>200</td>
<td>General population</td>
<td>Telephone</td>
<td>Payment card</td>
</tr>
<tr>
<td>5</td>
<td>112</td>
<td>General population</td>
<td>In-person</td>
<td>Auction (Vickrey auction)</td>
</tr>
<tr>
<td>6</td>
<td>1065</td>
<td>General population</td>
<td>Mail</td>
<td>Choice experiment</td>
</tr>
<tr>
<td>7</td>
<td>228</td>
<td>General population</td>
<td>Mail</td>
<td>Choice model and open-ended CV questions</td>
</tr>
<tr>
<td>8</td>
<td>437</td>
<td>Shoppers</td>
<td>Supermarket Mail</td>
<td>Choice experiment and open ended CV question</td>
</tr>
<tr>
<td>9</td>
<td>370</td>
<td>General population</td>
<td>Mail</td>
<td>Choice experiment</td>
</tr>
<tr>
<td>10</td>
<td>116</td>
<td>General population</td>
<td>Telephone</td>
<td>Open-ended CV questions</td>
</tr>
<tr>
<td>11</td>
<td>54</td>
<td>General population</td>
<td>Telephone</td>
<td>Auction (first and second price sealed bid auctions)</td>
</tr>
<tr>
<td>12</td>
<td>50 (32 + 18)</td>
<td>Students</td>
<td>In-person (university)</td>
<td>Auction (random nth price auction – combines Vickrey auction and BDM random price mechanism)</td>
</tr>
<tr>
<td>13</td>
<td>174</td>
<td>General population</td>
<td>In-person</td>
<td>Auction (BDM random price mechanism)</td>
</tr>
<tr>
<td>14</td>
<td>97</td>
<td>General population</td>
<td>In-person</td>
<td>Auction (BDM random price mechanism)</td>
</tr>
<tr>
<td>15</td>
<td>574</td>
<td>General population</td>
<td>Mail</td>
<td>Dichotomous choice CV questions (double-bounded)</td>
</tr>
<tr>
<td>16</td>
<td>141</td>
<td>General population</td>
<td>Mail</td>
<td>Dichotomous choice CV questions</td>
</tr>
<tr>
<td>17</td>
<td></td>
<td>Shoppers</td>
<td>Supermarket</td>
<td>Revealed preference in-store purchase</td>
</tr>
<tr>
<td>18</td>
<td>400</td>
<td>Shoppers</td>
<td>Supermarket</td>
<td>Dichotomous choice CV questions</td>
</tr>
<tr>
<td>19</td>
<td>400</td>
<td>Shoppers</td>
<td>Supermarket</td>
<td>Dichotomous choice CV questions</td>
</tr>
<tr>
<td>20</td>
<td>617</td>
<td>Students</td>
<td>In-person (university)</td>
<td>Dichotomous choice CV questions</td>
</tr>
<tr>
<td>21</td>
<td>450</td>
<td>General population</td>
<td>Telephone</td>
<td>Choice experiment</td>
</tr>
<tr>
<td>22</td>
<td>161</td>
<td>General population</td>
<td>Mail</td>
<td>Double-bounded CV questions</td>
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APPENDIX 6.3  RUGPLOTS OF MEAN WTP FOR GM WITHOUT CLEAR BENEFITS BY DISTRIBUTION METHOD
APPENDIX 6.4  BOXPLOTS OF MEAN WTP FOR GM-FREE BY ELICITATION TECHNIQUE
7. Valuing water quality changes in the Netherlands using stated preference techniques

Roy Brouwer

INTRODUCTION

This chapter provides an overview of the use of stated preference valuation methods in Dutch water policy and management. A number of large-scale contingent valuation (CV) surveys have been carried out since 2000, which aimed to inform public policy and decision-making at national level related to the revision of the European Bathing Water Directive (BWD), contaminated sediment clean-up and the implementation of the European Water Framework Directive (WFD). At regional level also, the ecological restoration of lakes in one of the most important recreational lake districts in the Netherlands was informed by a large-scale CV study. The use and acceptability of large-scale social surveys in the domain of water, including questions related to public willingness to pay (WTP), has increased significantly in the Netherlands over the past five years. The recent popularity of cost–benefit analysis (CBA) in water policy and decision-making, especially flood control policy (Brouwer and Kind, 2005), has certainly played a role in this. In Europe, the implementation of the WFD is expected to provide an important further impetus in view of the Directive's cost recovery requirements related to water use and water services. Also the emphasis in the WFD on public participation is expected to have a positive effect on the use of social survey methods to elicit public opinions and perceptions towards socially acceptable levels of water quality.

This chapter's main objective is to provide an overview of the economic values obtained in a number of large-scale stated preference surveys over the past three years and discuss the main findings. Currently, the use of stated preference surveys in Dutch national environmental policy is limited. Examples exist elsewhere in Europe where the inclusion of non-market values in public decision-making, and CBA more particularly, resulted
in significant protest and consequently some reflection about CBA as a decision support tool (e.g. ENDS, 1998; Moran, 1999; Bateman et al., 2000). In order to arrive at useful, authorized values in CBA, economic values have to be accepted by scientists, policy- and decision-makers and stakeholders involved, that is, those affected by public policy or decision-making and bearing the costs and benefits. An important first step in this authorization process is the institutional and political embedding of such values in official public policy documents.

The studies presented in this chapter were all commissioned by the responsible water authority. Although not sufficient, this in itself is an essential step towards their authorized use. In this chapter, the authorization process is mainly considered from the point of view of those who participated in the social surveys, that is, the public at large, as opposed to policy-maker authorization by explicitly assessing the validity, reliability and acceptability of the obtained values as a separate step in the survey design. The next section first provides a brief overview of the various studies presented in this chapter, including the policy context in which they are used, followed by a discussion of their main findings. The fourth section presents the results of a number of important validity and reliability checks, followed by the conclusions.

OVERVIEW OF STATED PREFERENCE STUDIES USED TO VALUE WATER QUALITY CHANGES

**Economic Valuation of Ecological Rehabilitation of the Frisian Lake District**

In August 2002, 500 face-to-face on-site interviews were conducted with local residents and visitors to one of the most important recreational lake districts in the Netherlands, the Frisian lake district in the north-east of the Netherlands. The Frisian lakes attract more than 700 000 domestic and foreign visitors annually and consist of more than 10 000 hectares of interconnected lakes and watercourses. Tourism and water-based recreation are the area’s most important sources of income.

Since 1970 the surface water levels in this area are maintained at an artificial level of 0.52 metres below sea level. This has had positive effects on agriculture, but detrimental effects on the area’s floodwater and nutrient retention capacity, natural habitat and biodiversity. The area’s infrastructure (e.g. height of bridges and the construction of houses and recreational bungalow parks near lakes) and commercial and recreational shipping (e.g. size of boats, accessibility of water courses and lakes) have fully adapted to
Natural resources

this situation over the past 35 years. Both the quantity and quality of reed and other helophytes have decreased (Iwaco, 2002). As a consequence, the aquatic ecosystem’s nutrient retention capacity has deteriorated, creating a favourable habitat for algae and fish species such as the freshwater bream, and less favourable living conditions for zooplankton, fish species like the pike, reed-dependent bird species like the reed warbler, and the otter (Coops, 2002).

To restore the area’s natural habitat and aquatic ecosystem functioning, a number of studies have proposed to introduce more flexible and sustainable water level management regimes (Coops, 2002; Claassen and Ietswaart, 2003). However, the costs of the associated compensatory measures to ensure that current living and working conditions are also sustained under such a management regime are considerable, ranging between €65 and €335 million (Wetterskip Fryslân, 2003). In order to support policy and decision-making regarding more flexible and sustainable water level management in the area, a pre-feasibility CBA was carried out (Brouwer et al., 2004). As part of this CBA, a cross-section of 170 local residents and 330 visitors were asked for their opinion, preferences and valuation – through WTP in an open-ended question – of two possible future situations in the lake district with different impacts on the area’s natural and living and working conditions (agriculture, living, commercial and recreational shipping): the expected future situation when the current constant water level of 0.52 metres below sea level is maintained or the expected future situation when water levels are allowed to fluctuate by approximately 0.40 metres (in winter 0.32 metres and in summer 0.72 metres below sea level). These two possible future situations were visualized with the help of carefully selected photographs provided by the area’s responsible water management authority (waterboard).

Economic Valuation of Bathing Water Quality Improvements as Proposed by the Revised European Bathing Water Quality Directive

In December 2002, a mail questionnaire was sent to 5000 randomly selected households in the Netherlands, asking for their perception and valuation of improved bathing water quality (BWQ) through a dichotomous choice WTP question. There are over 600 official bathing locations in the Netherlands where BWQ monitoring takes place during the bathing season (May–August). Non-compliance is currently limited: less than 5 per cent of the bathing sites are unable to comply with current standards for *Escherichia coli* and intestinal *Enterococci*. However, proposed new BWQ standards by the European Commission in a revised version of the original 1976 BWQ Directive (76/160/EEC) are expected to result in a substantial increase in the number of non-complying bathing sites to more than 30 per cent. Most
of these sites (> 95 per cent) concern inland waters, only a few are coastal bathing locations. At these sites, measures will have to be taken in order to comply with the new BWQ standards.

Improving BWQ is expected to have substantial recreational benefits. The estimated number of people swimming at non-complying sites (based on the proposed new BWQ standards) on a summer’s day is about 125,000. The health risks of bathing in open waters can be reduced by 50 per cent. Currently, on average one in every ten bathers runs a risk of getting one or more of the following health symptoms when BWQ standards are not met: infections to eyes, ears and throat and stomach upset (gastroenteritis) such as diarrhoea. Meeting the proposed new BWQ standards means that the health risks of bathing are reduced on average to one in every 20 bathers. The above-mentioned health risks are especially high when swimming, for example, during a hot day directly after heavy rainfall causing storm water overflow at or near bathing locations (i.e. discharge of excess rainwater together with untreated sewerage) or when swimming in standing waters with increased algae blooms during hot weather periods.

In order to support policy and decision-making regarding the revision of the existing BWQ Directive and the setting of more stringent standards for bacteriological water contamination, the extent and cause of the problem was investigated and measures identified in order to resolve expected future problems with BWQ. The costs and effectiveness of these measures were estimated along with the least cost way to achieve the new standards. The socio-economic benefits of the new standards were assessed in a large-scale household contingent valuation survey. A distinction was made between coastal water quality and freshwater quality as this appeared to result in significant differences in perceptions during the pre-testing. The costs and benefits were subsequently compared in order to assess the economic net benefits of the new proposed standards in a pre-feasibility CBA (Brouwer and Bronda, 2005).

**Economic Valuation of Good Water Status as Prescribed by the European Water Framework Directive**

In October 2003, another 5000 questionnaires were sent to a random selection of households, asking them about their knowledge, perception and attitudes towards current water quality in the Netherlands and the introduction of future water quality standards as a result of the implementation of the European Water Framework Directive (WFD) (2000/60/EC). Again, a distinction was made between coastal waters and inland freshwater bodies.
The WFD’s main objective is to improve the chemical and ecological status of European water bodies by reducing or eliminating the emission of polluting substances into these water bodies. By 2015 water bodies have to be in a so-called ‘good’ chemical or ecological state. If good status cannot be achieved, Member States can apply for derogation as detailed in Article 4 of the WFD. This may be for technical reasons (e.g. irreversible hydro-morphological changes of water bodies resulting in artificial or heavily modified water bodies such as polders or high natural background levels of arsenic in groundwater bodies or phosphorous in surface water bodies), or for economic reasons when the costs of additional measures to reach good status are considered disproportionately high.

The costs for water quality maintenance and improvement in the Netherlands currently total almost €3 billion annually (author’s own calculation). These costs mainly refer to the operation and maintenance costs incurred by waterboards, local governments (municipalities), agriculture and industry for wastewater collection and wastewater treatment, but also include ecological restoration projects of water bodies. Dutch households pay, on average, €200 per year for wastewater collection and treatment (author’s own calculation based on data provided by Statistics Netherlands). Between 2005 and 2010, a further increase in household taxes and levies related to wastewater collection and treatment is expected between 1.5 and 3 per cent annually (Gerritsen and Sterks, 2004). The large-scale survey aimed to assess public WTP for additional water quality improvements in the Netherlands (Brouwer, 2004a). This social value serves as a benchmark for future price and tax increases related to water quality improvements as a result of the implementation of the WFD.

**Economic Valuation of Contaminated Sediment Clean-up and Biodiversity Conservation**

In May 2004, 5500 questionnaires were sent to randomly selected Dutch households, asking them for their knowledge, attitudes and preferences to clean up the stock of contaminated sediments accumulated in watercourses (rivers, lakes etc) and protect and preserve aquatic biodiversity at contaminated sites. Approximately 9.5 million cubic metres (m$^3$) of sediments enter the Netherlands annually through the international rivers Rhine, Meuse and Scheldt (Osté, 2004). These sediments either end up in the North Sea or accumulate in the lower river delta, including sensitive natural areas such as the Biesbosch National Park (9000 hectares of wetlands including the largest bird population of blue throats in Western Europe). During the 1980s, it became clear that most of these sediments are contaminated with heavy metals and other toxic substances. Although enormous efforts have been
undertaken since then to clean up the stock of accumulated contaminated sediments (annually 19 million m$^3$ of sediments are dredged in coastal waters and more than 14 million m$^3$ in fresh inland waters), the accumulated stock of sediments has grown to 107 million m$^3$, of which 50 million m$^3$ are heavily contaminated. The stock of contaminated sediments has severe detrimental effects on the quality and diversity of plant and animal species in and around watercourses. Water quality is affected too and hence ultimately the entire food chain is at risk.

In 2003 the Ministry for Transport, Public Works and Water Management commissioned a study to assess the public benefits of increased and more rapid clean-up of the accumulated stock of contaminated sediments in Dutch water courses. The main objective of this study was to facilitate public policy and decision-making by explicitly addressing both costs and benefits of contaminated sediment clean-up. The costs of cleaning up the total stock of contaminated sediments are €2.3 billion (Kind et al., 2004). This includes the additional costs for dredging in order to reach an equilibrium between sediments entering the water system and sediments taken out of the water system each year.

Besides an evaluation of the benefits for commercial shipping, flood control, agriculture, recreation, fishery and human health, the ecological benefits were also assessed through expert models and judgement and a large-scale public survey investigating public perception and valuation of the benefits of contaminated sediment clean-up in and around sensitive natural areas in the Netherlands like the Biesbosch (Brouwer, 2004b).

**MAIN FINDINGS**

A summary of the main findings of these studies is presented in Table 7.1. All studies were thoroughly pre-tested through face-to-face interviews by professional interviewers before the final survey was implemented. Each questionnaire contains on average about 40 questions. Most questions (about 75 per cent) are closed (i.e., using predetermined answer categories). The information included in each study was carefully selected together with scientific experts (in some case up to ten different experts) and subsequently pre-tested and transformed by professional interviewers into a clear and understandable language for the lay public.

In all CV studies general taxation was used as the payment vehicle. In the first and third study in Table 7.1, respondents were also asked for their preferences for other payment vehicles (e.g. price of tap water, waterboard levy, local municipality tax etc) and whether they would be willing to pay if another payment mechanism were used. As expected, public WTP is
Table 7.1  Summary of large-scale stated preference studies related to water

<table>
<thead>
<tr>
<th>Study and Study Year</th>
<th>Public Good Benefits</th>
<th>Estimated Value</th>
<th>Original Sample Size</th>
<th>Target Population</th>
<th>Survey Format</th>
<th>Response Rate (%)</th>
<th>Average WTP(^a) (€/household/year)</th>
<th>Elicitation Format(^b)</th>
<th>Payment Vehicle</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. CV ecological rehabilitation Frisian lake district (2002)</td>
<td>Biological diversity and water recreation</td>
<td>Use and non-use value</td>
<td>670</td>
<td>Residents and visitors</td>
<td>Face to face</td>
<td>75</td>
<td>75 (58–93)</td>
<td>OE</td>
<td>General taxation</td>
</tr>
<tr>
<td>3. CV good water status EU WFD (2003)</td>
<td>All water functions (drinking water, water recreation, nature conservation etc)</td>
<td>Use and non-use value</td>
<td>5000</td>
<td>Households</td>
<td>Mail</td>
<td>27</td>
<td>105 (74–136)</td>
<td>DC</td>
<td>General taxation</td>
</tr>
<tr>
<td>4. CV biodiversity conservation and contaminated sediment clean-up (2004)</td>
<td>Biological diversity and human health</td>
<td>Use and non-use value</td>
<td>5500</td>
<td>Households</td>
<td>Mail</td>
<td>18</td>
<td>70 (51–88)</td>
<td>OE</td>
<td>General taxation</td>
</tr>
</tbody>
</table>

Notes:
\(^a\) 95 per cent confidence intervals around estimated mean value between parentheses.
\(^b\) OE: Open-ended; DC: dichotomous choice; PC: payment card.
highest, on average, for a programme of measures that aims to improve water quality in the Netherlands for all water-related activities or functions (including drinking water, recreation, nature conservation etc) as foreseen in the WFD. The water quality improvements proposed in the WFD cover the social benefits derived from all water-quality dependent aquatic ecosystem functions. On average Dutch households are willing to pay a maximum of 20 per cent for such extensive water quality improvements over and above what they currently already pay for drinking water, sewage collection and sewage treatment. No significant differences can be detected between average WTP in the two main river basins in the Netherlands Rhine and Meuse. An interesting finding is that a majority of households (61 per cent) have no clue how much they are currently paying for their wastewater collection and treatment. Of those households who state a money amount for their current water bill, more than 40 per cent say they were only guessing. One-third of all Dutch households also do not know how they are paying for wastewater collection and treatment or who they are paying for these water services.

For biodiversity conservation and contaminated sediment clean-up, average WTP varies between €50 and €70 per household per year depending on the elicitation format used (open-ended formats producing a significantly higher average WTP than the use of a payment card or a dichotomous choice question). This seems quite high. However, an important reason for this relatively high WTP is that 45 per cent of the respondents indicated that their WTP also relates to concerns about the health risks of contaminated sediments.

Average WTP is lowest for bathing water quality improvements (€35 per household per year). An interesting finding in this study is that bathers are willing to pay significantly more than people who never bathe in open waters in the Netherlands (coastal or inland waters). A similar result is found in the third study in Table 7.1 where people who take recreation in or near water are willing to pay significantly more than people who do not. Bathers are willing to pay, on average, approximately €40 annually to improve bathing water quality, while people who never bathe in open water in the Netherlands are willing to pay about €20 annually on behalf of their entire household.

A remarkable finding is furthermore that people perceive coastal water quality and inland freshwater quality to be significantly different. The quality of coastal bathing water is generally judged to be superior to that of inland freshwater bodies. On average, a third of the Dutch population believes that fresh inland waters are of poor quality and one in every five households believes this about coastal water. Vice versa, a third is convinced that coastal water is of good quality whereas 20 per cent think that fresh inland water bodies are of good quality. A large majority of people (> 90 per cent) state
that overall water quality should be improved in the Netherlands and consider this an important policy objective. When asking for the public’s perception of water quality changes over the past decade, most people (43 per cent) believe that freshwater quality has improved. Almost half as many people believe that coastal water quality has improved (22 per cent). A minority of 15 per cent of the Dutch population believe that fresh and coastal water quality has deteriorated over the past ten years. Slightly more people believe that the situation has remained the same for coastal water. Remarkable is the large number of people who said that they do not know whether water quality has changed (27 and 40 per cent for fresh and coastal water quality respectively). In reality, the concentration of nutrients and heavy metals in both coastal and inland surface waters has decreased significantly over the past 15–20 years (Ministerie van Verkeer en Waterstaat, 2004).

Compared with public WTP for all aquatic ecosystem functions as proposed in the third study in Table 7.1, the results found in Friesland for the ecological rehabilitation of the Frisian lakes are also high (average WTP is about €75/household/year). The relatively high average WTP value may partly be due to the use of the OE elicitation format. Contrary to previous studies (e.g. Bateman et al., 1995), tests carried out in the fourth study in Table 7.1 show that the OE format produces a significantly higher average WTP than the use of a payment card or a dichotomous choice WTP question. According to Desvousges et al. (1983), the OE format tends to produce a larger number of non-responses and protest responses compared with the other formats. An important reason for the high average OE WTP value found here is the uncertainty experienced by respondents in answering the OE question compared with the two other elicitation formats. Uncertainty was measured through the mean WTP’s standard error and the uncertainty respondents experienced when answering the WTP question measured on a scale from 0–100 per cent. The relatively high WTP amount is furthermore supported by the outcome of the travel cost study that was carried out together with the CV survey. The average economic value per visit (consumer surplus) is as high as €50 based upon the estimated individual travel cost model excluding the opportunity costs of travel time. This value is even higher when including the opportunity costs of travel time.

In order to test the temporal stability of the WTP responses in the first study in Table 7.1, 217 of the 500 respondents were contacted again two weeks later by telephone and asked whether they 1. remembered their stated WTP, 2. still agreed to pay this amount of money and 3. wanted to change their stated WTP. Two-thirds still remembered their stated WTP and 95 per cent of all 217 respondents stuck to their originally stated WTP. Hence, overall the stated WTP amounts are stable over this period of time, adding
to the reliability of the outcome. Only 5 per cent \((n = \text{ten respondents})\) changed their stated WTP during the follow-up telephone interview. Eight of these ten respondents said they want to pay less than they originally agreed to pay (of which two wanted to pay nothing anymore mainly because they felt they pay enough taxes already), while the two other respondents wanted to pay more.

**ASSESSING THE ACCEPTABILITY OF STATED PREFERENCE VALUES**

Each of the studies was scrutinized for its validity and reliability in a number of ways. Internal consistency and validity was thoroughly tested through conventional statistical analysis, confirming a priori expectations regarding the direction of influence of statistically significant explanatory factors of stated WTP such as bid and household income level. The explanatory power of the estimated models varies between 7 and 62 per cent.\(^1\) Consistency and validity was furthermore carefully checked by analysing protest bids and respondent understanding of the information provided in the survey and the WTP question. These results are summarized in Table 7.2.

Protest bidders are respondents who refuse to pay for different provision levels of the public good in question because they disagree with the proposed payment structure for other than a priori expected economic reasons. A high protest rate questions the validity of the valuation design (Arrow et al., 1993). Criteria to determine whether or not a respondent is a legitimate zero bidder to a WTP question or a protest bidder are often arbitrary, if reported at all in CV studies (Brouwer, 2000). No strict guidelines exist regarding acceptable protest rates. The notion of acceptability is highly subjective in this context. However, in order to help policy and decision-makers to judge the validity of CV findings, a protest rate less than 10 per cent should be aimed for in order to produce acceptable (valid and reliable) results. Protest rates between 10 and 20 per cent are still acceptable, but should be interpreted carefully, while a protest rate higher than 20 per cent might invalidate a CV study in my opinion. Clearly, other arbitrary threshold values may also apply depending upon the criteria used by the researcher or policy-maker to identify protest bidders. Judgements also depend on the extent of differences between the characteristics of protest and non-protest sample (i.e., are protest responses random?) More broadly, it is worth noting that assessing validity depends on the wider scrutiny of CV responses than just protests alone.

Examples of a priori expected economic reasons include low or no preference (e.g. respondents who attach no value to the good in question
<table>
<thead>
<tr>
<th>CV Study</th>
<th>Protest Response (%)</th>
<th>Difficult Answering WTP Question? (^a) (%)</th>
<th>Information Supply Sufficient? (^b) (%)</th>
<th>Clear What Exactly You Are Being Asked to Pay For? (^c) (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Ecological rehabilitation Frisian lake district</td>
<td>14.3</td>
<td>70 yes</td>
<td>70 yes</td>
<td>–</td>
</tr>
<tr>
<td>2. Bathing water quality improvements revised EU BWD</td>
<td>8.8</td>
<td>20 yes</td>
<td>93 yes</td>
<td>–</td>
</tr>
<tr>
<td>3. Good water status EU WFD</td>
<td>16.5</td>
<td>20 yes</td>
<td>92 yes</td>
<td>94 yes</td>
</tr>
<tr>
<td>4. Biodiversity conservation and contaminated sediment clean-up</td>
<td>13.5</td>
<td>33 yes</td>
<td>78 yes</td>
<td>96 yes</td>
</tr>
</tbody>
</table>

\(^{a}\) Share of the sample population who said they found it (very) difficult to answer the WTP question (measured on a five-point scale ranging from ‘not difficult at all’ to ‘very difficult’).

\(^{b}\) Share of the sample population who said they felt that the information provided in the questionnaire to answer the WTP question is (more than) sufficient (measured on a four-point scale ranging from ‘more than sufficient’ to ‘completely insufficient’).

\(^{c}\) Share of the sample population who said they know exactly what they are being asked to pay for (measured on a four-point scale ranging from ‘completely clear’ to ‘completely unclear’).
or who have no problem with the current situation), income constraints (respondents who say they have insufficient income to pay) or substitution effects (respondents prefer to spend money on other things than the good in question). The average protest rate of 13 per cent found in the four Dutch water studies reported and discussed here is considered too low to invalidate the results, but do indicate that the results have to be interpreted with the necessary care as approximately one in every eight respondents seems to object against the proposed market construct and valuation scenario. The most important reasons found in all four studies for protest are ‘the polluter should pay’ and ‘I already pay enough taxes’. These are typically reasons that do not say anything about the real value attached to the good in question (positive, zero or negative), but relate to the proposed market construct. Also in some cases respondent distrust that the money would actually be spent on the good in question played a role. However, the two first reasons were predominant in each study, even though it was emphasized for example in some studies that those responsible for the problem would pay based on the Polluter Pays Principle and respondents were offered the opportunity to pay in another way (e.g. through water prices or a one-off donation).

Following the study carried out by Spash and Hanley (1995), a number of additional questions were included in the biodiversity and contaminated sediment clean-up CV study about public perception and belief that biodiversity is an economic good or can be characterized by lexicographic preferences. For this purpose, Dutch households were presented with the following two statements:

Statement 1: Plant and animal species should be protected by law, not by asking people to pay for their protection.
Statement 2: Plant and animal species have a right to be protected irrespective of the costs to society.

Sixty per cent of Dutch households agree with the first statement and 50 per cent with the second statement (both measured on a five-point Likert scale ranging from ‘disagree strongly’ to ‘agree strongly’). These results can be used to indicate that a majority of respondents are expected to protest against a proposed market construct as in the CV study and that half of the respondents exhibit lexicographic preferences (as shown by Spash and Hanley based on a sample of students in their 1995 paper). As expected, statistical tests show that those respondents who said no to the question: ‘Are you willing to pay extra in principle for the protection of biodiversity in and around water?’ agree more with the first statement than those respondents who said yes to this question. On the other hand, those respondents who said yes to the question whether they are willing to pay in principle are
more agreeable to the second statement than respondents who said no to this question. However, 65 per cent of those respondents who agree with the first statement and 85 per cent of the respondents who agree with the second statement nevertheless state a positive WTP without question or protest. Only 20 per cent of those who agree with the first statement were identified as protest bidders and only 6 per cent of those who agree with the second statement.

Other indicators of the studies' validity include questions in the questionnaire about the ease with which respondents answer the WTP question, the amount and quality of the information provided in the questionnaire to answer the WTP question and in the last two studies also how clear respondents are about what exactly they are being asked to pay for (Table 7.2). When looking at the difficulty respondents experienced in answering the WTP question, the open-ended question in the first study was clearly more problematic than the dichotomous choice questions in the other studies. In all studies, the degree of difficulty to answer the WTP question appears to be a significant determinant of stated WTP in the multivariate regression analysis. The more difficulty respondents experience answering the WTP question, the lower, on average, stated WTP.

Finally, a majority of respondents feel that the information supplied suffices to answer the WTP question, while almost all respondents in the last two studies are clear what exactly they are being asked to pay for. These findings are also important additional indicators of the reliability of the CV results.

**DISCUSSION AND CONCLUSIONS**

The results from the stated preference studies presented in this chapter have a strong policy focus and were used in pre-feasibility cost–benefit studies. In the case of the implementation of the WFD in the Netherlands, the costs of additional measures are not yet known, and this study therefore aimed to provide an indicative upper limit for expected future increases in household water taxes as a result of the extra costs of WFD measures. Comparing the estimates for example with the results from a meta-analysis of international wetland contingent valuation studies (Brouwer et al., 1999), the value estimate for overall water quality improvement as proposed in the WFD appears to be fairly similar to the values found in other international studies. The average value in previous wetland valuation studies for water quality control was €90 per household per year.

The use of the estimated values for any purpose other than pre-feasibility CBA like price setting of water services, including their environmental and
resource costs as propagated in the WFD, is still under discussion. One important argument against their use in price setting is that the uncertainty underlying the estimation procedure currently prohibits their use in policy projects where the required degree of accuracy is high. For the calculation of costs of water projects, detailed guidelines and standard procedures have been developed by the central government. In these guidelines, standard cost categories are identified upon which water managers and accountants have to report in the explorative, planning and implementation phase of water-related measures. The required degree of accuracy and confidence of cost estimates differs depending upon the phase in the decision-making cycle. In the explorative phase of the policy or decision-making cycle, the variation coefficient ($\sigma/\mu$) cannot exceed 50 per cent and the required confidence level is 70 per cent, whereas the variation coefficient has to be less than 10 per cent in the implementation phase and the required confidence is 1 per cent (Directorate-General Rijkswaterstaat, 2004). Pre-feasibility cost–benefit studies are typically used in the explorative and planning phase of policy and decision-making.

Applying the existing guidelines regarding the required accuracy of cost estimates to the average benefit (WTP) estimates presented in this chapter (Table 7.1), the latter can be used in principle in the explorative and planning phase of policy and decision-making. In the case of the dichotomous choice estimate for biodiversity and contaminated sediment clean-up the estimate can even be used in the implementation phase of water policy. The variation coefficients of the stated preference-based estimates range between 8.5 (DC biodiversity and contaminated sediment) and 14.9 per cent (DC WFD). Even the open-ended WTP estimates remain under 15 per cent (11.4 per cent in the case of ecological rehabilitation of the Frisian lake district and 13.6 per cent in the case of biodiversity conservation and contaminated sediment clean-up). The variation coefficient of the estimate for bathing water quality improvement is 10.3 per cent. An important question obviously is how much confidence the researcher has in these estimates. Tests of the validity and consistency of the stated preference studies do not result in a rejection of the outcomes and justify the use of the valuation results in the pre-feasibility cost–benefit studies. The generic and spatially non-differentiated nature of the valuation scenarios used in three of the four case studies and the number of protest bidders found in each study, although not alarming, do require careful attention when translating the results into practical policy advice and management guidance. Especially in contexts where the required degree of accuracy is high such as price setting. However, the degree of accuracy and reliability of the results presented here are promising for future applications.
NOTE

1. The adjusted R-squared varied from 7 to 17 per cent in the first study in Table 7.2 (ecological rehabilitation Frisian lake district) while the pseudo R-squared ranged from 16 to 62 per cent in the fourth study in Table 7.2 (biodiversity conservation and contaminated sediment clean-up) depending on statistical model specification. The pseudo R-squared was 28 and 37 per cent in the second (BWD) and third study (WFD) respectively in Table 7.2.

REFERENCES


8. Measuring environmental externalities in the electric power sector

Patrik Söderholm and Thomas Sundqvist

INTRODUCTION

With the growing concerns about the impacts of air pollution in general and climate change in particular, energy policy and environmental policy have become closely integrated. A wide array of different regulations and economic incentives exist worldwide to address environmental problems associated with energy production and use, and promote the introduction of environmentally benign energy generation technologies. However, the implementation of such measures involves tough trade-offs: 1. what technologies should be considered environmentally benign, and 2. how does one identify a proper balance between the benefits of energy production and the costs of environmental degradation? Since environmental costs are not generally reflected in market prices, there exists a need to assist market processes by assigning monetary values to these costs, and integrate them into private and public decision-making.

In the early 1980s studies that attempted to assess and value environmental externalities in the electric power sector began to emerge (Schuman and Cavanagh, 1982). During the 1990s the number of externality analyses surged, in large part due to increased attention from policy-makers in Europe, with the ExternE project (European Commission, 1995; European Commission 1999), and in the US (Resources for the Future, 1994–98; Rowe et al., 1995 and Oak Ridge National Laboratory). The results and the methods used in these externality studies have been utilized as inputs in important modelling work and have served as vehicles in developing additional methodological work in the environment and energy field (Krewitt, 2002). For instance, studies on how different environmental regulation schemes affect national energy systems have made use of external cost estimates, as have a number of cost-benefit analyses of environmental policy proposals. However, the results from previous studies have affected
actual policy decisions only to a limited extent. Some authors argue that this is because electricity externality studies may have raised more questions than they have answered, and that there exist important limits to their usefulness in deriving policy-oriented recommendations (e.g. Stirling, 1997), while others (e.g. Pearce, 2002) stress a number of important policy uses for externality studies.

The main purpose of this chapter is to provide a brief survey of the approaches and the results of previous electricity externality studies, and analyse the usefulness of these valuation efforts for policy-making. The chapter addresses some important policy implications that follow from previous studies, but also discusses a number of remaining challenges for the effective integration of externality valuation efforts into the energy and environmental policy decision processes.

In contrast to earlier survey studies of the assessment of external costs in the power industry (see, for instance, Office of Technology Assessment, 1994; Kühn, 1996, 1998; Lee, 1997; Ottinger, 1997; Schleisner, 2000), which typically focus on the procedure of generating externality valuation estimates, we focus explicitly on the use of these estimates in policy-making. Moreover, while earlier surveys typically focus on a few selected studies we consider the results, methods and scope of more than 40 different externality studies. This permits us to draw more general conclusions about the overall results, policy implications as well as the limits of the work conducted in this research field.

Before proceeding, one important limitation of the analysis must be emphasized. In the past, external cost estimates (so-called ‘externality adders’) were largely intended for use in publicly owned utilities. Theoretically, externality studies permit the full social costs of different power generation technologies to be assessed, and for a public utility these could ideally be used to plan future capacity in a way that meets projected electricity demand at minimum social cost. However, in line with recent restructuring of electricity markets worldwide we focus primarily on the potential use of external cost estimates in a privatized and deregulated market. As we will see, in such a market environment the policy relevance of externality assessments is by no means weaker, but new important problems and issues also emerge.

THE ASSESSMENT OF ENVIRONMENTAL EXTERNALITIES IN THE POWER SECTOR

The theoretical basis of the economic valuation of environmental externalities is outlined in the welfare economics literature. This strand of
research recognizes that the economic value of a resource or service is a function of individual human preferences, and the tool for analysing welfare impacts is therefore utility theory. People are assumed to seek to satisfy their preferences, which are exogenously determined, complete, continuous and ethically unchallengeable. The environment is essentially treated as any other commodity, and people are generally willing to consider trade-offs in relation to the quantity or the quality of environmental ‘goods’. According to the welfare economic theory the appropriate role of policy in the field of energy-related externalities is to make sure that an optimal balance is struck between the monetary estimates of individual preferences for environmental quality and other (more tangible) economic benefits and costs. In sum, the economics of non-market valuation builds on: 1. clear and useful – but also relatively restrictive – behavioural assumptions; 2. a sense of society as the sum of the preferences of its individual members; and 3. a view of the task of public policy involving the internalization of external costs and benefits, and with utilitarianism as the ethical principle guiding social choice. In practice there are two basic methodological approaches used for the valuation of external costs in the energy sector: the abatement cost approach and the damage cost approach (Sundqvist and Söderholm, 2002).

The abatement cost approach uses the costs of controlling or mitigating damage or the costs of meeting legislated regulations as an implicit value of the damage avoided. The rationale behind this approach is that legislatures are assumed to have considered the willingness of the public to pay for alleviation of the damage in setting the standard, thus providing a revealed preference damage estimate no less reliable than the more direct valuation methods. Pearce et al. (1992) stress that one of the serious caveats related to this approach is that it relies on the strong assumption that these same decision-makers make optimal decisions, that is, they know the true abatement and damage costs. Another limitation of the abatement cost approach is that society’s preferences are assumed to change only as policies change. Hence, past revealed preferences might bear little relation to actual impacts today and their current value to society. For instance, the implicit value of CO₂ emissions indicated by a revealed preference analysis would in many cases be very low since there still exist relatively few regulations targeted towards this problem.¹ This built-in ‘tautology’ of the approach means that estimates need to be constantly revised as regulations and policies change, and since policy is (per definition) optimal the analysis provides no room for relevant policy implications. One must therefore question why the analysis is needed in the first place.

The damage cost approach aims at measuring the net economic damage arising from negative externalities by focusing more or less directly on explicitly expressed preferences. This approach can be subdivided into two
Measuring environmental externalities in the electric power sector

main categories: top-down and bottom-up. Top-down approaches make use of highly aggregated data to address the external costs of, say, particular pollutants. Top-down studies are typically carried out at the national or the regional level, using estimates of total quantities of pollutants and estimates of total damage caused by the pollutants. Specifically some estimate of national damage is divided by total pollutant depositions to obtain a measure of physical damage per unit of pollutant. These physical damages are then attributed to power plants and converted to damage costs using available monetary estimates on the damages arising from the pollutants under study. The main criticism of the top-down approach is that it is generic in character and does not take into account impacts that are site-specific; nor does it address the different stages of the fuel cycle. Another criticism is that the approach is derivative since it depends mostly on previous valuation estimates and approximations.

In the bottom-up approach, damages from a single source are typically traced, quantified and monetized through damage functions/impact pathways (see Figure 8.1). This method makes use of technology-specific data, combined with dispersion models, information on receptors, and dose-response functions to calculate the impacts of specific externalities. The bottom-up approach has also been criticized, primarily on the ground that application of the method has unveiled a tendency for only a subset of impacts to be included in the assessment (focusing on areas where data is readily available and where, thus, impact pathways can easily be established). Moreover, Bernow et al. (1993) caution that the bottom-up approach relies on models that may not adequately account for complexities in ‘the real world’, especially noting that there may be synergistic effects between pollutants and environmental stresses, and that there may be problems in establishing the timing of effects (i.e. between exposure and impact). Still, this is the approach that, due to its focus on explicit estimates of economic welfare, appears to be most in line with economic theory.

There exist several ways of placing a monetary value on externalities. Since externalities, by definition, are external to markets, impacts from externalities are often not reflected in existing market prices. Consequently any attempt to monetize an externality using bottom-up damage costing needs to rely on impact valuation methods. These methods can be subdivided into direct and indirect methods. Even if no information about public values is available from existing markets, it may be possible to derive values using direct methods that simulate a market. These methods are direct in the sense that they are based on direct questions about – or are designed to elicit – people’s preferences through willingness to pay (WTP) measures. An important advantage of the direct methods is that they can assess total economic values, that is, use as well as non-use values (such as
Natural resources

Well-known direct valuation methods include contingent valuation and stated preference methods (e.g. choice experiment). None of the indirect methods can assess non-use values; they are based on the actual (rather than hypothetical) behaviour of individuals. Either the external effects show up as changes in costs or revenues on observable markets or in markets closely related to the resource that is affected by the externality. The damage is thus valued indirectly using a relationship between the externality and some good that is traded in a market. Examples of indirect valuation methods are hedonic pricing, travel costs and replacement costs. Garrod and Willis (1999) provide a nice overview of the most commonly used methods to value the economic impacts of environmental change.

There exist also ‘methods’ that do not easily fit into the categories discussed above, but which may nevertheless prove useful. One important example is so-called benefit transfer, which does not involve any valuation


Figure 8.1 The impact pathway (bottom-up approach)
in itself. It instead makes use of the results of previous studies that have derived monetary estimates for the externality of interest. That is, a study may utilize the results from another valuation study and adjust them for use in the chosen context. Producing reliable benefit transfers is clearly an important, but also a difficult task (see, for instance, Smith et al., 2002).

AN OVERVIEW OF PREVIOUS ELECTRICITY EXTERNALITY STUDIES

As was noted above, a considerable number of externality studies were carried out during the 1980s and 1990s. The focus in this survey is on studies whose aim has been to assess the total external costs per kWh of different electric power technologies. Table 8.1 provides an overview of about 40 externality studies covered in the analysis. All monetary estimates presented in this chapter have been converted into US dollars (1998) using mean exchange rates and the US Consumer Price Index. This process has not always been straightforward since the base years used in the different studies are not always explicitly stated. Whenever this problem arose, the year of publication was used as a proxy for conversion. An inspection of the different externality assessments laid out in Table 8.1 reveals several conceptual issues of importance, out of which five will be stressed here.

First, most of the fuel sources available for power generation have been addressed in previous valuation efforts, including coal, oil, natural gas, nuclear, hydro, wind, solar, biomass and in a few cases lignite, waste incineration, geothermal, peat and orimulsion. However, most studies focus on the traditional fuels, such as coal and nuclear. There is thus a tendency that many studies address existing technologies rather than the technologies generally believed to play a significant role in the future (i.e. wind, biomass, solar etc). In many cases this is understandable given that empirical data clearly are more available for the existing (rather than for the emerging renewable) technologies. Nevertheless, an important goal of externality valuation in the power sector has been to 'level the playing field' in the selection between traditional and new generating technologies, and this would probably require a stronger focus also on promising but not yet commercialized technologies. Moreover, in order to set environmental priorities right there also exists a need to compare in more detail the external costs of the different renewable energy technologies.

Second, a majority of the studies have been carried out for the developed world (mostly for Western Europe and the US). Thus, only in some rare cases the focus has been on developing countries where the need for additional power capacity is by far the greatest (IEA, 1998). There are
### Table 8.1 Overview of previous externality studies

<table>
<thead>
<tr>
<th>Study</th>
<th>Country</th>
<th>Fuel</th>
<th>External Cost Range (US cents/kWh)</th>
<th>Method</th>
</tr>
</thead>
<tbody>
<tr>
<td>Schuman and Cavanagh (1982)</td>
<td>US</td>
<td>Coal</td>
<td>0.06–44.07</td>
<td>Abatement cost</td>
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<td></td>
<td></td>
<td>Nuclear</td>
<td>0.11–64.45</td>
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<tr>
<td></td>
<td></td>
<td>Solar</td>
<td>0–0.25</td>
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<tr>
<td></td>
<td></td>
<td>Wind</td>
<td>0–0.25</td>
<td></td>
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<tr>
<td>Hohmeyer (1988)</td>
<td>Germany</td>
<td>Fossil fuels</td>
<td>2.37–6.53</td>
<td>Damage cost (top-down)</td>
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<tr>
<td></td>
<td></td>
<td>Nuclear</td>
<td>7.17–14.89</td>
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<td></td>
<td></td>
<td>Wind</td>
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<td></td>
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<td></td>
<td></td>
<td>Oil</td>
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<tr>
<td></td>
<td></td>
<td>Gas</td>
<td>1.75–2.62</td>
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<tr>
<td>Bernow and Marron (1990);</td>
<td>US</td>
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<td></td>
<td>Gas</td>
<td>2.10–7.98</td>
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<td>Hall (1990)</td>
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<td>2.37–3.37</td>
<td>Abatement cost</td>
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<td>Friedrich and Kallenbach</td>
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<td>Coal</td>
<td>0.36–0.86</td>
<td>Damage cost (bottom-up)</td>
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<tr>
<td>(1991); Friedrich and Voss</td>
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<td>Nuclear</td>
<td>0.03–0.56</td>
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<td>(1993)</td>
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<td>Waste</td>
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<td>Abatement cost</td>
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<td>Method</td>
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also reasons to believe that externality estimates should differ substantially between developing and developed countries. In the former countries incomes are lower, and the environmental effects of power production may be fundamentally different. An important example of the latter is the environmental externalities stemming from hydropower. Hydroelectric development in a temperate climate may give rise to global warming impacts due to the mouldering of vegetation left in the reservoir, while hydroelectric development in colder climates will not (Moreira and Poole, 1993). This raises concerns about transferring values from studies conducted in, say, Western Europe, for use in a developing country context.

Third, examining the methodologies utilized over time reveals that over time the bottom-up damage cost approach appears to have become the dominant paradigm, while the abatement cost and top-down approaches were mostly used in the 1980s and early 1990s (see Figure 8.2). An important reason for this development is that the national implementation phase of the ExternE project (European Commission, 1999) relies solely on damage cost bottom-up models, and these studies together represent a large share of the total number of projects conducted during the latter part of the 1990s. This also indicates, however, that the bottom-up model has been accepted as the most appropriate method with which to assess power generation externalities. The ExternE project has largely served as a vehicle in the methodological development of externality valuation in the electricity and transport sectors. The scientific quality of the ExternE work as well as the methodologies used has been well accepted at the international level, and many followers rely heavily on the numbers and the methods presented in ExternE publications (Krewitt, 2002).

However, the above development also raises the question of whether the choice of methodological approach (between, say, abatement costs and damage costs) matters for the results. By using the results from over 40 electricity externality studies across eight fuel sources Sundqvist (2004) provides econometric evidence in support of the conclusion that methodological choice does matter for the results. He reports that the probability of obtaining a ‘low’ externality value is, ceteris paribus, generally lower when the abatement cost approach or the top-down damage cost approach are used while the opposite is true for the bottom-up damage cost approach. Joskow (1992) provides one important explanation for why the abatement cost approach tends to produce relatively high external damage estimates. Abatement costs will (theoretically) be representative of damage costs if and only if they are derived from the least cost strategy, but normally most studies employ the most commonly used (or mandated) abatement technology when assessing pollution abatement costs. For example, in deriving the external damages from SO₂ emissions in the USA Bernow et al.
Natural resources

(1991) make use of the costs for installing scrubbing equipment. At the time (1990–91) this indicated a cost per tonne of SO$_2$ of about USD 1500–2000, and this estimate corresponded fairly well to the projected prices of future SO$_2$ emission allowances in the tradable permit system soon to becoming implemented in the USA. However, the actual prices of SO$_2$ allowances for most of the period 1992–97 varied between USD 100 and USD 200 per tonne (Schmalensee et al., 1998), indicating that the compliance costs have been much lower than originally expected.

In practice, many of the US coal-fired plants chose to rely on low-sulphur coal in their production rather than to invest in scrubbers. Technical progress in the abatement technology field also contributed to lower sulphur prices. Thus, the failure of previous studies to identify the least cost abatement technologies or strategies tends to lead to an exaggeration of the damage costs involved.

One reason for why the top-down approach also tends to produce relatively high external damage estimates is that there may raise practical problems in attributing the ‘exact’ damage to each individual source, and the researcher is forced to rationalize and use standardized ‘rules’ for the attribution process. These ‘rules’ may fail to ascribe the aggregate damage to each and every individual source, especially smaller sources, thus producing estimates for specific power plants that are positively biased since these

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**Figure 8.2  Methodological choice over time**

(1991) make use of the costs for installing scrubbing equipment. At the time (1990–91) this indicated a cost per tonne of SO$_2$ of about USD 1500–2000, and this estimate corresponded fairly well to the projected prices of future SO$_2$ emission allowances in the tradable permit system soon to becoming implemented in the USA. However, the actual prices of SO$_2$ allowances for most of the period 1992–97 varied between USD 100 and USD 200 per tonne (Schmalensee et al., 1998), indicating that the compliance costs have been much lower than originally expected.

In practice, many of the US coal-fired plants chose to rely on low-sulphur coal in their production rather than to invest in scrubbers. Technical progress in the abatement technology field also contributed to lower sulphur prices. Thus, the failure of previous studies to identify the least cost abatement technologies or strategies tends to lead to an exaggeration of the damage costs involved.

One reason for why the top-down approach also tends to produce relatively high external damage estimates is that there may raise practical problems in attributing the ‘exact’ damage to each individual source, and the researcher is forced to rationalize and use standardized ‘rules’ for the attribution process. These ‘rules’ may fail to ascribe the aggregate damage to each and every individual source, especially smaller sources, thus producing estimates for specific power plants that are positively biased since these
Measuring environmental externalities in the electric power sector

plants, normally, are easily identifiable and represent significant sources of pollution.

Fourth, Figure 8.3 is based on the results from 63 externality studies and the numbers in brackets show the total number of observations for each fuel source. These studies include those outlined in Table 8.1 and a number of additional studies that are not presented here in detail (see, however, Sundqvist, 2000). Most of the latter observations build on secondary sources in which the details (i.e. methodology, scope etc) of the studies are not reported. As can be seen in Figure 8.3, the disparity of external cost estimates is considerable when compared across different studies (note the use of logarithmic scale). The ranges also intertwine making the ranking of various fuels with respect to externality impacts a difficult task. Still, some tentative conclusions can be drawn. For instance, the results suggest that fossil fuel fired power, in particular coal and oil, tend to give rise to the highest external costs, while some of the renewable energy sources, solar, wind and also hydropower, appear to incur the lowest. However, even among the renewable power sources the external costs can differ a lot.
and we discuss some of the implications of this in the next main section of the chapter.

Figure 8.3 also shows that even for a specific fuel source the difference between low and high values is substantial and this is also true if one looks at single studies; the ranges reported can often vary from a tiny fraction of electricity market prices and the private costs of producing power to a number that is way above these levels. Looking at, for example, coal and oil the results produced range from 0.004 to roughly 68 US cents per kWh for coal and from 0.03 to almost 40 US cents per kWh for oil. In comparison, the projected (private) lifetime generation costs for the cheapest new power plants (coal and natural gas) normally range between 2.5 and 7 US cents per kWh depending on country and site (International Energy Agency and Nuclear Energy Agency, 1998). The reported discrepancies in results for similar fuels may raise concerns about the validity and the reliability of the conducted valuation studies. Still, it must be made clear that there is no reason to question the general notion that to some extent the numbers should differ due to, for instance: 1. the use of different technologies (e.g. implying separate emission factors); 2. the characteristics of the specific site under consideration (e.g. population density, income, transport distances etc); and 3. differences in scope (e.g. a fraction of all externalities may be included). Overall, however, the question of whether the large ranges in estimates are motivated or not, is difficult to determine (as illustrated by the above discussion on the role of methodological choice), especially since there exists no objective truth with which to confront the empirical estimates.

Fifth and finally, Table 8.1 and Figure 8.3 do not display the different types of externalities covered in each study, but a closer examination of this also reveals important disparities among studies. The types and the classification of external cost impacts differ among studies (Sundqvist and Söderholm, 2002). There also exist important differences among the different studies with respect to the number of stages of the entire fuel cycle assessed. For instance, all the hydropower studies assess solely the construction and generation stage. For coal, on the other hand, a large part focuses on several stages of the fuel cycle. Sundqvist (2004) shows that the expected externality estimates of the studies focusing on the entire fuel cycle are higher, ceteris paribus, than those presented in generation only studies. This raises the question of what are the relevant scope and the appropriate externality classifications to use in these types of studies. Krewitt (2002) concludes in his evaluation of the ExternE project that it has provided some partial answers to these questions but many important issues remain unsolved. In the next section we touch on some of the critical
issues in externality assessment, and we focus in particular on the usefulness of previous valuation efforts for policy purposes.

USING EXTERNAL COST ESTIMATES FOR POLICY PURPOSES: IMPLICATIONS AND CHALLENGES

In this section we discuss the role of externality valuation in energy policy-making, as well as remaining obstacles to increased use of such assessments for policy purposes. Pearce (2002) identifies five important policy uses for external cost estimates, and we begin this section by elaborating briefly on these:

1. Estimates of specific external impacts (e.g. sulphur dioxide) can be used to determine the efficient levels of environmental taxes (so-called Pigovian taxes). This is a very direct way of addressing environmental externalities in a privatized electricity market.

2. Estimates of the total social (private plus external) costs can – in the case of public ownership of electric power sources – be used directly to plan future capacity additions. It should be added that also in privatized electricity industries, governments may choose to promote investment in specific technologies. One example is the use of tradable green certificate markets for the promotion of different renewable electric power sources. When implementing such schemes estimates of the social costs of power sources may provide important information about which power sources deserve to be included in the support system.

3. External cost estimates could be used to raise awareness among the public about the different environmental impacts of different electric power sources. In deregulated markets this could, for instance, facilitate the growth of an efficient market demand for ‘green’ electricity.

4. External cost estimates may in general assist in resolving difficult trade-offs in environmental policy. Although this notion is more or less implicit in all of the above-mentioned cases, it deserves special attention here, not the least since policy priorities also involve the allocation of public funds across different sectors of the economy (and not solely within one specific sector).

5. External cost estimates could also serve as inputs in the ‘greening’ of the national accounts, as a way to reflect the environmental damages arising from economic activity.

In the remainder of this section we address the first four of these potential uses, again with emphasis on deregulated and privatized electric power
markets. The use of environmental cost assessments for use in the national accounts will thus not be dealt with here.

The Design of Efficient Environmental Taxes in the Electric Power Sector

The use of external cost damages for, say, air pollution impacts to determine efficient environmental taxes is strongly endorsed in the environmental economics literature. Still, in practice this may be difficult and at least two important issues need to be resolved.

First, it has to be established whether there exists sufficient – natural as well as economic – scientific information to establish meaningful monetary damage values as the basis for a tax policy. The most important example is probably that of CO₂ emissions and taxes (e.g. Freeman, 1996). In the first phase of the ExternE project it was noted that the environmental damage cost estimates for greenhouse gas emissions presented in the literature spanned a range of several orders of magnitude (European Commission, 1995), and the main report concluded that all attempts to value these impacts require important normative judgements, and therefore the potential for synthesis or consensus is remote. After additional research efforts within the ExternE project (European Commission, 1999) these general conclusions largely appear to be still valid (Krewitt, 2002). It is, however, generally agreed that the external costs of CO₂ emissions are substantial and may therefore constitute a large share of the total value (Freeman and Rowe, 1995). The environmental economist faces a dilemma here; is it better to leave out potentially important external damages from the valuation and present biased estimates or should one make use of rough proxy estimates (e.g. mitigation costs) so as to provide (or at least approach) some kind of ‘full cost’ estimate? The ExternE study (EC, 1999) in the end chose the latter path and recommended the use of ‘minimum’, ‘central’ and ‘maximum’ estimates.

However, this raises a number of important issues. The choice of what externalities to include in the assessment cannot be done entirely objectively, and is largely a matter of judgement. The judgement that has to be made is essentially whether the externality under consideration is ‘mature’ enough to ‘undergo’ economic valuation. This is, however, not only a question of whether the scientific knowledge is more or less established; it also involves the issue of whether the public is sufficiently informed and, hence, able to form an opinion of its own about the issue at hand. Again, economic valuation is ultimately about measuring people’s given preferences towards goods, but if no relevant preference structure for a particular good, such as global warming, exists, the valuation effort may become arbitrary. We have also noted above that the use of abatement cost estimates (regulatory
revealed preferences) provide poor substitutes since such estimates rely on the notion that all relevant preferences have already been perfectly integrated into policy decisions. The recent research on the possible existence of hyperbolic (time-declining) discount rates shows that the structure of people's preferences towards time—in combination with the long-term nature of global warming—may have significant impacts on the external cost of carbon dioxide (as well as on the economics of nuclear power) (Pearce et al., 2003). So far, however, our knowledge about these time preferences is limited. In the case of carbon dioxide, therefore, the initial challenge of policy may not lie so much in ‘measuring’ and aggregating individual preferences but in specifying the conditions for public discourse over common ways of understanding what the pertinent issues are about.²

Second, for those impacts where it is safer to assume that people's preferences are fairly well established, there remains the question of which external cost estimates should be used as the basis for environmental taxes. It was implicit in Figure 8.3 that wide ranges of estimates may exist even for the exact same impacts, and as we also have noted, this may be due to several reasons. Here we will, however, focus on the simple fact that different estimates are based on different parameter input assumptions. There are essentially two different categories of parameter input assumptions made in electricity externality studies; technical assumptions (e.g. energy efficiency, dose-response functions, emission factors), and economic assumptions (e.g. monetary values elicited in different contexts, discount rates). Previous survey work in the field has spent a lot of time on these issues (Lee, 1997; Schlesinger, 2000). However, while past discussions have normally focused on what are the ‘best’ estimates (assumptions) to make, we will focus in more detail on the role of the above assumptions for providing useful policy implications.

A relevant example of the importance of parameter input assumptions concerns the assumptions made about the monetary values used to address mortality impacts. Many previous studies use the value of a statistical life (VOSL), and the assumptions made concerning this value tend to differ. For example, the ExternE core study (European Commission, 1995) uses a VOSL value of USD 2.6 million for Europe while van Horen (1996) relies on a value range of USD 2.9–5.6 million for (the poorer country) South Africa. In the national implementation part of the ExternE project (European Commission, 1999) the decision was made to introduce an alternative measure on which to base the valuation of mortality impacts due to air pollution. This is the so-called Years of Life Lost (YOLL) approach, which essentially assigns a WTP to the risk of reducing life expectancy rather than to the risk of death. The YOLL values attributed to the mortality impacts are, as is evident from Figure 8.4, reduced by
up to two orders of magnitude as compared with the values based on the VOSL method (see also Kühn, 1998). In addition, the core project (EC, 1995) that relied on the VOSL approach did not include values for chronic mortality impacts due to air pollution, something that the national implementation studies do.

Note: Estimates based on the 1995 ExternE Core Project.


Figure 8.4 External cost estimates in the ExternE core and national implementation projects: coal (C) and oil (O) fuel cycle

Overall, the preceding discussion implies very different messages to the policy-makers about mortality impacts depending on method used and the scope of the investigation. Schlesner (2000), who compares the ExternE core project (EC, 1995) and the Rowe et al. (1995) study, supports the view that the assumptions underlying the valuation of human health and mortality impacts as well as dose-response functions are major drivers of external cost estimates. This large sensitivity in results due to parameter input assumptions creates problems for policy-makers. For environmental tax purposes policy-makers prefer relatively ‘safe bets’ about the general impacts involved (as well as some rough ideas about any regional differences), but so far previous studies have often provided only wide ranges of estimates.

This discussion raises a fundamental issue in non-market valuation that is seldom raised in the electricity externality debate. Most environmental
economists would agree that meaningful environmental valuation efforts require that a relevant ‘project’ has been defined, and that in turn involves the choice between two or more pre-defined alternatives. In the case of electricity externalities these projects are normally the investments in different power plants. However, in externality studies these investment projects are often hypothetical, that is, they do not represent an existing (real) situation, and valuation estimates are also transferred from other studies. The problem here, though, is that according to the literature on non-market valuation (and indeed that on market valuation as well) economic values are context-dependent and project-specific. In other words, it may not make much sense to talk about a universal WTP for avoiding one tonne of SO\textsubscript{2} being emitted, even though that is just what many policy-makers would like to have as they prefer (if not for administrative simplicity) more or less harmonized standards and taxes across different regions.

Finally it is worth noting that examples where external cost estimates have formed the basis of environmental taxes in the energy industry are few, and Krewitt (2002) also notes that existing taxes on sulphur dioxide and nitrogen oxides in Western Europe (Sweden being the only exception) are generally not high enough to cover the expected damage costs (as estimated within the ExternE project). For carbon dioxide emissions, however, the opposite is true, and here estimated damage costs are about ten times lower than the marginal avoidance costs of meeting European Union reduction targets following the Kyoto Protocol (Krewitt, 2002). This latter result may of course partly reflect the above-mentioned uncertainties about the external costs of carbon dioxide emissions, but also the fact that the ‘Kyoto price’ and the ‘ExternE price’ reflect different (ethical) reasoning processes (Söderholm and Sundqvist, 2003). Within the ExternE project, hypothetical prices are established in advance, as one of the raw materials for calculating the ‘total’ cost of electricity. The Kyoto price, on the other hand, did not play a causal role in the negotiations but at most merely reflects the outcome of the process. In the latter case, it may therefore be the process (rather than the social net benefits) that defines the legitimacy of choice.

The Promotion of Renewable Energy

Policy efforts designed to promote directly the diffusion of new – primarily renewable – electric power technologies are very common, and include investment subsidies, feed-in tariffs and green certificate schemes. Such measures are often based on a wide variety of policy motives related to, for instance, employment, environmental benefits, energy self-sufficiency, resource scarcity etc. From an economic efficiency point it is generally best to target any environmental damages directly through a pollution tax or
a tradable permit scheme rather than to subsidize power sources that are
deemed to be environmentally benign, the most important reason being that
the use of subsidies makes power generation overall cheaper and therefore
may promote too much of electricity services (and also too much of
pollution). Nevertheless, in the renewable energy sector there may exist also
other – non-environmental – externalities, such as the presence of technology
learning effects, economies of scale and information externalities. These
provide an ‘infant industry’ argument for securing a certain market share for
environmentally benign – but not yet commercialized – power sources.

Some argue that the consumption of non-renewable natural resources,
such as fossil fuels and uranium, leads to external costs, and that – as a
consequence – renewable energy sources should be promoted simply because
they are renewable. Hohmeyer (1988) adds a resource depletion charge
and an external cost to public investment in R&D in his externality study.
According to his lower estimate, these two components together account
for more than 80 per cent of the external costs of the nuclear fuel cycle. The
classification of natural resource depletion as an externality is, however,
questionable. Hohmeyer (as well as others) rely on the concept of ‘backstop
technology’ in the development of external costs for depletion impacts. This
concept is based on the notion that the price for a given non-renewable
resource will increase over time as the resource becomes scarcer in line with
the so-called Hotelling rule (Hotelling, 1931), but only up to the point to
where a substitute (backstop) technology becomes more attractive (e.g.
renewable resources). However, historical data indicate that the real prices for
non-renewable resources have, due to technological developments, material
substitution and exploration, fallen over time, something that is in direct
contrast to the path predicted by the Hotelling rule (Radetzki, 2002). Thus,
for most natural resources the empirical data suggest decreasing (rather
than increasing) scarcity and that the backstop technology is not likely to
ever become economically viable. In addition, in contrast to the presence
of environmental externalities at the pollution stage the market is often
able to signal increased scarcity even in the absence of public intervention.
Of course, this may not be the case if property rights are poorly defined,
but lack of efficient property rights regimes remains perhaps an even more
severe problem for many renewable resources (such as forests and fish).

A related question to that of resource scarcity is whether one should
credit the avoided external costs from replacing existing power generation as
major benefits of ‘new’ investments in, say, wind, solar or biomass. However,
these avoided costs do not per se constitute externalities. Including these
‘avoided’ externalities of fossil fuels, as Lee (1997) notes, also gives rise to
double counting of externalities for these fuel sources (i.e. an external cost
for fossil fuels and an external benefit for the renewables). Clearly, if the
specific aim of a study is to evaluate the benefits and the costs of replacing existing power sources with new ones, it will be correct to include the avoided costs from replacing existing power sources, but such a research undertaking should not be confused with pure external cost studies.

While there clearly exist economic efficiency reasons (that is, environmental benefits) for promoting the development of new environmentally benign electric power sources, it is, however, doubtful whether the different renewable power sources are all equally beneficial from an external cost view. In spite of the considerable uncertainties involved, Figure 8.3 provides an indication that renewable power sources generally are more environmentally benign than those based on the combustion of fossil fuels, but it also suggests that bio-fuelled electric power appears to incur substantially higher external costs than do the remaining renewable power alternatives (wind, solar, hydro). Sundqvist (2004) shows that these tentative conclusions remain after having accounted for methodological choice, income, and for whether the entire fuel cycle (rather than only the generation stage) has been evaluated. This notion, if valid, questions some of the recent policy initiatives that attempt to encourage the use of renewable energy per se by providing the same support for all sources pre-defined as renewable through green certificates and competitive bidding systems.

Previous studies also support the notion that the public’s willingness to pay for a specific renewable power technology may differ considerably depending on the design and the location of the installation. For instance, Ek (2002) and Bergmann et al. (2006) show that both the Swedish and the Scottish public, respectively, are generally willing to pay more to avoid the negative environmental externalities associated with onshore wind power (in particular the negative impacts on the view of the landscape), compared with the corresponding externalities arising from offshore installations.

Raising Public Awareness about the Environmental Impacts of Power Generation

While external cost estimates ultimately rely on the elicitation of the public’s preferences towards environmental change, previous results from externality assessment studies can also raise awareness and fuel deliberations among the public about the welfare impacts of the overall and specific environmental externalities for different power generation alternatives. Such information efforts can be used both to affect any public resistance towards new investments in power plants, but also to stimulate the establishment of an efficient consumer market for ‘green’ electricity. In this section we focus solely on the use of external cost estimates for achieving the policy goal of stimulating demand for green electricity.
First it needs to be repeated that the assessment of environmental externalities in the power sector is motivated by the presence of perceived market failures, that is, the socially optimal level of green power is arguably higher than the level chosen by private investors. Even if consumers would be willing to pay a premium for green power and act accordingly in the electricity market, such green preferences would not in general induce the industry to approach the optimal level of environmentally benign power sources and in this way make externality assessments and related policy measures redundant. There exist several reasons for why, say, environmental taxes and green consumption are no policy substitutes (Brennan, 2001).

First, a higher demand for green power may be interpreted as a change in preferences in favour of green power sources, but since economists evaluate policy efficiency based on exogenously given preferences the very idea of preference change questions the foundations of economic policy analysis. In practice, of course, it is difficult to distinguish between activities that change people’s preferences and those that change behaviour by altering the available information.

Second, even if we would be able to distinguish between changes in behaviour due to preference change on the one hand and new information on the other, none of these alternatives support the conclusion that green demand can replace externality assessments. If consumers increase their demand for green power due to changing preferences the electricity production will still involve a market failure; the increase in green power demand simply implies that the optimal level of green power has increased but there will still exist a difference between the optimal and the actual level of green power capacity. Thus, the need for standard regulatory measures and thus for externality assessments remains. Even the very idea of green preferences as policy substitutes creates problems as it begs the question of how one would define the no policy alternative (i.e. the one corresponding to the free market solution in the welfare economics literature); preferences can not at the same time serve as both policy instrument and policy criterion (Brennan, 2001). If demand for green power in the past has been suppressed due to information failures we are essentially dealing with two different market failures: an environmental externality and incomplete information. If information becomes (in any sense) complete, principally the environmental externality problem would still be there even though the total environmental impacts may be less severe. Also in this case green power demand would not be able to replace externality assessments.

However, green electricity demand could make sense from a regulatory point of view if the policy goals are defined, not by the economic efficiency criterion, but by deliberations about the public good in which preferences are formed rather than considered as given (or, less favourably, by politicians’
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self-interests). However, even though one accepts the view that green preferences can serve as a substitute for taxes, regulations etc. there would still be a clear need for environmental valuation exercises. For instance, companies that wish to market their electricity services as green, need to understand how people perceive and value different aspects of their power generation portfolio. Valuation studies would provide important implications in that they indicate the willingness to pay for green electricity in general, and the extent to which households and/or other customers are willing to pay more for certain characteristics of the green power sources than for others. Such studies would imply a greater reliance on choice experiment applications, since they encourage people to consider different attributes of a good (e.g. green hydropower) rather than changes in the good as a whole (e.g. Bergmann et al., 2006). In the next section we provide an example of how the choice experiment approach has been used to value the environmental attributes of hydropower in Sweden, and how Swedish hydropower producers can use these results to increase the demand for their green power sources.

Assessing the Environmental Externalities of Hydropower in Sweden Using the Choice Experiment Approach

Following the deregulation of the Swedish electricity market in 1996 it is possible for households (as well as for other consumers) to freely choose their power supplier; this has also provided the possibility for consumers to voice their environmental preferences for the different power sources. This section summarizes the approach and the results of a study that aimed at estimating how different environmental attributes arising from existing hydroelectric production are valued by Swedish households. The starting point of the study is the criteria set up by the Swedish Society for Nature Conservation (SNF) according to which existing Swedish hydropower production can be labelled as green electricity (Swedish Society for Nature Conservation, 2001). The labelled electricity is sold on a voluntary basis at a premium to end-users. For hydroelectric producers who wish to market their production as green it is important to know to what extent electricity consumers value the specific environmental attributes of the production process, and what strategies can be used to mitigate the most important negative impacts.³ The use of a choice experiment approach to elicit consumer preferences towards hydropower permits the valuation of distinct (un-priced) environmental attributes.

The choice experiment reported here focuses on impacts that can be considered to be common for all types of hydroelectric stations where dams and reservoirs are present (which is typically the case in Sweden), as well
as impacts to which specific mitigation measures can be linked. The first two attributes used in the experiment describe the effects of changes in the water flow, while a third attribute focuses on impacts on fisheries. In addition, a cost attribute – changes in the electricity price – was included. The mitigating measures that can be undertaken can all be considered environmental improvements that will increase the costs of the producers, which, in turn, will raise the price that the consumers have to pay. The attributes and levels used in the experiment are summarized in Table 8.2. Each choice set consisted of two alternatives (A and B). The attributes and levels were varied in alternative A, while alternative B represented the status quo option, that is, the environmental characteristics of a standard hydropower facility in Sweden as of today. No opt-out alternative was included. Thus, respondents could only choose between existing and environmentally improved hydropower. The use of only two alternatives in the choice set was mainly motivated by the fact that hydropower is a very dominating power source in Sweden (normally accounting for about 50 per cent of total electricity supply), and also by the focus in the study on the relative valuation of the different attributes of hydropower rather than on hydropower production as such.

Data were gathered using a mail-out survey to 1000 randomly selected households in Sweden. Hydropower industry (Vattenfall) representatives provided information that was used in the design of the questionnaire, which was subsequently discussed in focus groups and pre-tested. In addition to the choice experiment the questionnaire included questions about the respondents’ general attitudes towards the environment and electricity production, as well as socio-economic questions. After two complete send-outs and one reminder the response rate was 48 per cent.

Results obtained by estimating a random effects binary probit model, pooled by individual, for the sample data indicated that the price attribute had the strongest impact on the utility of the respondents, thus indicating that environmental improvements must come at a low cost for the consumers. This is enforced by the other results of the estimated model, which indicate that the budget constraint may limit the average respondents’ willingness to pay for environmental improvements. For the non-monetary attributes the analysis indicates that erosion and fish life improvements are relatively important. In addition, the results of the econometric analysis imply that some of the Swedish households tend to prefer other green power sources than hydropower (i.e. wind, solar etc), and that for some government (rather than market-based) provision of green power is the preferred solution to the problem of increasing the share of green generation capacity.

The implicit prices derived from the pooled econometric analysis are presented in Table 8.3. These implicit prices reflect the relative importance
Table 8.2 Attributes and levels used in the choice experiment

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Definition</th>
<th>Levels</th>
</tr>
</thead>
<tbody>
<tr>
<td>Downstream water level</td>
<td>Effects due to changes in the downstream water level. Flora and fauna downstream of a dam may need a certain water level to survive. Hydro producers are mandated to keep a certain minimum flow</td>
<td>(1) 50% higher than minimum flow, (2) 25% higher than minimum flow, and (3) minimum flow (status quo)</td>
</tr>
<tr>
<td>Erosion and vegetation</td>
<td>Impacts due to water level variations in the reservoir causes erosion and affects beach adjacent vegetation. Changes in the release schedule may lower these types of impacts</td>
<td>(1) 50% lower erosion and damages on vegetation, (2) 25% lower erosion and damages on vegetation, and (3) existing erosion and damage to vegetation (status quo)</td>
</tr>
<tr>
<td>Fish</td>
<td>Impacts on fish life in the river due to hydro development. It is possible to adapt the hydro facilities so that the damages on fish life are limited</td>
<td>(1) Adapted to all fish species, (2) adapted to migratory fish species such as the salmon, and (3) unchanged situation (status quo)</td>
</tr>
<tr>
<td>Price change</td>
<td>Price increase due to cover the cost of implementing mitigation measures</td>
<td>Six levels ranging from 0 öre per kWh (status quo) up to an increase of 25 öre per kWh (about 3 US cents per kWh)</td>
</tr>
</tbody>
</table>

The respondents place on each of the non-monetary attributes, that is, the trade-offs the respondents would be willing to make among the non-monetary attributes included in the experiment. The negative signs for two of the levels are counterintuitive since they represent point estimates of willingness to accept (but none of these impacts were statistically significant). The other implicit prices reflect willingness to pay, which are positive as should be expected for environmental improvements. Statistical significance can, however, only be established for two of the attribute implicit prices. Overall no statistical significance can be established for the mid-levels, indicating that respondents tend to go for the most significant environmental improvements in the experiment. The implicit price estimates show that, within the experiment, respondents are, ceteris paribus, willing to pay 1.7

Swedish öre (about 0.2 US cents) per kWh extra for all fish species to be preserved or 1.5 öre per kWh for erosion and beach adjacent vegetation impacts to be lowered by 50 per cent. As a comparison, the premiums reported for the environmental improvements included in the choice exercise are ‘on the lower bound’ of the premium currently paid for green electricity in Sweden (0.3 to 6 öre per kWh).

Table 8.3  Implicit price estimates for hydropower attributes (Swedish öre per kWh)

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Mean</th>
<th>t-statistic</th>
</tr>
</thead>
<tbody>
<tr>
<td>Downstream water level (+50%)</td>
<td>-0.56</td>
<td>-1.15</td>
</tr>
<tr>
<td>Downstream water level (+25%)</td>
<td>0.54</td>
<td>0.95</td>
</tr>
<tr>
<td>Erosion and vegetation (–50%)</td>
<td>1.47***</td>
<td>2.66</td>
</tr>
<tr>
<td>Erosion and vegetation (–25%)</td>
<td>-0.26</td>
<td>-0.42</td>
</tr>
<tr>
<td>Fish (all)</td>
<td>1.66**</td>
<td>2.45</td>
</tr>
<tr>
<td>Fish (migratory)</td>
<td>0.61</td>
<td>1.09</td>
</tr>
</tbody>
</table>

Notes:
*** Statistically significant at the 1 per cent level.
** Statistically significant at the 5 per cent level.

In conclusion, the results of the choice experiment indicate that if the environmental costs arising from hydropower are to be minimized the mitigating measures that are taken should focus on limiting the impacts from reservoir erosion and on improving the situation for the fish that inhabit the affected river. However, from a green market demand point of view these measures must be taken at low costs since households appear to be rather sensitive to price increases.

DETERMINING ENVIRONMENTAL POLICY PRIORITIES: ADDITIONAL REMARKS

In one way or another all uses of external cost estimates aim at determining priorities between different environmental goals as well as resolving trade-offs between private and environmental goods. Environmental tax levels and subsidies for renewable energy generation signal different – explicit or implicit – policy priorities. Nevertheless, external cost studies may also provide important information that goes beyond the design of specific policy instruments. For instance, external cost assessments – and the problems
involved in conducting these – can highlight the importance of specific environmental problems or the presence of significant lack of knowledge about them. Such results may then motivate increased public-sponsored scientific research on, for instance, the identification of the true dose-response functions or on technical mitigation methods.

We have already noted that in many instances difficulties in attaining meaningful external cost estimates have spurred a lot of new research efforts; the external costs of carbon dioxide is one such example. However, in other cases previous external costs studies have also made important contributions towards understanding the economic importance of environmental externalities. Krewitt (2002, p. 843) notes in particular that the ‘assessment of health effects from air pollution was a key focus of ExternE, and certainly one of the most successful activities of the project’. The importance of the exposure to fine particulates for human health (mortality) was not a prioritized policy issue in the 1980s, but it is now a key air quality criterion in environmental policy. Still, the ExternE project also identified important knowledge gaps, not the least about the relationships between particulate emissions, exposure and health effects. Only before these issues can be resolved in more detail will we be able to provide reliable estimates of the negative welfare effects of the emissions of fine particulates from different power sources.

**FINAL REMARKS**

This chapter has analysed past research efforts on valuing the environmental externalities arising from power generation. In doing this we have raised a set of conceptual and, to some extent, unresolved issues, and in particular we have focused on the usefulness of previous valuation efforts for policy purposes. In terms of the policy context of the external cost estimates we have devoted special attention to policy implementation in essentially privatized and deregulated electricity markets.

People in general – and indeed policy-makers – often ‘have the expectation that external costs are as simple to understand as price tags in a store’, (Krewitt, 2002, p. 847), but in practice the empirical estimates of external costs may not provide strong general guidelines on how to allocate public funds (i.e. subsidies, R&D support) between different power sources. In addition, due to the context-dependent and site-specific characteristics of economic valuation estimates it may also be difficult to implement uniform taxes purely on the basis of previous external cost estimates. Nevertheless, the studies conducted have taught us quite a lot about the environmental impacts of power generation (in particular about the health effects) and
about the important uncertainties involved. Even if much of this knowledge cannot be transferred directly into a tax or a regulation it should have an impact on public deliberations in the energy field and ultimately on policy decisions. Results from externality assessment studies can raise public awareness about the overall and specific environmental externalities associated with different power generation alternatives, and as such stimulate the establishment of an efficient market for green electricity as well as assist in overcoming public resistance problems related to investment in new and environmentally benign power sources.

Finally, the usefulness of previous economic valuation efforts for policy purposes may be complicated by the fact that economic valuation of the environment typically relies on the notion that the ethical principle guiding social choice is economic efficiency. However, this view is not likely to be shared by all lay people and politicians. This means that, in contrast to many economists, the former are likely to be more inclined to promote: 1. a much broader definition of ‘externality’ than that available in the literature; and 2. the use of green power markets and green certificate schemes as substitutes for external cost assessment and implementation. Nevertheless, with its focus on rational choice over limited environmental resources, non-market valuation will continue to be a vital research area, but it also needs to be complemented by other forms of social intelligence about what should be the important criteria in energy policy decisions.

NOTES
1. One alternative that is often advocated (Ottinger, 1997) is to use control costs for existing but not necessarily required technologies (e.g. carbon sequestration in the case of CO2 emissions). However, these estimates may not bear any relation to people’s preferences towards the environment. The relevant policy question is whether people value the environment highly enough so that the use of these control methods can be motivated.
2. One may of course argue that even if people have less developed preferences towards global warming as such they may still be able to express their willingness to pay to avoid the consequences of global warming. Still, this notion disregards the fact that people’s preferences towards these consequences and their view on the ethical issues involved are likely to be dependent on the causes of these same effects.
3. The study was funded by the major Swedish power producer Vattenfall, which regularly undertakes investments aimed at improving the environmental performance of existing hydropower stations. Still, their ignorance about the increased market value of these investments spurred their interest in funding the research reviewed here. For more information about this choice experiment study, see Sundqvist (2002).
4. The implicit prices are given by the negative ratio of the coefficient of one of the non-monetary attributes and the coefficient of the monetary attribute (see, for instance, Alpizar et al., 2001). The t-statistics for the implicit prices in the model were calculated using the delta method. Since no opt-out alternative has been included the implicit prices need to be interpreted with some caution. Even if they reflect the willingness to pay for the environmental attributes of existing hydropower, it is assumed that all other attributes as
well as power sources are held constant. Thus, in real life, respondents, even those with positive implicit prices for the environmental improvements covered here, may choose to consume another type of electricity (e.g. green electricity from wind power).

REFERENCES


INTRODUCTION

An important element of energy policies in Sweden and the European Union is to promote the commercialization of renewable energy sources in the power sector. The recent wave of liberalization and deregulations of electricity markets may in itself benefit renewable energy as it allows for product differentiation; customers can choose among producers of electricity with different generation portfolios. If consumers are willing to pay a premium for electricity generated from renewable sources, such as wind power, the amount of renewable electricity capacity can be expected to increase.

Swedish consumers have had the opportunity to buy ‘green’ electricity since 1996, at the time when the electricity market was deregulated and the Swedish Society for Nature Conservation initiated a system for the labelling of ‘green’ electricity. The energy sources considered ‘green’ according to this scheme are existing hydropower, solar power, biomass power and wind power. All the major electricity distributors in Sweden offer ‘green’ electricity to their consumers, and some of them also offer electricity generated exclusively from wind.1 So far, wind power represents a small share of total electricity production in Sweden. In 2000 0.4 TWh wind power was generated, corresponding to about 0.3 per cent of total power generation in the country (Swedish National Energy Administration, 2001a). However, the political intention is to increase wind power production to 10 TWh by 2015 (Prop. 2001/02:143).

Sundqvist (2002) summarizes and compares the results of more than 40 different electricity externality studies, and his results indicate that wind power is an electricity source with relatively small negative impacts on the environment. However, although wind power may be considered a clean
electricity source that, for example, does not give rise to any emissions, there are negative environmental impacts involved in wind power generation as well. For instance, the presence of windmills can affect the view of the landscape in an undesirable way, and the generation of wind electricity creates noise pollution. The experience in Sweden and in many other European countries is that although the public opinion is, in general, positive towards wind energy, specific wind power projects often face resistance from the local population due to these negative impacts (Krohn and Damborg, 1999).

The above suggests that it is important to understand how the public, and not least the consumers of ‘green’ electricity, view the environmental effects related to wind electricity. The purpose of this chapter is to examine Swedish households’ valuation of the environmental attributes associated with wind power generation. In order to attain this purpose we undertake a choice experiment investigation. The major strength of the choice experiment approach, given the purpose of this chapter, is that it provides information about the respondents’ preferences over the different attributes (or characteristics) included in the scenario. Hence, for our purposes the choice experiment approach facilitates the analysis of the perceptions about the different attributes of wind power rather than the elicitation of preferences for the ‘service’ wind power as a package. In addition, the marginal rates of substitution for each included attribute relative to a monetary attribute are useful outputs from choice experiments since they indicate the relative importance of each of the attributes included in the experiment.

It is important to note, though, that the aim of the present study is not to evaluate whether the political target to increase wind power capacity is efficient from a welfare economics perspective, that is, we do not evaluate if and how much wind power capacity should be boosted. Instead, we are interested in how this politically desired expansion should be carried out so as to maximize the net social benefits associated with wind power. The results of this analysis should be able to provide important guidance in this respect. Specifically, the study provides an assessment of some of the external costs and the potential benefits associated with wind power. Improved information about the opinions for and against wind power is important to wind power producers as well. It is essential for the producers to know more about how the different characteristics of wind power are perceived by the consumers. This information could be used to differentiate their product as well as to market ‘green’ wind energy more efficiently. In addition, improved knowledge about the relative importance of the environmental impacts linked to wind power and the sources of the opinion for and against wind power would make the producers better equipped to respond to any opposition towards new wind power installations.
The remainder of this chapter is organized as follows. The next section presents some recent studies on the demand for ‘green’ electricity and the public’s attitude towards wind electricity. The theoretical and methodological framework is described in the third section. The fourth discusses the development of the choice experiment investigation and related survey design issues. In the fifth, sample descriptors are provided and the sample is compared with the relevant population. The results of the choice experiment are also presented and analysed. Finally, in the sixth section, the main findings of the study are summarized and some important policy implications are discussed.

PUBLIC ATTITUDES TOWARDS ‘GREEN’ ELECTRICITY AND WIND POWER: SOME EVIDENCE FROM THE LITERATURE

A number of willingness to pay surveys have demonstrated a significant potential market for ‘green’ electricity. It has also been recognized, however, that the stated willingness to pay differs from the level of actual contribution and participation in ‘green’ electricity schemes (e.g., Wiser, 1998). When Byrnes et al. (1995) compared the results of several different previous willingness to pay surveys with market simulations or real tariff schemes, they found that less than 10 per cent of those who stated that they were willing to pay a premium for renewable electricity could be expected to do so in practice when given the opportunity. Previous research efforts on the perception of renewable electricity also suggest that renewable electricity seems to be preferred over alternative energy sources (e.g., Farhar, 1996; Roe et al., 2001), and that willingness to pay for renewable electricity is positively related to income and to social group (e.g., Batley et al., 2001).

In general, public acceptance towards wind energy has been found to be high (e.g., Krohn and Damborg, 1999; Dudleston, 2000). However, this general acceptance does not seem to be valid when it comes to actual local projects; the occurrence of local resistance towards wind power developments is often explained by the NIMBY phenomenon (Not In My Back Yard). However, Wolsink (2000) claims that this NIMBY explanation is too simplistic. According to Wolsink the expression of NIMBY-behaviour is at most only a secondary issue for people opposing local wind power projects; instead institutional factors are highly important. Local resistance may, for instance, express suspicion towards the people or the company who want to build the wind turbines or a rejection of the process underlying
the decision to build new wind plants rather than a rejection of the wind
turbines themselves.

The existing qualitative literature on the attitudes towards wind power
and on how the related characteristics of wind power are perceived by
the public is extensive. The main lesson to be drawn from these previous
research efforts is that the visual impacts from wind power installations
seem to be of major importance (e.g., Hammarström, 1997; Collins et al.,
1998; Nordahl, 2000). Furthermore, although problems with noise pollution
are often mentioned when the environmental impacts of wind power are
discussed, the importance of this problem seems to be inconclusive (e.g.,
Dudleston, 2000; Pedersen and Persson Wayne, 2002).

Alvarez-Farizo and Hanley (2002) apply and compare the choice
experiment and the contingent ranking approach in a Spanish study on
household preferences over the environmental impacts of wind power
installations. They find that there are significant social costs involved in
wind farm developments. Respondents were asked to choose between (or
rank) three alternatives. The attributes included were whether to protect the
cliffs or not, whether or not to undertake measures in order to prevent the
loss of habitat on flora, and whether to protect the landscape or not. The
results show that the protection of flora and fauna were valued more highly
by Spanish households than the aesthetic impact on the landscape.

In sum, previous studies suggest that there seems to exist a relatively
strong willingness to support renewable energy sources, such as wind power.
However, we do not know whether we can expect that this willingness to
support renewable electricity is likely to be expressed in the electricity market
or not. Neither do we know much about whether the public considers some
characteristics of renewable energy as more ‘green’ than others.

The present study differs from most of the previous research on attitudes
towards wind power due to its quantitative approach. The output from the
choice experiment investigation will provide information not only about
whether the environmental effects included in the choice set are perceived
as improvements or deteriorations but also about the relative importance
of each environmental effect.

THEORETICAL AND METHODOLOGICAL
FRAMEWORK

Traditional microeconomic theory constitutes the basic theoretical foundation
of choice experiments. Consumers are assumed to seek to maximize utility
subject to a budget constraint. Specifically, the choice experiment approach
combines the characteristics theory of value (Lancaster, 1966) and the
random utility theory (McFadden, 1974). Choice experiments are commonly used in environmental applications as well as in marketing, psychology and transport research (see for example, Adamowicz et al., 1995; Boxall et al., 1996; Hanley et al., 1998 for environmental applications). The theoretical framework and the empirical model specification presented in this section draw heavily on this literature.

The basic assumption in choice experiment applications is that consumers derive utility from the different characteristics that a good possesses, rather than from the good per se. These characteristics are thus assumed to provide services to the individual (Lancaster, 1966). The utility function through which the individual is assumed to derive utility can be expressed as (Louviere et al., 2000):

\[ U_{iq} = V_{iq} + \epsilon_{iq} \]  

(9.1)

\( U_{iq} \) represents the utility to individual \( q \), derived from alternative \( i \). Assume further that the utility can be separated into two components: a systematic component, \( V_{iq} \), and a random component, \( \epsilon_{iq} \). The systematic component represents that part of utility that is provided by the attributes observed by the analyst; it is thus assumed to be equal across individuals. The random component is the utility provided by attributes unobserved by the analyst, which is assumed to be individual-specific and to reflect the individual idiosyncrasies of taste. Furthermore, \( V_{iq} \) can be written as:

\[ V_{iq} = \hat{\alpha}X \]  

(9.2)

where \( X \) is a vector of levels of observable attributes, socio-economic characteristics, attitudes towards the environment and policies interacting with these attributes while \( \beta \) is a vector of utility parameters to be estimated.

Utility maximization postulates that individual \( q \) will choose alternative \( i \) over alternative \( j \) if and only if:

\[ U_{iq} > U_{jq}, \forall i \neq j \in A \]  

(9.3)

So far in this representation, the theoretical relationships between the selection of alternatives and the sources of utility have been specified. The random utility model will now be related to a more operational econometric specification. Assume that we have a binary choice situation where the
individual $q$ has the option to choose between alternative $i$ and alternative $j$. Let us define the binary variable $y_{iq}$, which is equal to 1 if the individual chooses alternative $i$. The choice probability can then be expressed as:

$$
P(y_{iq} = 1) = P(e_{iq} > -V_{iq}(\hat{X}_{iq})).
$$

(9.4)

However, in order to calculate these choice probabilities some assumptions about the distribution of the random component have to be made. In the commonly used multinomial logit model the random components are assumed to be independently and identically distributed. Since the respondents in our case make repeated choices (see the next section for details), the assumption of statistical independence between observations may be violated; the random component may well be correlated within the individual choices. Following Butler and Moffit (1982) and Hammar and Carlsson (2001), the error term is therefore specified as:

$$
e_{iq} = u_{iq} + v_{iq}; \quad u_{iq} \sim N(0, \sigma_u^2); \quad v \sim N(0, \sigma_v^2)
$$

(9.5)

where $u_{iq}$ is the unobservable individual-specific random effect, $v_{iq}$ is the remainder disturbance and $\sigma^2$ represents the variance in $u$ and $v$, respectively. The components of the error term are consequently independently distributed across individuals as follows:

$$
Corr(e_{iq}, e_{jq}) = \rho = \frac{\sigma_u^2}{\sigma_u^2 + \sigma_v^2}
$$

(9.6)

This specification of the error term gives us the standard random effects binary probit model, which assumes equal correlation across choices for each individual. The implications for the choice experiment are that it assumes no learning or fatigue effects over choice sets and that the preferences are stable. These assumptions should, however, hold reasonably well in this experiment since respondents are confronted with relatively few attributes and choice sets in the experiment (Hanley et al., 2002). In this study a test of one aspect of preference stability is provided and the results of this exercise are presented in the fifth section.

The estimation of the random effects binary probit model will generate parameter estimates as specified in equation (9.2) above according to the following underlying indirect utility function:
Quantifying the environmental impacts of renewable energy

\[ V_x = \beta_1 X_1 + \beta_2 X_2 + \ldots + \beta_k X_k \]  

(9.7)

Hence, estimation of the random effects binary probit model yields utility parameter estimates for each attribute included in the experiment. From the parameter estimates the rate at which the respondents are willing to trade off between the attributes can easily be calculated. For a linear utility function, the marginal rate of substitution between two attributes is simply the ratio of their coefficients (e.g., Alpizar et al., 2001; Louviere et al., 2000). If a monetary attribute is included in the experiment the willingness to trade off between the attributes can be interpreted as the implicit price for attribute \( k \), \( IP_k \), which equals:

\[ IP_k = -\left( \frac{\beta_k}{\beta_p} \right) \]  

(9.8)

where \( \beta_k \) is the coefficient of attribute \( k \) and \( \beta_p \) is the coefficient of the monetary attribute. If the implicit price turns out to be positive it can be interpreted as the marginal willingness to pay for a change in the attribute, within the experiment. However, this is theoretically correct only if a status quo option is included in the experiment (Alpizar et al., 2001; Bennet and Blamey, 2001). In the present study, these estimates will primarily indicate the relative importance of the attributes included in the experiment.

Hence, in this study the choice experiment approach allows us to estimate the preferences over the environmental effects of the different characteristics of wind energy generation rather than the value of wind electricity as such. For instance, the output of the analysis will facilitate a comparison of the public’s perception of the relative importance of the noise pollution from windmills and the visual impacts. However, in order to be able to estimate these utility parameters and implicit prices, the relevant attributes and their levels have to be selected and defined. These issues are discussed in detail in the next section.

SURVEY CONSTRUCTION AND DESIGN ISSUES

Within the choice experiment in this study, respondents were asked to choose between two different wind power alternatives, A and B, each associated with different environmental attributes and prices.\(^2\) Hence, respondents were asked to choose between two alternatives of perfectly homogeneous electricity (in terms of output per kWh) although differentiated with respect to environmental quality and cost.
The choice scenario was formulated in a way that it would mimic the decision that the respondent normally faces when choosing an electricity supplier. In each choice set, respondents were asked the following question: ‘Given that you could only choose among the two alternatives below the last time you chose an electricity supplier, which alternative, A or B, would you have chosen?’ The aim was to construct a reasonably realistic choice task in order to trigger respondents to act as consumers in the electricity market when stating their most preferred wind power alternative. The actual choices in the experiment were followed up with a debriefing question (in which they were asked why they chose as they did).

The different attributes associated with wind power and its levels varied in alternative A while alternative B represented the attributes and levels of wind power generated in Sweden today, that is, alternative B was the status quo option. There was no opt-out option included in the experiment. Therefore, since the respondents were only allowed to choose between two different wind power options, they were ‘forced’ to choose a wind electricity alternative. The motive for omitting the opt-out option is that if it had been included it would likely have been the preferred alternative for many of the respondents. This would have made the task of identifying the attitudes towards the environmental attributes of wind power more difficult. In addition, given the relatively ambitious political goal in Sweden to increase wind power capacity, the opt-out option is, in some sense, of minor interest. The policy-relevant question examined in this study is, thus, how the introduction of more wind power capacity can be facilitated by altering its characteristics and in this way increase the public acceptance of wind power. However, the exclusion of the opt-out alternative implies that we cannot interpret our results as estimates of the respondent’s willingness to pay for changes in wind power quality. Nevertheless, the study will provide an assessment of the relative importance of wind power attributes.

Defining Attributes and Levels

Clearly, choosing the attributes to be included in the choice set is a task of crucial importance. First, the attributes included in the experiment should, in one way or another, be relevant for the policy-making process as well as for the wind power producers. This implies, in general, that attributes included in the experiment should ideally be associated with actual potential measures or choices. For instance, the location of windmills is likely to be a highly relevant attribute. If wind power producers are interested in differentiating and developing their product in accordance with what electricity consumers actually prefer they should locate new wind power schemes where the perceived negative environmental impacts are relatively
Quantifying the environmental impacts of renewable energy

small. Similarly, if the noise pollution from the windmills is judged to constitute a significant negative attribute, the energy companies will have an incentive to lower this impact. Clearly, these choices could also be influenced by the policy-making process through regulation and/or different economic instruments. Second, the respondents must also perceive the attributes as relevant. This implies that the environmental impacts that are considered important by the public should also be included as attributes in the choice experiment. Furthermore, the attributes should vary across levels that are considered realistic by respondents. If the included attributes or the levels of the attributes are not perceived as relevant by respondents or if an attribute considered as being important is excluded, this might influence the responses negatively and the number of valid responses would decline (Garrod and Willis, 1999; Bennett and Blamey, 2001).

When the attributes and the levels of attributes included in the present study were chosen, the previous research efforts outlined in section two on the public attitudes towards wind power constituted an important input (e.g., Hammarström, 1997; SOU, 1999:75; Nordahl, 2000; Pedersen and Persson Wayne, 2002). According to the previous research on the attitudes towards wind power and the environmental impacts of wind power, the amenity effect seems to be of major importance. The attributes included in the experiment that aimed at representing the visual impacts from wind power installations were the location, the height, and the grouping of windmills.

In 2002 the majority of the existing wind power capacity in Sweden consisted of separately located windmills, onshore near the coastline and on the islands Gotland and Öland. However, the wind potential is also good offshore and in the mountainous areas and these areas might be of interest for future wind power installations. In order to identify how the public views the different location options, ‘onshore’, ‘offshore’ and ‘in the mountains’ were included in the experiment representing different qualitative levels of the location attribute. The levels of the location attribute were illustrated with colour photographs visualizing windmills located offshore, onshore and in the mountains, respectively. The pictures were chosen so that the windmills appeared to be of the same size and so that the weather conditions appeared similar.

Windmills are relatively high objects, generally located in open areas, and are often visible over far distances. For instance, in a Danish study 60 metre-high windmills located on a very flat and open area were found to be clearly visible at a distance of at least 7 kilometres (Miljö- og Energiministeriet, 1996). The most common height of windmills in Sweden is at present about 60 metres, although significantly higher ones are becoming increasingly common. The following two levels of the height attribute were included
Natural resources

in the experiment: the most common height at present (60 metres) and a level that represents higher windmills (100 metres). To make comparisons of height easier, a few high well-known buildings and their heights were referred to as reference objects. These reference objects were a flagpole (10 metres), a ten-storey building (30 meters), and the tower of Stockholm City Hall (112 metres).

Although separately located windmills have been the most frequent, large wind parks including up to 50 windmills or more have been developed. The world’s largest offshore wind park is sited in Denmark and consists of 80 wind turbines. To facilitate the analysis of whether the average Swedish electricity consumer considers wind parks as being something positive or negative, compared with individually located windmills, one attribute for the grouping of windmills was incorporated in the choice experiment. Three levels of the grouping attribute were included: separately located windmills, small groups (less than ten windmills) and large groups (between ten and 50 windmills). The levels of the grouping attribute were described in words in the questionnaire.

Although the relative importance of the noise pollution impacts from wind power is inconclusive according to previous studies, a noise attribute was included in the experiment. The noise attribute was included because problems with noise are often claimed to be important sources of disturbances in the discussions and the debate in the media. Noise pollution was also mentioned as an important aspect in the focus group deliberations (see below). Two levels of the noise attribute were included. The status quo level was the highest level allowed outdoors in residential areas in Sweden (40 decibels), and the other level represented a reduced noise level (30 decibels) (SOU 1999:75). To facilitate comparisons between noise levels, different sounds at similar levels as the ones included in the choice experiment were described and used as reference objects. These reference sound sources were the following: the ticking from a clock (20 decibels), rustling leaves (30 decibels), a new refrigerator (40 decibels), and a normal conversation (65 decibels) (Clayman, 2000; Electrolux, 2002). However, the perception of noise may differ with respect to the source of the noise. For example, to have to put up with soughing leaves at the same sound level as a distant windmill is probably perceived as a much more pleasant and tolerable experience than the mechanical sound from the rotor blades of a windmill.

The electricity price facing the households represents the cost attribute in the choice experiment. Only for the noise attribute we would expect that a change from the status quo level represents an unambiguous improvement since it is reasonable to expect that a lower noise level is preferred to a higher one. For this reason the respondents were confronted with both
increased and lowered prices of the cost attribute. Six price levels were included, three that represented a higher electricity price, two a lower price and one level representing the status quo option with a zero price change. An approximate average electricity price (including taxes) was presented to make comparisons easier. In addition, two examples where the effects on household expenses from changes in electricity prices (per month and per annum) were described, were included in the questionnaire to facilitate comparisons between different prices. The first example described the change in expenditures for a low-consuming household (without electricity heating) and the second example outlined the corresponding change for a high-consuming household (with electricity heating). The included attributes, their levels and coding are summarized in Table 9.1. The levels of the qualitative attributes (i.e., all included attributes except the price attribute) were effect-coded.

Table 9.1 Attributes, corresponding variables and levels

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Variable</th>
<th>Levels and coding</th>
</tr>
</thead>
<tbody>
<tr>
<td>Noise</td>
<td>Noise</td>
<td>1 if 30 dB, –1 if 40 dB</td>
</tr>
<tr>
<td>Location</td>
<td>Mountain</td>
<td>1 if mountain, 0 if offshore, –1 if onshore</td>
</tr>
<tr>
<td></td>
<td>Offshore</td>
<td>1 if offshore, 0 if mountain, –1 if onshore</td>
</tr>
<tr>
<td>Height</td>
<td>Height</td>
<td>1 if higher than 60 metres, –1 otherwise</td>
</tr>
<tr>
<td>Group</td>
<td>Small</td>
<td>1 if small group, 0 if large group, –1 if separate</td>
</tr>
<tr>
<td></td>
<td>Large</td>
<td>1 if large group, 0 if small group, –1 if separate</td>
</tr>
<tr>
<td>Price</td>
<td>Price</td>
<td>5 levels ranging between –10 öre/kWh and +15 öre/kWh¹</td>
</tr>
</tbody>
</table>

Note: ¹ 10 öre corresponds to about 1 US cent, the average household electricity price in Sweden is about 50–65 öre per kWh (or about 5–6.5 US cents) including taxes (depending on electricity supplier and in which part of the country the consumer lives).

Since all the qualitative variables were effect-coded, as with dummy variables, the main effect of a qualitative variable can be defined by \( L - 1 \) effects-coded variables that represent \( L - 1 \) of its levels. That is, if an attribute has \( L \) levels, \( L - 1 \) will be included as variables in the model (Louviere et al., 2000).

The Development of the Questionnaire

The questionnaire was developed by using the experiences from: 1. an early test on a group of graduate students; 2. a pretest involving about
30 respondents (of which a few were people active in the wind power industry); and 3. a focus group deliberation in the concluding stage of the development. The early check on the group of students aimed primarily at testing the relevance of the attributes chosen. In the pre-test of the questionnaire and in the focus group, the formulation of the questions, the descriptions of the attributes in the choice sets and the levels of the price attributes were tested.

The general impression after these different exercises was that the task of choosing the most preferred alternative in the choice sets seemed to be manageable. There were no indications that some attributes were missing or were in any other way inadequate. For instance, no participant argued that impacts on wildlife (such as birds colliding with wind turbines) or employment effects were important aspects that should have been included in the experiment. A spontaneous comment from some of the participants in the focus groups and in the early check on students was, however, that it was a quite demanding questionnaire to answer. When the pictures illustrating the three levels of the location attribute were discussed, some of the participants stated that they considered the pictures to be beautiful, although nothing indicated that any of the location attributes were considered as being more or less beautiful than the other. Some clarifications in the descriptions of the included attributes were made as a result of the early student test.

During the focus group deliberation, some of the participants argued that wind power is already a power source with a relatively small impact on the environment and that none of the negative effects on the environment are irreversible. Consequently, these participants argued that their overall willingness to pay for improvements were somewhat limited and that the price changes would have to be relatively modest for them to consider choosing anything other than the status quo option. The price changes were also kept relatively small (the highest price change represents about 25 per cent of the average electricity price including taxes) and in the pre-test the choices were distributed fairly even between the two alternatives in the choice sets. Finally, some minor changes in the formulations of some of the attitudinal questions were brought about as a result of the pre-tests, primarily due to the focus group discussion.

The first part of the questionnaire contained questions about the respondents’ attitude towards the environment, towards electricity production in general, and towards wind power generation in particular. In the second part, the attributes and their levels were described. Respondents were then asked to state their choices in six different choice sets, and the choices were followed up with a question about why the respondents had answered the way they did. The third and last part of the questionnaire collected socio-economic information.
We present the alternatives included in the experiment in a generic (unlabelled) form, that is, the respondents were not told that alternative A represented ‘changed attributes of wind power’ and alternative B the ‘characteristics of existing capacity’. Generic alternatives should be used when the major focus in the analysis is on the marginal rates of substitution between the attributes. Otherwise, the respondent may focus on the label of each alternative rather than on the attributes associated with the alternative (Alpizar et al., 2001; Bennett and Blamey, 2001). One example of a choice set to which respondents were confronted in the questionnaire is given in Figure 9.1.

If you only had been able to choose between alternative A and B the last time you chose electricity supplier, which alternative would you have chosen? Mark with a cross.

<table>
<thead>
<tr>
<th>Noise</th>
<th>Alternative A</th>
<th>Alternative B</th>
</tr>
</thead>
<tbody>
<tr>
<td>Decibels</td>
<td>40</td>
<td>40</td>
</tr>
<tr>
<td>Height</td>
<td>100 metres</td>
<td>60 metres</td>
</tr>
<tr>
<td>Grouping</td>
<td>Individual</td>
<td>Individual</td>
</tr>
<tr>
<td>Location</td>
<td>Mountains</td>
<td>Onshore</td>
</tr>
<tr>
<td>Price change per kWh</td>
<td>−5 öre</td>
<td>0 öre</td>
</tr>
</tbody>
</table>

**Figure 9.1 Choice set example**

The levels of the attributes included varied over all choice sets in alternative A, while alternative B represented the status quo option, and thus the characteristics of existing wind power capacity. These characteristics are a maximum allowed noise level of 40 decibels outdoors in residential areas, wind turbines with a height of 60 metres, separately located, situated onshore, and no change in the electricity price (SOU, 1999:75; Swedish National Energy Administration, 2001b).

**Experimental Design**

Experimental design deals with how to create choice sets in an efficient way, that is, how to combine attribute levels into alternatives and choice sets. The most common approach in economic applications has been to use orthogonal designs, in which the levels of the attributes of the different alternatives are uncorrelated in the choice sets.

The five attributes included in the experiment, which can take between two and five different levels, resulted in a full factorial with 180 combinations ($2 \times 3 \times 2 \times 3 \times 5$). This would be more than the respondents could be expected to cope with. Although the main effects are of primary interest
in this study, the presence of at least some interactions is likely and the design should permit the testing for some of these potential interactions. For example, it seems reasonable to expect that the noise perception would differ with respect to where the wind power capacity is located. In order to facilitate the estimation of all main effects and at least some of the two-way interactions, a main effect orthogonal design was combined with an endpoint design (following Louviere et al., 2000, pp. 94–6). This means, in short, that randomly drawn choice sets from the full factorial were combined with randomly drawn choice sets from another factorial where only the lowest and highest levels of each attribute were included. The experimental design was accomplished by using the statistical software SPSS. After reducing identical combinations and combinations that seemed unreasonable, 30 combinations remained. These 30 choice sets were then randomly assigned to five blocks such that each single respondent would be confronted with six choice sets.

In order to permit a test of whether the order of the attributes within the choice sets may have affected the outcome, that is, one aspect of stability of preferences, the ordering of the attributes was varied. Fifty per cent of the respondents received choice sets with the attribute noise described and placed first in the choice sets and the other 50 per cent received choice sets with the noise attribute described and placed last. If the cognitive burden on respondents was too heavy they may have used some simplifying decision rule when they stated their most preferred alternative in the choice sets rather than choosing after a comprehensive judgement of all the included attributes. For instance, one such simplifying strategy could be to give more weight to the noise attribute when it was the first attribute that respondents faced in the choice set than on the subsequent ones.

**Questionnaire Logistics and Sample**

In the present study, a postal survey was chosen over an interview approach, primarily since it was considered cost-efficient. In the literature on non-market valuation it is generally recommended that personal interviews should be used over postal surveys (e.g., Arrow et al., 1993). There are, however, pros and cons associated both with postal surveys and with personal interviews. Personal interviews permit the interviewer to a greater extent to use visual material to help respondents if necessary, but interviews are relatively high cost. Also, personal interviews normally generate high response rates, although they may be subject to ‘interviewer bias’. Postal surveys are relatively low cost and provide the respondents with time to contemplate their answers more, but can also lead to low response rates and also consequently sampling selection biases (e.g., Bennett and Blamey, 2001).
In early March 2002, the questionnaire, together with an introductory letter, was mailed to 1000 Swedish residential homeowners, randomly selected from the Swedish Official Register of Persons and Addresses. About two weeks after the questionnaire had been sent out, a follow-up reminder was sent out to non-respondents. Within an additional three weeks, a second reminder was sent to the remaining non-respondents. The second reminder was, however, not complete in the sense that it included only a short reminder note and no new copy of the questionnaire was included.

The reason for limiting the survey solely to people living in owner-occupied houses is that they have the opportunity to actively and freely choose among different electricity suppliers. Consequently, they are familiar with the choice situation to which they are confronted within the questionnaire. Of course, this also implies that the results of the study reflect the attitude of the average Swedish homeowners rather than the attitude of the average Swedish electricity consumer or household.

RESULTS

The Response Rate

In the present study 1000 questionnaires were sent out, and 547 completely or partially usable answers were returned. There were thus 453 non-responses to the questionnaire. Adjusting for the 15 respondents that were unable to answer due to unknown addresses, severe illness or death, the overall response rate was 56 per cent.

A few of the respondents stated, as a general comment, that it was a quite demanding task to complete the questionnaire. Among the responses that were incomplete, the majority refused to state their preferred alternatives in some or all of the choice sets. The respondents to three of these non-complete questionnaires stated explicitly that they had refused to answer because they were negative towards wind power and they did not accept having to choose between two wind power options only. In other words, at least these three respondents refused to participate in the study because no opt-out alternative was included in the experiment (see, however, Note 2).

Testing for the Presence of Sample Selection Bias

A potential problem associated with postal surveys is that the presence of non-responses can lead to a bias caused by sample self-selection. For instance, it is reasonable to expect that those with a strong positive or
negative opinion towards ‘green’ electricity and wind power are more likely to answer and return the questionnaire. Hence, if homeowners with a strong interest in the environment are over-represented in the sample, it could be interpreted as an indication of the presence of sample self-selection bias. In Table 9.2 the characteristics of the respondents within the sample are compared with the characteristics of the typical Swedish home owner.

**Table 9.2 Sample characteristics**

<table>
<thead>
<tr>
<th>Variable</th>
<th>Sample</th>
<th>Typical Swedish Homeowner</th>
</tr>
</thead>
<tbody>
<tr>
<td>Age (share of &gt; 65)</td>
<td>28%</td>
<td>24%(^a)</td>
</tr>
<tr>
<td>Average income (per month)</td>
<td>33 000 SEK</td>
<td>32 000 SEK(^a)</td>
</tr>
<tr>
<td>Membership in environmental organization</td>
<td>14%</td>
<td>4.2%(^a)</td>
</tr>
<tr>
<td>Family situation (share of sample with at least two adults with children)</td>
<td>34%</td>
<td>30%(^a)</td>
</tr>
</tbody>
</table>

*Source:* \(^a\) Statistics Sweden (2002).

Within the sample, 14 per cent of the respondents stated that they were members of an environmental organization, and this is a significantly higher share than among Swedish homeowners in general. The null hypothesis that these two estimates are equal can be statistically rejected at the 1 per cent significance level.\(^8\) In 1992, 8.5 per cent of the population between 16 and 84 years reported that they were members of an environmental organization. By 2000 this share had decreased to 4.2 per cent. However, an additional 9 per cent declared that they belong to recreational organizations, and since the definition of environmental and recreational organizations may overlap, it is not possible to make direct comparisons of the averages of this sample and the national averages reported in the study by Statistics Sweden (2002). That is, some of the respondents in the sample that declared membership of environmental organizations might belong to organizations that are categorized as recreational organizations by Statistics Sweden.\(^9\) Consequently, respondents who are members of environmental organizations are likely to be over-represented in the sample, although the difference may be less pronounced than is indicated in Table 9.2.

The share of respondents older than 65 years, those with children in the household and the share with a university degree are slightly higher in the sample compared with the estimates reported by Statistics Sweden. For both
the share of older than 65 and the share with children in the household, the null hypothesis that the share in our sample is equal to the estimate reported by Statistics Sweden can be rejected at the 5 per cent significance level. However, the interval in which the average income for the sample is found coincides with the average income for the population. Although the share of respondents with a university degree is higher in this sample than in the estimate for Swedes on average, this difference is not statistically significant at the 5 per cent significance level. Hence, when comparing the socio-economic characteristics of the realized sample with the estimates of Statistics Sweden, the respondents seem to be slightly older than the relevant population, and respondents with children seem to be somewhat over-represented in the sample.

Results of the Choice Experiment

The results of the choice experiment are based on the responses of 488 individuals and 2928 observations. We received 547 more or less complete answers; 18 of the respondents refused to participate in the choice experiment and 26 additional respondents did not participate completely and stated their most preferred alternative in less than six of the choice sets. After removing these incomplete answers and the 15 additional ones that were incomplete (due to respondents refusing to state their income and/or age and gender), 488 individuals remained in the sample. Descriptive statistics for the variables included in the random effects binary probit model and their coding are given in Table 9.3.

There were no a priori expectations about whether the attributes related to location, height and grouping would be considered by respondents as improvements compared with the present situation or as a change for the worse. For instance, large groups of high windmills may be considered to have a negative visual impact while large wind parks may also be considered more efficient than smaller separately located windmills. Furthermore, the average respondent could consider the mountainous area and the archipelago either as areas worth protecting from exploitation (since they are widely used for recreation) or as being suitable for wind power developments (since they are in general at a far distance from more densely populated areas). The coefficient for the noise attribute, however, was expected to have a positive sign since the change represented a lower noise level than presently allowed.

It would be reasonable to expect that the perception of at least some of the environmental characteristics included in the model would be different with regard to location. For instance, groups of windmills located offshore or in the mountainous area may be perceived differently by the average
### Table 9.3 Descriptive statistics

<table>
<thead>
<tr>
<th>Variable</th>
<th>Coding</th>
<th>Mean</th>
<th>Std</th>
<th>Min</th>
<th>Max</th>
</tr>
</thead>
<tbody>
<tr>
<td>Choice</td>
<td>1 for alternative A</td>
<td>0.44</td>
<td>0.50</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Noise</td>
<td>1 if reduced noise level, –1 otherwise</td>
<td>–0.07</td>
<td>0.99</td>
<td>–1</td>
<td>1</td>
</tr>
<tr>
<td>Mountain</td>
<td>1 if located in mountainous area, 0 if offshore, and –1 if onshore</td>
<td>0.18</td>
<td>0.79</td>
<td>–1</td>
<td>1</td>
</tr>
<tr>
<td>Offshore</td>
<td>1 if located offshore, 0 if in the mountainous area and –1 if onshore</td>
<td>0.09</td>
<td>0.75</td>
<td>–1</td>
<td>1</td>
</tr>
<tr>
<td>Height</td>
<td>1 if higher than 50 metres, –1 if not</td>
<td>0.11</td>
<td>0.98</td>
<td>–1</td>
<td>1</td>
</tr>
<tr>
<td>Small</td>
<td>1 if small group, 0 if large, and –1 if separately located</td>
<td>0.03</td>
<td>0.75</td>
<td>–1</td>
<td>1</td>
</tr>
<tr>
<td>Large</td>
<td>1 if large group, 0 if small, and –1 if separately located</td>
<td>0.18</td>
<td>0.82</td>
<td>–1</td>
<td>1</td>
</tr>
<tr>
<td>Large offshore</td>
<td>1 if large groups offshore</td>
<td>0.07</td>
<td>0.63</td>
<td>–1</td>
<td>1</td>
</tr>
<tr>
<td>Small mountain</td>
<td>1 if small groups in the mountains</td>
<td>0.02</td>
<td>0.61</td>
<td>–1</td>
<td>1</td>
</tr>
<tr>
<td>Visit mountains</td>
<td>1 if visited mountains and located in mountains</td>
<td>0.06</td>
<td>0.44</td>
<td>–1</td>
<td>1</td>
</tr>
<tr>
<td>Price change</td>
<td>–10, –5, +5, +10, +15 per kWh</td>
<td>2.75</td>
<td>10.08</td>
<td>–10</td>
<td>+15</td>
</tr>
<tr>
<td>Environmental organization</td>
<td>1 if member of an environmental organization</td>
<td>0.14</td>
<td>0.35</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Near</td>
<td>1 if windmill exists in sight of residence or summerhouse</td>
<td>0.11</td>
<td>0.31</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Age</td>
<td>Age of respondent</td>
<td>55</td>
<td>13</td>
<td>22</td>
<td>86</td>
</tr>
<tr>
<td>Social choice</td>
<td>1 if choices are based on what is best for society as a whole</td>
<td>0.53</td>
<td>0.49</td>
<td>0</td>
<td>1</td>
</tr>
</tbody>
</table>
respondent compared with groups located onshore. The interaction variables included in the model were therefore ‘large groups’ ‘offshore’ and ‘small groups in the mountains’.

Some socio-economic and attitudinal variables were also included in the analysis, interacting with the attributes or as shift variables. The socio-economic variables included were ‘age’ and ‘environmental organization’.\(^{10}\) There were no a priori expectations about the signs of any of these parameter estimates. The ‘age’ variable was included so as to test whether the probability of choosing alternative A over alternative B differs with respect to the age of respondents. The variable ‘environmental organization’ is included to capture a general interest in environmental issues; this interest is expected to imply a higher probability for respondents to choose the green alternative. However, since there was no clear-cut green alternative in the experiment and since it was not known in advance whether the varied environmental characteristics included in alternative A would be interpreted as improvements or deteriorations, any specific expectation about the sign of the parameter estimate for the environmental organization variable was not deemed relevant. The variable ‘visit mountains’ indicates whether respondents have visited the mountainous areas during the last year. Respondents that visit mountainous areas regularly for recreation could be expected to be less likely to accept wind power facilities located in these areas. Therefore the coefficient for ‘visit mountains’ was expected to be negative. The variable ‘near’ shows whether respondents have wind power installations in sight of their residence or summerhouse or not, and it was included to facilitate a test of whether respondents familiar with wind power generation were more or less likely to choose wind power with different characteristics (i.e., alternative A) than the average respondent.\(^{11}\) Furthermore, the variable ‘social choice’ is based on answers given in the debriefing question. The variable is equal to one if the respondents stated that they had chosen their most preferred alternatives in the choice sets based on what they considered was best for society as a whole (in contrast to the alternative that gave them, as electricity consumers, most value for the money).

The results obtained by estimating the random effects binary probit model, pooled by individual, are reported in Table 9.4. The estimated correlation between the error terms (Rho) is 0.57 and highly statistically significant, which implies that we cannot reject the random effects model in favour of a more restrictive model that assumes no correlation between the error terms. A likelihood ratio test of the hypothesis that all coefficients are equal to zero was performed. With a chi-squared value of 488, the hypothesis of all coefficients being equal to zero could be rejected at the 1 per cent significance level. Given that the alternatives were generic (not
Natural resources

labelled), and, since there was no systematic difference between the two alternatives included, no constant was included in the model.

Table 9.4 Random effects binary probit model results

<table>
<thead>
<tr>
<th>Variable</th>
<th>Coefficient</th>
<th>t-statistic</th>
</tr>
</thead>
<tbody>
<tr>
<td>Noise</td>
<td>0.06*</td>
<td>1.68</td>
</tr>
<tr>
<td>Mountain</td>
<td>-0.19***</td>
<td>-3.93</td>
</tr>
<tr>
<td>Offshore</td>
<td>0.31***</td>
<td>5.95</td>
</tr>
<tr>
<td>Height</td>
<td>0.02</td>
<td>0.60</td>
</tr>
<tr>
<td>Small</td>
<td>0.13**</td>
<td>2.38</td>
</tr>
<tr>
<td>Large</td>
<td>-0.15***</td>
<td>-2.85</td>
</tr>
<tr>
<td>Large offshore</td>
<td>0.06</td>
<td>0.92</td>
</tr>
<tr>
<td>Small mountainous area</td>
<td>-0.12*</td>
<td>-1.91</td>
</tr>
<tr>
<td>Visit mountains</td>
<td>-0.25***</td>
<td>-3.28</td>
</tr>
<tr>
<td>Price</td>
<td>-0.09***</td>
<td>-26.47</td>
</tr>
<tr>
<td>Environmental organization</td>
<td>0.17**</td>
<td>2.35</td>
</tr>
<tr>
<td>Near</td>
<td>0.41**</td>
<td>2.03</td>
</tr>
<tr>
<td>Age</td>
<td>-0.01***</td>
<td>-3.12</td>
</tr>
<tr>
<td>Social choice</td>
<td>0.41***</td>
<td>3.22</td>
</tr>
<tr>
<td>Rho</td>
<td>0.57***</td>
<td>21.63</td>
</tr>
</tbody>
</table>

Notes:
- Sample size: 488 individuals.
- Restricted log-likelihood: –1668.
- Chi-squared: 488.
- ** Statistically significant at the 1% level.
- *** Statistically significant at the 5% level.
- **** Statistically significant at the 10% level.

In the choice experiment, respondents chose alternative A in 44 per cent of the choice sets. The behaviour in the experiment was partly ‘lexicographic’ in the sense that 9 per cent of the respondents always chose alternative A and 20 per cent always chose the status quo option, alternative B. In the end of this section we analyse to what extent this behaviour may have affected the outcome of the experiment.

Estimated coefficients with a positive sign imply that a change from the status quo option to the corresponding attribute increases the probability of choosing alternative A over alternative B, and a negative sign implies consequently a reduced probability of choosing alternative A. Hence, each estimated attribute coefficient with a negative sign is perceived by the average respondent as an environmental deterioration compared with the status quo.
option. Inversely, positive coefficients indicate that the related attributes are viewed as environmental improvements.

The positive sign of the coefficient for the ‘noise’ attribute indicates that a reduced noise level is considered to be an environmental improvement, as was expected. This coefficient is statistically significant at the 10 percent significance level. The results indicate further that windmills located ‘offshore’ are considered by the average respondent to be an environmental improvement while a location in the ‘mountainous area’ is considered to be a change for the worse, all compared with wind power capacity located onshore. The coefficients representing the two location attributes, ‘offshore’ and in the ‘mountainous area’, are both highly statistically significant. The positive sign of the parameter for the location attribute offshore should, however, be interpreted with some caution since the environmental impacts from offshore wind power developments are not well known. For instance, offshore wind facilities may have negative impacts on fish life; our investigation though, focuses primarily on the visual impacts. Additional research on this issue is thus necessary.

The positive sign of the coefficient for the ‘height’ attribute indicates that the average homeowner considers higher wind turbines as an improvement compared with lower. However, the parameter estimate of the ‘height’ coefficient is not significant from a statistical point of view. Hence, we are unable to present any reliable evidence that the height of windmills do affect the utility of the average Swedish homeowner.

Furthermore, separately located windmills are, according to the results, preferred over ‘large’ wind parks while ‘small’ wind parks seem to be preferred over separately located windmills. One possible explanation to this somewhat puzzling result may be that respondents dislike the impact on the landscape from large parks while small parks are considered as not affecting the landscape much more than separately located turbines. Both the group coefficients are statistically significant.

It is interesting to note that although large groups onshore are considered to be a change for the worse, there is no evidence in this study that ‘large groups offshore’ do affect the utility of the average respondent since the coefficient for the interaction effect for ‘large groups offshore’ is insignificant from a statistical point of view. This is, however, not the case for small groups of wind turbines in the mountainous area, since ‘small groups in the mountains’ is perceived to be an environmental deterioration, compared with separately located windmills onshore. This coefficient is also statistically significant. Respondents who recently had visited the mountains also seem to be more negatively affected by the presence of wind turbines in these areas compared with the average homeowner since the sign of this coefficient is negative. The ‘price’ coefficient has a negative sign and is clearly significant.
from a statistical point of view. This means, as expected, that respondents prefer low electricity prices to high ones.

The positive sign of the coefficient for ‘environmental organization’ indicates that members in these organizations are, in general, more likely to choose the alternative with different wind power characteristics than the present, that is, they were more likely to choose alternative A. This coefficient was also statistically significant. Elderly respondents were, as indicated by the negative sign of the ‘age’ coefficient, less likely to choose wind electricity with other characteristics than the present. The ‘age’ coefficient was also significant from a statistical point of view.

Respondents who stated in the debriefing question that they made their choices on the basis of what they considered was best for society as a whole were more likely to choose the alternative with changed wind power attributes, that is, alternative A. The same result was found for respondents with existing wind turbines in sight of their residence or summerhouse (‘near’). Both these coefficients were statistically significant.

From the parameter estimates, the rate at which respondents are willing to trade off costs for changes in any of the other attributes, were calculated, that is, the implicit price. However, in the present study, the implicit price should be interpreted as an indication of the relative importance of the attributes included. The relative importance coefficient for the noise attribute, for instance, is the ratio of the ‘noise’ coefficient and the ‘price’ coefficient (see the third section). These relative importance coefficients derived from the above parameter estimates are presented in Table 9.5. The corresponding t-statistics were calculated using the delta method (Greene, 2000). The 95 percent confidence intervals were in turn estimated using the Krinsky and Robb (1986) procedure with 5000 random draws from the asymptotic normal distribution of the parameter estimates.

For instance, according to these results, the relative importance of the negative perception of windmills in the mountains is higher than the negative perception of large wind parks, all compared with the characteristics of present capacity.

The most essential information provided from these implicit prices is thus whether the average Swedish homeowner considers these changes as improvements or as deteriorations and the relative importance of each of these attributes. Also, since we did not include any opt-out option in the choice experiment, respondents were forced to choose to buy wind power. Given the option, it is likely that some respondents with positive implicit prices in this experiment would have preferred to buy electricity not stemming from wind or not paying any premium at all for electricity labelled as green.14
We now discuss some of the potential drawbacks related to choice experiments and test whether there is any evidence that the results presented above suffer from any of these. A large number of potential errors are brought up in the literature. Here we discuss whether the cognitive burden on respondents may have been too heavy, that is, if choices have been made after some simplified strategy rather than after a comprehensive judgement of the levels and attributes to which the respondents were confronted in the choice sets. Specifically, we will analyse whether the ordering of the attributes in the choice sets may have affected the results of the choice experiment and also whether the presence of lexicographic behaviour may have affected the outcome.

**Does Order Matter?**

The application of stated preference techniques, such as choice experiments, requires respondents to undertake a number of tasks. For instance, in a choice experiment the respondent is required to understand the attributes of the alternatives in general terms, the way in which attribute levels vary across alternatives and thus to make a number of choices between two or more alternatives. The complexity of the task facing choice experiments respondents is thus likely to exceed that of conventional contingent valuation studies (Bennett and Blamey, 2001). The complexity of the choice task depends on the number of alternatives in each choice set, on the number of attributes and levels of attributes used to describe the alternatives and on the number of repetitions.
One aspect of task complexity is related to whether preferences are stable, that is, learning and fatigue effects. Although individuals may become more proficient after completing a few choice sets, and thus become more familiar with the choice situation, a point may be reached when fatigue effects occur. This may be associated with the occurrence of status quo biases in which respondents simply give up the choice task and opt to stay with the status quo option. Carlsson and Martinsson (2001) test for stable preferences in a study on the validity of choice experiments. When testing whether responses were affected by the order of the choice sets – half of the respondents received choice sets in the order (A, B) and the other half received the choice sets in the order (B, A) – they could not reject the null hypothesis that preferences are stable.

In the present study, however, the order of the attributes was varied. Half of the respondents received a questionnaire in which the noise attribute was described first in the informative part preceding the choice experiment, and the noise attribute was also the first attribute that respondents faced in the choice sets. The other half received a questionnaire in which the noise attribute was described last and was placed as the last of the environmental attributes included in the choice sets, only succeeded by the price attribute.

To facilitate a test of whether the order of the attributes affects the parameter estimates, a model with a dummy variable for order was estimated (coded as one when the noise attribute was the first attribute, and as zero if not). The results from estimating the random effects binary probit model with the dummy variable for order included are given in Table 9.6 together with the results without the order dummy, in the first and the second columns of the table.

The parameter estimate for the order dummy is equal to 0.07; the positive sign indicates that noise was considered as having a greater impact on the utility of the average respondent when the noise attribute was presented first. However, since this coefficient is not statistically significant, we cannot reject the hypothesis that the parameter estimates are independent of the order in which the attributes have been presented and located within the choice sets. Moreover, the parameter estimates of the other included attributes and socio-economic and attitudinal variables proved to be relatively stable across the two model specifications.

Analysing the Presence of Lexicographic Behaviour
Lexicographic behaviour may arise in choice experiments: 1. if the included alternatives are not sufficiently different to ensure trade-offs; 2. as a result of ‘yea’ saying (where the respondent, for instance, consistently chooses the green alternative); or (3.) as an indication of strategic behaviour or
<table>
<thead>
<tr>
<th>Variable</th>
<th>Original Specification</th>
<th>Order-dummy Included</th>
<th>Restricted Sample</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Coefficient</td>
<td>t-statistic</td>
<td>Coefficient</td>
</tr>
<tr>
<td>Noise</td>
<td>0.06*</td>
<td>1.68</td>
<td>0.06*</td>
</tr>
<tr>
<td>Mountain</td>
<td>-0.19***</td>
<td>-3.93</td>
<td>-0.19***</td>
</tr>
<tr>
<td>Offshore</td>
<td>0.31***</td>
<td>5.95</td>
<td>0.31***</td>
</tr>
<tr>
<td>Height</td>
<td>0.02</td>
<td>0.60</td>
<td>0.02</td>
</tr>
<tr>
<td>Small</td>
<td>0.13**</td>
<td>2.38</td>
<td>0.14**</td>
</tr>
<tr>
<td>Large</td>
<td>-0.15***</td>
<td>-2.85</td>
<td>-0.15***</td>
</tr>
<tr>
<td>Large offshore</td>
<td>0.06</td>
<td>0.92</td>
<td>0.06</td>
</tr>
<tr>
<td>Small mountainous area</td>
<td>-0.12*</td>
<td>-1.91</td>
<td>-0.16*</td>
</tr>
<tr>
<td>Visit mountains</td>
<td>-0.25***</td>
<td>-3.28</td>
<td>-0.24***</td>
</tr>
<tr>
<td>Price</td>
<td>-0.09***</td>
<td>-26.47</td>
<td>-0.09***</td>
</tr>
<tr>
<td>Environmental organization</td>
<td>0.17**</td>
<td>2.35</td>
<td>0.25</td>
</tr>
<tr>
<td>Near</td>
<td>0.41**</td>
<td>2.03</td>
<td>0.42**</td>
</tr>
<tr>
<td>Age</td>
<td>-0.01***</td>
<td>-3.12</td>
<td>-0.007***</td>
</tr>
<tr>
<td>Social choice</td>
<td>0.41***</td>
<td>3.22</td>
<td>0.41***</td>
</tr>
<tr>
<td>Rho</td>
<td>0.57***</td>
<td>21.63</td>
<td>0.57***</td>
</tr>
<tr>
<td>Order dummy</td>
<td>0.07</td>
<td>1.50</td>
<td></td>
</tr>
</tbody>
</table>

Notes:

- Original specification
  - Sample size: 488.
  - Restricted log-likelihood: –1668.
  - Chi-squared: 488.

- Order dummy included
  - Sample size: 488.
  - Restricted log-likelihood: –1663.
  - Chi-squared: 488.

- Restricted sample
  - Sample size: 390.
  - Log-likelihood: –1227.
  - Chi-squared: 156.

- *** Statistically significant at the 1% level.
- ** Statistically significant at the 5% level.
- * Statistically significant at the 10% level.
genuinely lexicographic preferences. If the choice situation is too complex, lexicographic behaviour may also be a result of respondents simplifying the choices by using some lexicographic decision rule. If respondents use a simplifying decision rule, it might lead to biased results while genuine lexicographic preferences would not (although it would not provide much information either) (Alpizar et al., 2001). One such simplifying decision rule can be, for instance, to stick with the status-quo option in all of the choice sets. In the experiment, 9 per cent of the respondents always chose alternative A and 20 per cent always chose the status quo option, alternative B.

In order to test whether this lexicographic behaviour appears to have affected the results the random effects binary probit model was also estimated for a restricted sample in which respondents that consistently chose alternative B were removed. The results from this exercise are given in Table 9.6, together with the results in the original model specification and the results with the order dummy included.

The parameter estimates proved to be relatively stable between these different specifications. The signs of all the attributes and the socio-economic and attitudinal variables were unchanged while there were some changes with respect to statistical significance in this alternative model specification. For instance, the coefficient for the ‘small group’ attribute did not prove to be significant within this second model specification (with lexicographic choices removed), nor did the coefficients for the variables ‘environmental organization’, ‘near’ or ‘age’. However, the coefficients for the levels of the location attribute are highly significant within this model specification as well. Overall this suggests that the results appear to be quite robust with respect to alternative model specifications.

CONCLUSIONS

The purpose of this study has been to employ a choice experiment investigation to examine how the public values different environmental attributes associated with wind power. Overall, the estimated model appears to have performed well. For instance, the model proved to be fairly robust, with respect to alternative model specifications.

Among the included attributes in the experiment the visual impact in general, and the location of wind power capacity in particular, appears to have a significant impact on the utility of Swedish homeowners. According to the results, wind power offshore is considered as an environmental improvement, compared with wind power located onshore while a location of wind capacity in the mountainous areas is considered an environmental deterioration. In addition, reduced noise levels would increase the utility
of respondents, small wind farms are considered a change for the better while large wind farms are perceived as changes for the worse compared with separately located windmills. According to the results of the choice experiment the electricity price also has a significant impact on the utility of the respondents.

Thus, if an expansion of wind power capacity in Sweden is to be accomplished in a way that minimizes the environmental external costs associated with wind power development and, consequently, gains support from the public, new schemes should primarily be located offshore rather than in the mountainous area. Therefore, if wind power producers are interested in differentiating their product further and market wind power as a green electricity source, they should primarily give prominence to offshore installations and avoid large wind farms (if not located offshore) rather than investing in the development of less noisy wind turbines. However, if the aim is to increase the market share of wind electricity these measures should also be taken at a low cost; according to the results the Swedish homeowners are cost-conscious and clearly prefer low electricity prices over higher.

Finally, this cost consciousness may, however, limit the potential for future offshore expansions since the production of offshore facilities are more costly than installations onshore (e.g., Hartnell and Milborrow, 1998). However, since the results presented here suggest that the external costs from offshore wind power facilities are significantly lower than onshore installations, this may compensate, at least partially, for the higher production costs offshore.

NOTES

1. If a consumer chooses to buy wind electricity, the supplier guarantees that the amount of electricity the consumer uses will be generated from wind power. Even though this would not imply that the electricity delivered to a specific consumer would be produced from wind it would imply an increase in the demand for wind power and thus in wind power capacity.
2. For more details about the survey, see Ek (2002).
3. The exclusion of the opt-out option may have bothered respondents with a negative attitude towards wind power since they did not have the option to refuse to buy wind power. In order to find out to what extent this was the case respondents were asked about their general attitude towards wind power (and some of its related effects) and the choices made in the choice experiment were followed up with a debriefing question. The general impression from this analysis is that the majority of the respondents seemed to be positive towards wind power. When asked to mark their general attitude towards wind power on a scale ranging from 1 to 5 (where 1 represented a negative attitude and 5 a positive) only 10 per cent marked 1 or 2 while 64 per cent marked 4 or 5.
4. Seven individuals participated in the focus group. The age, occupation, gender and social status of the participants varied.
5. For instance, noise impacts from wind turbines are essentially local, and if the operation of the turbines is stopped these noise impacts will entirely disappear. This is in sharp
contrast to the long-lasting waste from the nuclear fuel chain, with which society has to deal over thousands of years after the shutdown of the plant.

6. Although there were six levels of the price attribute, the price change was never set equal to zero in alternative A. Therefore in the alternative with varying characteristics of the attributes, there were only five levels of the price attribute, and the zero price change was only used in the status quo option, that is, alternative B.

7. For instance, combinations in which the only change in alternative A compared with the status quo option was a lowered noise level in combination with a lowered electricity price, implying an economic compensation for an unambiguous improvement, were considered unreasonable and were thus removed.

8. The sample sizes for Statistics Sweden’s estimates on age, membership in environmental organizations, family situation and education are not known. For this reason, our sample estimates for each of these variables was compared with the estimates of Statistics Sweden based on the assumption that these latter estimates reflect the ‘true’ values.

9. In a contingent valuation study on Swedish households Vredin (1997) found that 18 per cent of the respondents declared membership of environmental organizations.

10. Alternative specifications were also estimated, with more socio-economic variables included (such as for instance gender, income, education). However, these specifications were not preferred since none of these socio-economic variables proved to be significant from a statistical point of view.

11. An alternative specification was estimated in which the variable ‘near’ was interacted with each of the attributes in the choice set. This specification was not preferred since none of these interaction terms were statistically significant.

12. When a constant was included in the model, the hypothesis that it was equal to zero could not be rejected at the 10 per cent significance level. In addition, the estimated slope coefficients proved to be rather stable between the two different specifications.

13. The perception of wind power offshore did not differ with respect to whether the respondent visited the archipelago during the previous year, since the coefficient for this interaction was not statistically significant when included in the model. It is therefore not included in Table 9.4.

14. When the respondents that had not previously bought electricity labeled ‘Bra Miljöval’ were asked why they had not, 19 per cent stated that they were not interested or that they did not see any positive environmental effects associated with ‘Bra Miljöval’ electricity.

15. In a lexicographic preference relation one of the commodities in the consumption bundle has the highest priority in determining the preference ordering (e.g., Mas-Colell et al., 1995).

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Quantifying the environmental impacts of renewable energy


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PART II

Amenity and Water Quality
10. Underground or overground? Measuring the visual disamenity from overhead electricity transmission lines

Giles Atkinson, Brett Day and Susana Mourato

INTRODUCTION

The statutory obligation to provide households in the United Kingdom (UK) with electricity necessitates the ongoing construction of high-voltage transmission lines (HVTLs) used to transmit electric power over relatively long distances, usually from a central generating station to main substations. Typically, HVTLs are carried overhead suspended from steel lattice towers commonly known as ‘pylons’. Alternatively it is possible, though costly, to lay such cables underground. The construction of new HVTLs has important implications for the visual amenity of landscapes in rural and urban areas. Proposals to construct new HVTLs have typically been scrutinized within the UK planning system and a recent development within this process has been a demand from local authorities, environmental and other groups for pylon designs that are less visually intrusive than the familiar lattice towers or for the undergrounding of HVTLs. A reasonable response to such demands would be to actually evaluate the strength of public preferences for these alternatives. Moreover, finding out how much households prefer alternatives would furnish decision-makers with information about the value of the benefits that households enjoy from having a new HVTL constructed using a new tower design or from transmission lines being placed under the ground. This would make it possible to state whether the benefits of the action exceed its costs.

One way of valuing these benefits is to use stated preference techniques such as contingent valuation (CV), a survey-based method that elicits preferences, in monetary terms, for changes in non-market goods and
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services (see, for example, Mitchell and Carson, 1989; Bateman et al. 2002). In the current context, a household may believe that the lattice towers, currently used in HVTL construction, generate a greater visual disamenity than one of the proposed alternative designs. In that case, it is plausible that the household will be willing to pay something to see HVTLs constructed using that alternative design. That is, the placing of towers (of the current design) in the vicinity might diminish their use of a local area as a source of visual amenity (amongst other concerns). In addition, the construction of new towers in unique landscapes such as those designated as national parks might be associated with a diminution of value experienced by both users (e.g. tourists) and non-users (e.g. the knowledge that the visual appearance of a unique amenity will be diminished). A CV survey is designed so as to elicit from households statements of this willingness to pay (WTP). Economic theory indicates that statements of WTP provide a valid measure of the economic benefits of a particular course of action. In this chapter, using a CV study, we focus on preferences in the case of the former example: namely, the loss of well-being experienced by local residents, which is attributable to the visual disamenity caused by electricity transmission towers in ‘typical’ rural and urban landscapes.1

A number of issues arise in exploring this valuation problem. First, it is entirely possible that a proposed change – such as the provision of a new type of pylon design – might constitute a ‘good’ for some households and a ‘bad’ for other households. If so, it is worthwhile assessing the extent to which those with negative preferences are willing to pay to retain the status quo in order to obtain measures of a proposal’s benefits that are biased upwards.

Second, one prediction might be that WTP for visual amenities conceivably declines the further the respondent lives from the source of the amenity. More generally, this spatial dimension to the elicitation of preferences for environmental improvements has been termed ‘distance decay’. It would be interesting to know if ‘distance decay’ exists for the current study problem.

Third, an important application of the findings of CV studies, such as described in this chapter, is in cost–benefit analysis. However, the question as to how our WTP findings on the benefits of changing or removing pylons can be used to evaluate proposals for either existing or new pylons on cost–benefit grounds itself raises some interesting issues that are worth considering in more detail.

The remainder of this chapter is organized as follows. First, we describe the design of the current study and discuss the socio-economic and demographic characteristics of respondents. Second, we outline the main findings of the CV study including a summary of household WTP for options that entail
Measuring the visual disamenity from overhead electric transmission lines

either changing the design of transmission towers or placing lines under the ground. Next we investigate in more detail the importance of negative WTP and ‘distance decay’ respectively as well as provide a discussion of the use of these findings in cost–benefit analysis. Lastly, we offer some concluding remarks.

STUDY DESIGN AND SAMPLE CHARACTERISTICS

Figure 10.1 shows the various HVTL design options investigated in our CV study illustrated in a ‘typical’ rural setting in England and Wales. These options include five new pylon designs and the current design (lattice) as well as the case of lines buried underground. Each new pylon design in Figure 10.1 satisfies a set of engineering and design parameters determined by National Grid plc, which is the corporate body responsible for bulk transmission of electricity in England and Wales. Visual representations of pylons in a ‘typical’ urban setting were also prepared and respondents were either shown pictures of the rural setting or of the urban setting, depending on their area of residence.

In practice, the visual impacts of new tower designs and undergrounding options will primarily be relevant to newly constructed transmission routes. A CV question directly addressing this issue might ask households to state their WTP to avoid the visual disamenity brought about by a new route being constructed in their locality. For a number of reasons, however, it was decided that responses to a question of that nature could prove unreliable. First, it seemed likely that households unfamiliar with living in the proximity of HVTLs might overstate their visual disamenity. Second, asking questions about new transmission routes would tend to compound impacts of visual disamenity with a whole range of other perceived disamenities resulting from proximity to HVTLs.

As a result, the final survey was administered solely to households living in the vicinity of current HVTLs. Respondents were asked to express not only their WTP to have lattice towers along a stretch of that line replaced with towers of a new design, but also their WTP to have the HVTL buried underground. Since underground HVTLs effectively impose no visual disamenity on households, expressions of household WTP to underground existent HVTLs indicate the value of the entire visual disamenity of overhead transmission lines carried on traditional lattice towers. Accordingly, such values also provide an accurate indication of the value of the visual disamenity resulting from the construction of a new overhead HVTL using the traditional design of towers. Moreover, differences in the relative
One pole

Lattice (current design)

Double pole with arms

Single pole with arms
Figure 10.1 Current and proposed designs for electricity transmission towers
value of the visual disamenity resulting from different tower designs can be deduced from the expressions of WTP to change from the lattice towers to towers of the various proposed designs.

As illustrated in Figure 10.2, the CV questionnaire progressed through two main stages: Stage 1 measured preferences for new transmission tower designs while Stage 2 measured preferences for undergrounding.

STAGE 1:
PREFERENCES FOR NEW TRANSMISSION TOWER DESIGNS

Ranking exercise
Rank six pylon designs (the current design + five new designs) according to their visual appearance

For each design worse than current one
For each design preferred to current one

Willingness to oppose
Which of these (increasingly costly and time-consuming) actions would you take to oppose the replacement of current local pylons with pylons of each new design that you like less than the current one

Willingness to pay
How much would you be prepared to pay to replace current local pylons with pylons of each new design that you prefer to the current one?

STAGE 2:
PREFERENCES FOR UNDERGROUNDING

Likelihood of support
Would you be in favour of removing local pylons and put electricity cables underground?

If yes

Willingness to pay
How much would you be prepared to pay to remove local pylons and put electricity cables underground?

Figure 10.2 Valuation sequence
Measuring the visual disamenity from overhead electric transmission lines

In Stage 1, respondents initially were presented with visual representations of each of the six tower designs, including the current lattice design, in either a ‘typical’ rural or urban setting. Respondents were then asked to rank the six designs according to their visual appearance within the (specified) landscape alone. As well as providing useful information about public opinion towards current and new designs, this exercise enabled further exploration of household preferences for new designs.

To the extent that a respondent ranked any new design as being preferable to the current one, he or she was asked to express his or her WTP to see a specified stretch of towers in their local area changed to this new design. The specific location of the stretch of towers to be changed (about 2 kilometres long) was identified via especially produced maps shown to respondents at each sampling point. The payment vehicle for this change was a one-off increase in the standing charge component of the household’s electricity bill. This payment was justified to respondents as a contribution to the costs of replacing the towers. WTP was elicited by means of a payment ladder where respondents were asked to tick the amounts that they would be willing to pay for each new design that they preferred to the current one. Respondents who were willing to pay less than £1 were subsequently asked whether their WTP was zero or a very small amount between 0 and £1. Hence, information about the lower end of the (positive) WTP distribution was also collected. Overall, the estimated values represent a measure of the visual benefits associated with new tower designs.

To the extent that a respondent preferred the current (lattice) design to some or all new tower designs, the valuation procedure was less straightforward. Such respondents can be said to have a negative WTP for the proposed change. (See below for a more detailed discussion about how these values were elicited.)

Respondents then proceeded to Stage 2 of the valuation sequence where they were shown visual representations of either a ‘typical’ rural or urban setting, with and without electricity transmission towers (of the current lattice design). They were then asked whether they would be in favour of removing local pylons and putting local electricity cables underground. If they were in favour of undergrounding, they were subsequently asked for their WTP for undergrounding. This was again elicited by means of a payment ladder where respondents were asked to indicate which value most closely matched their maximum WTP for burying HVTLS in the ground. Payments were framed as an annual addition to the standing charge on household electricity bills lasting for one to three years. While information was collected regarding preferences for the status quo (i.e. by identifying those people who preferred the status quo to undergrounding) this was not followed up by an attempt to elicit negative willingness to pay for the
undergrounding option (the implications of this are discussed in more detail below).

The study itself was carried out in 34 locations (17 urban and 17 rural) in England and Wales bordering current HVTLs. In total, 785 face-to-face interviews were conducted in August and September of 2001. Respondents were sampled from four geographical bands in order to investigate the effects of distance on preferences for the different HVTL options: 1. Band 1: living within 500 metres of the HVTL; 2. Band 2: living between 500 metres and 1 kilometre from the HVTL; 3. Band 3: living between 1 kilometre and 2 kilometres from the HVTL; 4. Band 4: living between 2 kilometres and 5 kilometres from the HVTL. Using maps and photographs, respondents were shown the location and appearance of the HVTLs in their area. In addition to eliciting WTP for new tower designs and undergrounding of lines, the questionnaire also collected respondents’ socio-economic details and extensive information on attitudes towards and exposure to HVTLs.

Table 10.1 Urban and rural sample characteristics

<table>
<thead>
<tr>
<th></th>
<th>Rural</th>
<th>Urban</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total number of individuals</td>
<td>391</td>
<td>394</td>
</tr>
<tr>
<td>Demographic variables</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Males (%)</td>
<td>40</td>
<td>45</td>
</tr>
<tr>
<td>Average age (years)</td>
<td>48</td>
<td>45</td>
</tr>
<tr>
<td>Average family size</td>
<td>2.9</td>
<td>2.9</td>
</tr>
<tr>
<td>Education:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Primary school (%)</td>
<td>15</td>
<td>10</td>
</tr>
<tr>
<td>GCSE/’O’ levels/CSE (%)</td>
<td>53</td>
<td>58</td>
</tr>
<tr>
<td>‘A’ levels or vocational training (%)</td>
<td>12</td>
<td>17</td>
</tr>
<tr>
<td>Professional degree (%)</td>
<td>9</td>
<td>5</td>
</tr>
<tr>
<td>College or university (%)</td>
<td>9</td>
<td>8</td>
</tr>
<tr>
<td>Higher degree (Masters, PhD etc) (%)</td>
<td>2</td>
<td>2</td>
</tr>
<tr>
<td>Economic variables</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average monthly personal income (using mid-points of intervals)</td>
<td>£1049</td>
<td>£1041</td>
</tr>
<tr>
<td>Average monthly household income (using mid-points of intervals)</td>
<td>£1666</td>
<td>£1657</td>
</tr>
<tr>
<td>Personal income non-response (%)</td>
<td>24</td>
<td>18</td>
</tr>
<tr>
<td>Household income non-response (%)</td>
<td>30</td>
<td>28</td>
</tr>
</tbody>
</table>

*Note:* Percentages of certain variable categories may not sum to 100% due to rounding.
Table 10.1 provides details of the demographic and socio-economic characteristics of the rural and urban samples. There are few notable differences between the two samples. The main exception is in the percentage of males; that is, it seems that males were somewhat 'under-sampled' in rural and, to a lesser extent, urban locations. Of course, our sampling strategy was based on the locations of properties with relation to HVTLs and not on the demographics of respondents. One possibility is that there are actually less males living in areas close to HVTLs, though this seems unlikely. Rather the under-representation of males in the sample may simply reflect differing roles in the household. For example, it may be the case that females are more likely to be non-working and hence more likely to be picked up in a door-to-door survey. Of course, our survey elicits household WTP. So, to the extent that it should not matter which particular member of the household responds on behalf of that household, we do not anticipate that this demographic skew will influence our findings unduly.

MAIN FINDINGS

Before turning to the WTP estimates for each tower design and the undergrounding option, it is instructive to look at the proportion of people in the sample that were and were not prepared to pay to replace the current lattice tower design. For the new designs, Table 10.2 shows that 69 per cent of respondents (543 people) preferred at least one of the five new pylon designs to the current design. However, preferring a new design does not necessarily translate into being willing to pay to ensure that it is put in place. Indeed, only just over half of this group actually expressed a positive WTP to replace the current pylon design. Clearly, there is a substantial degree of indifference with respect to paying for changing the design of pylons. The remaining 46 per cent of the people in this group (32 per cent of the total) said they would not pay anything to change the current lattice design. These responses are split between valid zeros (26 per cent of the total; e.g. those who considered pylon appearance as relatively unimportant) and protest zeros. The latter represent 6 per cent (or 48) of all respondents and include those who objected to some aspect of the contingent market: for example, an objection primarily made on the grounds that a respondent believed it was the responsibility of the electricity company or government to fund changes to the design of HVTLs.

Of the 242 respondents that preferred the current tower design to any of the five alternatives, 63 per cent said they would not do anything to prevent a new design. This we take to correspond to a zero WTP to maintain the current design. The remaining respondents in this group (89 people)
Amenity and water quality

Table 10.2 Aggregate responses to WTP questions

<table>
<thead>
<tr>
<th>Response</th>
<th>Willingness to pay</th>
<th>New Designs Scenario</th>
<th>Undergrounding Scenario</th>
</tr>
</thead>
<tbody>
<tr>
<td>Preferred new situation</td>
<td>Positive WTP for new situation</td>
<td>37.6 %</td>
<td>54.6 %</td>
</tr>
<tr>
<td></td>
<td>Valid zero WTP for new situation</td>
<td>25.5 %</td>
<td>22.7 %</td>
</tr>
<tr>
<td></td>
<td>Protest zero WTP for new situation</td>
<td>6.1 %</td>
<td>10.1 %</td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td>69.2 %</td>
<td>87.4 %</td>
</tr>
<tr>
<td>Preferred current situation</td>
<td>Zero WTP for current situation</td>
<td>19.5 %</td>
<td>7.8 %</td>
</tr>
<tr>
<td></td>
<td>Protest zero WTP for current situation</td>
<td>0.0 %</td>
<td>2.5 %</td>
</tr>
<tr>
<td></td>
<td>Positive WTP for current situation</td>
<td>11.3 %</td>
<td>2.3 %</td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td>30.8 %</td>
<td>12.6 %</td>
</tr>
<tr>
<td></td>
<td>No. of respondents</td>
<td>785</td>
<td>785</td>
</tr>
</tbody>
</table>

Note: All percentages are calculated with reference to the total number of respondents.

indicated that they would take action to prevent the replacement of their preferred lattice design, depending on which design was to replace the current one. These people are assumed to have a positive WTP for the current design (or, conversely, a negative WTP for new designs) expressed via their contributions of time and money to actions that would prevent the change to a new design.

Table 10.2 also contains a summary of the responses regarding the decision to pay or not to pay for the undergrounding option. This shows that undergrounding was supported, in principle, by a large majority: 87 per cent of respondents. Unlike the proposed changes in tower design, the undergrounding option was met with significantly less indifference from the sample population: these data indicate a majority of people stating a positive WTP for this option (55 per cent). Overall, a total of 239 respondents registered a zero WTP with a further 99 protesting against the undergrounding scenario. Finally, 2 per cent of respondents opposed the undergrounding option on the basis that laying electricity cables
underground might bring a disruption to their lives and increase possible dangers from these cables. These respondents might be interpreted as having a positive WTP for the current situation although its value was not assessed in the questionnaire.

Figure 10.3 illustrates the distribution of WTP for each new design by estimating the relevant survivor functions. These exclude (a relatively small number of) protests on the basis that the true preferences of these respondents were not revealed by the survey. That is, while these respondents may well experience a positive benefit from the proposed change, our survey was not able to uncover this benefit. In addition, WTP for the undergrounding option excludes those 18 respondents who preferred the status quo. This has implications for mean (and median) WTP as it introduces a small element of upward bias into these estimates.

Each survivor function in Figure 10.3 shows the probability that a respondent in the sample will have a WTP that is greater than a certain amount. All of the survivor functions clearly show that the majority of respondents were indifferent towards replacing the current design with each of the new designs and that the proportion of people willing to pay some amount of money quickly declines as that amount rises. For some designs such as the single pole with arms or the double pole with arms, most of the non-zero WTP lies on the positive side of the distribution. Conversely, for least liked designs such as the windmill or the V-pole there is a more even split between WTP amounts on the positive and negative side of the distribution. Lastly, for the undergrounding option, the difference in the distribution between this option and options that simply change pylon design is clear to see. That is, the proportion of people willing to pay some positive amount of money, to secure the change, declines relatively slowly as that amount rises.

Table 10.3 (column 2) reports estimates of the sample mean WTP for each of the new tower designs and the undergrounding option. The single pole with arms design is found to be the most valued of the new designs, conferring average benefits of £6.60 per household. Closely following are the one-pole and double pole with arms designs with values of £4.86 and £4.67, respectively. The V-pole and the windmill designs are the least valued by respondents. Table 10.3 also shows the 95 per cent confidence intervals associated with the estimates of mean WTP. As can be inferred from the confidence intervals shown in the table, the value of the three most preferred designs is significantly different from zero (although given the overlap between confidence intervals, in all likelihood, the difference of value between them is statistically insignificant). In turn, the value of the V-pole and the windmill designs is not statistically different from zero, again as indicated by their confidence intervals. It should also be noted that
Proportion of respondents

Willingness to pay (£)

0.9 0.8 0.7 0.6 0.5 0.4 0.3 0.2 0.1

0

‘One-pole’ design

‘V-pole’ design

‘Single pole with arms’ design

‘Windmill’ design
Figure 10.3  Survivor functions for new tower designs
Table 10.3 Mean and median WTP for replacing current pylons

<table>
<thead>
<tr>
<th></th>
<th>Whole Sample</th>
<th>Urban Sample</th>
<th>Rural Sample</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean (95% confidence interval)</td>
<td>Median (95% confidence interval)</td>
<td>Mean (95% confidence interval)</td>
</tr>
<tr>
<td>One-pole</td>
<td>£4.86 (£3.32–£6.54)</td>
<td>£0</td>
<td>£6.50 (£4.20–£9.11)</td>
</tr>
<tr>
<td>V-pole</td>
<td>£0.45 (£–0.69–£1.61)</td>
<td>0</td>
<td>£1.46 (£–0.10–£3.07)</td>
</tr>
<tr>
<td>Single pole with arms</td>
<td>£6.60 (£5.11–£8.42)</td>
<td>0</td>
<td>£5.45 (£3.59–£7.72)</td>
</tr>
<tr>
<td>Windmill</td>
<td>–£0.40 (£–1.31–£0.35)</td>
<td>0</td>
<td>–£0.46 (£–1.33–£0.22)</td>
</tr>
<tr>
<td>Double pole with arms</td>
<td>£4.67 (£3.40–£6.06)</td>
<td>0</td>
<td>£4.14 (£2.43–£6.31)</td>
</tr>
<tr>
<td>Undergrounding</td>
<td>£65.53 (£54.79–£76.16)</td>
<td>£8.53</td>
<td>£58.12 (£44.06–£72.88)</td>
</tr>
</tbody>
</table>
Measuring the visual disamenity from overhead electric transmission lines

median WTP is zero for each of the new designs (column 3). That is, if it was put to a vote, a majority of respondents would choose not to have the current design replaced by any of the new ones.

Table 10.3 (column 2) also presents the mean WTP for placing electricity cables underground. This value (£65.53) is significantly higher than the mean WTP for even the most preferred pylon design. However, median WTP for undergrounding (£8.53) is significantly lower than the mean (column 3). This considerable difference between the mean and median WTP indicates that some respondents have stated very large WTP amounts: that is, unlike the median, mean WTP is inflated by the presence of very large values.

As can be seen from Table 10.3 (columns 4 and 5), the economic benefits of placing electricity cables underground appear to be larger in rural than urban areas at least insofar as is indicated by a simple comparison of the mean WTP of each respectively. By contrast, there is no obvious pattern to the distribution of mean WTP across rural and urban locations for new designs. Some designs were more valued in urban locations (i.e. one-pole and V-pole) while others attained a higher value amongst the rural subsamples (i.e. single pole with arms, windmill and double pole with arms).

ELICITATION OF NEGATIVE WTP

It was noted earlier in discussing Table 10.2 that a non-trivial proportion of respondents felt that at least some of new designs were considered sufficiently unsightly that they felt the surrounding landscape would be visually poorer for their installation. In such cases, in principle, CV practitioners should allow respondents to express either a welfare gain or welfare loss for any particular change (see, Bohara et al. 2001 and Atkinson et al. 2004). However, it is not necessarily a straightforward task to devise a payment vehicle that can credibly be used to elicit values for both gains and losses.

One candidate approach might be to ask respondents preferring the current design for their willingness to accept (WTA) a reduced standing charge as compensation for the disamenity of viewing towers of the new design. A reduction in the standing charge can be explained as reflecting reductions in the maintenance costs of the more modern design perhaps at the behest of the electricity regulator (i.e. OFGEM). Unfortunately, the credibility of this approach is not maintained in a survey, such as in the current context that seeks separate values for each of a number of different changes. Here a particular respondent might prefer one change to the status quo whilst ‘dispreferring’ another. Even the least sceptical of respondents would find implausible a scenario in which preferred changes happened to
trigger increases in bills but less preferred changes resulted in reductions in bills. More generally, for many policy interventions entailing the valuation of multiple options for say landscape (or land-use) change from the status quo, it is plausible that similar problems will arise.

As a result, an alternative approach was taken in the current study. In order to estimate possible welfare losses, for those people preferring the current design to some or all new tower designs, respondents were asked to state which of a number of increasingly arduous tasks they would perform to avert the replacement of the current towers with towers of a new design. These tasks are described in the first column in Table 10.4 and involved signing petitions, writing complaint letters or making donations to protest groups. Each intended action can then be given a monetary dimension by relating it to the associated value of time lost (writing letters, signing petitions) or loss of money (donations).

Table 10.4 Translating intended actions into WTP estimates

<table>
<thead>
<tr>
<th>Intended Action</th>
<th>Assumed WTP to Retain the Current Design</th>
</tr>
</thead>
<tbody>
<tr>
<td>I wouldn’t do anything as I don’t really care</td>
<td>WTP = 0</td>
</tr>
<tr>
<td>I would sign a petition complaining to my MP and local council</td>
<td>0 &lt; WTP &lt; c</td>
</tr>
<tr>
<td>I would sign a petition and independently write to my local council and/or MP and/or electricity company in order to complain</td>
<td>c ≤ WTP &lt; £10 + c</td>
</tr>
<tr>
<td>As well as signing a petition and writing letters of complaint I would be prepared to donate £10 to a group coordinating protest</td>
<td>£10 + c ≤ WTP &lt; £30 + c</td>
</tr>
<tr>
<td>As well as signing a petition and writing letters of complaint I would be prepared to donate £30 to a group coordinating protest</td>
<td>WTP ≥ £30 + c</td>
</tr>
</tbody>
</table>

Note: \( c \) is the value in money terms of the time, effort and expense involved in writing a letter of complaint.

The second column in Table 10.4 describes the results of imputing WTP values to each of the possible actions to avoid replacing the current design, where the value in money terms of the time, effort and expense involved in writing a letter of complaint is described by \( c \). A respondent who indicated that he or she would not do anything was assumed to be stating indifference, that is, a zero WTP to retain the current design. A respondent stating that
they would sign a petition but not go as far as writing a letter to their Member of Parliament (MP) was assumed to be indicating that they were not indifferent but would not suffer a sufficient welfare loss to invest the time, effort and expense in writing a letter. Hence, their WTP was larger than zero but less than \( c \). A respondent stating they would write a letter but would not pay £10 to a protest group was indicating that their welfare loss lay in the interval between \( c \) (inclusive) and \( c + £10 \) (exclusive). Respondents stating they would write a letter and pay £10 to a fighting fund but not pay £30 were indicating that their welfare loss lay in the interval above or equal to \( c + £10 \) but below \( c + £30 \). For those willing to donate £30, it can be inferred that their maximum WTP is above or equal to \( c + £30 \).

Given that \( c \) is of an unknown magnitude, the assumption was made that it takes an hour to produce and mail such a letter. Put another way, \( c \) is the value the household places on one hour of its time. Following some frequently used assumptions concerning the value of non-labour time (Cesario, 1976), \( c \) is calculated from the annual after-tax income. Specifically, the value of time is taken as a third of the wage rate, which is approximated as a two-thousandth of the annual after-tax income of the household.

Table 10.5 contains the estimated mean (household) WTP for each of the new tower designs (with 95 per cent confidence intervals in parentheses). Columns 2 (reproducing the relevant data from Table 10.3) and 3 offer a comparison of mean WTP taking account of, and ignoring negative WTP respectively. Not surprisingly, ignoring negative WTP in all cases leads to an overestimate of WTP for the new design. However, the extent of this overestimate varies considerably as illustrated by the final column in Table 10.4. The smallest absolute errors are for the single pole with arms, the double pole with arms and the one-pole designs. For these designs, it is arguable that, in practical terms, the error entailed in ignoring negative WTP is acceptable. For example, for these three designs, the mean values estimated when negative WTP is ignored fall well within the 95 per cent confidence intervals of the mean values estimated from the complete WTP range. For the V-pole and windmill designs, however, the error arising from not taking proper account of those who preferred the status quo is far more significant. In the case of the former, ignoring these preferences gives an estimate of mean WTP that is more than four times as great as ‘true’ mean WTP. For the latter, ignoring those who preferred the status quo entails a change on the sign of the estimated mean WTP amounts. In other words, these results show that ignoring the existence of welfare losses can result in an overestimation of the benefits of particular policy options.
Table 10.5  WTP for new tower designs: with and without negative WTP (95% confidence intervals in parentheses)

<table>
<thead>
<tr>
<th>Design</th>
<th>WTP ≤ £200</th>
<th>0 ≤ WTP ≤ £200</th>
<th>Error (absolute difference between means)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Single pole with arms</td>
<td>£6.60</td>
<td>£7.03</td>
<td>£0.43</td>
</tr>
<tr>
<td></td>
<td>(£5.11–£8.42)</td>
<td>(£5.23–£8.78)</td>
<td></td>
</tr>
<tr>
<td>One-pole</td>
<td>£4.86</td>
<td>£6.03</td>
<td>£1.17</td>
</tr>
<tr>
<td></td>
<td>(£3.32–£6.54)</td>
<td>(£4.50–£7.72)</td>
<td></td>
</tr>
<tr>
<td>Double pole with arms</td>
<td>£4.67</td>
<td>£5.17</td>
<td>£0.50</td>
</tr>
<tr>
<td></td>
<td>(£3.40–£6.06)</td>
<td>(£3.95–£6.52)</td>
<td></td>
</tr>
<tr>
<td>V-pole</td>
<td>£0.45</td>
<td>£2.51</td>
<td>£2.06</td>
</tr>
<tr>
<td></td>
<td>(–£0.69–£1.61)</td>
<td>(£1.61–£3.46)</td>
<td></td>
</tr>
<tr>
<td>Windmill</td>
<td>–£0.40</td>
<td>£1.85</td>
<td>£2.25</td>
</tr>
<tr>
<td></td>
<td>(–£1.31–£0.35)</td>
<td>(£1.22–£2.47)</td>
<td></td>
</tr>
</tbody>
</table>

DISTANCE, FAMILIARITY AND WTP

One of the most important determinants of a household’s WTP to remove a stretch of pylons and have the HVTL run underground will be the frequency and the duration with which that household encounters those pylons. Households that encounter the pylons rarely and briefly are unlikely to benefit a great deal from the removal of the visual disamenity. Households that encounter the pylons frequently and for long periods are likely to benefit a lot. Accordingly we anticipate WTP to fall as the frequency and duration of encounters with pylons declines. As a matter of fact, we did collect data on the frequency (though not the duration) of encounters with pylons and we shall return to consider these data shortly. However, collecting measures of the frequency and duration of encounters in our sample and providing details of the relationship between this measure and WTP, would provide little guidance in benefits transfer exercises. In particular, without collecting new primary data, analysts would have no way of estimating the frequency and duration of encounters endured by households at a transfer location.

For practical purposes, we might propose that we choose to proxy frequency and duration of encounters by the distance a household lives from the stretch of pylons under consideration. Indeed, decreases in WTP with distance lived from an environmental good or amenity – ‘distance decay’ – have been observed in a number of studies (Sutherland and Walsh, 1985; Bateman and Langford, 1997; Pate and Loomis, 1997). In the United
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Kingdom, distance decay has been examined in a number of studies with interest arising from a crucial question for cost–benefit analysts, namely, the appropriate population across which to aggregate estimates of household or individual WTP across users and even non-users of an amenity (Moran, 1999; Bateman et al. 2000; Hanley et al. 2002).

In the current study, we might assume that those living closer to an HVTL will encounter the pylons more frequently and for longer than those living further away. As such, we also anticipate WTP to fall as distance from the pylons increases. The practical benefit of using distance is that this can be simply measured for any household and so facilitates benefits transfer exercises. Imagine we wished to estimate the benefits of a scheme proposing the undergrounding of an HVTL in a location not included in our sample. We could use the relationship between distance and WTP revealed by our sample data, to estimate which households in the transfer location would benefit from the scheme and by how much.

Does distance influence the values that households placed on either new pylon designs or undergrounding? In the case of the former, given that there is substantial indifference to options involving new designs, it is likely that less scope for testing standard predictions concerning the relationship between WTP and distance (see, for a more detailed discussion, Atkinson et al. 2004). For undergrounding, however, this option removes the source of the disamenity altogether. As such, respondents are apparently far less indifferent towards this proposal and it should be feasible to detect any distance decay effect should this exist.

The value of placing electricity cables underground exhibits a distance decay pattern as can be seen in Figure 10.4. As distance increases, away from the stretch of pylons being considered, mean WTP for undergrounding decreases. It should also be noted that even amongst those living furthest from the pylons (2–5 kilometres away), average WTP for undergrounding is significantly positive in Figure 10.4. We presume that WTP will continue to decline with increasing distance but have no data to support this claim. An important question is whether or not these differences between mean WTP across the distance bands that we can observe are statistically significant?.

To test the significance of the differences observed between the means of these four subsamples of the data we use a simple bootstrap procedure. These tests indicate statistically significant mean differences (i.e. with a 95 per cent level of confidence) between the (higher) mean WTP of respondents living in Band 1 (within 500 metres of the HVTL) and the (lower) WTP of respondents living in Band 4 (between 2 kilometres and 5 kilometres from the HVTL). Similarly, there are statistically significant mean differences in the WTP of those who live in Band 2 (between 500 metres and 1 kilometre from the HVTL) and Band 4. Mean differences for comparisons of WTP
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between other distance bands are not significant (with a 95 per cent level of confidence). On balance, however, there appears to be good evidence in Figure 10.4 that WTP declines with distance.

Nevertheless, the question remains as to how well distance approximates the frequency and duration of encounters with pylons. Figure 10.5 provides at least some answer to this question. Here: ‘often’ is defined as a respondent stating that he or she encounters the specified stretch of pylons everyday; ‘sometimes’ refers to encountering at least once a week (but not everyday); ‘rarely’ refers to encountering between once a fortnight and once a month; and, ‘never’ refers to never or seldom encountering the specified stretch of pylons. Notice first that, in Band 1, over 60 per cent of respondents state that they see pylons frequently. As we might expect, progressively fewer respondents state that they encounter pylons ‘sometimes’ (15 per cent), ‘rarely’ (12 per cent) and ‘never’ (11 per cent). Respondents in Band 2 live further away from the pylons and hence we would expect them to encounter the pylons less frequently. Encouragingly, this is the pattern we observe, rather fewer respondents indicate that they see the pylons often than did so in Band 1 (45 per cent as compared with 60 per cent), whilst rather more indicate they only encounter pylons sometimes (30 per cent as compared with 15 per cent). By the time we reach Band 3, the sometimes category contains more respondents (36 per cent) than the often category (32 per cent). Finally, in Band 4 the often category has fallen into third place (12 per cent) behind both the sometimes category (36 per cent) and the rarely category (35 per cent).

Broadly speaking these results are in line with our expectations. The percentage of respondents stating that they see pylons frequently
Measuring the visual disamenity from overhead electric transmission lines

Progressively falls with distance whilst those stating that they see pylons rarely progressively increases with distance. As we might expect, those stating they encounter pylons only sometimes first increases with distance then appears to begin falling off. Noticeably the percentage of respondents stating that they never see pylons remains stable at around 10 per cent for all four distance bands. The data suggest that there is something qualitatively different about the ‘never’ category. For example, one might conjecture that 10 per cent of the population seldom leave the confines of their homes. This 10 per cent will indicate that they never encounter pylons no matter how close they live to the pylons. Clearly, distance on its own is unable to capture all those aspects of behaviour that result in encounters with pylons. Nonetheless, Figure 10.5 provides enough evidence to indicate that distance provides a reasonable indicator of encounter frequency and hence we can be reasonably confident that using distance as a basis for benefits transfer exercises does not introduce undue bias.

Figure 10.5 Percentage of households encountering pylons at four different frequencies by distance lived from pylons
COST–BENEFIT ANALYSIS AND WTP TO REPLACE TRANSMISSION TOWERS

An important application of the findings discussed in this chapter is in the cost–benefit analysis of proposals concerning either *existing* or *new* pylons. A cost–benefit appraisal would seek to indicate whether (or not) such proposals were socially desirable. At present, formal cost–benefit assessments play little part (if any) in decisions regarding where to locate pylons in England and Wales. Rather this process is dealt with as part of the planning process, which observes the statutory obligation to supply electricity to households (otherwise known as Class G: Electricity undertakings – see, for example, Lichfield, 2003). This planning process entails rounds of discussion with stakeholders and local authorities and the avoidance (in general) of constructing HVTLs across certain designations of land (e.g. National Parks). However, an authoritative review of development rights by Lichfield (2003) discusses the finding that nearly a quarter of local authorities were dissatisfied with development outcomes, with at least some of this dissatisfaction arising from concern about the visual impact of overhead lines in rural and urban areas. Assessing the monetary benefits of, for example, lessening these visual impacts, could offer an additional input to evidence-based policy-making in this domain. In this section, therefore, we set out the basic steps that a cost–benefit appraisal would take and comment on how our estimates of WTP for visual amenity fit into this framework.

Turning first to the costs of implementing new pylon designs or undergrounding options, a proximate guide to these costs is presented in National Grid Company (2000). This indicates that erecting a 2 kilometre stretch of pylons costs £500 000 while 2 kilometres of underground cables costs £20 million. While it is not wholly clear whether these are present values and include all construction and maintenance costs, these data (taken at face value) are at least crudely indicative of the differences in costs between (current) pylons and undergrounding options.

Any exact estimate of cost depends on whether the proposal is replacing current pylons or is a wholly new transmission line project. For example, if the project is in a location where towers or overhead lines currently exist, the relevant baseline is the current lattice design. That is, ‘costs’ refer to the economic resources needed to remove the current pylons and replace them with some transmission alternative. Alternatively, if the project is in a location where *no* towers or overhead lines currently exist, the relevant baseline could still be the current lattice design when there is a statutory obligation to incur at least the cost of HVTLs supported by pylons of the current design. For example, in the case of the cost incurred if power lines
are buried underground minus the cost incurred if power lines supported by a stretch of lattice towers are erected.

A comparison of benefits with these cost estimates, in principle, can be facilitated by using the WTP findings outlined earlier in this chapter. However, where the project is the construction of wholly new pylons in a particular location – that is, the loss of a visual amenity relative to the status quo – then this amounts to an implicit assumption that the typical household values equivalent gains (e.g. WTP for more of an amenity) and losses (e.g. WTA compensation for less of an amenity) are approximately the same. Yet, there is a plethora of evidence across a range of environmental goods that WTA exceeds WTP perhaps by a factor of four or possibly considerably more for various environmental goods that have public good characteristics (Horowitz and McConnell, 2002).

The issue as to whether WTA or WTP is the correct measure of the welfare change essentially boils down to a judgement about property rights (see, for example, Mitchell and Carson, 1989). On the one hand, if households in the vicinity of the proposed transmission tower project have a property right to the current level of visual amenity then arguably it is WTA that should be used to measure benefits. If so, then in such cases, our findings are a lower bound on the loss that households place on the construction of new pylons and, for example, sensitivity analysis should take this into account. On the other hand, the statutory obligation to supply electricity to households might be taken to imply that property rights reside with households that require the transmission line to be erected. In this case, it is arguable that WTP can be used to measure the benefits of instead say laying the transmission line under the ground.

The total or aggregate benefit of replacing or removing a pylon is calculated as the product of a quantity and a price, where the former refers to the number of households affected by the project and the latter is household WTP (subject to the caveat previously noted). While, a precise estimate of the number of households for a particular location using GIS (Geographical Information Systems) would be desirable, precision in assessing exact household numbers is unlikely to make much difference to the probable finding that, for ‘typical’ landscapes, undergrounding options will not pass a cost–benefit test. That is, the number of households that would need to be affected by a project in order to make its net benefits zero (e.g. £19.5 million divided by £65.53) is likely to exceed substantially the actual number of households particularly in rural locations. It is worth noting at least two caveats to this conclusion. First, it is assumed that WTP for greater visual amenity is zero for all people living more than 2 kilometres away from a stretch of towers, which might be a rather strong assumption (e.g. from the evidence in Figure 10.4). Second, our WTP values do not
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capture possible benefits accruing to those who do not live near a stretch of pylons but encounter them regularly, say, whilst commuting or visiting the area. Third, it assumes that there is no non-use value arising from improving visual amenity. While it is inconceivable that there are non-use values associated with all land, at least some land is likely to be of relatively high marginal value perhaps because the surrounding landscape or its visual appearance is unique.

It is worth noting a final issue of some importance. The mean WTP values reported in Table 10.3 are heavily influenced by a minority of respondents who have strong and positive preferences for particular design options. Thus, these mean values disguise the fact that, especially with regards to different tower designs, a large proportion of respondents are indifferent to the proposed changes. Indeed, the median values reported in Table 10.3 indicate that the majority of respondents would not be prepared to pay anything in order to see a change in the design of transmission towers. So, whilst the WTP values indicate that there are social benefits realized from changes in the design of transmission towers, any policy that actually sought to recoup expenses for such changes through, for example, increasing electricity bills, would be deemed unacceptable by a majority of respondents.

CONCLUSIONS

This chapter has presented a number of issues related to alternatives to existing electricity pylons. First, we have examined to what extent rural and urban households prefer and are willing to pay for new tower designs and a scenario that would replace overhead transmission lines – and towers or pylons – with underground lines. Second, we examined the empirical significance of including negative WTP in estimates of the welfare effects of changing pylon design. Third, we sought to explore the influence of household location on WTP to replace pylons by placing transmission lines under the ground. Fourth, we have sketched some of the implications of our findings for cost–benefit appraisals, or evidence-based planning more generally, of new electricity transmission line proposals.

Our findings suggest that many people think positively about new designs in that a majority of respondents chose at least one new design in preference to the current design. That said, preferring a new design (to the current one) did not necessarily translate into being willing to pay to see that new design replace the old. Indeed, the majority of respondents were not actually willing to pay anything for new designs. Put another way, these people can be thought of as being ‘indifferent’ between maintaining and replacing the current design. In addition, a smaller but still notable
number of respondents stated that they would actually suffer a loss in their well-being if the current design was replaced by a new design. All of these factors were taken into account in this study in estimating WTP for new pylon designs. There was significantly more support for undergrounding and this did translate into a majority of respondents actually being willing to pay some (positive) amount for this option.

In order to elicit negative WTP, we sought indicators of how inconvenienced respondents would feel if the current towers were replaced by a less preferred tower design. That is, the time and expenditure that respondents were prepared to commit to opposing the change. This indirect ‘payment vehicle’ was designed to overcome problems with explicitly asking how much respondents were willing to pay to preserve the status quo or how much they are willing to accept in compensation. The results showed that ignoring the existence of welfare losses can result in some overestimation of the benefits of particular policy options. Whether these absolute differences, in this instance, conceivably could alter outcome of a cost–benefit assessment of the preferred course of action is debatable. Even if negative WTP is ignored, there remains considerable indifference towards options that entail changing pylon design rather than eliminating the source of the visual disamenity altogether.

We also find some evidence of the impact of household location on WTP for undergrounding: WTP declines with distance of a household from the site. This is reassuring from the perspective of, for example, benefits transfer exercises. Distance that households live from HVTLs at a transfer location can be relatively straightforwardly observed whereas frequency and duration of encounters with pylons cannot. Moreover, we argue that the former is a reasonable proxy for latter although distance is not able to capture all those aspects of behaviour that result in encounters with pylons.

Again, whether proposals such as undergrounding transmission lines themselves would pass a cost–benefit test is another matter. Important practical issues surround: whether residents living within an area to be affected by the proposed erection of new pylons have a property right to the status quo or prevailing level of visual amenity; and, in the face of a wide distribution of willingness to pay values whether mean willingness to pay (what the ‘average’ person would pay) and median willingness to pay (what the majority of people would pay) is the better indicator of the social value of the proposal. A cost–benefit test would typically use the former. However, the latter is arguably a more telling summary statistic with respect to the acceptability of a project that actually sought to capture the value of the benefit accruing to households arising from placing transmission lines under the ground. On the balance of evidence, our findings arguably indicate that for typical rural and urban landscapes the costs of novel proposals
for routing electricity transmission lines are unlikely to be exceeded by the (visual) benefits. There are, of course, caveats to this speculation not least about what is ‘typical’. Indeed, this specific issue points to at least one interesting avenue for future research: that is, visual benefits for ‘unique’ land of a likely higher marginal value.

NOTES

1. The focus of the chapter is on the estimation of the visual disamenity imposed by pylons of different designs; additional costs (real or perceived) possibly associated with HVTLs (high-voltage transmission lines) such as health impacts are not estimated.

2. Twenty-seven out of the 48 respondents that registered protest responses to changing tower designs also registered protest responses to the undergrounding option.

3. To calculate the survivor function at a particular value, say \( x \), first count the number of respondents with a value greater than this amount. An obvious estimate of the probability of having a value greater than \( x \), is given by dividing this count by the total number of respondents in the sample. Note that, in estimating the survivor functions in Figure 10.3, the sample data is simply used to dictate the specific form taken by the function.

4. The data collected from the payment ladder indicates the interval of values that contains each respondent’s maximum WTP. We use the lowest value in this interval to calculate a lower bound estimate of the sample’s true mean WTP.

5. For example, Day et al. (2001) analyse the CV data that we have been discussing using a semi-parametric regression model. Their findings indicate that, controlling for a range of other determinants, there is a higher likelihood that WTP is higher if a respondent encounters pylons relatively frequently.

6. By drawing with replacement from the original data, we generate 1000 bootstrap data sets each of the same size as the original sample. For each bootstrap replication we calculate and record the difference in subsample means. We then sort these recorded differences in ascending order and estimate the 95 per cent confidence intervals of the difference in means to be the values of the 25th and the 975th observations in this list. Furthermore, the probability that the mean of the first subsample actually exceeds that of the second can be estimated as the proportion of observations in the list with values falling below zero.

7. Of course, to the extent that currently erected pylons have a scrap value this item should also be included in the assessment of costs and benefits.

8. A (relatively) sophisticated approach would distinguish (if appropriate) a project’s location (i.e. rural or urban) as well as the number of households in each distance band and socio-economic or demographic characteristics of households in these locations.

9. For options that seek to replace designs, the conclusions are even plainer; that is, these proposals result in extremely small aggregate benefits for even the most favoured designs, while the least favoured designs could apparently lead to an overall loss in well-being if these replaced the current lattice design.

REFERENCES


11. Using choice experiments to value urban green space

Craig Bullock

INTRODUCTION

Estimates of the value of environmental goods help to ensure that these are properly taken into account once they are compared with costs in a cost–benefit analysis. This in turn improves the prospects of decision-makers paying more attention to environmental impacts when new policies or projects are implemented. Much has been written on the relative theoretical merits of alternative valuation techniques. Typically, researchers have provided examples of applications that demonstrate the relevant author’s assertion of the appropriateness of the method or of a particular approach. However, sufficient consideration is not always given to the ultimate use to which such values are put.

Some methods of environmental valuation expose their limitations where decision-makers require information on a variety of project scenarios, rather than just a single strategy or outcome. For instance, it is cumbersome to include a variety of different scenarios within a contingent valuation study. Environmental valuation requires that survey respondents are given as much of the full context of information as possible before they are asked to express a willingness to pay. While frequently argued in theory, in practice this is difficult to achieve in a questionnaire-based survey. There is a risk that the information provided will be insufficient and will contribute to any other of the numerous potential biases that the diligent researcher is desperately attempting to avoid. Where respondents are asked to consider various scenarios, the risk of information overload is increased. As an alternative, the researcher could prepare different versions of a questionnaire each describing a different scenario, but as reliable contingent valuation requires an adequate (usually large) sample size, this is time-consuming and costly to achieve.

Consequently, valuation is often limited to information on a discrete ‘policy on/policy off’ scenario. This, of course, is valuable where decision-
makers need a monetary measure of the benefits (or costs) of a policy. Indeed, there have been numerous instances where policy-makers have commissioned a valuation study simply to justify an existing policy or set of plans. However, there will be other occasions where valuation methods can be used to help design policy.

**CHOICE MODELLING**

**Origins**

The use of discrete choice experiments in the context of environmental valuation is based on the premise, expounded by Lancaster (1966), that the utility of goods depends on their constituent characteristics. This assumption is applied within conjoint analysis whereby people are asked to indicate their preferences for various goods using a ranking or rating scale. Conjoint analysis has frequently been used in market research to demonstrate consumer preferences for alternative products that vary only in terms of key characteristics. One of these characteristics can, of course, be product price.

Unfortunately, the application of conjoint analysis within economics has been stalled by economists' natural scepticism of any technique that elicits information by means of an ordinal ranking scale, or by a rating scale where relative distances between successive rating levels are ill-defined. Admittedly, this has not prohibited economists from being tempted to use rankings or ratings to provide supplementary information where conventional valuation methods have been used.

**Theoretical Foundations**

A breakthrough came when Louviere and Woodworth (1983) demonstrated how orthogonal factorial designs can be used to underpin various scenarios defined in terms of different characteristics, or attributes. Orthogonal factorial designs can be used to represent attribute combinations while removing all collinearity between attributes. These attribute levels can then be presented as choice sets where people are asked to choose between two or more alternatives and the fixed status quo. This they do by mentally trading off the benefits of one set of attributes, or attribute levels, against another.

As with contingent valuation, the exercise is still one of stated preference based on hypothetical scenarios. Actual preferences will vary within the population. For instance, people's perception of the same attributes will differ, as will their awareness of the full set of alternatives. These differences
between hypothetical and actual behaviour, or between different individuals’ preferences, represent a source of random error. From the researcher’s perspective, choice therefore becomes a function of both a deterministic element and a random element \( e \). If the latter is assumed to have a linear distributional form, utility can be represented as

\[
U_i = V_i + e_i
\]  

(11.1)

where \( V_i \) is the conditional utility for alternative \( i \) and is comprised of additively separable components with respective parameter values \( \beta \).

A framework for linking choice to behaviour is provided by the ‘random utility model’ (McFadden 1974). As \( e \) is unknown, a random utility model can be referenced to predict choice depending on the probability that \( V_i > V_j \). If it is assumed that all attributes influence choice in the same way,\(^1\) then the probability of choice can be estimated from the difference in the parameter values for alternative \( i \) and the composite alternative \( j \):

\[
P(i) = \frac{\exp(V_i)}{\sum \exp(V_j)}
\]  

(11.2)

Assuming an additive linear model, a multinomial logit analysis provides information on the odds, or the probability, of choosing one set of alternatives over another. The output also provides for an estimation of the marginal relative value of any one attribute in terms of another. This is in contrast to the discrete changes in environmental goods typically measured by contingent valuation.

**Survey Approach**

By means of a questionnaire, each recipient is presented with a number of choice sets in which each alternative is defined by a different set of attribute levels. To prevent respondent fatigue, it is typical to present no more than ten choice sets depending on the complexity of the exercise. Presenting a series of choice sets also allows the researcher to check the consistency of responses.\(^2\)

As well as presenting varying choice sets to each individual, the combination of attribute levels in each choice set varies from one survey recipient to the next. With anything more than a handful of attributes, the number of prospective combinations increases dramatically. Indeed, in a full factorial design, where every attribute level is combined with every other attribute level, hundreds of people can each receive a unique set of attribute pairs.
Consequently, more data is available than can be achieved through the use of contingent valuation. The huge amount of information collected on each respondent’s choices means that a well-designed survey can often arrive at significant values for most or all attribute level parameters.

Unlike rankings or ratings, economists are comfortable with the notion of choice. It is through the process of choice that goods are purchased or most policies decided upon. Economists can therefore respect a technique in which marginal attribute values can be derived from a large amount of choice data. If one of the attributes is a pricing attribute, such as an entry fee or package tour cost, then it is possible to measure the relative value of each of the other attribute levels in the same monetary terms. These attribute values become meaningful where one alternative is a reference level, for instance, the status quo. In principle, it is also possible to add these values together to arrive at a compensating variation measure of the benefit of one alternative over another.

There is still the same dependence on a payment vehicle (i.e. an entry fee) as occurs with contingent valuation, but choice experiments avoid the need to request an overt expression of willingness to pay. Not only is it easier for a respondent to choose an alternative than to express a willingness to pay, but the payment attribute is one or several attributes. This can ‘mask the true purpose of the exercise’ and make it more difficult for respondents to give a strategic response to influence the outcome of a study (Alpizar et al. 2002; Bristow and Wardman, 2004). It also reduces non-response by recipients who may object to such a willingness to pay question.

Estimated attribute values alone provide information to policy-makers. If, furthermore, these attribute values can be reliably combined into a package, then the policy-maker can be given information that demonstrates the relative value of taking one particular action over another. Taking the example of agri-environmental policy (e.g. Hanley et al., 1998), the policy-maker could be given information on whether to design a strategy where farmers are paid relatively more for actions that help to protect water quality than for landscape features (or vice versa). Furthermore, the results can be extended to demonstrate the level of water quality that people are prepared to pay for. It would appear, therefore, that the method provides policy-makers with more useful information with which to design a strategy than the relatively blunt valuation estimates supplied by other valuation methods.

**Relationship with Revealed Preference Methods**

Choice experiments have another major advantage. The process of asking people to choose between alternative packages of attributes is only fundamentally different from revealed preference techniques (e.g. the travel
cost method) in terms of the means of elicitation, namely the reliance on surveys (stated preference). Increasingly, researchers have been applying the same random utility approach to hedonic pricing or to travel cost (e.g. Earnhart, 2001) to arrive at estimates of the probability of choosing a property, or of travelling to a recreation site, based on that property or location's attributes. The approach allows for more flexible application than before. For example, with travel cost, more information is provided on the factors that make the destination attractive in the first place.

As an extension, the attribute-based data from hedonic pricing or travel cost can potentially be merged with data from a stated preference choice experiment. The error variance within each data set will vary, but this can be overcome by normalizing one data set's scale to unity. Where data is merged in this way, it has been argued that the revealed preference information reduces the hypothetical effect of the stated preference data by grounding the experiment in reality, for instance via a common baseline scenario. In turn, the stated preference data is not dependent on an existing scenario and can be used for forecasting the benefits of alternative scenarios, whereas revealed preference data may depend on a small number of actual destinations or property sales. Being based on an orthogonal design, the stated preference data also reduces the influence of the multi-collinearity that inevitably occurs with revealed preference data where attributes may be physically related to one another. Furthermore, in environmental economics, it is not uncommon to find that the influence of environmental variables on, for example, property prices can be weak and difficult to identify. In the choice experiment, there are less restrictions. By being listed as an explicit attribute, the environmental attribute's value can more easily dissected from the estimates of other attribute values.

APPLICATION: URBAN GREEN SPACE

Experimental Design

Dublin has a good number of open spaces, but much of this could be described as being either of two extremes, that is, formal parkland or rather featureless open space. Large green areas were set aside during the development of new housing estates up to the 1990s, but unimaginative design, combined with the low funding priority given to parks, has meant that many of these spaces have become little more than open landscapes of grass with few facilities. More recently, government guidelines have encouraged local authorities to reverse the former policy of low-density suburbanization by providing for the infill development of some of these
open spaces. Simultaneously, private apartment development is removing the few remaining pockets of semi-natural green areas, so further reducing the diversity of space.

The Greenspace study first applied a factor analysis to 40 attributes to help identify the motivations for park use. This was then followed by the core choice experiment to determine what attributes of green space provide people with the most utility. Eight key green space attributes were selected, namely park size, maintenance, vegetation, water features, play facilities, other facilities (seating, paths and trails), number of users and journey time. Each of these was described in detail before being presented at two or three levels in the choice sets. For example, for vegetation, these levels were ‘mostly mown grass few trees’, ‘parkland with scattered trees’ and ‘mostly woods and meadows’.

In all, 500 householders were interviewed. Each was presented with eight choice sets (see Figure 11.1) in which the attribute combinations were almost unique for each respondent. They were then asked to choose which of two hypothetical green spaces they would be most inclined to visit. This was followed by a ranking question in which respondents were asked to rank the two parks in relation to their usual park destination (or their nearest park). In this context, the ranks could also be analysed one level at a time using logit according to the so-called exploding rank method (Chapman and Staelin, 1982). Supplementary questions were asked about nature of park visits, the attributes of the respondent’s usual park, mode of transport and personal characteristics.

The choices that respondents made between combinations of attributes were modelled to provide a quantitative estimate of the marginal value of any one attribute level. In addition, because ‘Journey time’ can be given an opportunity cost in terms of the alternative use of leisure time, it was possible to quantify the value of each attribute level in monetary terms for the purposes of a cost–benefit analysis.

Results

The analysis provided coefficients for each of the attributes in relation to the base level. Most parameters were significant. Table 11.1 also shows that positive coefficient values are associated with a moderate level of facilities, as well as with a ‘mix of quiet and busy areas’, ‘advance play facilities’, ‘scattered trees’, and ‘riverside walks’. ‘Adventure play facilities’ have a consistently high positive coefficient. ‘Scattered trees’ too have a positive influence on choice, although this does not extend to ‘woods and meadows’. The negative coefficient on woods and meadows appears to arise
A LARGE PARK

Moderate maintenance
Plenty of scattered trees and grass
Riverside walk
No playground
Plenty of seating, paths, trails and cycle paths
Mix of busy and quiet areas
45 minutes by foot/20 minutes by car

B SMALL LOCAL PARK

High maintenance
Woods and meadows
Natural-looking lake
Small playground
Only a few benches and paths
Tends to be quite busy
45 minutes walk/20 minutes by car

Comparing just parks A and B which would you prefer to visit? Tick one

A = □□□□□ □□□□□ B = □□□□□ □□□□□ Not go/Do something else □□□□□

Compared with your usual park which do you consider best (1st), second best (2nd) and third best (3rd)?

A = □□□□□ □□□□□ B = □□□□□ □□□□□ Your usual park □□□□□ Rank

Figure 11.1 Choice set example
from respondent fears over personal security, although this same attribute level becomes slightly positive when the data is restricted to large parks only. 'Journey time' is also significant with a negative coefficient as is to be expected where utility decreases with distance.

Table 11.1  
Attribute coefficients and marginal economic values  
(multinomial logit)

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Coefficients</th>
<th>Attribute Value Per Visit</th>
</tr>
</thead>
<tbody>
<tr>
<td>Size (large)</td>
<td>-0.205</td>
<td>-€3.07</td>
</tr>
<tr>
<td>Maintainance (high)</td>
<td>0.013</td>
<td>€0.06</td>
</tr>
<tr>
<td>Woods and meadows</td>
<td>-0.197</td>
<td>-€0.95</td>
</tr>
<tr>
<td>Scattered trees</td>
<td>0.140</td>
<td>€0.67</td>
</tr>
<tr>
<td>Mostly mown grass, few trees</td>
<td>-0.057</td>
<td>€0.27</td>
</tr>
<tr>
<td>Riverside walks</td>
<td>0.071</td>
<td>€0.34</td>
</tr>
<tr>
<td>Natural-looking ponds/lakes</td>
<td>-0.028</td>
<td>-€0.13</td>
</tr>
<tr>
<td>Small man-made pond</td>
<td>-0.043</td>
<td>€0.21</td>
</tr>
<tr>
<td>Adventure play facilities</td>
<td>0.135</td>
<td>€0.65</td>
</tr>
<tr>
<td>Small playground</td>
<td>0.146</td>
<td>€0.70</td>
</tr>
<tr>
<td>No playground</td>
<td>-0.281</td>
<td>-€1.33</td>
</tr>
<tr>
<td>Plenty of surfaced paths, seating, trails and cycle paths</td>
<td>0.288</td>
<td>€1.38</td>
</tr>
<tr>
<td>Surfacd paths, some seating</td>
<td>0.186</td>
<td>€0.89</td>
</tr>
<tr>
<td>Few paths and seating</td>
<td>0.474</td>
<td>-€2.27</td>
</tr>
<tr>
<td>Can be quite busy</td>
<td>-0.307</td>
<td>€1.47</td>
</tr>
<tr>
<td>Mix of busy and quiet areas</td>
<td>0.197</td>
<td>€0.95</td>
</tr>
<tr>
<td>Few people around</td>
<td>-0.110</td>
<td>€0.53</td>
</tr>
<tr>
<td>Journey Time (four levels)</td>
<td>-0.033</td>
<td></td>
</tr>
<tr>
<td>Log-likelihood (parameters)</td>
<td>-1941.61</td>
<td></td>
</tr>
<tr>
<td>Log-likelihood (no parameters)</td>
<td>-2196.13</td>
<td></td>
</tr>
<tr>
<td>Adjusted $p^2$ (significance)</td>
<td>0.113</td>
<td></td>
</tr>
</tbody>
</table>

The results in Table 11.1 represent average parameter values. However, it can be expected that the utility that people associate with visits to green space varies depending on the purpose of their visit or their personal characteristics. A random parameters, or mixed logit, approach was therefore used to identify the sources of variation in the data. In many cases, this variation was indeed related to the respondents’ socio-demographic characteristics such as age or gender. Using this approach raised model significance to up to 0.433.
For example, more frequent visitors appear to appreciate the presence of a playground, a result that is consistent with the observation that people with children are amongst the most frequent visitors to parks. Indeed, for the subset of respondents with dependent children it was no surprise to find that the coefficients on play facilities were higher. Although, the coefficient for ‘natural-looking ponds/lakes’ was negative, possibly due to parents’ concern with safety despite the familiar image of children feeding ducks. Other significant differences and interactions were found between size and journey time, socio-economic class and journey time, age and woodlands, and gender and woodlands.

ISSUES IN RELATION TO THE APPLICATION

The analysis of the choice experiment data resulted in a good level of significance for most attribute parameters, and model fit improved once the data was split into particular subsets based on use or socio-demographics. The following provides some discussion of issues that should be borne in mind by researchers who wish to use choice experiments for environmental valuation.

Journey Time and Model Fit

The habitual assumption of a linear model is rather restrictive, but linearity does permit a straightforward monetary estimation of the value of each attribute by dividing the coefficient values for the physical parameters by that for ‘Journey Time’. Table 11.1 gives these values per park visit. However, ‘Journey Time’ proved to be both an interesting and awkward parameter to include in the experiment. An earlier question had estimated the value that respondents place on their leisure time by means of a contingent rating exercise. However, the true value of leisure time remains more difficult to pin down than might be the case for work time, which has a clearer opportunity cost in that the alternative of earning an income provides a more reliable yardstick (at least for those respondents who do work).

Further investigation of the data revealed a relationship between the disutility of journey time and journey inconvenience, such that lower socio-economic classes placed a high value on short journey times, particularly in cases where they had children. Interestingly, journey time ceased to have a high coefficient for some socio-economic classes at weekends. At these times, it appears that people have made a commitment to a family outing to a more distant park.
Model Significance

The significance of the ‘Journey Time’ parameter was such that, for some socio-economic or user groups, model significance was greatly improved where people held a similar disutility for journey time. Likewise, model significance was also higher within groups who could be expected to hold similar preferences, such as frequent users or those with dependent children. However, for the most part, variation in preferences could not easily be identified by the usual socio-demographic variables, but appeared to be more inherent to the respondent’s own personality and motivations. Model significance could probably have been improved had an analysis (i.e. latent class) been used to link the choice experiment to the results of the earlier factor analysis. Similarly, model significance was improved where more information was available on the context in which respondents visit parks, for example, dedicated trips, exercise, passing through, etc.

Numbers of Attributes

A critical consideration for the application of choice experiments is that respondents must be able to perceive a good in terms of its attributes rather than just in its entirety. A related limitation is that rather few attributes can be included to describe the good before the underlying statistical design becomes unmanageably large. In this study, more attributes could have been included if two attribute levels had been described rather than three. However, offering a choice of just no park facilities or many park facilities, is arguably of little practical use to decision-makers. Including more attributes means that there is also less information on interactions between attributes. If the design does not allow for an analysis of these interactions, then the coefficients for these two attributes in isolation could be inaccurate. There is also the important risk of an exercise becoming too difficult for respondents if many attributes are included. This would lead to a risk of error and consequently poor parameter significance.

A promising approach is suggested by attempts to restrict the design (D-optimal designs) to those attribute levels between which people are more likely to make trade-offs. This requires pre-testing and some judgement. Potentially, though, these more efficient non-linear designs could allow more attribute combinations to be examined.

Merging Stated Preference with Revealed Preference

In an earlier question people had been asked which parks they visited most frequently and which they did not. In addition, the second question
in the choice set in Figure 11.1 calls for a similar comparison between the hypothetical alternatives and the respondent’s usual park destination. These questions are therefore asking respondents about their revealed behaviour. Analysis of these two questions resulted in a far better model fit than the analysis of the stated preference data alone. However, this better fit is illusionary and arises largely from the actual collinearity that exists between attributes in the usual park choices. Typically, popular parks are well-managed, contain good facilities including playgrounds, and are busy. While it has been argued that the merger of revealed and stated preference data helps to ground an exercise in reality, the benefits of the latter’s orthogonal design are much diluted. In practice, there are many instances of high collinearity and researchers must therefore consider whether the nature of the good merits merging the two databases.

In the Greenspace study, a high proportion of respondents preferred their usual park to the hypothetical alternatives. It is not unusual to find a bias towards the status quo. More reliable results were, though, achieved where those respondents who chose the usual park alternative in most of the eight choice sets were removed from the analysis. This left only those cases where respondents were prepared to trade off attributes.

CONCLUSION

Choice experiments have a number of advantages over other valuation methods. Not least, they are useful for providing decision-makers with information of the utility that people attach to various environmental attributes. This information can be used by decision-makers to adapt policy in a way that provides the greatest public welfare.

A fundamental consideration is whether survey respondents are able to perceive goods in terms of their attributes. While this may be true of some applications, for instance climbing (Grivalja et al. 2002), fishing (Hauber and Parsons, 2000) or hunting (Bullock et al., 1998), for park use much depends on information on context and user type. Contingent valuation has been argued to be a preferable approach where discrete scenario changes are being considered (Kriström and Laitila, 2002).

There are also issues as to whether the number of attributes that can be included in a choice experiment can sufficiently describe the good in question and whether the omission of certain interactions or consideration of varying preferences can provide an unrepresentative result. If such factors are taken into account, then choice experiments can, indeed, provide a reliable alternative method of environmental valuation.
NOTES

1. The assumption that all random elements are independently and identically distributed (IID) is a key feature, and weakness, of multinomial logit approach to estimation.
2. There is a risk of serial correlation between factors leading to each respondent’s choices.

REFERENCES


12. Valuing the environmental benefits of water industry investment in England and Wales

Bruce Horton and Jonathan Fisher

INTRODUCTION

The Environment Agency, the main environmental regulator in England and Wales, has undertaken assessments of the benefits of water quality and water resource improvements in order to identify what action might be required by the UK private water companies to address environmental problems. The purpose was to inform UK government ministers on environmental requirements for the 4th Periodic Review of the water industry.

This chapter outlines the benefit assessment work undertaken. The next section provides some policy background and introduction to the process, outlining the economic appraisal approach adopted for different types of schemes. The third section details the methodologies employed to assess the environmental benefits associated with 1. the overall environment programme, and 2. a subsection of discretionary schemes that were subject to a detailed cost–benefit analysis. The common and consistent methodology was based on applied benefit transfer techniques. The fourth section presents the main results of the different types of assessment. The fifth section reflects on what went well and what problems were encountered during what was a complex process, the like of which had not been attempted on such a scale in the UK before. Section six then takes forward some of the key lessons learnt for the future.

POLICY BACKGROUND

Every five years, the Office of Water Services (OFWAT), which regulates the private water sector, reviews the business plans and assesses the future revenue needs of water and sewerage companies in England and Wales. OFWAT also sets annual price limits for each company for the following
five years, reflecting assumptions about what they need to spend to meet
their capital expenditure programmes and to finance their operations. The
4th Periodic Review of the water industry (PR04) covers the period from
2005 to 2010. The Environment Agency provides advice to the Secretary of
State for the Department for Environment, Food and Rural Affairs (DEFRA)
in England and the Minister for Environment, Planning and Countryside in
the Welsh Assembly Government (WAG) on the environmental requirements
for the review.

To inform this process, the Environment Agency (EA) drew up a list of
potential water quality and water resource improvement schemes affecting
some 6000 assets where it was believed that water company action was
required to address an environmental problem. This was known as the
National Environment Programme (NEP). In January 2003, ministers issued
initial guidance that specified the following types of schemes relating to the
types of decisions that had to be made within the NEP and the extent of
discretion that ministers had on them (DEFRA, 2003):

- Type 1: ‘essential and clear’ (DEFRA) or ‘established requirements’
  (WAG);
- Type 2: ‘essential when clarified’ (DEFRA) or ‘expected requirements’
  (WAG);
- Type 3: ‘choices will be made’ (DEFRA) or ‘subject to policy decisions’
  (WAG).

Table 12.1 outlines the types of economic appraisal techniques carried out
for different types of scheme, in accordance with the principles set out in the
guidance. Further details can be found in Environment Agency (2003a).

Table 12.1 Economic appraisal techniques for types of schemes

<table>
<thead>
<tr>
<th>Type of Scheme</th>
<th>Economic Appraisal Technique Applied</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Cost-effectiveness</td>
</tr>
<tr>
<td>Type 1</td>
<td>X</td>
</tr>
<tr>
<td>‘Essential and clear’ (DEFRA) or ‘established requirements’ (WAG)</td>
<td></td>
</tr>
<tr>
<td>Type 2</td>
<td>X</td>
</tr>
<tr>
<td>‘Essential when clarified’ (DEFRA) or ‘expected requirements’ (WAG)</td>
<td></td>
</tr>
<tr>
<td>Type 3</td>
<td>X</td>
</tr>
<tr>
<td>‘Choices will be made’ (DEFRA) or ‘subject to policy decisions’ (WAG)</td>
<td></td>
</tr>
</tbody>
</table>
The EA worked closely with companies to ensure that our proposed programme was as cost-effective as possible. This reduced the total number of schemes by about 48 per cent and cut costs by over 50 per cent compared with those of the initial programme in water company draft business plans. The risks of non-compliance were then assessed in cases where clarification regarding the need for schemes or outcomes was required. Cost–benefit analysis was used to help determine whether or not 500 or so Type 3 schemes should proceed. In this chapter, we describe the process of assessing the benefits associated with: 1. All three types of scheme (the overall programme); 2. Type 3 schemes only. Environment Agency (2004a, 2004b) describe these processes in more detail.

METHODOLOGY

Detailed guidance was prepared to enable the environmental benefits of all schemes in the programme to be assessed consistently (Environment Agency, 2003b). This Benefits Assessment Guidance (BAG) was developed for use by both EA and water company planners, most of whom are non-economists. Using the BAG, the benefits of schemes impacting upon various types of water body and their associated ecosystems, including groundwater, rivers, freshwater fisheries, bathing and shellfish waters, were assessed.

The BAG is based on valuation where as many as possible of the impacts (both positive and negative) are measured in monetary terms. Where the use of monetary values was not considered to be robust, or where original monetary estimates did not exist, the BAG uses qualitative descriptions and semi-quantitative assessment. Monetary valuation has the advantage of allowing the environmental and social impacts of a scheme to be summed and compared with the capital and operating costs. Benefits transfer (BT) was used to derive monetary values. BT takes an estimate of the value of an environmental attribute from an existing study or studies at one or many sites, and transfers it to a new site. It is a widely used valuation method but recognized as appropriate for valuing non-market benefits, especially where the number of schemes to be assessed would preclude original valuation surveys, as in this case. BT may introduce various kinds of bias (see Bateman et al. 2002), but we attempted to minimize these through the use of the most temporally, spatially and methodologically appropriate studies to derive a value for each environmental attribute that could be applied in different contexts.

The BAG also sets out how to derive average or typical estimates of the number of people or households who would gain from a particular type
of benefit in a particular type of location. This is clearly important when aggregating monetary valuations to derive total benefit estimates.

The BAG was thoroughly reviewed and tested. Peer reviewers included relevant policy stakeholders, academics and representatives from the water companies. A seminar to discuss the BAG (Environment Agency, 2003c) noted the uncertainties associated with benefits transfer and provided advice on ways to address these. Overall, it concluded that our work was serious and creditworthy and that the BAG made the best use of available data and was the best that could be done within the time frame. A further workshop, attended by all key stakeholders and some of the main authors of the original studies, discussed and reviewed some of the most important and uncertain valuations used to assess the environmental benefits of schemes, particularly those associated with non-use and bathing water benefits (Environment Agency, 2003d).

The methodology used for calculating the benefits of both the overall programme and the smaller set of Type 3 schemes was broadly similar and involved the following steps:

1. Identification of the environmental driver related to the scheme and estimation of the environmental outcome of a water quality or water resource improvement scheme. This may be in terms of kilometres of water body improved by a river ecosystem level (RE), a classification we use to assess the chemical quality of water bodies with five categories, from RE1 (very high) to RE5 (very poor), or reduction in size and frequency of low flow events. When assessing the overall programme, we assessed the overall environmental impact of all schemes on each stretch rather than simply adding up the separate impacts of individual drivers or schemes. This approach minimized the risk of double counting.

2. Identification of the relevant benefit categories associated with the improvement. The full list is as follows:

- informal recreation;
- angling;
- commercial fisheries;
- in stream recreation;
- amenity, property prices;
- abstractions;
- groundwater resources;
- heritage, archaeology and landscape;
- coastal bathing;
- shell fisheries;
- ecosystems and natural habitats.
3. Qualitative description of impacts and assessment of whether they were likely to be significant enough to affect a scheme’s justification.

4. Quantitative assessment of impacts where they were significant, involving calculation of the area affected, number of users or visits per annum, etc.

5. Monetary valuation of benefits (disbenefits) where possible, using values from the BAG. To ensure that the costs and benefits were in a comparable format, we calculated the present values of both over a common time horizon of 25 years using the Treasury recommended discount rate for public project appraisal of 3.5 per cent.

6. Application of sensitivity analysis to monetary estimates to provide ranges of values (low, central and high).

7. Aggregation of individual schemes and drivers to provide total estimates of the value of environmental benefits over each water company region.

So that the costs and benefits of Type 3 schemes could be compared in detail, water companies provided estimates of the financial and environmental costs of these schemes in their draft business plans. OFWAT reporters then analysed and reviewed these costs and OFWAT applied low and high sensitivities. More detailed explanations of the methodologies used in the benefits assessment process are available in Environment Agency (2004c, 2004d).

RESULTS

This section first discusses the benefits associated with the overall environment programme, and then looks at the benefits of those schemes that were subject to a detailed cost benefit assessment.

Overall Programme

Table 12.2 shows the overall benefits estimated to accrue from both the statutory (Type 1 and 2) and discretionary (Type 3) parts of the environment programme.

The overall environmental benefits from the proposed PR04 programme are significant and were estimated to be between £3.1 and £5.6 billion with the full implementation of all three types of scheme. These benefits are equivalent to around £218 million to £397 million per year, or between about £10 and £18 per household in England and Wales per year. They do not include the environmental disbenefits of Type 3 schemes (e.g. external
Valuing the environmental benefits of water industry investment

Costs of energy for sewage treatment or congestion from road works), which water companies were able to assess using Part 5 of the BAG. However, only one company assessed any disbenefits for schemes of this type, estimating disbenefits to be less than 1 per cent of the total environmental benefits.

Table 12.2 Overall benefits of Type 1/2 and 3 schemes

<table>
<thead>
<tr>
<th>Benefits per year (million)</th>
<th>Type 1/2 Schemes (£)</th>
<th>Type 3 schemes (£)</th>
<th>All schemes (£)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Benefits per household per year</td>
<td>211–383</td>
<td>8–14</td>
<td>218–397</td>
</tr>
<tr>
<td>Capitalized benefits (million)</td>
<td>2988–5425</td>
<td>102–204</td>
<td>3091–5,629</td>
</tr>
<tr>
<td>Benefits per household per year</td>
<td>9.62–17.49</td>
<td>0.36–0.64</td>
<td>9.98–18.13</td>
</tr>
</tbody>
</table>

Overall Benefits by Category

Table 12.3 shows the total benefits under each benefit category for all types of scheme.

Table 12.3 Total benefits by category

<table>
<thead>
<tr>
<th>Category</th>
<th>Total Benefits (£ million per year)</th>
<th>Benefits per Household (£ per year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Informal recreation</td>
<td>5.0–11.1</td>
<td>0.23–0.51</td>
</tr>
<tr>
<td>Angling</td>
<td>16.2–16.9</td>
<td>0.74–0.77</td>
</tr>
<tr>
<td>Amenity</td>
<td>13.5</td>
<td>0.62</td>
</tr>
<tr>
<td>Bathing</td>
<td>15.8</td>
<td>0.72</td>
</tr>
<tr>
<td>Groundwater</td>
<td>30.4</td>
<td>1.39</td>
</tr>
<tr>
<td>Ecosystems and natural habitats</td>
<td>137.3–309.8</td>
<td>6.27–14.14</td>
</tr>
<tr>
<td>Total</td>
<td>218–397</td>
<td>9.98–18.13</td>
</tr>
</tbody>
</table>

The highest single benefit is attributable to improvements and protection of ecosystems and natural habitats (ranging from about £137 million to over £300 million per year). These are ‘non-use’ benefits and indicate the very real importance attributed to environmental quality by the public. The importance of environmental quality to the public is shown in many economic valuation studies. In this case we have used values relating to
Amenity and water quality

ecosystems and improved water quality in a nationwide study for England and Wales (Willis and Garrod, 1996, cited in FWR, 1996). This found that, across the country as a whole, the value for improved water quality and associated ecosystems was 0.2 to 0.5 pence per household per year per kilometre improved. Box 12.1 provides some examples of these types of benefits that will be delivered by the PR04 programme.

BOX 12.1 BENEFITS TO ECOSYSTEMS AND NATURAL HABITATS

Clumber Park SSSI

Clumber Park SSSI (Site of Special Scientific Interest) is one of the largest areas of mixed habitat in Nottinghamshire and contains an area of open water, Clumber Lake, which is surrounded by alderwood, open sedge marsh and reedswamp. The lake itself is notable for its colonies of short leafed water-starwort (*Callitriche truncata*). The Park also supports a diverse bird community including wintering birds such as mallards, gadwall, tufted duck and pochard, which feed on the emergent aquatic plants.

A reduction in nutrient levels is needed at Clowne, Creswell, Whitwell and Langwith sewage treatment works (STW) to help to maintain a diverse assemblage of aquatic plants in Clumber Lake. Nutrient removal at the STWs would also benefit the following additional sites: Welbeche Lake SSSI, which supports a notable bird community, including little grebe, sedge warbler and reed bunting; the River Poulter network, which supports a number of important species including water voles, as well as a coarse fishery.

River Wensum

The River Wensum in Norfolk is one of the best examples in the UK of a naturally enriched chalk lowland river. The river catchment is low-lying and largely rural, with intensive arable farming in many places and a few large urban settlements. STW and other discharges, if not properly regulated, can cause poor water quality and increase nutrient concentrations. Fertilizer applied to farmland may wash into rivers and elevate nutrient concentrations. These factors can be exacerbated by abstraction for both public water supply and crop irrigation, which reduce flows and water levels in the river.
The results of a joint customer survey undertaken as part of the PR04 work plan confirm the significance of environmental quality (MVA, 2003). This found ecological benefits to be one of the top four priorities for customers. A similar survey found that improving ecological quality of rivers such that they support rich aquatic life ranked second only to reductions in leakage (Yorkshire Water, 2002).

Table 12.3 shows that more tangible use type benefits related to informal recreation, angling (the most popular participatory sport in the UK) and cleaner bathing waters, were also important. An example of the benefits associated with bathing water improvement schemes is shown in Box 12.2.

The Wensum is a riverine SSSI, approximately 70 kilometres in length. It supports over 100 species of plants, a rich invertebrate fauna and a relatively natural corridor. It is probably the best whole river of its type in nature conservation terms in the UK. The upper reaches support typical chalk stream vegetation, whilst the washland comprises managed grassland with areas of fen, wet grassland, wet woodland and reedbed. The nationally rare Desmoulin’s whorl-snail, Vertigo moulinisiana, a species of stonelly, Amphinemura standfusii, more usually associated with upland rivers, and the flatworm, Crenobia alpina, which is noteworthy as it is a relic in Southern England where it is confined to coldwater springs, are also present.

The river is also a cSAC (candidate Special Area of Conservation). It is one of the best areas in the UK for the white-clawed crayfish, Austropotamobius pallipes. It supports a significant presence of the bullhead, Cottus gobio, and the brook lamprey, Lampetra planeri. It also supports a significant presence of Vertigo moulinisiana, and is one of the best areas in the UK for water-crowfoot, Ranunculon fluitantis. The river is also a part of the Broadland RAMSAR wetland and SPA (Special Protection Area) designation.

However, river pollution, particularly through nutrient enrichment, and changes to natural flow regimes is a threat to their continued success. Our planned programme for PR04 will build on work already scheduled. Ongoing improvements to the larger STW will improve water quality and ensure that nutrient concentrations begin to fall, but more work still needs to be done. By targeting eight smaller discharges, a further reduction in nutrients, especially when combined with enhanced river flows, the proposed measures in PR04 are expected to result in additional improvements to plant and animal communities.
In addition, there are specific benefits for local residents from an improved water environment. These are less tangible than recreational or fishing benefits, but contribute to the broader quality of life of people in the locality. We have therefore included estimates of these amenity benefits on the basis of the documented increase in value of riparian properties with good water quality as an indicator of improved amenity from an enhanced water environment.
Lastly, under the groundwater drivers of PR04, one of the effects of cleaner discharges from STW is to reduce water pollution that would otherwise adversely affect groundwater quality, thus increasing the availability of good quality groundwater resources. We estimated the benefits of these improvements based on resource values for groundwater of £1 million per megalitre per day, the value used in water resource plans by the Agency. These values are calculated on the basis of reduced costs of treatment and the increase in quality and quantity of groundwater.

Non-monetized Benefits

Improvements to water quality can also contribute towards wider economic gains and social benefits, through regeneration and stimulation of economic activity in deprived areas. No attempt was made to put a monetary value on these economic and social benefits as they very much depend on the local context and it is difficult to determine the importance of water quality improvements independently of other drivers. In addition, if benefits accruing in one area result in some displacement of economic activity elsewhere then net national gains may be limited. However, this does not detract from the importance of such benefits to many local economies and communities.

Improvements to the environment can lead to regeneration or wider economic benefits in a number of ways:

- In areas of decayed urban infrastructure and poor environment, improving the water environment is an essential component of programmes for urban regeneration.
- Enhancing and maintaining a good water environment sustains socio-economic activity in an area, and is important to underpin future developments.
- Enhanced water quality is important for creating and sustaining economic activities based on leisure and recreation and tourism.
- Improving and maintaining good water quality can help to diversify economic activities, for example in rural areas where traditional activities are at risk.

There are a number of other types of benefits for which monetary values could not be estimated. These included:

- Benefits arising outside the actual identified stretch where schemes are undertaken. For example, schemes affecting rivers will have
downstream impacts on estuaries and coastal zones, but we have not been able to estimate the value of these benefits.

- Benefits to shell fisheries that lie mainly in commercial gains to the shellfish industry, information on which is not in the public domain.
- Benefits in estuaries or coastal stretches other than beaches. For example, in the Southwest, reductions in unsightly discharges to coastal waters will enhance enjoyment of walking by the coastal path.
- Benefits from improvements to in-stream recreation, since this may have led to double counting with informal recreation benefits.
- Benefits associated with schemes aimed at reducing the impacts of discharges into the Thames Tideway. These schemes are being assessed independently and are outside the scope of this analysis.
- Investigations. These are undertaken to substantiate the risks and impacts of discharges or abstractions. Once the investigations are completed, cost-effective solutions can be developed. Even if investigations do not find a problem, or find that its cause is something other than water company activity, the benefit of investigations will be in terms of saving unnecessary expenditure by the water companies.

Schemes Subject to Cost–Benefit Assessment

Type 3 schemes were assigned to one of six categories on the basis of their benefit cost (B/C) ratio and additional important non-monetary benefits as follows:

1. Schemes identified by local stakeholders or us as being of high regional priority but which had a B/C ratio of < 1.2. Such schemes might, for example, be important in regional economic development or regeneration strategies, or there might be other significant non-monetary benefits that the BAG was not able to capture (see above);
2. Monetary benefits at least twice as great as costs.
3. Monetary benefits less than double but > 1.2 times costs. This latter ‘bound’ was selected to take account of the level of uncertainty in the benefit assessment methodology.
4. Monetary benefits < 1.2 times but > 0.8 times costs. We paid particular attention to this category and provided additional information and justification for those borderline cases that were local or regional priorities – these were included in category 1 above. We recommended that the remaining borderline cases in category 4 should be deferred.
5. Monetary benefits < 0.8 times costs.
6. Monetary benefits > 1.2 times costs but which the EA recommended should be deferred.

Categories 1 to 3 comprised the schemes that we recommended for implementation in PR04. Categories 4 to 6 comprised the schemes recommended for deferral. Figure 12.1 shows the total number of water quality and water resource schemes across England and Wales falling into each of six categories. Figure 12.2 shows the total cost of schemes in each category. Table 12.4 shows the overall number of schemes and their total costs and benefits for categories 1 to 3 (recommended for implementation) and categories 4 to 6 (recommended for deferral).

![Figure 12.1 Number of Type 3 schemes in each cost–benefit category](image1)

![Figure 12.2 Total cost of Type 3 schemes in each category](image2)
Figures 12.1 and 12.2 and Table 12.4 show that 272 schemes (62 per cent of the total) were proposed for implementation under PR04, which would yield benefits with a net present value of £1.2 billion. This significantly exceeded the cost of £645 million. Figure 12.1 shows that most of these schemes had benefits that were double their costs. It was proposed that schemes costing more than £1 billion be deferred. The proposed schemes would therefore deliver about 80 per cent of the total environmental benefits at 38 per cent of the total costs, achieving good value for money from the programme.

Table 12.4  Summary findings of costs and benefits of Type 3 schemes

<table>
<thead>
<tr>
<th>Categories</th>
<th>1 to 3 (% of total)</th>
<th>4 to 6 (% of total)</th>
<th>All categories</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total number of schemes</td>
<td>272 (62%)</td>
<td>166 (38%)</td>
<td>438</td>
</tr>
<tr>
<td>Total costs (£ million)</td>
<td>645 (38%)</td>
<td>1040 (62%)</td>
<td>1685</td>
</tr>
<tr>
<td>Total benefits (£ million)</td>
<td>1154 (80%)</td>
<td>289 (20%)</td>
<td>1443</td>
</tr>
</tbody>
</table>

Type 3 Schemes by Category

Figure 12.3 shows the proportion of all (water quality and water resources) Type 3 scheme benefits by benefit category.

Figure 12.3  Total benefits of Type 3 schemes by category
Figure 12.3 clearly shows that the non-use category dominated the calculated benefits for Type 3 schemes. As discussed above, this high proportion is in line with other study and survey findings showing that people attach most importance to the ecological benefits of improved aquatic ecosystems and natural habitats. Other important categories were bathing waters, informal recreation, angling and amenity.

DISCUSSION

There were a number of areas where the assessment of benefits at both the overall level, and for Type 3 schemes only, was successful and some other areas where problems were encountered. These are discussed in this section.

What Went Well?

The successful features of the approach were:

• BAG delivered the assessments and analyses on time and to budget, thereby optimizing the use and efficiency of public funds.
• The EA collaborated well with DEFRA, OFWAT and other stakeholders in developing the BAG and carrying out and reviewing both the overall and the individual assessments.
• The EA liaised well with area and regional staff who used their professional and technical expertise to estimate the incremental environmental outcomes that the PR04 schemes would achieve. EA also reviewed and quality-assured all assessments to ensure that the benefits in qualitative, quantitative and monetary terms had been appraised consistently.
• BAG documented transparently the basis behind the detailed Type 3 benefits assessments, using appraisal summary tables (ASTs). This approach enabled the EA to aggregate diverse benefits and accordingly derive a well-balanced proposed programme.
• The individual B/C ratios for Type 3 schemes provided a useful summary indicator of whether or not a scheme was worthwhile. Alternatives such as multi-criteria analysis do not indicate so readily whether an individual scheme is worthwhile.
• For Type 3 schemes, EAs technical and policy experts successfully and sensitively used the B/C ratios as a plausibility test in reviewing the scheme assessments and checking whether the findings regarding the relative importance of different schemes fitted with their local
knowledge, experience and expertise. We discussed the findings with local stakeholders and, where all parties considered that the B/C ratio based on monetized costs and benefits alone did not adequately represent a scheme’s benefits, further examined the schemes and assessments. Where necessary, we set out additional qualitative information on factors not adequately considered by the BAG assessment, leading to the identification of regional priority schemes (category 1) where B/C < 1.2 or deferred schemes where B/C > 1.2 (category 6).

- As outlined above, the results of the Type 3 assessments were used to reduce significantly the number of schemes in the proposed programme and enhance the value for money of our recommendations.

**What Problems Were Encountered?**

Considerable time and resources were spent clarifying the exact nature of improvement schemes and their expected environmental outcomes. It was difficult to ensure that the costs of the schemes related to precisely the same outcomes as the benefits, partly because water companies had not always correctly interpreted the environmental requirements. The EA published the benefits assessment findings on the EA’s website but commercial confidentiality considerations prevented publication of estimated costs and benefit cost ratios. There were also difficulties in presenting consistently the findings for England and Wales together and separately, since there are differences in governance and subsequent terminology, timing and process between the two.

In some cases, uncertainties were due to incorrect specification or interpretation of the scheme requirements (see above). In other cases, the cost estimates appeared excessive. Estimates in companies’ final business plans, once they had been scrutinized by company reporters and the EA, were around 40 per cent lower than those in draft business plans.

Inevitably with such a process, there were differences of opinion amongst stakeholders regarding the methodology and findings of the analyses. Reviewers of the BAG noted the well-known uncertainties associated with BT (benefits transfer) and provided advice on ways to address these (Environment Agency, 2003c). They also recognized and agreed that BT was the only practical means by which the EA could consistently assess the benefits of 500 Type 3 schemes. The EA therefore focused attention on achieving the most valid and reliable BT values and applying these consistently to all schemes (Environment Agency, 2003d).

Whilst uncertainty was addressed through the use of sensitivity analyses and provision of additional information where necessary, there
are counterbalancing reasons why use of the BAG might have resulted in under- or overestimates of benefits.

**Reasons for underestimates of benefits**

- We were unable to assign monetary values to a number of intangible and indirect benefits such as those discussed above.
- Non-use values make up the majority of overall benefits and benefits for most Type 3 schemes. Locally bounded values, based on Georgiou et al. (2000), were used to calculate low and central benefit estimates of water quality schemes and nationally derived estimates by Willis and Garrod (1996) (cited in FWR, 1996) to calculate high estimates. Values derived using the former study are conservative because:
  - In estimating the mean values, the top 5 per cent of individuals’ valuations were excluded. This allows for the possibility that these high values might not adequately allow for income constraints of respondents – even though they may be reflecting individuals’ real and high concerns about the environment.
  - It is unlikely that all ecosystem impacts are fully included in the values (see Environment Agency, 2003d for full details).
- The use of values from the Georgiou et al. (2000) study may give particular rise to underestimation for rural water quality improvements, where there are few people living in the bounded zones around water bodies.
- In addition, for Type 3 schemes in areas where tourism is a major factor, the number of beneficiaries may increase significantly during the tourist season. Whilst we allowed for this in respect of estimation of beneficiaries for informal recreation and angling, it is difficult to make such adjustments for non-use benefits.

**Reasons for overestimates of benefits**

- The Georgiou et al. (2000) non-use values were derived for a single river and the bounded zones used to estimate beneficiaries for non-use values could result in overestimates where the population within the zone is large or where there are other PR04 schemes within the zone. However, the study used to calculate non-use benefits of water resource schemes effectively took account of the latter issue since it first elicited values for the environmental benefits of all schemes in an area and then sought values for an individual scheme’s share of these total benefits. It is therefore interesting to note that the proportion of non-use benefits of water quality schemes was in line with that of water resource schemes, suggesting that the extent of any overestimation was probably limited.
It was not always easy to identify the circumstances to which the valuations of benefits used by the studies in the BAG related. There was a general lack of clarity in the reporting of many of the available studies and they generally did not relate well to the types of marginal environmental change relevant to the schemes proposed under the PR04 programme. The benefits associated with such marginal changes may be lower than the types of change considered in the original studies.

CONCLUSIONS AND LESSONS FOR THE FUTURE

Many within the environmental economic and water policy communities recognize that the benefit assessment process described in this chapter is a welcome and significant step forward in terms of environmental policy appraisal. It also represents the largest such appraisal of its kind undertaken in the UK to date. It is therefore inevitable that, despite the progress made, some lessons can be learnt for the future. The EA is addressing these in the collaborative research programme on economic appraisal for the forthcoming European Union Water Framework Directive (WFD) (see DEFRA, 2004). The key lessons relate to a number of specific areas.

The WFD will encompass more numerous and diverse schemes, since it covers all water-related environmental pressures, not only those that are specific to the water industry. This particularly includes diffuse sources of pollution (from agriculture for example) and a range of innovative solutions, such as economic instruments and wider catchment options including land and strategic flood risk management, will be required to tackle these. Consequently, there will need to be considerable prioritization in drawing up a draft programme of measures for appraisal so that most time and resources can be focused on those where the balance of costs and benefits is likely to be least obvious. This will require greater central discipline and scrutiny by all parties to ensure that schemes are appropriately specified and properly costed before the cost and benefit assessment process begins.

The PR04 timetable was tight and ambitious. But the scale and complexity of the task for the WFD necessitates even higher organizational and operational standards. The critical path for the WFD has the following important features that will help, provided that each stage of the work is completed well and on time. First, the economic appraisal will be carried out over a longer period (2005 up to 2008/9). Second, cost-effectiveness analysis and supporting research needs to begin at the start of this period so that sufficient time is available for costs and benefits assessment later
on. Third, and related to this, a key lesson is that the cost-effectiveness of all schemes must be carefully validated before any benefits assessments are carried out, and certainly before DEFRA has to make any policy decisions on the environmental requirements for the WFD.

Investigations should be undertaken early in the WFD process so that any resulting necessary actions can be costed, assessed and implemented as necessary either as part of future periodic reviews or as part of the WFD river basin management plans (RBMP) cycle. They need to focus on assessing the value added information provided in terms of specifying the problem and its causes, the extent to which it could result in failure to achieve good ecological status (and hence requiring a derogation), as required for water bodies under the WFD, and possible solutions.

By applying the BAG to both water quality and water resource improvement schemes, considerably more consistency and integration was achieved compared with assessments for previous water industry reviews. However, this has also presented difficulties in adequately taking account of the different issues faced by these different types of schemes. For example, whilst PR04 helped to reduce the risks of sewer flooding arising from intermittent discharges, greater integration with flood risk management strategies is required to implement the WFD with respect to heavily modified water bodies (water bodies that have been substantially altered through human use).

It is also important to clarify the decision-making process for the various WFD measures. For example, how many schemes will need to be considered? Which sectors should be considered? Who decides on them and at what level (national or at river basin)? Who will pay for the schemes and whom they will affect? What are the key WFD and non-WFD milestones affecting them? Due consideration needs to be taken of the resource implications for all parties (agencies, government bodies and stakeholders) when carrying out assessments.

The work has highlighted a number of key priority research areas for which a coherent, collaborative, long-term phased research programme is needed. In close collaboration with DEFRA, the EA outlined such a programme for 2004 to 2008 (DEFRA, 2004). A first task is to set out the overall decision-making and supporting appraisal processes needed for the WFD. The next key priority is to specify clearly the schemes needed and to improve methods for assessing costs and effectiveness of options. In particular, this needs to provide even-handed assessments of costs and economic impacts for diverse sectors such as agriculture and the water industry, which are distinctly different in terms of their ability to pass through any cost increases. The extent and nature of likely undecided cases, on which
assessment of costs and benefits will be needed to determine whether or not costs are disproportionate, will need to be determined. Finally, it will also be necessary to determine how these assessments should be carried out and what new studies are needed to improve estimation of numbers of beneficiaries and the valuation of these benefits. Any new research needs to cover the following three essential steps:

1. **Qualitative description and assessment of impacts.** In PR04, EA staff carried out assessments based largely on their local knowledge, ensuring that the schemes were focused on the local situation and needs and incorporating the views of local stakeholders. In particular, they discussed the findings of the assessments with their regional advisory groups. For the WFD, it will be necessary to have greater input of local stakeholders’ views at an earlier stage to specify concerns and impacts, identify possible solutions and the costs, effectiveness and benefits of these.

2. **Quantification of impacts.** Better information is needed on the numbers of beneficiaries of proposed solutions and better collation and use of data on visitors and users (e.g. anglers), using geographical information systems (GIS) data techniques where possible.

3. **Monetary valuation where possible and needed.** There are likely to be many contentious cases in the WFD where the measures required to achieve ‘good ecological status’ might be ‘disproportionately expensive’ as defined under the WFD. Disproportionate cost is a permitted reason for not ensuring that good ecological status is achieved, but has to be demonstrated as such. In many cases, it will be too expensive and impractical to carry out original benefits valuation surveys for each of these cases and hence there is a need to scope and characterize them. Moreover, the EA will need to explore a range of alternative assessment methods (e.g., contingent valuation, choice experiments) and review which is most appropriate and ‘fit for purpose’ for the scope and nature of the cases encountered. These alternative methods will then be tested using virtual analyses and pilot trial applications, identifying any significant gaps in data, the value of obtaining additional information and the most appropriate way in which it could be obtained and readily used by our technical staff.

The PR04 process has shown that the EA is at the leading edge of the use of valuation in public policy appraisal. But the EA is not committed to the use of BT (benefits transfer) for the WFD. Whilst more work on BT is needed and better transfer functions would be useful, it is recognized that this may not lead to large improvements in the consistency or reliability of values. Hence the EA will, in conjunction with the policy, academic community...
and others, continue to explore whether improved understanding of values through the use of BT is most worthwhile or whether some other decision-making technique would be preferable.

If BT is used, it will be necessary to examine how to make the best use of the existing studies, supplemented by carefully targeted new studies, so as to apply their findings to all the potentially disproportionately costly cases. Any new studies should therefore be designed ex ante to incorporate a BT function and fit the key characteristics of the likely cases where costs might be disproportionate so that they are readily usable outside of their original context. Key features of any new studies are that they will need to:

- be based on a sound methodology (best practice guidance);
- be peer reviewed;
- report methodology and findings in a clear and transparent fashion;
- be interdisciplinary (e.g., incorporating economics, psychology and science);
- be carefully designed and provide clear information on the context and purpose of the study;
- address carefully the issue of income constraints and the opportunity costs for a respondent associated with their valuations, particularly important where a number of substitute sites may be available;
- provide clear and full information on the environmental benefits in question as defined by analysis of people’s specification of these benefits (from qualitative assessment above);
- explicitly take account of uncertainties;
- use GIS to facilitate development of distance-decay functions for the values and estimates of beneficiaries at various distances from the sites and for the application of the transferred values to alternative sites and schemes;
- be achievable within the resources obtained by the collaborative research programme.

NOTES

1. Jonathan Fisher is from the Environment Agency and Bruce Horton was at the Agency when this chapter was written.
REFERENCES


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Valuing the environmental benefits of water industry investment


INTRODUCTION

Public regulation of water and sewerage companies is accepted because they are regional monopolies and because they create environmental externalities (through water abstraction and waste-water disposal). Moreover, their service delivery exhibits ‘public good’ elements (everyone on a supply network receives the same level of service). The economic regulator for water and sewerage companies in England and Wales is OFWAT (Office of Water Services). This government agency sets standards to which water companies must conform, and every five years sets prices that water and sewerage companies can charge their customers over the ensuing quinquennium.

Water companies have to conform to legal minimum standards set by the UK government,¹ and to EU Directives.² The regulator OFWAT allows water companies discretion to vary service levels provided beyond these minimum standards, provided water companies can demonstrate to the regulator that benefits to customers and the general public exceed costs to customers.

This study demonstrates the use of random utility models (RUMs) in evaluating water company customers’ preferences; and in estimating the benefits of changes in service levels associated with attributes of water supply, water quality and waste-water disposal, which exceed statutory minimum standards. The study also illustrates the importance of using RUMs to value non-statutory environmental improvements that water companies might be called upon to make by the Environment Agency (EA), which regulates environmental standards in England and Wales.
STATED CHOICE AND ECONOMIC THEORY

The analysis of stated choice (SC) data is based upon consumer demand theory, particularly the theory of consumer behaviour following Lancaster (1966) and Rosen (1974). This assumes that utility to consumers of any good (e.g. water supply, water quality and waste-water disposal, or an environmental good) is derived from the characteristics of the good.

In SC experiments customers are presented with different alternative combinations of characteristics of a good (i.e. water supply, water quality and waste-water disposal, etc) and asked to choose their most preferred alternative. Repeated choices by customers from sets of different alternatives reveals the trade-offs customers are willing to make between the characteristics or attributes of a good. This choice is modelled as a function of the good’s characteristics using random utility theory. Random utility theory is based on the hypothesis that individuals will make choices based on the characteristics of the good (an objective component) along with some degree of randomness (a random component). This random component arises either because of randomness in the preferences of the individual or the fact that the researcher does not have the complete set of information available to the individual. The utility function is specified as:

\[ U_{ij} = V_{ij} + \epsilon_{ij} \]  (13.1)

where \( V_{ij} \) is the deterministic component of the utility function and \( \epsilon_{ij} \) is its random component. If it is assumed that \( V_{ij} \) is a linear utility function then \( V_{ij} = \beta'x_{ij} \). The conditional logit model (CLM) is derived by placing restrictive assumptions of the random component element of the utility: error disturbances are assumed to have a Type 1 extreme value distribution

\[ \exp(-\exp(-\epsilon_{ij})) \]  (13.2)

It is assumed that alternative \( j \) is selected by individual \( i \) if

\[ U_{ij} = \max\{U_{ik}\} \]  (13.3)

From the Type 1 extreme value distribution with scale parameter \( \lambda \), the probability of choosing an alternative \( j \) among \( n_i \) choices of individual \( i \)

\[ = \exp [(\beta\lambda)'x_{ij}] / \Sigma_k \exp [(\beta\lambda)'x_{ik}] \]  (13.4)
where each coefficient is scaled by the parameter $\lambda$, which is inversely proportional to the variance of the unobserved portion of utility, and cannot be identified in a single data set.

A property of the conditional logit (CL) model is the independence of irrelevant alternatives (IIA). IIA implies that all cross-effects are equal; so, for example, if an attribute gains in utility it draws shares from other attributes in proportion to their contribution to current utility.

Other forms of choice modelling relax this undesirable assumption, and adopt different distributions for the error term, and different structures in decision-making. The nested logit (NL) model assumes that decisions are taken sequentially following a decision tree. NL models assume a generalized extreme value distribution for the error term $\varepsilon_{ij}$, where the distribution of $\varepsilon_{ij}$ is identically correlated across utilities of alternatives in the same nest, but independent for utilities of alternatives belonging to different nests. As a consequence, the IIA property is retained within nests but not between nests.

There is support for this decision structure from behavioural observation: customers appear to decide whether to stick with the status quo position or seek a change (Samuelson and Zeckhauser, 1988), and if they decide on a change then to consider the alternatives. Thus, in the context of this study customers might be assumed to consider whether they are satisfied with current water supply, quality and waste-water disposal service standards of a water company, and if not then to consider what service factors (SFs) they wished to see changed.

In the mixed logit (MXL) model

$$U_{ij} = \tilde{\beta}'x_{ij} + \varepsilon_{ij}, \quad (13.5)$$

where $\tilde{\beta}$ is a vector of utility parameters containing some random elements, and in this case these are each distributed according to a parametric distribution $f(\beta|\theta)$ with parameters $\theta$. The MXL model trivially collapses to a MNL model if the variances of the distributions of all the random coefficients are simultaneously zero.

**APPLICATION TO WATER SUPPLY**

Water demand and price elasticities are typically estimated on the basis of actual consumption and price charged (see Taylor et al., 2004). However, SC (stated choice) experiments have been applied in a number of studies to investigate the change in benefits from varying water service delivery to
Valuing water service level changes

Households. They have also been used to value the environmental impacts of water abstraction and disposal to the general public.

Water abstraction and disposal creates environmental impacts. Willis and Garrod (1998) used an RUM model to investigate the public’s WTP to improve flow levels in rivers in South-west England. River flows could be improved by reducing abstraction from rivers and underlying aquifers, but this would incur increased water prices to secure alternative sources of supply. In this study improved flow levels were evaluated in the context of the public’s trade-offs and WTP for other environmental improvements: extending the length of rivers with good water quality, and also the number of bathing beaches meeting water quality standards. Gordon et al. (2001) assessed options for water supply in Canberra in terms of river flow levels, species habitat loss, sprinkling of public lawns in urban parks, in relation to reductions in household water use, increased use of recycled water, and increased water charges. Water abstraction can also affect neighbouring wetlands, reducing their extent and biodiversity content.

In a study of the Macquarie and Gwydir marshes in New South Wales, Bennett et al. (2001) investigated these issues in terms of the trade-offs between irrigation related employment, wetlands area, water birds breeding, endangered and protected species and water prices, for use and non-use benefits of these marshes.

A similar situation was evaluated by Willis et al. (2002) in Sussex, England, where the local water company proposed to abstract river water in winter to artificially recharge the underlying aquifer. It was estimated that 90 per cent of the water could be recovered in the summer through boreholes into the aquifer. The study appraised the use and non-use benefits of the potential effect of this reduced river flow on neighbouring wetlands in terms of bird and plant species, in relation to risk of summer supply interruptions if the project was not implemented.

RUM models have also been used to evaluate the service performance of water companies and customer preferences for attributes of water supply and quality. Asthana (1997) used a multinomial logit (MNL) model to predict households’ choice of water supply system (private pipe potable water; public stand post; hand-pump; dug-well; surface water) in Bhopal, India. Haider and Rasid (2002) used a simple RUM model to investigate customer preferences for variations in two attributes: water pressure and water taste (both expressed qualitatively: worse, same, improved) in relation to a percentage change in water prices, for Thunder Bay, Ontario.

Hensher et al. (2003) assessed households’ willingness to pay (WTP) to avoid outages (defined in terms of frequency, timing and duration) and information (defined in terms of notification of interruption and telephone response to queries) for water supply and disposal, in Canberra, Australia,
using a series of RUM and MXL models. A similar study was undertaken by MacDonald et al. (2003), who also used a multinomial logit (MNL) model, and a random parameter logit (RPL) model, to assess implicit prices associated with urban water supply attributes in Adelaide (in terms of duration of interruption; frequency of future interruption; communication about interruption [phone call/knock on door]; alternative supply [central location/bottle of water]; and price change).

THE STUDY

This chapter reports a study of Yorkshire Water customer preferences and WTP for service level changes. It differs from the studies above in that it considers many more characteristics: 15 (14 service factors plus price change). Yorkshire Water (YW) supplies water to some 1.8 million residential customers (households) and approximately 100,000 businesses throughout Yorkshire in England. Its water supply, water quality, and waste-water disposal, are characterized by a number of attributes, hereafter called service factors (SFs). The aim of the research project was to estimate the benefit YW customers derive from marginal changes to the level of service provided with respect to 14 SFs. The results were used by YW to determine whether the benefits exceeded the costs of improving particular service levels, and if so the extent to which SFs should be improved.

Experimental Design

The SFs were 1. security of supply (SOS), 2. interruptions to supply (ITS), 3. drinking water biological and chemical quality (DWB), 4. drinking water discolouration (DWD), 5. leakage (LKG), 6. inadequate mains pressure (IMP), 7. lead in drinking water (LD), 8. sewage flooding into properties (SFP), 9. areas flooding by sewage (AF), 10. nuisance from odour and flies from sewage treatment works (OF), 11. pollution incidents (PI), 12. ecological quality of rivers (RQ), 13. ability to use inland waters for recreation (AM), 14. bathing beach water quality (BB). Table 13.1 provides a list of these SFs, with their descriptions, and feasible changes to their levels.

Service factors (SFs) 1 to 7 reflect supply and quality of water to households; SFs 8 to 10 represent the external disbenefits of waste-water disposal to households; whilst SFs 11 to 14 represent environmental factors that the household may consume or have concerns about and are impacted to some extent by wastewater disposal. Each SF exhibits the characteristics of a local ‘public good’ to some degree. Every customer on a particular
<table>
<thead>
<tr>
<th>Table 13.1 Service levels (SF) for residential and business customers</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Sf</strong></td>
</tr>
<tr>
<td>-------</td>
</tr>
<tr>
<td>1</td>
</tr>
<tr>
<td>2</td>
</tr>
<tr>
<td></td>
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<tr>
<td>3</td>
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<td>4</td>
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<td>10</td>
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<tr>
<td>11</td>
</tr>
<tr>
<td>12</td>
</tr>
<tr>
<td>13</td>
</tr>
<tr>
<td>14</td>
</tr>
</tbody>
</table>
Amenity and water quality network is subject to the same SF levels. Different customers on the same network cannot choose to purchase different SF levels. Changes in SFs were valued across all customers. The valuation thus encompasses use, option, non-use and altruistic utility; that is, the benefit each customer directly derives from consumption of the SF (use value); the individual's assessment of the option value of the SF; the benefit she derives from knowing that the SF is being improved even though as a customer she may not have been directly affected by it (non-use value); and the utility the customer receives from knowing that other customers would benefit from the SF improvement (altruistic value).

The large number of SFs raised issues of experimental design, cognitive ability of respondents and econometric analysis. Cognitive limitations preclude respondents from simultaneously trading off large numbers of factors, without adopting some heuristic rule. The greater the number of factors, the greater the cognitive difficulty in simultaneously trading off one factor against another across alternative bundles of factors (Timmermans, 1993; DeShazo and Fermo, 2002). The maximum number of factors respondents can handle and produce reasonably consistent choices satisfying the economic axioms of non-satiation, transitivity, continuity, is quite small, typically four or five factors at most.

Three ways of dealing with the problem of large numbers of SFs were considered in turn. A 'headline' approach would have a 'headline' block of generic SFs (e.g., one factor representing all SFs 1 to 7 [supply and quality]; a second factor SFs 8–10 [waste-water disposal], and third SFs 11–14 [environmental factors], with bill change as the fourth factor). Analysis of choices in this block would reveal respondents' WTP for broad generic areas. Then, separate blocks of three or four specific SFs, without money (e.g., one block for water supply and quality; a second for waste-water disposal etc.) would establish utility for each specific SF. The value of each specific SF within a block would be scaled relative to the amount of money for that generic area established in the 'headline' block. It is doubtful whether respondents would, in this 'headline' approach, appreciate all the possible changes that might take place to 'water supply and quality' when this is presented as a single generic factor.

A second approach would be to have an SF common to all blocks, to scale utility between blocks. However, the scale parameter problem in SC (stated choice) experiment models means that it is difficult, if not impossible, to link coefficients across different logit models in this way.

The third way of dealing with the problem of large numbers of SFs is to block them into small groups of SFs with a price change attached to each block. This was the approach adopted as it was deemed the most suitable and practical.
The Scale Parameter Problem

In MNL models only differences in utility matter; the scale of utility is arbitrary. But, utility needs to be scaled relative to something and since the scale of utility and the scale of the variance of the error term are inversely related, normalizing the variance of the error term (typically to 1) is equivalent to normalizing the scale of utility. In the standard logit model, with independently and identically distributed (iid) random variables, the error term has a Gumbel distribution with variance equal to $\pi^2/6$. Thus the coefficients (or marginal rates of substitution [MRS]) in an RUM model are scaled by this amount, that is scaled proportional to the standard deviation of the unobserved factors that affect respondents’ choices.

Hence a ‘block’ design necessitating comparisons between blocks is only justified when the variance of the unobserved elements of the SFs in each block is the same across all blocks. Since the variance of the unobserved factors or error term is unlikely to be the same across all the blocks, biased estimates of the MRS of SFs are likely to be obtained.

This can be avoided if a price factor is assigned to each block. The price factor levels for each block in the YW study were approximated on the basis of a large pilot survey. The analysis of the pilot survey data permitted the relative importance of each SF to be assessed in relation to the total amount respondents said they would be willing to pay for the improvement to all SFs to their highest level (Scarpa and Willis, 2002).

Each of the five blocks of SFs comprised an orthogonal design that permitted the investigation of the ‘main’ effect of each SF. The five ‘blocks’ were: ‘Main’ (SOS DWB SFP PI MON); ‘WS1’ (IMP ITS MON); ‘WS2’ (LKG LD DWD MON); ‘WW1’ (AF RQ OF MON); ‘WW2’ (AM BB MON). The ‘main’ block permitted customers to trade off some principal factors in water supply and quality (SOS and DWB) with waste-water disposal factors (SFP and PI); whilst the other blocks investigated customers’ marginal rates of substitution (MRS) between different water supply and quality factors (in WS1 and WS2), and waste-water factors (in WW1 and WW2).

Sample

The survey comprised a stratified random sample that covered 1000 residential customers, and was undertaken in June 2002. The sample reflected the geographic distribution of population; the approximate rural–urban population split; the approximate age distribution; and the approximate socio-economic status of YW customers. Survey administration was
Amenity and water quality computer-based and conducted in person in the homes of the sampled households by personnel trained in market research.

Stated Choice Analysis

The SC analysis estimated the MRS for each SF and the MRS for money. The implicit monetary benefit of changes in each SF level, or WTP, is determined by the ratio of these MRS: between SFs and money.

Goodness-of-fit of the logit models can be assessed in terms of conformity to economic and theoretical expectations (e.g. appropriate sign on coefficient, reasonable MRS, etc); statistical significance (e.g. in terms of the significance of the coefficients; and significance and explanatory power of the model in terms of log-likelihood, pseudo R^{2}, etc criteria); and behavioural consistency (how individuals make decisions for example between all options simultaneously (RUM model), or between the status quo (‘as is’) and ‘change’). For example, we focus here on the standard multinomial logit model. For this a pseudo R^{2} value of 0.12 is considered average for RUM models; so that the model, reported in Table 13.2 – which pertains to four factor levels – shows a pseudo R^{2} of 0.2835 and this represents a quite good fit to the data.

The marginal rates of substitutions (MRS) between the parameters of each SF and the increase in cost provide an estimate of the implicit price or value of each SF. For example, MRS for SFs in the ‘main’ block are reported in Table 13.2 column 2. All the coefficients are statistically significant at the 1% level. Column 3 provides estimates of the implicit price of each SF. Thus, residential customers were ‘on average’ willing to pay £0.32 for each 1 per cent increase in SOS, and £0.03 for each unit reduction in DWB (each sample of drinking water no longer failing to meet the biological or chemical standard), SFP (each property no longer subject to sewage flooding), and PI (every reduction in pollution incidents) from current standards.

MNL estimates for SFs in blocks WS1, WS2, WW1 and WW2, are not reported for the sake of brevity. However, all were satisfactory in terms of expected signs on coefficients, and in terms of goodness-of-fit measures with pseudo R^{2} measures of 0.3759, 0.2811, 0.3427 and 0.1865, respectively. The implicit prices provided by these models for each SF are reported in Table 13.3. Thus households were WTP £2.27 per year for every 1000 less properties subject to a water supply interruption of 7–12 hours during the year from the current level of 4000; £0.78 per year for every 1000 less properties complaining about discoloured drinking water from the current level of 15000. Conversely, these values can be interpreted as the amount that households would require their water bills to be reduced to compensate for an equivalent deterioration in water service levels.
Table 13.2  Models of utility estimates for SFs: SOS, DWB, SFP, PI (‘MAIN’ block)

<table>
<thead>
<tr>
<th>Coefficient</th>
<th>MNL Linear</th>
<th>Implicit Price (£)</th>
</tr>
</thead>
<tbody>
<tr>
<td>SOS</td>
<td>0.02724</td>
<td>0.317</td>
</tr>
<tr>
<td></td>
<td>(0.00259)</td>
<td></td>
</tr>
<tr>
<td>DWB</td>
<td>-0.00218</td>
<td>0.025</td>
</tr>
<tr>
<td></td>
<td>(0.00012)</td>
<td></td>
</tr>
<tr>
<td>SFP</td>
<td>-0.00219</td>
<td>0.025</td>
</tr>
<tr>
<td></td>
<td>(0.00011)</td>
<td></td>
</tr>
<tr>
<td>PI</td>
<td>-0.00256</td>
<td>0.030</td>
</tr>
<tr>
<td></td>
<td>(0.00015)</td>
<td></td>
</tr>
<tr>
<td>MON</td>
<td>-0.08571</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(0.00275)</td>
<td></td>
</tr>
<tr>
<td>Log-likelihood</td>
<td>-3162.14</td>
<td></td>
</tr>
<tr>
<td>Pseudo-R²</td>
<td>0.2835</td>
<td></td>
</tr>
<tr>
<td>N</td>
<td>4020</td>
<td></td>
</tr>
</tbody>
</table>

Note: Standard error (in parentheses). All coefficients were statistically significant at 1% level.

Table 13.3  Implicit prices for specified changes in SFs and the effect of changes in SF levels on the odds of selecting an alternative (improvement or status quo scenario)

<table>
<thead>
<tr>
<th>SF</th>
<th>Unit Change</th>
<th>WTP</th>
<th>Δx</th>
<th>Odds Ratio: Residential</th>
</tr>
</thead>
<tbody>
<tr>
<td>SOS</td>
<td>1%</td>
<td>0.317</td>
<td>20% ↑</td>
<td>1.72</td>
</tr>
<tr>
<td>DWB</td>
<td>1 sample failure</td>
<td>0.025</td>
<td>250 sample failures ↓</td>
<td>0.58</td>
</tr>
<tr>
<td>SFP</td>
<td>Per property</td>
<td>0.025</td>
<td>140 properties ↓</td>
<td>0.73</td>
</tr>
<tr>
<td>PI</td>
<td>Per incident</td>
<td>0.030</td>
<td>240 incidents ↓</td>
<td>0.54</td>
</tr>
<tr>
<td>IMP</td>
<td>100 properties</td>
<td>1.536</td>
<td>100 properties ↓</td>
<td>0.74</td>
</tr>
<tr>
<td>ITS</td>
<td>1000 properties</td>
<td>2.275</td>
<td>1000 properties ↓</td>
<td>0.64</td>
</tr>
<tr>
<td>LKG</td>
<td>1%</td>
<td>0.697</td>
<td>9% ↓</td>
<td>0.33</td>
</tr>
<tr>
<td>LD</td>
<td>Per year</td>
<td>0.148</td>
<td>6 years ↓</td>
<td>0.85</td>
</tr>
<tr>
<td>DWD</td>
<td>1000 properties</td>
<td>0.783</td>
<td>1000 properties ↓</td>
<td>0.25</td>
</tr>
<tr>
<td>AF</td>
<td>1%</td>
<td>0.106</td>
<td>65% ↑</td>
<td>2.59</td>
</tr>
<tr>
<td>RQ</td>
<td>1%</td>
<td>0.637</td>
<td>15% ↑</td>
<td>3.71</td>
</tr>
<tr>
<td>OF</td>
<td>100 properties</td>
<td>0.935</td>
<td>450 properties ↓</td>
<td>0.57</td>
</tr>
<tr>
<td>AM</td>
<td>Per area</td>
<td>0.415</td>
<td>12 (areas) ↑</td>
<td>2.23</td>
</tr>
<tr>
<td>BB</td>
<td>Per</td>
<td>0.081</td>
<td>100% (improvement) ↑</td>
<td>3.71</td>
</tr>
</tbody>
</table>
Economic theory would predict a non-linear function over large changes in SFs. We found some evidence for this when we estimated some RUM models with quadratic terms in the utility specification for a small subset of SFs (principally SOS) (see Willis and Scarpa, 2003). But generally the conventional linear models had a goodness-of-fit and more statistically significant coefficients than the quadratic models.

Variations in WTP by Socio-economic Characteristics of Customers

Utility of changes in factor levels can vary by the socio-economic characteristics of customers. How utility varies by socio-economic group can be practically determined in many different ways. For example, 1. by undertaking separate RUM models for different sub-groups of customers; or 2. by interacting the socio-economic characteristics of respondents with either the cost factor (if the emphasis of the study is on marginal WTP or WTA compensation for changes from the *status quo*), or the ‘*status quo*’ alternative (if the aim of the study is to capture any socio-economically related ‘*status quo*’ effects).

The inclusion of socio-economic characteristics of households revealed that the probability of selecting the status quo levels of SFs SOS, DWB, SFP and PI increased if households had a male head, and/or were Bangladeshi. Conversely, respondents were less likely to choose the *status quo* position of SF levels if the household had a water meter, was in social class B, C1, or C2, or aged 25–34.

In terms of price, rural households, those with hosepipes, and those with more than three cars were all more likely to agree to pay more to improve SF levels, whilst male heads of household/respondents were more likely to pay less. Respondents were more likely to select an increase in SF levels with increasing income. These interaction patterns were evident throughout the different blocks of SFs, with some minor variations.

Households with an income of < £10000 are willing to pay less for improvements to SOS, DWB, SFP and PI than households with an income of > £10000 per year. ‘White’ customer households with a household income of > £10000 per annum were WTP an additional £0.378 per year per 1 per cent increase in reservoir capacity (SOS), compared with £0.317 for the ‘average’ customer in the sample as a whole. The analysis also revealed that ‘non-white’ customers were willing to pay less than the average ‘white’ customer, even after standardizing for ‘non-white’ customers on low incomes.

Effect on Customer Acceptability of Changes in Factor Levels

Customer acceptance of changes in SFs can be assessed by estimating the proportion of customers willing to change from the *status quo* position.
The dependent variable in an RUM model is the log of odds ratio. If alternative \( k \) is the status quo and the new SF level of provision is alternative \( j \), then \((x_{ij}^* - x_{ik}) = \Delta_{ij}^* x_i\). Thus, \(\exp (\Sigma_i b_i \Delta_{ij}^* x_i)\) shows the odds ratio of the probability of choosing alternative \( j \) over the baseline \( k \) when an SF is changed from \( x_{ik} \) to \( x_{ij}^* \). Such analysis indicates how the proportion of customers choosing the status quo or current situation would change with a change in a particular SF.

Table 13.3 lists the change in market acceptability for different SF improvements. It shows that if DWB failures were reduced from their current level (275 failures out of 250,000 samples per year, i.e. 99.89 per cent compliance) to the maximum possible envisaged (level +2, with only 25 failures out of 250,000 samples, i.e. 99.99 per cent compliance) an odds ratio of 0.58. This means that the odds of choosing the improvement in DWB would be 58 per cent of the status quo alternative, ceteris paribus: that is, the odds of choosing the status quo would fall by 42 per cent.

The effect of improvement in some SF levels is much greater than in others in altering the odds of choosing either the improvement or the status quo. The maximum possible improvement (from the current situation to level +2) has the greatest impact for DWD on the odds of choosing the alternative over the status quo: the odds of choosing the status quo would fall by 75 per cent. Large changes in the odds of choosing the improved water quality alternative vis-à-vis the status quo are also noticeable for PI, ITS, LKG and OF. Least impact on the odds ratio is the reduction in time to reach the new 10µg per litre LD content standard for all drinking water.

**Effect on Welfare of Changes in Factor Levels**

Parameter estimates represent the MRS or how important ‘on average’ respondents weigh each SF. Utility from a change in SF levels is calculated from the difference between some level of SF provision (e.g. the status quo) and a prescribed change in SF provision.

Utility is measured by the bill change \( \Delta \ell \) customers are willing to pay for (or accept) a (positive or negative) change in SF levels associated with moving from \( V^0 \) to \( V^1 \). It is the amount that sets the two utility levels equal to each other. So if \( \Delta V = V^1 - V^0 \), then \( (V^1 - \Delta \ell) - V^0 = 0 \). The general formula is:

\[
\Delta \ell = \left[ \Sigma_i \beta_i(x_i^0 - x_i^1) \right]/\alpha = \left[ \Sigma_i (\beta_i/\alpha)(\Delta x_i) \right] = \Sigma_i (\beta_i/\alpha)(\Delta x_i) \quad (13.6)
\]

where: \( \beta \) is the parameter estimate for the attribute; \( x \) is the numerical service level for the attribute; \( \alpha \) is the parameter estimate for money; the superscripts 0 and 1 refer respectively to the initial and final state; the
subscripts \( i \) refers to the generic \( i \)-th SF. Thus a change in value is given by
the sum of the products between part-worths of each SF and the respective
change in provision \( \Delta x_i \).

For example, the utility to the ‘average’ YW customer of moving from
the current SF provision \( (V^0) \) where SOS = 30 per cent, DWB = 275, SFP
= 540 and PI = 320; to \( (V^1) \) where SOS = 50 per cent, DWB = 25, SFP =
400 and PI = 80, assuming an RUM linear function as specified in Table
13.2, the \( \Delta £ = £23.46 \). This is the annual amount that can be taken away
from the ‘average’ customer to leave the customer indifferent between SF
provision \( V^0 \) and \( V^1 \).

The benefits of various combinations of SF level changes can be
calculated, and these can be compared with the costs of investment, so
that investment in maintenance and improvements to infrastructure only
occurs where benefits exceed costs.

**Summary**

The above analysis of benefits changes attributable to SF changes was
used, along with estimates of the costs of SF improvements, by Yorkshire
Water to formulate its business plan to OFWAT for the 2004 Periodic
Review (PR04).

**ENVIRONMENT AGENCY**

As part of the periodic (quinquennial) price reviews, the Environment
Agency (EA) evaluates various schemes, where an environmental
improvement could be made as a result of a change in water company
operations (e.g. improvements in the chemical content of water discharged
into rivers or the sea from sewage treatment works [STW]). These are non-
statutory cases in the sense that STW discharges already meet regulatory
standards. However, if the EA can demonstrate the benefits that the public
will derive from the environmental improvement exceed the cost of achieving
the improvement, the EA can recommend to the minister (in charge of
water policy at DEFRA)\(^5\) that such schemes be incorporated in a water
company’s five-year business plan. OFWAT determines the price rise in
water charges that the company can impose on customers to pay for these
environmental ‘choices to be made’ (CTBM) schemes, in addition to other
improvements to meet regulatory requirements, and any other discretionary
improvements to service levels the company has identified where benefits
exceed costs to customers.

The economic feasibility of the CTBM schemes is appraised by the EA,
which evaluates the benefits and costs of each scheme. The EA identified
Valuing water service level changes

over 500 CTBM schemes throughout England and Wales, all of which had to be evaluated within a six-month time frame. Approximately five working days were devoted by the EA to evaluating each scheme, a meagre resource input considering the capitalized expenditure on these CTBM schemes was envisaged to amount to over £3 billion.

To undertake these appraisals the EA developed a benefit assessment guidance (BAG) procedure. BAG is based upon 1. a benefit transfer (BT) approach: BT selects a representative study that has valued a particular environmental attribute and applies the value from that study to all schemes falling into that category; 2. distance-decay functions: non-use values are assumed to decline with distance from the CTBM site; and 3. an independent valuation and summation (IVS) appraisal structure: the benefits from each CTBM scheme were estimated and aggregated under the ceteris paribus assumption.

The main problems with the EA’s BAG approach is that BT has neither proved to be a reliable nor accurate method of estimating environmental benefits; moreover, no study has yet been able to show under what conditions environmental value transfer is generally valid. Distance-decay effects in BAG are largely subjective and are not statistically valid. And IVS is known to overestimate the aggregate total benefit of schemes, leading to too many schemes passing the ‘benefit–cost’ test (Hoehn and Randall, 1989).

Brouwer (2002) found that BT errors could be as large as 56 per cent for simple transfers of mean WTP values. Generally a benefit function transfer (BFT) approach is regarded as more likely to provide a more accurate estimate of the value of the environmental attribute at the new site than simple transfer of mean WTP (Loomis, 1992). However, some studies, for example, Bergland et al. (2002), have shown that even BFT may not be reliable, especially where there are differences in environmental quality between the two sites (Barton, 2002). Meta-analysis invariably explains less than 50 per cent of the variation in the estimates of the value of a particular environmental good derived by different studies, indicating that the researchers have not been able to model all the factors accounting for differences in environmental values between sites.

Distance-decay effects for non-use values used by the EA are neither accurate nor reliable. The studies from which they are derived can typically only explain 0.04 per cent of the variation in WTP even taking distance from the site into account. Hence arbitrary cut-off values and distances are applied by the EA.

IVS is an inappropriate basis on which to derive aggregate values for all CTBM schemes in total. This occurs because values derived from studies valuing a particular environmental attribute under a ceteris paribus framework, ignore substitution effects between attributes and other
CTBM schemes. IVS also ignores substitution effects between statutory environmental improvements, water company discretionary schemes and EA CTBM schemes, as well as the greater impact on a household’s budget from implementing all the CTBM schemes instead of the single scheme under investigation. All of these factors produced inaccurate and unreliable estimates, overvalued benefits and resulted in too many CTBM schemes passing the CBA test (Hoehn and Loomis, 1993; Randall and Hoehn, 1996).

The distance-decay approach does not necessarily eliminate the part–whole bias IVS imparts in aggregating non-use values across different schemes. Hanley et al. (2003) calculated WTP for improving flows in all 30 River Thames tributaries in need of improvement, by applying the distance-decay function of the River Mimram to all the tributaries. The Mimram study survey asked respondents’ WTP to improve the Mimram, and in a separate question their WTP to improve all 30 Thames tributaries in need of improvement. Aggregate WTP based on single site values (i.e. applying the Mimram distance decay function to all tributaries) was approximately 4.1 times higher than aggregate WTP by respondents for the all-Thames tributaries scenario: a stark discrepancy.

Table 13.4 documents the outcome of appraising different CTBM schemes (column 1) in the Yorkshire Water (YW) area under the EA’s BAG approach (column 2) compared with the MNL model outlined earlier in this chapter (column 3). The MNL estimates are not based on BT: they are households’ WTP for marginal improvements in the ecological quality of rivers (RQ), improvements to rivers for recreational use (AM), and improvements to bathing water quality (BB). The MNL estimates are not estimates for each individual scheme ceteris paribus in isolation from all other improvements for which customers will have to pay; nor do they employ distance-decay functions, but are the value that customers are willing to pay for all schemes in the Yorkshire Water area.

It is obvious from Table 13.4 that the EA’s BAG results in very large estimates of the value of environmental improvements in CTBM schemes, compared with estimating the benefits of these CTBM schemes on the basis of the MNL model (differences are attributable to differences in the estimates of benefits rather than differences in the estimates of costs). Indeed, of the 33 schemes in the YW area that the EA appraised, it identified 17 as having B/C ratios > 1.2 (listed in Table 13.4), which it recommended to the Minster for inclusion in YW’s business plan to OFWAT, and 16 schemes as having B/C ratios < 1.2 which it did not recommend for implementation (of which some are listed in Table 13.3 for information). The fact that the minister at the end of the day only approved three CTBM schemes for inclusion in
Valuing water service level changes

YW’s business plan might be interpreted as a sign that DEFRA viewed the EA’s BAG method with a considerable degree of scepticism.

Table 13.4  Comparison EA cost–benefit ratios for ‘CTBM’ schemes with MNL model estimates of the benefit of these schemes in the Yorkshire Water area

<table>
<thead>
<tr>
<th>Scheme</th>
<th>EA B/C ratio</th>
<th>YW B/C ratio</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>0.80</td>
<td>0.66</td>
</tr>
<tr>
<td>B</td>
<td>2.54</td>
<td>0.07</td>
</tr>
<tr>
<td>C</td>
<td>5.71</td>
<td>0.01</td>
</tr>
<tr>
<td>D</td>
<td>17.30</td>
<td>0.20</td>
</tr>
<tr>
<td>E</td>
<td>2.28</td>
<td>0.07</td>
</tr>
<tr>
<td>F</td>
<td>1.70</td>
<td>0.00</td>
</tr>
<tr>
<td>G</td>
<td>3.57</td>
<td>0.00</td>
</tr>
<tr>
<td>H</td>
<td>7.01</td>
<td>0.05</td>
</tr>
<tr>
<td>I</td>
<td>1.28</td>
<td>0.29</td>
</tr>
<tr>
<td>J</td>
<td>17.75</td>
<td>0.36</td>
</tr>
<tr>
<td>K</td>
<td>1.25</td>
<td>0.00</td>
</tr>
<tr>
<td>L</td>
<td>8.12</td>
<td>0.19</td>
</tr>
<tr>
<td>M</td>
<td>66.18</td>
<td>0.02</td>
</tr>
<tr>
<td>N</td>
<td>0.99</td>
<td>0.04</td>
</tr>
<tr>
<td>O</td>
<td>2.80</td>
<td>0.05</td>
</tr>
<tr>
<td>P</td>
<td>24.57</td>
<td>0.13</td>
</tr>
<tr>
<td>Q</td>
<td>0.91</td>
<td>0.05</td>
</tr>
<tr>
<td>R</td>
<td>16.40</td>
<td>0.09</td>
</tr>
<tr>
<td>S</td>
<td>37.40</td>
<td>0.48</td>
</tr>
<tr>
<td>T</td>
<td>1.43</td>
<td>0.46</td>
</tr>
</tbody>
</table>

Notes:
EA = EA estimate of benefits with OFWAT estimate of cost.
YW = MNL model estimate of benefits with Yorkshire Water estimate of costs.

Source: Yorkshire Water.

It is clear from the above analysis that benefit assessment of water services should avoid benefit transfer (BT), distance-decay functions and a lack of context or a ‘bottom-up’ IVS valuation approach, as far as possible, if accurate and reliable estimates of benefits are to be derived. Conversely more accurate and reliable estimates of SFs (including environmental SF levels) are likely to be derived if the benefits of different SF improvements...
Amenity and water quality are evaluated simultaneously, within the context that consumers have to pay for all SF improvements in their area. One way of doing this is to adopt an SC experiment approach outlined in the first part of this chapter.

CONCLUSIONS

This chapter illustrates how customer preferences and willingness to pay for service level changes by a water and sewerage service provider can be evaluated. This customer research enabled Yorkshire Water to identify which areas of service are most important to customers and to estimate the value that customers place on the benefits of specific changes in service provided. These benefits were subsequently compared by Yorkshire Water with the costs of maintaining and improving service provision to identify economic levels of service and investment, thus maximizing benefits to a company’s stakeholders: customers, shareholders and regulators.

The chapter also illustrates the importance of allowing for substitution effects between SFs and the erroneous results that are likely to ensue by appraising environmental schemes such as those proposed by the EA outside the ambit of other improvements to water supply, quality and disposal for which water company customers will have to pay.

NOTES

1. The UK Drinking Water Inspectorate is responsible for regulating the quality of drinking water; and the Environment Agency for protecting and improving the quality of rivers, estuaries and coastal waters.
2. For example, the EU Water Framework Directive (WFD) (Directive 2000/60/EC) requires all inland and coastal waters to reach ‘good ecological status’ by 2015. This constrains the chemical and biological content of sewerage company discharges into rivers and coastal waters.
3. A good that is non-rival in consumption (any number of people can consume it without reducing its availability to others) and non-excludable (people cannot be excluded from consuming/using it: for example everyone on a local distribution network experiences the same service level; for example supply interruption, number of days with discoloured drinking water, etc).
4. The non-satiation axiom requires a consumer to choose a choice set that is superior in all respects to another choice set. Transitivity requires that if a consumer prefers alternative \(A\) over alternative \(B\), and alternative \(B\) over alternative \(C\), then he or she must prefer alternative \(A\) over alternative \(C\). The continuity axiom requires that two alternatives that are similar to each other in terms of attributes are ranked close to each other in the preference ordering. This axiom requires the absence of lexicographic ordering. These axioms are discussed by Foster and Mourato (2002).
5. Department for Environment, Food and Rural Affairs.
6. Meta-analysis is a technique to assess how much of the estimated outcome of a study (e.g. environmental value) can be explained by a set of causes of that outcome, characteristics of the good, socio-economic composition of the population sample in the study, research methods used, time period covered and location of the study.
REFERENCES


Willis, K.G. and R. Scarpa (2003), *Stated Choice Methods to Estimate Customers’ Utility Changes from Water Service Level Improvements*, Report to Yorkshire Water Services, University of Newcastle upon Tyne: Centre for Research in Environmental Appraisal and Management (CREAM), School of Architecture, Planning and Landscape.

INTRODUCTION

The River Thames is well known worldwide as the body of water that weaves its way through Central London and past some of its most famous views: Greenwich, Tower Bridge, St Paul’s Cathedral, the Houses of Parliament and Hampton Court, to name but a few. One-third of the river – the section that runs through all of London, from Teddington in West London out to the seaward limit – is tidal and known as the Thames Tideway.

The Thames Tideway has suffered from severe pollution many times over the past 200 years. In the 1950s there was virtually no life in the river. However, thanks to a number of efforts since the 1970s, most importantly in the treatment of sewage, there has been a steady improvement in water quality. Today, the river supports a diversity of fish life, including 120 different fish species. The waters naturally appear brown in colour, not because they are dirty, but because of the sand that is being continually stirred up from the river bed with the fast flowing water. In fact there are claims that the Thames Tideway can now be considered one of the cleanest metropolitan rivers in the world (Thames Water, 2002).

However, untreated sewage still finds its way into the Thames on a regular basis. This is because London’s complex drainage system, which dates back to the late nineteenth century, is ‘combined’. This means it carries both human waste and rain water as there are no separate storm water sewers to carry the rain. Water runs off streets and roofs into the main sewer network and mixes with the sewage coming from buildings. While much of this will flow to the sewage treatment works, when the sewers are full – which occurs not only at times of heavy rain and storm conditions but also during moderate rainfall – the excess flows are diverted through 63 discharge
points into the river. Known as combined sewer overflows (CSOs), these discharge events mean that sewage litter and pollutants are deposited in the river without any treatment. During the summer low flow periods, the CSOs can have adverse effects on river water quality with the potential of some fish being killed – particularly fry and juveniles – as well as posing a threat to human health from recreational use of the river. In addition, there can be visual disamenity from the presence of unsightly litter in the river from items flushed down toilets or washed from the city’s streets (Thames Water, 2001). Typically, the overflows discharge on a weekly basis totalling some 60 CSOs per year (Brown, 2004).

As a response, Thames Water – the regional water and sewerage company – undertook a strategic study – the Tideway Strategy – to investigate and identify possible solutions to the problem of CSOs, for implementation post-2005. The strategy comprised: 1. estimation of suitable water quality objectives for the tidal Thames; 2. development of practical engineering solutions to meet the objectives set; and 3. a cost–benefit analysis of the engineering solutions, including a stated preference study.

Major engineering works would be required to implement any of the various technical solution options being considered, ranging from a massive new tunnel, that could be up to 35 kilometres long and 12 metres wide, to be built deep down underneath the river bed (Brown, 2004), to litter screening filters. Their costs range from around £1 billion to £3 billion (Thames Water, 2002). If works do go ahead, this will be the biggest sewage project in London since the current intercept system was built during Victorian times. Hence, having accurate estimates of the benefits – market and non-market – accruing from the various solutions is crucial for justifying such complex and costly works. This chapter presents the results of the stated preference study that was undertaken to estimate the non-market environmental benefits associated with solving the combined sewage overflow problem in the Tideway.

Specifically, the study was designed to identify individuals’ preferences for the different types of improvements brought about by the range of potential engineering solutions to the CSO problem in the River Thames – that is, reductions in sewage litter, in the risk to human health and in the risk to fish populations – and to estimate the economic value of a marginal change in each of these improvements. Because they allow the estimation of preferences for individual attributes of the change in question, choice experiments was the valuation technique chosen.

The rest of this chapter is structured in the following way. The next section describes the methodology used, including the choice of key attributes and levels for the choice experiment. In the third section the survey results are analysed, discussed and aggregated. Validity tests are conducted in the
fifth while the final section presents some conclusions and a discussion of the use of the study in decision-making.

METHODOLOGY

Choice experiments – one of several available survey-based stated preference techniques – assume that the change to be valued can be described in terms of its attributes, or characteristics, and the levels that these take (Louviere et al., 2000; Hanley et al., 2001; Bateman et al., 2002). In a choice experiment (CE), respondents are presented with several scenarios, each described by a number of attributes, varying at different levels. Respondents are then asked to choose their most preferred scenario, in each set of options they are presented with. Each scenario has a price tag attached from which willingness to pay (WTP) for each of the other attributes can be indirectly inferred. As an attribute-based approach, CE is particularly useful when valuing complex and multidimensional situations such as the impacts of the CSOs in the Thames Tideway: it was therefore the chosen technique to estimate the non-market benefits in this study.

Furthermore, the decision-making context surrounding the study, at the time it was implemented (2002), was one of uncertainty. Firstly, there was uncertainty about the exact frequency of the CSO events. Secondly, the volume of raw sewage entering the river in each overflow event was not precisely known. Thirdly, there was significant uncertainty regarding the scale of the environmental and amenity effects that result from a CSO event – for example, although there is a theoretical potential for fish kills after an overflow these are not easily observable or verified. Finally, there was also uncertainty relating to the efficacy of the various technical solutions proposed as is inherent in such large-scale and long-term engineering projects. All this meant that, at the time of the study – and to this day, to a lesser extent – it was not known with any degree of precision, how the different characteristics of the River Thames were affected by the CSOs. Therefore, in such an uncertain world, CE seemed to be the most adequate technique to use as it presents a range of levels over which the various attributes of interest may vary.

Selection of Attributes

The first step in designing a CE study is the choice of attributes that, in this case, are the key characteristics of the Thames Tideway that are affected by the CSOs, alongside a monetary cost. In order to identify the relevant attributes extensive consultations were held with experts in ecology, engineering and
economics, and other interested stakeholders of the Thames Tideway such as sporting clubs and businesses operating on and along the river.

Three attributes were chosen to reflect the environmental impacts of CSOs on river water quality: ‘sewage litter’ (such as human excrement and toilet litter) expressed as a percentage of general litter that can be found in the river and on the foreshore; ‘risk to human health’ from the practice of water sports, represented by number of days in a year when contact with water would not be advisable due to elevated risk of minor symptoms, such as stomach upsets; and ‘risk to fish population’, denoted as number of potential fish kills during which a number of fish may die due to lack of oxygen. Subsequently, the piloting stages of the questionnaire revealed that respondents often confused sewage litter with other general litter (such as plastic bags, cans, bottles or trolleys). To clarify the difference between sewage and non-sewage litter, a fixed-level attribute named ‘other litter’ was added: its levels remained constant and unaffected by any changes in CSOs as the purpose for its inclusion was solely to facilitate the focus on sewage litter. Finally, the monetary attribute chosen was the ‘cost’ of financing the solutions to the CSO problem, expressed in terms of increased annual water bills. The list and definition of attributes used in the survey is presented in Table 14.1.

Assignment of Levels

The consultation process that aided the selection of attributes was also used to identify their current levels and how these might change under alternative future scenarios. The selected levels are presented in the last column of Table 14.1, with those corresponding to the current situation appearing in italics. Thus, it is currently estimated that 10 per cent of all litter in the tidal Thames is sewage litter; due to increased health risks it is not advisable to come into contact with the river water during 120 days per year; there may be four or eight potential fish kills per year; and the extra cost to households is zero because no additional action is presently taken to address these impacts of CSOs. Note that due to extreme uncertainty regarding the number of fish kills occurring in any given year two alternative baseline levels were considered (four or eight fish kills) and the description of the wording reflects the fact that these are ‘potential’ and not actually verified events.

Although many solution scenarios are possible to counter the CSO problem, for the purposes of this study three basic scenarios were considered, each reflecting a different package of engineering solutions, ranging from overflow tunnels to litter filters: a ‘maximum’, a ‘medium’ and a ‘minimum’ intervention scenario, corresponding, respectively, to solutions where 100
<table>
<thead>
<tr>
<th>Attribute</th>
<th>Definition Used in the Questionnaire</th>
<th>Description of Levels</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sewage litter (% of total litter)</td>
<td>Sewage overflows can bring human excrement and other toilet litter such as condoms and sanitary towels that can be visible in the river and can wash up along the foreshore. Because this part of the river is tidal it can take up to three months for litter to flow out to the sea, so items of sewage litter are present most of the time at some point along the tidal Thames, although not necessarily visible.</td>
<td>10%, 3%, 1%, 0%</td>
</tr>
<tr>
<td>Other litter</td>
<td>Sewage litter is separate from other litter, such as cans and plastic bags that are thrown into the river. Sewage litter only makes up 10% of total litter. The presence of other types of litter in the Tideway would not change as a result of the proposed engineering works.</td>
<td>Present, present</td>
</tr>
<tr>
<td>Watersports – health risk (no. of days when health risks are increased due to overflows)</td>
<td>Sewage overflows can increase the risk of suffering minor illnesses (such as a mild stomach upset) from contact with the water, especially near sewage outfalls following moderate or heavy rainfall. A small number of people use the tidal Thames for contact watersports, so only a few people are currently at risk. Currently, it is estimated that there are 120 days in the year when there is an increased risk of suffering minor illnesses (such as a mild stomach upset) due to the effects of sewage overflows on water quality. During these days participation in watersports is not advisable.</td>
<td>120, 60, 10, 4, 0 days in a year</td>
</tr>
<tr>
<td>Fish population (number of times in a year an overflow event is big enough to pose a risk to fish populations)</td>
<td>Currently 45 species of fish are present in the river at any one time. However, there is a risk that when severe overflow events occur, especially in the summer and autumn, large numbers of fish fry may be killed off from lack of oxygen. Experts estimate that currently there are about eight overflows a year that are big enough to pose a risk to fish populations, potentially killing all the young of a particular species born in that year.</td>
<td>8, 4, 2, less than 1, 0 potential fish kills per year</td>
</tr>
<tr>
<td>Cost (annual increase in water bills)</td>
<td>Currently your water bills pay for sewage treatment and disposal and the maintenance of the existing sewerage system, as well as for the supply of clean drinking water to your home. In order to reduce sewage overflows substantial new investments would have to be made, which would need to be financed through an increase in annual water bills. The extra amount would be paid by all Thames Water customers.</td>
<td>£0, £5, £15, £23, £36, £45, £77, £115</td>
</tr>
</tbody>
</table>
per cent, 95 per cent or 80 per cent of storm events are captured. The range of levels that each of the attributes might take under each of the intervention scenarios was also determined through expert consultation. For example, the ‘maximum intervention’ scenario is assumed to eliminate all impacts of CSOs; hence, the corresponding attribute levels are 0 per cent sewage litter, zero fish kills and zero days when contact with water is not advisable. Note that even under maximum intervention there would be non-sewage-related litter visible in the river since its presence is unrelated to CSO impacts.

Overall, four levels were assigned to sewage litter (10 per cent, 3 per cent, 1 per cent and 0 per cent), a single fixed level for other litter (‘present’), five levels for potential fish kills (8, 4, 2, less than 1 and 0), five levels for elevated risk to human health (120, 60, 10, 4, 0 days in a year) and eight price levels (£0, £5, £15, £23, £36, £45, £77 and £115).

**Experimental Design**

Statistical design theory was used to combine the levels of the attributes into a number of alternative scenarios to be presented to respondents. Combination of the four environmental attributes at their various levels created a total of 80 \((4 \times 1 \times 4 \times 5)\) or 100 \((4 \times 1 \times 5 \times 5)\) possible scenario combinations, depending on the number of fish kills in the current situation – that is, if the current situation was assumed to be eight potential fish kills, with additional levels 4, 2, less than 1 and 0, then there would be a total of five levels in this attribute; but if four potential fish kills was assumed to be the current situation, then the total number of attribute levels would be only four. The price attribute was not included in the experimental design but randomly added to the scenarios afterwards; this simplified the experimental design and ensured that no unfeasibly priced scenarios occurred.

Clearly, respondents would not able to cope with such large numbers of scenarios. In order to reduce the number of scenarios, while still maintaining the possibility of estimating ‘main effects’ – that is, the effects of the attributes on respondents’ choices, which typically explain 80 per cent of all the variation in choices made – a fractional factorial design, available in statistical software packages, was adopted. This narrowed down the set of scenarios to 25.

**Construction of Choice Sets**

The scenarios identified by the experimental design were then grouped into choice sets to be presented to respondents. Each choice set contained the current situation (in which CSOs happen as they currently do) and two improvement scenarios (in which different engineering solutions are
implemented to address the impacts of CSOs). The current situation was included in each choice set to give respondents the opportunity to choose the ‘no change’/zero WTP option so that they were not forced to choose costly improvement scenarios.

The 25 scenarios generated by the experimental design were paired randomly with their fold over design – that is, the mirror image of the original design, where the high level is replaced by the low level and vice-versa – to create 25 pairs of improvement scenarios (Louviere, et al., 2000). To allow for rationality tests (Foster and Mourato, 2002) a number of pairs were then added to the design: some contained a dominated scenario, to check whether respondents chose the dominant option; others a repeated pair of scenarios to check whether respondents were consistent in their choices. This resulted in a total of 32 different pairs of scenarios to which the current situation was added creating 32 triplet choice sets. Finally, the 32 choice sets were further divided into four groups of eight choice sets each, to be presented to respondents (as the piloting stages indicated respondents would not cope well with more than eight choice cards). A sample choice card is depicted in Figure 14.1.

**Questionnaire Design**

Focus groups, peer review and field pilot surveys were used to develop and pre-test the choice experiment questionnaire. A total of six focus groups and two field pilots took place in August and September 2002 in several locations situated at different distances from the river. Each pilot survey consisted of about 50 face-to-face interviews. The final questionnaire comprised not only a detailed valuation section, but also sections on experience and use of the River Thames, attitudes towards the river, health information on those who thought they had become ill as a result of contact with the river water and demographic information.

The final valuation section provided respondents with some concise background information on the history of environmental pollution and clean-up in the River Thames; the CSOs and their impacts on the Tideway, accompanied by maps depicting the tidal Thames and overflow points; and the intentions of Thames Water to reduce the impacts of the CSOs. The need to involve the public in undertaking a preference-revealing exercise for the different investments and their outcomes for the river was then explained. Following this, the choice cards and the river attributes and levels were presented and interviewers worked through the first choice card with respondents to ensure that they fully understood the process. As noted above, each respondent was asked to complete eight choice cards.
<table>
<thead>
<tr>
<th>SHOWCARD Q</th>
<th>Current Situation</th>
<th>Alternative Option A</th>
<th>Alternative Option B</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sewage litter</td>
<td>Some items visible (10% of total litter)</td>
<td>Items almost never visible (1% of total litter)</td>
<td>Not present (0% of total litter)</td>
</tr>
<tr>
<td>Such as human excrement, condoms and sanitary towels</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Other litter</td>
<td>Present</td>
<td>Present</td>
<td>Present</td>
</tr>
<tr>
<td>Such as plastic bags and cans</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Water sports: health risk</td>
<td>120 days per year of increased health risk</td>
<td>4 days per year of increased health risk</td>
<td>0 days per year of increased health risk</td>
</tr>
<tr>
<td>Days when water sports are not advisable due to increased health risk (such as mild stomach upset)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fish population</td>
<td>8 potential fish kills per year</td>
<td>8 potential fish kills per year</td>
<td>Less than 1 potential fish kills per year</td>
</tr>
<tr>
<td>Number of potential threats to fish populations due to overflows</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Annual cost</td>
<td>No additional cost</td>
<td>£15</td>
<td>£36</td>
</tr>
<tr>
<td>Additional water bill payment</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Preferred option**

*Figure 14.1  Sample choice card*
Population and Sample

The relevant population for the study were Thames Water customers – that is, those provided with sewage and/or water services – as they would be the main beneficiaries of any future improvements to the Tideway and would also have to finance the cost of any engineering solutions through increased water bills. This corresponded to a population of approximately 4.5 million households.

Of course, there are other groups of (potentially) affected populations. One group, which can be loosely termed ‘non-users’, includes households living outside the Thames Water customer base and the River Thames catchment area, but which could still potentially benefit from the knowledge that the Tideway had been improved. In the case of these non-users, it was not clear how their preferences, if proved to exist, could be captured, as there is no current mechanism that allows for financing of a project by one water company by the customers of another. This group was therefore not included in the survey sample.

Another population group of interest can be loosely termed ‘users’ referring to those who use the River Thames in general – and the Tideway in particular – for recreational purposes. It is possible that these users, particularly those who participate in in-stream recreational activities (as opposed to river bank dog-walkers, cyclists etc), could have stronger preferences for river water quality, including ecology, appearance and associated health risks, and hence could have a higher WTP for the proposed improvements to the Tideway. As in-stream users were suspected to form a relatively small group within the affected population, a sample representative of Thames customers would be unlikely to include a sufficiently large number of observations to provide statistically different WTP results for this subsample. But, once more, there is no separate mechanism to capture the potentially higher WTP of the users (e.g. by a user fee) and consequently, the in-stream river users were not sampled separately.

A representative sample of 1214 Thames Water customers aged 18+ was selected and personally interviewed, at home, by a professional survey company (MORI). All respondents had responsibility or joint responsibility for utility bill payment in their households. Two hundred and seven sampling locations were selected within the Thames Water area according to distance from the River Thames, and in a selection of areas according to their socio-economic characteristics. Interviews were conducted during the period 21 October to 10 December 2002.
RESULTS AND DISCUSSION

Despite the relative length and complexity of the survey, the questionnaire appeared to work well in the field. The majority of respondents (72 per cent) rated the questionnaire as either interesting or educational and only 4 per cent considered the scenarios presented to be unrealistic. Moreover, only 7 per cent of the sample found the questionnaire difficult to understand.

Demographics

The survey sample was representative of the population in the Southeast of England and London. Fifty-one per cent of the sample was female (compared with 52 per cent in the Southeast and London) and the average age of the sample was 48 (compared with a regional average of 47). Average household income was £26,762 per year (compared with £28,269 per year regional average) and 40 per cent of the sample was in full-time employment (compared with 40 per cent of the regional population). The proportion of respondents who did not reveal their income was 19 per cent, which is low compared with similar studies.

Twenty-eight per cent of respondents did not know the size of their water and sewage bill while the remainder of the sample revealed an average bill of £215 per year. This result compares favourably with official estimations of average annual bills of between £198 and £236 (OFGWAT, 2002).

Experience, Perceptions and Health

The majority of respondents (74 per cent) had at least some experience of the river, either because they came within view of it as they went to work or were out on errands, or because they made purposeful recreational or leisure visits to the river. But regular use of the River Thames – that is, visiting the river for recreation at least once a month – was limited to 27 per cent of the sample. And only a very small percentage of respondents engaged in activities that involved potential contact with the river water – 1 per cent mentioned boating and recreational fishing and 1 per cent occasional paddling and swimming.

Public perceptions of the river were assessed via a rating exercise of the various characteristics of the River Thames – with 2 being ‘very good’ and –2 very poor. Most attributes were rated at the middle of the scale, with the landscape along the river receiving the highest average score (0.6) and water quality receiving the lowest (0.09). These ratings differed according to the area of the river that was being evaluated: as expected, the part of the river that encompasses metropolitan London – as well as 32 of the 63 overflow
The value of a tidier Thames

points – scored lowest on all of the environmental attributes. Interestingly, the spread of scores for water quality along the river matched the ecological profile of the river drawn by scientific studies, showing that respondents were aware of (at least the visual manifestation of) water quality issues.

Most respondents (92 per cent) stated they had seen ordinary litter such as plastic bags, cans and bottles when visiting the river (often, occasionally or rarely). A significant proportion of the sample also encountered sewage litter, with 36 per cent seeing condoms, 22 per cent sanitary towels and 20 per cent excrement. But notably, only 36 per cent of respondents had been aware of the overflow situation before the survey; that is, the large majority of respondents were unaware of the mechanisms through which sewage reached the river.

Finally, in terms of health, only 13 respondents said that they, or a family member, had been ill as a result of contact with water in the Thames: ten had suffered gastroenteritis and three had suffered other ailments. Only two required any medical treatment.

Choice Experiment Results

The standard economic model that underlies the choice experiment used in this study is based on the following indirect utility function \( U_{ij} \) that represents the satisfaction that individual \( i \) receives from the state \( j \) of the Tideway:

\[
U_{ij} = V_{ij} + \varepsilon_{ij} = \text{ASC}_j + b_1 \text{SEWAGE\_LITTER}_{ij} + b_2 \text{HEALTH\_RISK}_{ij} + b_3 \text{FISH\_KILLS}_{ij} + b_4 \text{PRICE}_{ij} + \varepsilon_{ij}
\]  

(14.1)

where \( V_{ij} \) is a deterministic element that is a linear index of the attributes of the \( j \) different scenarios; and \( \varepsilon_{ij} \) is a stochastic element that represents unobservable influences on individual choice. In this model, the attributes are percentage of sewage litter present (\( \text{SEWAGE\_LITTER} \)), health risk of coming in contact with the water (\( \text{HEALTH\_RISK} \)), potential fish kills in a given year (\( \text{FISH\_KILLS} \)) and the cost to the household of achieving a certain level of improvements to the Tideway (\( \text{PRICE} \)). The coefficients \( b_1, b_2, b_3 \) and \( b_4 \) describe the influence of the various attributes on \( U_{ij} \). Note that the presence of general litter used as an attribute in the choice experiment does not enter the model above since it does not vary across the scenarios – it was included in the choice cards simply to clarify the distinction between general and sewage litter. Finally, alternative specific constants (\( \text{ASC}_j \)) for the various scenarios can also be added to the model to capture any variation in choices that is not explained by either the choice
Amenity and water quality

attributes or respondent-specific socio-economic variables. An alternative specific constant representing the baseline situation was used in this case.

Formally, the probability that scenario $g$ is chosen by a respondent $i$ can be expressed as a conditional logit model:

$$P(U_g > U_h, \forall h \neq g) = \frac{\exp(V_g)}{\sum_j \exp(V_j)}$$

(14.2)

This model is estimated using maximum likelihood procedures and the log-likelihood function is:

$$\log L = \sum_i \sum_j y_{ij} \log \left( \frac{\exp(V_j)}{\sum_j \exp(V_j)} \right)$$

(14.3)

where $J$ is the total number of scenarios, $N$ is the total population, and $y_{ij}$ is an indicator variable that takes a value of one if respondent $i$ chose river improvement scenario $j$ and zero otherwise.

In the case of this study, the conditional logit model explains respondents’ choices between Thames Tideway scenarios, as a function of river attributes. The total number of usable responses in the model is less than the total sample size of 1214 as protest answers and those who failed the dominance test were removed from the sample. Specifically, respondents who chose the current situation in every choice set were considered to have zero willingness to pay for the alternative CSO improvement scenarios. Protesters were considered to be those who, when explaining why they always chose the current situation, only stated protest reasons ('I object to paying higher water bills'; 'The government/council/water company should pay for this'; 'I don’t believe these improvements would actually happen'). Thirteen per cent of respondents always chose the current situation: 25 per cent of these were classified as protest answers and removed from the estimation. Overall, this is only 3 per cent of the total sample. Furthermore, as noted above, each respondent was presented with one choice set where one of the improvement options was better than the other in all respects. Failing the dominance test implied choosing the dominated (worse) option. Nine per cent of respondents failed this test and were consequently dropped from further analysis. The model results are presented in Table 14.2.

Overall, the model performed well. All coefficients are negative as expected; lower levels of sewage litter are desirable, reductions in fish kills and reductions
in health risk all have a positive impact on utility, whereas increases in water bills are undesirable, lowering respondents’ utility. The negative coefficient on the baseline constant indicates that there is some utility gain associated with moving away from the current situation, that is, choosing an improvement scenario. All coefficients are significant at the 5 per cent level.

**Table 14.2 Conditional logit model – all usable responses**

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Coefficient</th>
<th>Standard Error</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>ASC_BASELINE</td>
<td>-0.133</td>
<td>0.063</td>
<td>0.034</td>
</tr>
<tr>
<td>SEWAGE_LITTER</td>
<td>-0.035</td>
<td>0.004</td>
<td>0.000</td>
</tr>
<tr>
<td>HEALTH_RISK</td>
<td>-0.007</td>
<td>0.000</td>
<td>0.000</td>
</tr>
<tr>
<td>FISH_KILLS</td>
<td>-0.029</td>
<td>0.008</td>
<td>0.001</td>
</tr>
<tr>
<td>PRICE</td>
<td>-0.019</td>
<td>0.001</td>
<td>0.000</td>
</tr>
<tr>
<td>Log-likelihood</td>
<td>-8414.950</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pseudo R²</td>
<td>0.078</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Number of observations</td>
<td>8311</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The coefficients in this model can be used to estimate marginal rates of substitution between attributes. Specifically, the rate at which respondents are willing to trade off money (in this case an increase in annual water bills) for any given attribute is equivalent to their willingness to pay for a marginal change in that attribute. It is calculated as the ratio of the coefficient of that attribute to the coefficient of the cost variable (e.g. \( b_1/b_4 \) is the WTP for changes in sewage litter). This trade-off is also known as the implicit price, or part-worth of the attribute. The implicit prices of each of the attributes are presented in Table 14.3.

**Table 14.3 Willingness to pay results**

<table>
<thead>
<tr>
<th>Attribute</th>
<th>WTP (£/household/year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>(a) Sewage litter</td>
<td>1.82</td>
</tr>
<tr>
<td>(per percentage point of total litter)</td>
<td>(1.41–2.24)</td>
</tr>
<tr>
<td>(b) Health risk</td>
<td>0.38</td>
</tr>
<tr>
<td>(per day of increased health risk)</td>
<td>(0.34–0.43)</td>
</tr>
<tr>
<td>(c) Fish population</td>
<td>1.51</td>
</tr>
<tr>
<td>(per potential fish kill)</td>
<td>(0.66–2.35)</td>
</tr>
<tr>
<td>Number of observations</td>
<td>8311</td>
</tr>
<tr>
<td>Number of respondents</td>
<td>1039</td>
</tr>
</tbody>
</table>

*Note:* 95 per cent confidence intervals in parentheses.
Amenity and water quality

The results show that respondents were willing to pay £1.82 per year for a 1 per cent reduction in sewage litter, £0.38 per year for one less day of increased health risk and £1.51 per year for one less potential fish kill. Interestingly, the most popular reasons stated for being willing to pay positive amounts for improvements in the Tideway related to non-use values (e.g., for everyone to enjoy the Tideway, for future generations, to protect the wildlife or to protect the Thames as a river of national and cultural importance).

Finally, no statistically significant difference (at the 5 per cent level) emerged between WTP estimated in the subsample presented with a baseline of eight fish kills and that presented with a baseline of four fish kills: it appears that changing the potential fish kills baseline does not significantly affect preferences. This can be regarded as evidence of scope insensitivity. But on the other hand, it is perhaps unsurprising given that the precise number of fish kills due to CSOs is actually unknown to scientists and the questionnaire thus referred to ‘potential’ fish kills. Furthermore, the impact of a fish kill on the ecology of the river is also unknown and so was not described in the questionnaire. Hence, the real difference between eight and four potential fish kills may itself not be significant.

Aggregation Over Population and Time

In order to use the valuation estimates in appraisal they need to be aggregated. There were three basic steps in the aggregation process.

Firstly, the implicit prices for each attribute were aggregated to estimate the annual household WTP for three river scenarios of interest – those corresponding to the maximum, medium and minimum intervention levels attained by potential engineering solutions (Table 14.4). Essentially, this involved multiplying the marginal WTP for each attribute by the level of change occurring in that attribute, brought about by a particular scenario (e.g., from the current situation to the maximum intervention scenario), and summing these. Hence, the welfare change associated with the maximum intervention scenario, where the highest level of improvement in all attributes is achieved, was calculated to be £76.43 per household per year, assuming a fish kills baseline of eight. The results for all three scenarios are depicted in the fourth row of Table 14.4.

Secondly, the annual household WTP for particular scenarios was aggregated across the affected population – that is, Thames Water’s customer base, which is estimated to be approximately 4.8 million households. This implied multiplying the household welfare change calculated for each scenario (Table 14.4, row 4) by the total number of households affected. The results can be seen in the fifth row of Table 14.4.
Thirdly, the population annual WTP is aggregated over time, where the present value of a particular scenario (or engineering solution mix) is calculated based on the project lifetime and discounting assumptions. Three alternative time frames are used – 10, 30 and 50 years – to test the sensitivity of the benefit estimates to the time over which respondents are assumed to pay higher water bills. The discount rate used was 3.5 per cent, as recommended by HM Treasury in the latest version of the *Green Book* (HM Treasury, 2003). These estimates are contained in the last three rows of Table 14.4.

Inspection of Table 14.4 shows that annual household WTP for the three improvement scenarios ranges from about £64 to £76. This is quite a narrow range, reflecting the fact that all three potential solution scenarios considered offer significant and similar improvements relative to the baseline. The analysis also shows that the final aggregated benefit estimates depend to a large extent on the period over which it is assumed that payments last. For example, the estimated benefits for the maximum intervention scenario vary from just over £3 billion, if a ten-year period is assumed, to about £9 billion if a 50-year timescale is chosen instead. Note that the questionnaire did not stipulate a time limit for the payment of the higher water bills: this was because, in the piloting stages of the study, it was consistently found that people assumed that any water rate increases would be permanent.² Hence, it is reasonable to assume a stream of payments spreading over a long period.

**VALIDITY TESTING**

As the benefit estimates produced by the choice experiment survey were to be used in the appraisal of a major engineering project it was important to investigate to what extent they were valid, that is, to what degree they succeeded in accurately measuring the welfare change associated with addressing the problem of CSOs in the Thames Tideway. Various kinds of validity tests were performed on the data and are discussed below.

**Content Validity**

Content validity refers to whether a study asked the right questions in a clear, neutral, understandable and meaningful manner appropriate for obtaining a valid estimate of willingness to pay. Many of the judgements about content validity are essentially subjective. Nonetheless, the responses to a number of questions in the survey provide some way of assessing content validity.
Table 14.4 Aggregated benefits of three improvement scenarios

<table>
<thead>
<tr>
<th>Possible Levels of Intervention</th>
<th>Maximum Intervention</th>
<th>Medium Intervention</th>
<th>Minimum Intervention</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sewage Litter</td>
<td>None</td>
<td>Almost none</td>
<td>Some</td>
</tr>
<tr>
<td>(WTP = £1.82 per percentage reduction per household per year)</td>
<td>0% total litter</td>
<td>1% of total litter</td>
<td>3% of total litter</td>
</tr>
<tr>
<td>Water sports: health risk</td>
<td>0</td>
<td>4 days</td>
<td>10 days</td>
</tr>
<tr>
<td>(WTP = £0.38 per day of reduced risk, per household per year)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fish population (assuming baseline = eight fish kills)</td>
<td>No fish kills per year (under normal conditions)</td>
<td>Less than one potential fish kill per year</td>
<td>Two potential fish kills per year</td>
</tr>
<tr>
<td>(WTP = £1.51 per fish kill per household per year)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total household WTP (£ per household per year)</td>
<td>£76.43 (£66.73–£86.12)</td>
<td>£71.56 (£62.73–£80.39)</td>
<td>£64.11 (£56.34–£71.87)</td>
</tr>
<tr>
<td>(Sum of WTP per attribute × change in that attribute)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total WTP over all households (£ million per year)</td>
<td>£371.63</td>
<td>£347.95</td>
<td>£311.73</td>
</tr>
<tr>
<td>(Total household WTP × number of households)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total WTP over all households over:</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>10 years (present value, £ million) (r = 3.5%)</td>
<td>£3198.9</td>
<td>£2995.0</td>
<td>£2683.3</td>
</tr>
<tr>
<td>30 years (present value, £ million) (r = 3.5%)</td>
<td>£7074.3</td>
<td>£6623.5</td>
<td>£5934.0</td>
</tr>
<tr>
<td>50 years (present value, £ million) (r = 3.5%)</td>
<td>£9021.9</td>
<td>£8447.0</td>
<td>£7567.7</td>
</tr>
</tbody>
</table>
Firstly, only 4 per cent of respondents considered the questionnaire to be unrealistic and only 7 per cent found it difficult to understand. This is an indicator that, in general, the survey design was successful in creating a credible and meaningful hypothetical market. Secondly, the occurrence of item non-response (refusal to answer one or more questions) was very low. In fact, the only significant item non-response occurred for the income question, which 19 per cent of respondents refused to answer. This percentage is relatively low when compared with other valuation surveys undertaken in the UK. Thirdly, only 3 per cent of the sample were classified as ‘protesters’, that is, stated an unwillingness to pay for improvements to river attributes due to invalid reasons. This is a very small percentage, implying that leaving these respondents out of the analysis would not bias the results. Protesters are unlikely to be a particular group, but randomly distributed among the sample.

Additionally, the survey instrument was developed in consultation with experts in ecology, economics and engineering and also user groups of the Thames, to ensure accuracy and neutrality in the facts presented. It was also extensively pre-tested through focus groups and field pilots, to certify that it was workable and well understood by respondents.

Construct Validity

Construct validity is concerned with whether or not the results are consistent with other studies (convergent validity) and with economic theory, intuition and prior expectation (expectations-based validity). In convergent validity testing, no measure can automatically claim to be superior in terms of being a naturally closer approximation of the value held by respondents. Likewise, the result that two approaches deliver similar or logically related measures does not mean that those measures are valid; instead they may be equally invalid. Nevertheless, it is clear that a large and unexpected difference between estimates would show that at least one measure is invalid or two different questions are being addressed.

Convergent Validity

Since, as we are dealing with non-market impacts, there is no observable market data to verify the accuracy of the choice modelling estimates, the results of the study can be compared with those of similar studies. Over the past 25 years, a considerable body of applied literature has investigated the non-market benefits of water quality improvements, both of marine and freshwater bodies, in developed and developing countries. Both the recreational and health impacts associated with polluted waters have
been measured. Examples of recent studies in Europe include Willis and Garrod (1998), Spurgeon et al. (2001), Georgiou et al. (2002), Machado and Mourato (2002) and Hanley et al. (2003).

Of that literature, two stated preference studies were identified as being particularly relevant to the focus (river water quality improvements), policy change (reduction in sewage overflows), time (2002) and geographical context (UK) of the Thames Tideway study. The first is a contingent valuation study by McMahon (2001) that estimated the preferences of Southern Water customers in the UK for connection to mains sewage collection, as a means to reduce the adverse environmental and amenity impacts associated with private waste-water systems (such as cesspools and septic tanks). The WTP for mains sewer connection was estimated to be £3433 per user household and £0.40 per non-user household (one-off payments). ‘Non-users’ in this context were those already connected to mains sewers; these respondents were asked if they would be willing to pay for the connection of others as a means to avoid the environmental and amenity impacts associated with inadequate private drainage systems. While the improvements discussed in the Thames Tideway study share similarities with the environmental improvements of having mains sewers over private options, they do not deliver any of the private household benefits such as avoided damage to property or nuisance. Therefore, WTP estimated for the Tideway could be expected to fall in between the two estimates of McMahon (2001). This was indeed found to be the case.

The other study of relevance is eftec (2002), a large-scale choice experiment study measuring the preferences of the population of England and Wales for several improvements resulting from the revised Bathing Water Quality Directive. Some of the beach attributes valued by the eftec (2002) study are broadly comparable with river attributes estimated in the current study. In particular, the WTP for reducing ‘unsafe to swim’ days at coastal beaches by one day was estimated to be about £1 per household per year (eftec, 2002), while the WTP to reduce the same risk for river water in the Thames was estimated to be around £0.40 per household per year (see Table 14.3). The difference between the two results is remarkably small, showing consistency of results, while the difference can be explained by the fact that bathing activity on coastal beaches is much more common than on the River Thames.

In addition, eftec (2002) also estimated the WTP for elimination of ‘some litter/dog mess’ on coastal beaches at between £6–£11 per household per year. By comparison, WTP for eliminating sewage litter in the Tideway was found to be around £18 per household per year. Again these figures are of similar magnitude and the difference can be explained by the content of
Finally, the choice experiment questionnaire also included a contingent valuation question, asked at the end of the CE exercise, which offers a within-sample convergent validity test. Respondents were presented with a showcard containing the current situation and the ‘maximum intervention’ solution in which sewage litter was eliminated, there were no fish kills and zero days of elevated health risk due to CSOs. Willingness to pay to attain the maximum intervention scenario was then elicited by means of a single-bounded dichotomous choice question. Eight different bid amounts were used, ranging from £5 to £115 per year, to match those used in the choice experiment. Mean WTP was estimated to be about £59 per household per year (with a 95 per cent confidence interval of £51.91 to £65.97). Reassuringly, this result is not statistically different from the WTP estimate produced by the choice experiment model for the maximum intervention scenario (£76 per household per year).

Expectations-based Validity

This type of validity is usually assessed by estimating a model that explains variations in respondents’ choices in terms of their underlying socio-economic characteristics, experience or attitudes. Owing to the statistical specification of the logit model, these individual-specific regressors need to be interacted with one of the scenario attributes or the ASC; otherwise they would drop out of the model as they do not vary across scenarios (Louviere, et al., 2000).

Table 14.5 presents the extended conditional logit model, including attitudinal and socio-economic regressors, which have been interacted with the ASC for the baseline situation. The interpretation of the interacted terms is the following: a positive coefficient sign indicates that a particular characteristic increases the likelihood of choosing the current situation over alternative scenarios, while a negative sign indicates that a particular characteristic increases the likelihood of choosing alternative scenarios over the current situation.

Inspection of Table 14.5 shows that the extended model performs well according to prior expectations – that is, all the attribute coefficients have negative signs and the coefficients of the interactions have signs in accordance with expectations. Specifically, being male, having higher income, belonging to an environmental group, being concerned about destruction of animal and plant life and being frequently exposed to the Thames (at least once a month) significantly increases the likelihood of choosing an improvement
(and more costly) scenario over the current situation. Conversely, older people are more likely to choose the current situation.

Table 14.5  Extended conditional logit model: with individual-specific interactions

<table>
<thead>
<tr>
<th></th>
<th>Coefficient</th>
<th>Standard Error</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>ASC_BASELINE</td>
<td>0.152</td>
<td>0.128</td>
<td>0.236</td>
</tr>
<tr>
<td>SEWAGE_LITTER</td>
<td>−0.036</td>
<td>0.005</td>
<td>0.000</td>
</tr>
<tr>
<td>HEALTH_RISK</td>
<td>−0.008</td>
<td>0.001</td>
<td>0.000</td>
</tr>
<tr>
<td>FISH_KILLS</td>
<td>−0.044</td>
<td>0.009</td>
<td>0.000</td>
</tr>
<tr>
<td>PRICE</td>
<td>0.019</td>
<td>0.001</td>
<td>0.000</td>
</tr>
<tr>
<td>SEX</td>
<td>−0.155</td>
<td>0.062</td>
<td>0.012</td>
</tr>
<tr>
<td>AGE</td>
<td>0.009</td>
<td>0.002</td>
<td>0.000</td>
</tr>
<tr>
<td>INCOME</td>
<td>−0.007</td>
<td>0.001</td>
<td>0.000</td>
</tr>
<tr>
<td>ANIMALS</td>
<td>−0.502</td>
<td>0.093</td>
<td>0.000</td>
</tr>
<tr>
<td>MEMBERSHIP</td>
<td>−0.736</td>
<td>0.091</td>
<td>0.000</td>
</tr>
<tr>
<td>SEE_RIVER</td>
<td>−0.456</td>
<td>0.064</td>
<td>0.000</td>
</tr>
<tr>
<td>Log-likelihood</td>
<td>−6629.707</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pseudo R²</td>
<td>0.101</td>
<td></td>
<td></td>
</tr>
<tr>
<td>No. of obs.</td>
<td>6716</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Notes:
All individual specific variables were interacted with the ASC for the baseline.
SEX is coded as 1 for men and 0 for women; AGE is the exact age of respondents; INCOME is annual household gross income; ANIMALS is a dummy variable indicating whether respondents considered protecting animals to be a priority; MEMBERSHIP is a dummy variable indicating whether respondents are members of any environmental groups; and SEE_RIVER is a dummy variable indicating whether respondents saw the Thames at least once a month.

Further analysis revealed that distance of residence from the river was not a significant determinant of WTP (eftec, 2003), that is, no clear ‘distance-decay’ function appeared to exist. In fact, it can be argued that respondents’ place of residence is not always a reflection of their exposure to or use of the river; commuters, for example, may live a long way away from the Thames, but work close by, thereby having regular exposure to it. Thus, it was hypothesized that exposure to the river might be the key variable in determining a declining pattern in WTP. This can be measured by the frequency of seeing the river and, as expected, the negative coefficient of SEE_RIVER in Table 14.5 suggests that those who see the river more often will tend to choose improvement scenarios over the current situation.
Similarly a ‘frequency-decay’ function can be estimated for WTP (see eftec, 2003, for further results).

CONCLUSIONS

The Thames Tideway CE survey aimed to provide robust monetary estimates of the non-market benefits associated with addressing the environmental impacts caused by combined sewage overflows on the tidal Thames, expressed in terms of the presence of sewage litter, health risks and potential fish kills. At the time of the study, there was significant uncertainty surrounding a number of key parameters of the CSOs: the exact number of CSOs occurring in a typical year was unknown, as were accurate estimates of the environmental impacts of the sewage discharges or the environmental standards that needed to be attained for the various impacts of interest. Furthermore, the available technical solutions were likely to perform significantly differently in terms of addressing the environmental impacts of interest, that is, sewage litter, potential fish kills and elevated risk to human health. Thus, the choice experiment questionnaire was designed to cater for the multitude of possible combinations of intervention approaches resulting in a variety of reductions in the river environmental impacts.

Based on a representative sample of Thames Water customers, the results presented in this chapter showed that, although the majority of respondents had experience of seeing the river and of encountering items of sewage and other litter, only about a third was aware of the CSOs situation. But when faced with the choice between various improved river scenarios and the current situation, only 13 per cent of the sample expressed a preference for the current situation.

On average, a reduction of 1 per cent in sewage litter as a percentage of total litter was valued at £1.82 per household per year, a reduction in one day of increased health risk was worth £0.38 and a reduction in one potential fish kill was valued at £1.51. This translated in a WTP of £76.43 per household per year, in increased water bills, for the maximum intervention scenario, which reduced the impacts of the overflows to their lowest possible levels. The final aggregate benefit estimate was seen to depend to a large extent on the period over which it was assumed that payments lasted. For example, the estimated benefits for the maximum intervention scenario varied from about £3.2 billion, if a ten-year period is assumed, to about £9 billion if a 50-year timescale is chosen instead. But, as qualitative research undertaken in the pilot stages of the survey showed, respondents typically assumed that water rate increases would be permanent, lending support to the use of a long payment period.
In parallel with the stated preference study reported in this chapter, a market evaluation study was also conducted to analyse the environmental benefits of the Tideway solutions on commercial activities, recreational activities or house prices. The study showed that water quality in the Thames Tideway was unlikely to have a major impact on the market uses of the river, amounting to only between 0.01 per cent and 0.03 per cent of the non-market benefits. Other studies focused on the cost side and estimated the financial and environmental costs associated with implementing the various engineering options.

All the relevant benefits (environmental market and non-market benefits) and costs (financial and environmental) were subsequently incorporated into a cost–benefit analysis performed on a selected number of potential technical solutions for the CSO problem. The baseline case, against which the technical options were assessed, was a ‘do nothing to address the CSO problem’ scenario. A 3.5 per cent discount rate was used, in line with Treasury recommendations and the project lifetime assumed to be 50 years. The net present value associated with the three levels of intervention presented earlier was found to be clearly positive and in the region of £3 to £5 billion. Solving the CSO problem, the biggest sewage project in the capital since the nineteenth century, appears to be therefore justified, on economic grounds.

The non-market environmental benefits associated with solving the problem of sewage overflows in the Thames, as estimated in this chapter, were a key component of the cost–benefit analysis of potential engineering solutions and played a crucial role in balancing off the large infrastructure costs involved. Monetizing these large non-market impacts is clearly a relevant procedure as their omission would significantly affect the cost–benefit analysis and, consequently, the decision-making process. Obviously, the accuracy of the estimates of both cost and benefits is crucial for an accurate economic judgement to be made. We have shown in this chapter that the CE results were seen to perform well according to various tests of validity performed. But also highlighted was the problem of scientific uncertainty regarding the occurrence and ecological impacts of the CSOs. Future work should develop the environmental impact analyses to meet the requirements of an economic study and ensure better quality information, and, hence, a more efficient use of resources.

Economic analysis is an important element of the decision-making process of large infrastructure projects, such as the one described in this chapter. But making the right decision involves complex trade-offs, not all of which can be justified on purely economic grounds. Unfortunately, this does imply that the job of decision-makers is sometimes more difficult than making ‘yes/no’ decisions on the basis of the results of a valuation study or
cost–benefit analysis. Perhaps as an indication of this, three years after this CE study and two years after the subsequent CBA, work is still continuing to improve scientific and engineering information and an investment decision is yet to be taken.

NOTES

1. More specifically, the Tideway Strategy study was undertaken in close collaboration between Thames Water and the industry regulators – Environment Agency, OFWAT and the Department for Environment, Food and Rural Affairs (Thames Water, 2002).

2. In reality, this is a simplifying assumption, as even a maximum intervention engineering solution would not achieve 100 per cent capture of storm events (it is technically not possible). But the number of CSOs occurring would be very close to zero.

3. The sample was Enumeration District (ED) based, which means that all EDs within each selected area were ranked according to the proportion of social grading ABs, and any with less than 60 addresses excluded. The sample was then selected at random from the remaining EDs. Within each ED, quotas were set by work status, sex and age. Target quotas reflect 2000 population profile estimates obtained from the Office of National Statistics (these estimates are based on 1991 census data, but modelled to take into account estimations of how the population has changed since then).

4. Note that WTP estimates should not be extrapolated beyond the bracket of values that were tested in the questionnaire and for which no information was uncovered.

5. Although it is, at least in principle, possible that some respondents might assume paying the higher water rates over shorter periods – for example, taking into account a shorter life expectancy or the likelihood of moving out of the area in the near future – in practice, we found no evidence to support this.


7. The contingent valuation and the choice experiment results were estimated on the same set of usable answers (i.e. sample excluding protesters and those who failed the dominance test in the CE exercise).

REFERENCES


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15. Cost–benefit analysis and the prevention of eutrophication

Ian Bateman, Brett Day, Diane Dupont, Stavros Georgiou, Nuno Gonçalo Matias, Sanae Morimoto and Logakanthi Subramanian

INTRODUCTION

The eutrophication of water bodies is a natural phenomenon, but over the past few decades, this pollution problem has increased significantly due to the large additions of nutrients from human sources. Eutrophication can affect a receiving ecosystem in a number of ways, especially with respect to the quality of water and the uses to which that water can be put. Policy-makers are thus increasingly faced with the question of what, if anything should be done about this problem. With respect to any policy analysis of the eutrophication issue, the economic perspective can help policy-makers and society more generally to identify allocations of economic resources that maximize social welfare and design policies that achieve these allocations. Social welfare is defined here in terms of individuals' utility or 'preference satisfaction'. Since people value environmental as well as consumer goods, then economic analysis seeks to allocate resources such that welfare is maximised over both types of goods. The economic approach involves trying to balance the opportunity costs of resources used for environmental protection, with the social welfare benefits provided by such protection. A cost–benefit analysis of any pollution control policy requires information both on baseline loadings and on the various changes in these loadings and associated costs and benefits.

This chapter undertakes a cost–benefit analysis of eutrophication prevention measures for rivers and lakes in East Anglia. The remainder of the chapter is organized as follows. The next section considers the eutrophication problem in more detail, in particular looking at the pressures giving rise to the problem, policies to deal with it and the costs associated
with no change. The third section comprises the major analytical focus of this chapter, an assessment of the benefits of measures to prevent eutrophication, undertaken using a contingent valuation study that incorporates a novel method of value elicitation. The final section presents the outcome and conclusions from the cost–benefit analysis.

THE EUTROPHICATION PROBLEM: PRESSURES, POLICY AND COSTS

One of the most significant problems facing the ecology of freshwater in the UK is eutrophication as caused by the addition of nutrient salts to these water bodies. Eutrophication is defined as the accelerated fertility or ageing of a water body by nutrients. These nutrients are mainly in the form of nitrogen and phosphorus compounds. These critically determine the growth of primary production in water bodies. Nutrients are naturally introduced to water bodies by plant and animal decomposition, excretion, air–sea exchange and oceanic mixing processes. However, these inputs are supplemented by human sources of nutrients including domestic wastes, agricultural run-off of excess fertilizer, animal wastes from intensive livestock rearing, aquaculture, industrial effluents and atmospheric deposition of pollutants.

Although both phosphorus and nitrogen contribute to eutrophication, increased loadings of phosphorus are usually considered to have played a central role in accelerated eutrophication of rivers and lakes (Moss, 1996, 1998; Reynolds and Davies, 2001). Generally, phosphorus (as orthophosphate) is the limiting nutrient in freshwater aquatic systems. That is, if all phosphorus is used, plant growth will cease, no matter how much nitrogen is available. Although phosphate is an essential element for plant growth and a fundamental element in the metabolic reactions of plants and animals, excessive inputs will result in changes in primary production and species composition resulting in intense algal blooms and oxygen depletion due to the plant material from the increased blooms and algae decomposition. The lack of oxygen causes the death of organisms requiring high oxygen levels such as fish. Hydrogen sulphide is also produced, which is poisonous and can lead to ‘dead’ water bodies.

Within the UK, patterns of increasing phosphorus concentrations in freshwater are well documented (e.g. DoE, 1976; Heathwaite et al., 1996; Neal et al., 2000) and suggest that phosphorus concentrations have increased to elevated levels that affect freshwater ecology. The resulting increased fertility through its effect on destabilizing the ecosystem, causes the growth of algae, in the form of slime, mats and blooms, certain rooted aquatic
plants and weeds. This results in various anthropocentric impacts upon the quality of water supplies causing taste and odour problems and clogging industrial water supply filters resulting in large clean-up costs. Eutrophication also causes deleterious effects upon recreation activities, aesthetic quality and commercial and sport fishing\(^2\) and has been associated with seafood contamination by neurotoxins leading to human health problems. It is immediately apparent then that eutrophication can affect an ecosystem in numerous ways and that there may be substantial economic costs as a result of such pollution.

The Environment Agency’s 1998 report on the state of freshwater in the UK identified nutrient enrichment as one of the key issues that needs to be addressed in order to achieve a more sustainable balance between the needs of society and that of freshwater ecosystems. Phosphates are introduced into freshwater from both domestic sewage and agricultural run-off. In contrast to agricultural waste and soil run-off, sewage and urban waste-waters arise in far greater quantity; and the potential for adverse environmental impact from the phosphates they contain are by far the most serious (Moss, 2001).

Estimates for the UK put the annual production of domestic sewage at around 40 million tonnes. These wastes are estimated to contain around 45 000 tonnes of phosphorus, largely derived from human wastes and other household activities, including contributions from household detergents (CEFIC/CEEP, 1998).

The stresses put upon the integrity of freshwater resources are likely to be further exacerbated as a result of pressure from global climate change and population growth. Over the past 50 years or so, there has been an increasing pattern of climate variability in the UK with a trend towards greater extremes, resulting in lower summer baseflow and higher winter flows (Marshe and Sanderson, 1997). The trends reduce the capacity for sewage dilution resulting in elevated nutrient concentrations (Neal et al., 2000).\(^3\) Consequently there are concerns over nutrient inputs, that is, lower summer baseflows, higher water residence times and elevated nutrient concentrations may promote the development of algal blooms (Jarvie et al., 2002), with some evidence of regional climate change impact on nutrient cycling in the UK (Osborn et al., 2000). Horne and Goldman (1994) report that alterations to the nutrient cycle through climate change will exacerbate existing water quality problems such as eutrophication. Climate change is also expected to have a secondary effect upon water quality through its role in increasing human demand for water services (McCarthy et al., 2001), an effect that may well be exacerbated by population growth. For example, the East Anglian case study area considered in this chapter has one of the highest population inflows in the UK (National Statistics, 2001), but is not particularly well prepared for such changes. Of a total of 1200 sewage
works in the East Anglian region, only 91 have nutrient treatment facilities for removing phosphates.

Given the importance of the water environment to society and the economy, there are statutory obligations and strong political pressure for greater control of phosphorus levels in freshwater ecosystems. The recently agreed EU Water Framework Directive (WFD) together with the EU Urban Waste Water Treatment Directive (UWWT) provide an important impetus for the general control of phosphorus. The 1991 UWWT Directive (CEC, 1991) is concerned with the collection, treatment and discharge of urban and certain industrial waste-waters, and addresses the waters suffering, or at risk from eutrophication. Various measures have been taken in the UK to achieve compliance, including increasing the number of designated sensitive areas and the increased treatment of urban waste-water through phosphate stripping, dredging, etc. Although significant progress has been achieved through the UWWT Directive, it has not achieved all of its stated goals. Given the increasing demand for cleaner water environments, the European Parliament and Council adopted a comprehensive Water Framework Directive that entered into force on 22 December 2000 (CEC, 2000). It requires all member states to adapt the Directive into their national and regional water laws by December 2003. An important part of compliance with this Directive is the completion by December 2004 of an analysis of the impact and pressures upon states’ waters, along with an economic analysis. Such waters need to meet ‘good’ status by December 2015. For surface waters, the definition of ‘good’ is based on a new concept of ‘ecological quality’ taking into account biology, chemistry and their physical features.

Turning now to the costs of preventing eutrophication, a significant problem here in estimating such costs concerns the fact that no absolute definition exists of when nutrient enrichment causes adverse impacts. Rather threshold levels vary such that the nature of the relationship between nutrient enrichment, resulting effects and costs is uncertain. Accepting this caveat, Pretty et al. (2003) provide some useful cost estimates by carrying out an assessment of the policy response costs of preventing eutrophication of freshwaters for the whole of England and Wales. The costs incurred in responding to eutrophication damage and changing practices to meet legal obligations include compliance control costs and other direct costs incurred by agencies. The compliance costs incorporate sewage treatment costs, the costs of treatment of algal blooms and in-water preventive measures and the costs of adopting new farm practices that result in lower emissions of nutrients. The direct costs incurred include the costs incurred by statutory agencies for monitoring of water and air, investing in eutrophication control policies and strategies and enforcing any solutions to eutrophication.
With regards to estimating these various costs, the authors note a number of limitations with the economic data on eutrophication. Firstly, there are a number of valuation methodologies and these are not necessarily comparable across cost categories. Second, the limited data for England and Wales necessitated the use of information from elsewhere in the UK and the world. Thirdly, some cost data relate to the wider problem of sewage treatment and hence only a proportion relates specifically to eutrophication. Finally, there is some degree of uncertainty regarding the temporal, spatial and geographic extent of the problem and associated costs. Despite these problems in estimation, Pretty et al. (2003) find that the current annual policy response costs of addressing eutrophication damage in England and Wales amount to £54.8 million. We adapt those figures for use in our subsequent cost–benefit analysis.

THE BENEFITS OF PREVENTING EUTROPHICATION

In order to carry out a cost–benefit analysis exercise it is obviously necessary to estimate the economic benefits that individuals derive from preventing excess algae (eutrophication) impacts upon open water in rivers and lakes in East Anglia. It is this estimation that forms the major empirical focus of this chapter. Given the public goods nature of rivers and lakes in the East Anglia region, estimates of the economic benefits lost through damage to these goods as a result of eutrophication are not available from market price data. In order to obtain the values needed to undertake a cost–benefit analysis, a survey questionnaire based on the contingent valuation method (Mitchell and Carson, 1989; Bateman et al., 2002) was used to estimate an individual’s willingness to pay (WTP) for a scheme to prevent excess algae in the rivers and lakes in order to ensure continued access to the amenity and recreation facilities that each site provides. In the present case, respondents were being asked to pay for a sewage treatment programme that would remove phosphorous and reduce eutrophication. As such the study seeks to estimate an equivalent loss welfare measure (Boadway and Bruce, 1984; Bateman et al., 2000). Furthermore in undertaking the valuation exercise it is necessary to define the accounting stance, or relevant target population, with which to compare the economic benefits and costs. Since the present study was considering the ‘rivers and lakes of East Anglia’, only the regional population was considered in the valuation and subsequent aggregation exercise, though it is acknowledged that those who benefit may include user households living outside the area as well as a significant population holding non-use values (who may be quite dispersed and difficult to determine).
CV Survey Design and Data

A contingent valuation survey requires that the change in the provision of the good that respondents are being asked to value is communicated to and understood by them. A procedure to elicit respondents’ values is then required (elicitation method), as well as a mechanism by which respondents are told that they will have to pay for the change in provision (payment vehicle). One needs to be confident that respondents are actually valuing the specific change in provision and not some other more general change. These elements are usually contained within an information statement, a valuation scenario and questions and debriefing questions. In designing the contingent valuation survey questionnaire used in the present study there were a number of stages in its development, including focus group discussions (Morgan, 1993; Krueger, 1994) and a pilot survey of roughly 100 respondents.

The aim of the focus groups was to help determine how much information to present and how to convey it within the specific context of this study, such that the valuation scenario was credible and realistic, as well as to refine the questions used in the valuation exercise survey (e.g. Desvousges and Frey, 1989). Four focus group sessions of between six and seven people were thus organized and conducted during March 2003 with groups of East Anglian residents. The focus group protocol paid particular attention to participants’ level of understanding of issues such as the concept of ‘open’ waters; the uses given to ‘open’ water; the frequency of these activities (uses); the characteristics of good rivers and lakes; the concept of ‘algae’; the causes of algal growth; the problems caused by excess algae; what defines a ‘bad’ river or lake; how ‘bad’ qualities are rated; how both good and bad open water is defined.

The findings from the focus groups were used in the development of a draft contingent valuation survey questionnaire. This comprised a number of sections including respondents’ assessment of their present use of water bodies; reactions, including belief indicators, regarding the process by which water bodies may be affected by eutrophication; assessments of how such changes might impact upon usage of those water bodies; valuation scenario section outlining the proposed scheme (solution to the phosphorus problem); valuation task to determine households’ willingness to pay to avoid the specified eutrophication impacts; further belief indicators regarding perceived credibility of the proposed phosphorus reduction scheme.

Since the study was concerned with the prevention of eutrophication of rivers and lakes caused by domestic sewage as a result of future climate and population changes, the valuation scenario paid particular attention to the features and uses of rivers and lakes, the problem of phosphates,
their sources and impacts in terms of algal blooms, future increases in the
population of East Anglia and consequent pressure on sewage treatment
works, and the effects of changing future weather patterns on water quality.
Pictograms and photographs were used to explain the scenario, these being
detailed in the Appendix to this chapter. These were tested and revised in
the focus group discussions. Regarding the constructed market solution to
the problem of eutrophication, survey respondents were asked to consider
a plausible phosphate removal scheme at sewage works. They were told
that such treatment would increase their annual household water bill. Such
bills have desirable properties as a payment vehicle since their compulsory
payment nature reduces any potential free-riding behaviour (Bateman et
al., 2002). Water bills are also likely to be the method by which actual
payments would be made.

After the presentation of the valuation scenario and payment vehicle,
the elicitation question asked respondents how much they would pay to
purchase the good, under specified terms and conditions. In particular,
the question was based on the application of a novel ‘one-and-one-half-
bound’ (OOHB) dichotomous choice (DC) method for the elicitation of
WTP values that has recently been developed (Cooper et al., 2002). This
differs from a standard single bound DC approach (based on a single yes/
no question) or the double bound DC approach (based on a first yes/no
question followed by a second yes/no question conditional upon the answer
given to the first question), and instead presents survey respondents with
upper and lower bound limits of cost per household (or per individual)
associated with provision of the good in question. These amounts, which are
presented simultaneously as a range, are denoted as £\(_U\) and £\(_L\) respectively.

Once the range has been defined, a typical treatment might ask a respondent
initially if they are prepared to pay the amount £\(_L\). A negative response
to this question terminates the value elicitation process and is assumed to
indicate that the respondent has a WTP less than or equal to £\(_L\). However,
a positive response to this initial question triggers a second valuation task
in which the respondent is asked whether they are prepared to pay £\(_U\). Here
a further positive response is assumed to indicate that WTP is greater than
or equal to £\(_U\) whereas a negative response denotes WTP lies somewhere
between £\(_L\) and £\(_U\). The precise value of £\(_L\) and £\(_U\) varies across the sample
in order to permit the estimation of survival functions and associated mean
and median WTP. The present study is the first to apply this technique
within a public goods, non-market valuation context.

The OOHB procedure thus generates a mixture of either single or double
bounded responses from individuals. The approach has greater statistical
efficiency than the single bound dichotomous choice approach, and whilst
not as efficient as the double bounded format, does not suffer from the
Amenity and water quality

charge of lacking incentive compatibility regarding the second of the double bounded responses (Carson et al., 2000).

The elicitation questions were followed by a debriefing question that asked respondents to state their reasons for accepting or refusing the bid amounts. This was followed by questions about the respondents’ beliefs regarding the plausibility of the scenario presented. Finally, the survey included questions concerning demographic and socio-economic characteristics of the respondents.

The draft questionnaire was administered to a sample of around 100 respondents for the purposes of pre-testing and in order to derive information with which to specify a bid vector for use in the main survey. Results from the pilot survey helped to fine-tune the questionnaire, especially in regard to the refinement of the valuation scenario budget reminder and the inclusion of ‘cheap talk’ text. Cheap talk refers to the process of explaining hypothetical bias to individuals prior to asking a valuation question, and has been found to help in reducing such bias when administered with due care (Cummings and Taylor, 1999; List, 2001; Aadland and Caplan, 2003).

Following on from the pilot testing of the questionnaire, the main survey was carried out for a period of five weeks in and around the city of Norwich, centrally located in the East Anglia region. The survey was carried out using a face-to-face interview approach in accordance with the NOAA Panel recommendation (Arrow et al., 1993). Randomly selected starting bid levels were administered to the sample. Information gleaned from the focus group and pilot survey exercises suggested a bid vector ranging from £10 to £200. Each respondent was randomly allocated to one of possible bid pairs: £10–£50; £25–£100; £50–£100; £75–£100; £100–£150; £100–£200; £48.50–£98.50. With the exception of the latter pair (£48.50 to £98.50, which was only offered in a ‘lower to upper’, ascending order), all bid pairs were offered either in an ascending or descending order generating a total of 13 different bid treatments. Although households were randomly allocated to bid pairs care was taken to ensure that roughly similar proportions were allocated to each pairing of bid amounts.

Results

The results of the contingent valuation survey are presented under five headings. The first section provides information on the general characteristics of the sample; the second and third sections consider respondents’ perceptions and level of awareness of the issue, while the fourth section presents the results of the valuation exercise to assess the benefits of reducing eutrophication. The fifth section aggregates these benefit estimate values for
the relevant population to obtain total benefits from reducing the damage caused by eutrophication to rivers and lakes in East Anglia.

Survey participation and refusal
Although most contingent valuation studies undertake extensive analysis of WTP responses, relatively few consider the important issue of sample self-selection. In total, 2321 households were approached for interview at their place of residence. Of this, 1067 refused to participate in the survey, thus giving a response rate of 54 per cent. Analysis of the decision to participate in the survey was undertaken in order to examine whether there was any systematic bias in the survey sample collected. This analysis was naturally limited to those variables passively observable by interviewers for those who were approached for interview (since it was not possible to collect more extensive information on those who refused to participate in the survey). These variables included the approximate age of the person approached; their gender; and the type of housing they lived in. It was found that, as expected, there was a high degree of correlation between the type of housing a person lived in and their age. Further multivariate logistic regression analysis examined the influence of the available variables, including the gender of the interviewer, on the decision to participate in the survey. Given the correlation between house type and age, as well as the ease of interpreting the age variable, the former variable was excluded from this analysis. Table 15.1 shows the coefficients from the estimated model that explains respondent participation in the survey. All the coefficients were highly significant and indicate that the probability of participating in the survey is a parabolic function of age; female respondents are less likely to participate in the survey compared with men; and respondents of either sex are more likely to participate in the survey when questioned by a female rather than a male.

Table 15.1  Multivariate logistic analysis of decision to participate in survey

<table>
<thead>
<tr>
<th>Variable</th>
<th>Coefficients</th>
<th>s.e.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Respondent age</td>
<td>–0.9491**</td>
<td>0.1627</td>
</tr>
<tr>
<td>(Respondent age)^2</td>
<td>0.0818**</td>
<td>0.0138</td>
</tr>
<tr>
<td>Female respondent</td>
<td>–0.1933*</td>
<td>0.0859</td>
</tr>
<tr>
<td>Female interviewer</td>
<td>0.3514**</td>
<td>0.0877</td>
</tr>
<tr>
<td>Constant</td>
<td>2.7065**</td>
<td>0.4527</td>
</tr>
<tr>
<td>LLF</td>
<td>–1572.9364</td>
<td></td>
</tr>
</tbody>
</table>

Note:  Confidence levels are: **99%; *95%.
Figure 15.1 illustrates these findings more clearly by showing the probability of participating in the survey according to age, the gender of the respondent and the gender of the interviewer. We observe a U-shaped relationship between age and participation. This relationship contrasts with the inverted-U relationship between WTP and age sometimes observed in CV studies.\footnote{7} Taken together one might argue that these results suggest that those most likely to participate in CV studies are those with lower values for the good in question. This need not be considered a paradox if we consider that those with lower values may be participating so as to ensure that their values are not over-ridden in the decision process resulting in an overcharging for services relative to the value they generate for such respondents.

Figure 15.1 Analysis of survey participation

Figure 15.1 also illustrates the interactions between age and the gender of both the (potential) respondent and the interviewer. Those most likely to participate in the survey are male respondents being interviewed by female interviewers, whilst those least likely to participate are female respondents being interviewed by male respondents. In summary these findings suggest that there is significant self-selection within our sample, a result that we suspect may apply to many other CV studies that typically do not test for this problem. We return to this issue subsequently.

Respondents' characteristics

If we now consider the sample that actually completed the survey, their general socio-economic characteristics are shown in Table 15.2 along with the population demographics of the East Anglia target population (National Statistics, 2001). There were significant differences in gender ($p < 0.05$) and employment ($p < 0.05$) between the study sample and the target population.
However, no significant differences were found in the distribution of income between the study sample and the target population.

Table 15.2  Economic and socio-demographic characteristics of the sample and target population

<table>
<thead>
<tr>
<th>Respondent's Characteristics</th>
<th>Total Sample</th>
<th>East Anglia Population</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean household income per annum (£)</td>
<td>£22,732</td>
<td>£26,520</td>
</tr>
<tr>
<td>Mean age (years)</td>
<td>47.2</td>
<td>N/A</td>
</tr>
<tr>
<td>University degree (%)</td>
<td>26.8</td>
<td>N/A</td>
</tr>
<tr>
<td>Upper secondary (%)</td>
<td>57.2</td>
<td>N/A</td>
</tr>
<tr>
<td>Female (%)</td>
<td>57.0</td>
<td>51.0</td>
</tr>
<tr>
<td>Mean size of the household</td>
<td>2.6</td>
<td>2.4</td>
</tr>
<tr>
<td>Employment (% of sample with some type of employment)</td>
<td>52.6</td>
<td>79.2</td>
</tr>
<tr>
<td>Membership to Greenpeace/National Trust/RSPB (%)</td>
<td>24.1</td>
<td>N/A</td>
</tr>
</tbody>
</table>

Awareness, perceptions and beliefs regarding eutrophication and the valuation scenario

Prior to the valuation questions, respondents' awareness of the algae problem was sought, as well as their perceptions of how algae might affect their enjoyment of any visit made. The responses to these questions are presented in Table 15.3. It can be seen that 82.5 per cent of the sample surveyed had previously seen algae in the water bodies, whilst 69.1 per cent considered it would considerably reduce their enjoyment of a visit.

Table 15.3  Awareness and perceptions regarding eutrophication

<table>
<thead>
<tr>
<th>Awareness</th>
<th>Have you ever seen algae before in any lake or river you have visited?</th>
<th>Response (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>No</td>
<td></td>
<td>16.3</td>
</tr>
<tr>
<td>Yes</td>
<td></td>
<td>82.5</td>
</tr>
<tr>
<td>Unsure</td>
<td></td>
<td>1.2</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Perception</th>
<th>How will seeing algae in the water affect your enjoyment of the visit?</th>
<th>Response (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>No change in enjoyment</td>
<td></td>
<td>4.9</td>
</tr>
<tr>
<td>Slight reduction in enjoyment</td>
<td></td>
<td>25.9</td>
</tr>
<tr>
<td>Considerable reduction in enjoyment</td>
<td></td>
<td>69.1</td>
</tr>
</tbody>
</table>
Respondents were also asked some questions concerning the plausibility of the constructed CV scenario. These included questions about whether they believed the information that was presented to them regarding population growth in East Anglia; the change in weather patterns in recent decades; the possibility of excess algae occurring in the future; and the success of the proposed scheme to prevent algal growth. Responses to these questions are shown in Table 15.4. In all cases, the majority of respondents believed the information that was provided. However, a substantial minority expressed uncertainty regarding the effectiveness of the proposed scheme. We return to this issue subsequently.

Table 15.4  Belief in the valuation scenario

<table>
<thead>
<tr>
<th>Statement on Valuation</th>
<th>Don’t Believe (%)</th>
<th>Unsure (%)</th>
<th>Believe (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Population of East Anglia is increasing</td>
<td>0.8</td>
<td>5.0</td>
<td>94.2</td>
</tr>
<tr>
<td>Weather patterns are changing</td>
<td>3.5</td>
<td>17.5</td>
<td>78.9</td>
</tr>
<tr>
<td>Possibility of excess algae occurring in the future</td>
<td>0.6</td>
<td>14.8</td>
<td>84.6</td>
</tr>
<tr>
<td>The scheme would prevent excess algae from occurring</td>
<td>1.4</td>
<td>33.0</td>
<td>65.6</td>
</tr>
</tbody>
</table>

Economic valuation results

A total of 1254 households completed the survey questions containing one of the 13 bid treatments, selected randomly so as to ensure equal sample sizes for each bid level. The percentages of respondents agreeing to accept payment of the bid level amount offered to them are presented in Figure 15.2 and Table 15.5 for each of the nine possible bid amounts used. It can be seen that, as expected, the rate of acceptance falls from 93.2 per cent of the sample when the bid amount is £10 to 17.8 per cent acceptance when respondents face a £200 bid level amount. Acceptance rates are also shown for the subsample of respondents who answered positively to the questions on the plausibility of the valuation scenario (see previous section). Once again, the proposed bid amount had a negative relationship on acceptance of the bid level. Comparing acceptance rates across the two samples, it can be seen that, as expected, higher acceptance rates were found at all bid levels (with the exception of the highest bid level of £200) for the subsample who expressed no uncertainty regarding the plausibility of the valuation scenario.
<table>
<thead>
<tr>
<th>Bid Amount (£/household/year)</th>
<th>Total Sample</th>
<th></th>
<th>Believers in Scenario&lt;sup&gt;a&lt;/sup&gt;</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Total asked&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Acceptance of bid amount</td>
<td>Percentage</td>
<td>Total asked&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>10</td>
<td>178</td>
<td>166</td>
<td>93.2</td>
<td>93</td>
</tr>
<tr>
<td>25</td>
<td>169</td>
<td>145</td>
<td>85.8</td>
<td>76</td>
</tr>
<tr>
<td>48.50</td>
<td>46</td>
<td>88</td>
<td>52.3</td>
<td>35</td>
</tr>
<tr>
<td>50</td>
<td>341</td>
<td>177</td>
<td>52.0</td>
<td>174</td>
</tr>
<tr>
<td>75</td>
<td>167</td>
<td>71</td>
<td>42.5</td>
<td>78</td>
</tr>
<tr>
<td>98.50</td>
<td>88</td>
<td>18</td>
<td>20.5</td>
<td>35</td>
</tr>
<tr>
<td>100</td>
<td>836</td>
<td>323</td>
<td>38.6</td>
<td>409</td>
</tr>
<tr>
<td>150</td>
<td>163</td>
<td>55</td>
<td>33.7</td>
<td>82</td>
</tr>
<tr>
<td>200</td>
<td>174</td>
<td>31</td>
<td>17.8</td>
<td>92</td>
</tr>
</tbody>
</table>

<sup>a</sup> Believers in scenario represent the subgroup of the sample who responded positively to the questions used to test the plausibility of the valuation scenarios.

<sup>b</sup> Total asked numbers include responses to the initial value presented and when it was presented as a follow-up value. Hence, we do not have equal numbers of responses for each bid value.
In order to obtain estimates of the mean and median WTP values for the phosphorous removal scheme, parametric modelling of the OOHB responses is undertaken. The OOHB model begins by assuming that the individual’s WTP is a random variable with a cumulative distribution function (CDF) denoted \( G(C_i; \theta) \), where \( C_i \) is the individual’s true maximum WTP and \( \theta \) represents the parameters of the distribution, which are to be estimated on the basis of the responses to the CV survey. As outlined earlier, the OOHB format presents respondents with an initial range of bids \([L, U]\) such that \( L < U \). One of the two bids \((L, U)\) is selected at random and the respondent is asked if he or she would be willing to pay this amount. A follow-up question is asked only if it is compatible to the response to the initial bid. Thus, if the lower bid amount \( L \), is drawn randomly as the initial bid, a respondent would only be asked a second question about willingness to pay the higher bid amount \( (U) \) if they have said yes to the initial lower bid. If the lower bid amount \( (U) \) is offered as the starting bid, the possible response outcomes are (yes), (no, yes to the second lower bid amount) and (no, no to the second lower bid amount). Following Cooper et al. (2002) the corresponding response probabilities are denoted by \( \pi_i^N, \pi_i^{YN}, \pi_i^{YY} \). Conversely, if the higher bid amount \( (U) \) is offered as the initial bid, the possible response outcomes are (yes), (no, yes to the second higher bid amount) and (no, no to the second higher bid amount).
Again, the response probabilities can be denoted as $\pi_i^Y$, $\pi_i^{YN}$, $\pi_i^{NN}$. As these probabilities are such that $\pi_i^N = \pi_i^{NN}$ and so forth, then:

$$
\pi_i^N = \pi_i^{NN} = \Pr(C_i \leq L) = G(L; \theta),
$$

$$
\pi_i^{YN} = \pi_i^{NY} = \Pr(L \leq C_i \leq U) = G(U; \theta) - G(L; \theta),
$$

$$
\pi_i^{YY} = \pi_i^Y = \Pr(C_i \geq U) = 1 - G(U; \theta).
$$

The log-likelihood function for responses using the OOHB format is given by:

$$
\ln L^{OOHB}(\theta) = \sum_{i=1}^n [d_i^Y \ln [1 - G(U; \theta)] + d_i^{YN} \ln [G(U; \theta) - G(L; \theta)] + d_i^{NN} \ln [G(L; \theta)]]
$$

where,

- $d_i^N = 1$ if either the starting bid is $L$ and the response is (no) or the starting bid is $U$ and the response is (no, no), and 0 otherwise;
- $d_i^{YN} = 1$ if either the starting bid is $L$ and the response is (yes, no) or the starting bid is $U$ and the response is (no, yes), and 0 otherwise; and
- $d_i^{YY} = 1$ if either the starting bid is $L$ and the response is (yes, yes) or the starting bid is $U$ and the response is (yes), and 0 otherwise.

The parameters of the WTP distribution are estimated by maximizing the above log-likelihood function using maximum likelihood techniques programmed in GAUSS©. In the empirical application described below, we assume that the distribution of WTP, $G(\cdot)$, is log normal. Accordingly, values for mean and median WTP can be simply recovered by plugging the parameter estimates into well-known closed-form formulas that describe the mean and median of a log-normal distribution (Bateman et al., 2002). Moreover, 95 percent confidence intervals for mean and median WTP can be estimated using a bootstrap procedure (Efron and Tibshirani, 1993). In our application, the 95 percent confidence interval is taken as the 2.5th and 97.5th percentiles of the distribution of mean (or median) WTP values calculated from 1000 bootstrap samples of the original data.

In order to assess the theoretical validity (Mitchell and Carson, 1989; Bateman et al., 2002) of our findings, statistical analysis procedures, as set out above, were employed to estimate a bid function in which the mean of the distribution was able to be shifted by the socio-economic characteristics of respondents. The variables included in the model were respondents’ sex, age, income and their annual frequency of visits to lakes and rivers in East
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Anglia. The estimated model, which is described in Table 15.6, makes the conventional assumption that the WTP distributions underpinning responses to the first and second bound are identical (irrespective of the order in which amounts were presented).9 Results from the estimation process show the expected negative relationship between a positive response and the cost of the scheme, with the latter providing a superior fit when specified as a natural logarithm (ln BID). Expected positive relationships are also with respondents’ household income and the annual number of visits to lakes and rivers. WTP also declines significantly with age, but is not significantly determined by the sex of the respondent. These findings accord with prior economic-theoretic expectations and thereby support the theoretical consistency of our results. The degree of correlation that exists between the two responses from the same individuals is captured by the parameter RHO, which here is positive and significantly different from zero, implying that there is positive correlation between individuals’ responses.10

Table 15.6  Coefficients from constrained model

<table>
<thead>
<tr>
<th>Variable</th>
<th>Coefficients (s.e.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ln BID</td>
<td>-0.7783 (0.0584)*</td>
</tr>
<tr>
<td>INCOME</td>
<td>0.0133 (0.0024)*</td>
</tr>
<tr>
<td>AGE</td>
<td>-0.0040 (0.0019)**</td>
</tr>
<tr>
<td>FEMALE</td>
<td>0.0652 (0.0680)</td>
</tr>
<tr>
<td>VISITOR</td>
<td>0.0009 (0.0004)**</td>
</tr>
<tr>
<td>CONSTANT</td>
<td>3.0917 (0.2777)*</td>
</tr>
<tr>
<td>RHO</td>
<td>0.4263 (0.1134)*</td>
</tr>
<tr>
<td>LLF</td>
<td>-1092.157</td>
</tr>
</tbody>
</table>

Note: Figures in parentheses are standard errors. Confidence levels are: *99%; **95%.

The coefficients from the above model were used to produce estimates of the mean and median WTP as shown in Table 15.7. The mean household annual WTP for the total sample (n = 1112) was found to be £75.41. Although analysis of protest responses was undertaken, this did not have any impact on the valuation estimates.11 Further analysis was also undertaken on the subsample of users (according to whether respondents visited the rivers and lakes in the East Anglia region). Thus 1013 users were identified. It was found that the users did have a slightly higher WTP than the total sample, though the difference was not statistically significant. These results are in line with the consistent finding of somewhat elevated WTP amongst user groups (Desvousges et al., 1987; Dupont, 2003) showing that users have...
higher WTP than non-users. Estimation of WTP for non-users was not possible due to insufficient sample size.

WTP estimates were also estimated for the sample of respondents who 'believed' (see Table 15.4 for definition) in the plausibility of the valuation scenario. These results are also presented in Table 15.7. For the sample of 'believers', the mean annual household WTP was now £66.39. Once again when only users of rivers and lakes were included for the 'believers' sample mean household WTP was £67.06 per year. Again the mean WTP of the non-users was not estimated due to insufficient sample size.

### Table 15.7  Mean and median WTP for the total sample and 'scenario believers' subsample

<table>
<thead>
<tr>
<th></th>
<th>Total Sample</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Total (n = 1112)</td>
<td>Users (n = 1013)</td>
</tr>
<tr>
<td>Mean WTP (£)</td>
<td>75.41</td>
<td>77.75</td>
</tr>
<tr>
<td>Median WTP (£)</td>
<td>69.07</td>
<td>71.87</td>
</tr>
<tr>
<td>95 per cent C.I.</td>
<td>69.41–84.36</td>
<td>70.43–86.96</td>
</tr>
<tr>
<td>Believers only</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>(n = 537)</td>
<td>Users (n = 497)</td>
</tr>
<tr>
<td>Mean WTP (£)</td>
<td>66.39</td>
<td>67.06</td>
</tr>
<tr>
<td>Median WTP (£)</td>
<td>63.86</td>
<td>64.66</td>
</tr>
<tr>
<td>95% C.I.</td>
<td>58.98–74.97</td>
<td>58.93–76.89</td>
</tr>
</tbody>
</table>

### Aggregation of benefit estimates

In aggregating the benefits estimates for application within a cost–benefit analysis (CBA) only the population of the East Anglian region was included as the relevant population that benefits from the prevention of eutrophication. However, as already discussed, such values exclude those held by others living outside the region and so can be considered lower bound benefit estimates.

The results of the benefits aggregation exercise are shown in Table 15.8. The estimates found differ according to the precise WTP values assumed to apply at the population level. In this respect, two possible alternatives were identified. For our first estimate, no adjustment is made to the sample mean WTP to take account of those respondents who refused to be interviewed (i.e., the sample mean WTP is applied to the target population). In this case, the population aggregate WTP was estimated by multiplying the sample mean WTP (£75.41) by the number of households in East Anglia (2.253 million) to obtain annual benefits of £170 million. Our second estimate assigns a zero WTP to those who refused to be interviewed (as per Bateman...
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and Langford, 1997). As such, the inclusion of the 1067 respondents who refused to be interviewed results in a lower sample mean household WTP per year of £38.48 giving an aggregate annual benefits estimate of £87 million.

Table 15.8  Aggregate benefits of the phosphate removal scheme for East Anglia region

<table>
<thead>
<tr>
<th>Aggregation Approach</th>
<th>WTP Per Household (£/year)</th>
<th>Aggregate Benefits (£)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aggregation with sample mean WTP 75.41 (excluding non-response) and East Anglian population – total sample</td>
<td>75.41</td>
<td>169.89 million</td>
</tr>
<tr>
<td>Aggregation with sample mean WTP 38.48 (including non-response) and East Anglian population – total sample</td>
<td>38.48</td>
<td>86.70 million</td>
</tr>
</tbody>
</table>

COMPARISON OF COSTS AND BENEFITS AND CONCLUSION

Comparing the annual aggregate benefits of preventing eutrophication by a phosphate removal scheme for the East Anglian region (shown in Table 15.8) with the current annual policy response costs of addressing eutrophication damage for the whole of England and Wales (reported earlier at £54.8 million), it is clear that net benefits are positive. Even for the lower bound benefit estimate value of £86.7 million/annum, the costs of addressing eutrophication for the whole of England and Wales are still less than the benefits of preventing eutrophication just for the East Anglian region.

One source of uncertainty in our analysis is the self-selection bias noted at the start of our discussion of results. This issue, commonly ignored in many CV studies, proved significant with participation in the survey being linked to age and the interplay between interviewer and potential respondent. However, our analysis provides admittedly circumstantial evidence to suggest that, if anything, it may be those with lower WTP values who are self-selecting into the sample of respondents at a greater rate than others. This would tend to deflate aggregate values and lower cost–benefit ratios. Certainly we have no evidence here to suggest that total values have been inflated by the self-selection process. Consequently the above conclusions remain valid in our opinion.
Although there are still uncertainties associated with both the cost and benefit estimates, freshwater resources would appear to be highly valued, and policies and schemes to protect them can be considered to be worthwhile investments. This point is driven home by the fact that the benefit estimated in this study was only for prevention of eutrophication by a phosphate removal scheme. Other benefits linked to the protection of rivers and lakes, such as the value of water bodies for commercial and other uses, make the investment in phosphate removal equipment an even more compelling project.

To conclude, CBA has been previously used in a number of water resources management schemes (Whittington et al., 1994, Bateman et al., 1995; Eisen-Hecht et al., 2002). In terms of project analysis, no method other than CBA enjoys as widespread application or analytical power. The analysis of the relative costs and benefits of preventing eutrophication undertaken in this study has shown how even a ‘crude’ cost–benefit analysis can play an important role in environmental decision-making. In the wake of the new EU Water Framework Directive and the increasing pressure on public finances, it becomes even more important to correctly allocate resources with respect to freshwater resource protection. Such investments would certainly appear to be worthwhile since these resources provide positive net benefits for people in the United Kingdom.

Finally, recall that the benefit values estimated in this study were based on the novel one-and-one-half-bound elicitation method proposed by Cooper et al. (2002). This approach is designed to address some of the problems associated with single bound and conventional double bounded elicitation approaches. Nevertheless, more rigorous testing of the methodological robustness of this technique is warranted and further investigation is ongoing (see Bateman et al., 2004).

NOTES

1. From an ecological point of view, large-scale eutrophication can be summarized as ‘oxygen depleted “dead” bottoms, excessive planktonic and epiphytic algal growth along with a shrinking of the littoral fucus habitat (shift in vegetation and ecosystem)’ – *Ambio* (1990).

2. In fact fish production may increase, though species composition will change, for example trout may decline in favour of fish of lower value.

3. The impact of these changes is most severe in the South and East of England, which experience low rainfall and high evapo-transpiration (Jarvie et al., 2002).

4. For convenience we adopt the term WTP to refer to this value measure, although it should be noted that in so doing we are not referring to the compensating gain measure to which the WTP term more properly applies.

5. Which has been endorsed by the NOAA Blue Ribbon Panel (Arrow et al., 1993) on the grounds that it is an incentive compatible format.
6. The lack of incentive compatibility arises since respondents are told that the first of the double bounded bids is the cost of the scheme. This is then (conditional upon the first bid response) sequentially changed to a second bid. Arguably this lack of incentive compatibility underpins the observed inconsistency between first and second response distributions in DB exercises (Cameron and Quiggin, 1994; Kanninen, 1995; Bateman et al., 2001). The OOHB approach attempts to address this problem by presenting a range of prices at the outset of the valuation exercise so that the plausibility and incentive compatibility of both $EU$ and $EL$ are assumed to hold for both elicited responses.


8. While Table 15.1 is compiled from responses from the full set of 1254 households, the regression models are restricted to a smaller set of some 1112 households that provided responses in terms of the socio-economic and demographic covariates considered. The extent of this change is dictated by item non-response to the income question for which 142 households (11.3 per cent of those interviewed) failed to provide a response.

9. In further work we examine the implications of relaxing this assumption (see Bateman et al., 2004).

10. More intriguingly, the correlation is far from perfect; RHO is highly significantly different from a value of 1 ($t$-stat 8.38). Again this is investigated further in Bateman et al. (2004).

11. A total of 232 protest bids were identified based on the answers to questions regarding the reasons for acceptance/refusal of a bid amount. These were identified as proposed by Bateman et al. (2002).

REFERENCES


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APPENDIX 15.1  INFORMATION USED TO CONVEY CERTAIN OF THE SCENARIOS PRESENTED TO SURVEY RESPONDENTS

Excess algae can change rivers and lakes from this …
Excess algae forms a scum in and on top of the water.
This cuts out light and reduces the amount of oxygen in the water.
This can kill larger water plants and the fish and wildlife that depend on them.
It creates an unpleasant odour (a musty or mouldy smell).

Algae prevents swimming and may affect other recreation, such as enjoying the view.
The water environment in East Anglia is facing two pressures: increasing population and changes in climate.

- Increase in population
- Increase in houses
- Increase in water demand and sewage
- Increase in algae in rivers and lakes
- Increase source of phosphates
- Increase in algae in rivers and lakes

- Earlier spring
- Longer summers
- Decrease in summer temperatures

Susana Mourato, Allan Provins, Ece Özdemiroğlu, Stavros Georgiou and Jodi Newcombe

INTRODUCTION

In December 2000, the European Commission released a Communication (COM(2000)860) proposing a number of changes to the Bathing Water Directive 76/160/EEC.¹ The changes were initiated by a combination of improved scientific knowledge, the need for more active quality management of bathing waters and a desire for improved public information. The revised Bathing Water Directive is expected to tighten microbiological standards, harmonize beach management regimes across the European Union, require that more information on water quality be made public, and widen the scope of designated bathing waters to include waters used by other water contact sports such as surfing, windsurfing and kayaking.

The costs of achieving the Commission’s expected minimum bathing water quality standards are estimated at around £2.5–£3.9 billion, over 25 years, for England and Wales (DEFRA, 2002). Most of these costs are attributed to reducing agricultural diffuse microbiological pollution of bathing waters, generating costs directly to farmers or in terms of agricultural subsidies. There are also some costs to the water industry from further work on the sewerage infrastructure. In light of these significant costs, the relevant policy question is: do potential benefits justify the costs arising from the revised Directive?

Many of the benefits that the public currently enjoy at beaches, and which may be augmented through implementation of a revised Directive, are non-market in nature. Moreover, the impact of these improvements on individuals’ enjoyment of bathing waters and beaches is uncertain. In this context, non-market economic valuation techniques can be used to
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elicit public preferences for the proposed improvements to bathing water quality. This chapter presents the results of a stated preference study that estimates the economic value of changes that may occur in several beach attributes in England and Wales as a result of revisions to the Bathing Water Directive.

The results of the study may be used to estimate the total benefits from the implementation of a revised Bathing Water Directive to England and Wales. The estimate of total benefit can then be compared with the total cost of the implementation of the revisions in a cost–benefit analysis (CBA) framework. As well as enabling government decision-making to be undertaken on the basis of CBA, the findings of this study are also useful in encouraging further debate over the potential revisions of the Bathing Water Directive.

The rest of this chapter is structured in the following way. The next section describes the methodology used, including the choice of key attributes and levels for the choice experiment. In the third section the survey results are analysed, discussed and aggregated. The fourth section concludes the chapter and discusses the use of the study in decision-making.

METHODOLOGY

This study was designed to measure individuals’ preferences for different types of benefits occurring in beaches and marine waters, brought about by the implementation of a revised Bathing Water Directive in England and Wales. Potential benefits may include improved water quality, beach cleanliness, safety measures and amenities. These changes may be categorized as either ‘use’ benefits (derived by beach users who may swim, relax or sunbathe on the beach, for example) or ‘non-use’ benefits (derived by those who do not make use of a beach but may still value its upkeep per se, or so that it can be used by others and by future generations).

Standard economic theory defines a ‘benefit’ as a change that increases human well-being. Likewise a ‘cost’ is a change that decreases human well-being. Human well-being is determined by the preferences held by individuals, which may be measured by eliciting the maximum willingness to pay (WTP) of an individual for a specified change in the level of provision of a given good. The rate at which a particular individual is prepared to trade off goods and services against one another and against money corresponds to the value of a change in the price or quality of a good or service. Where well-functioning markets exist, the price of a good provides an indication of its value to individuals. However, the various attributes of beaches, such as clean bathing water, are typically not traded in markets. Consequently
There exists no price mechanism to inform on the preferences individuals have towards beaches and the benefits they provide.

There are several economic valuation techniques that may be used to uncover preferences in situations where markets do not exist. This study uses a survey-based stated preference technique known as choice experiments (CEs) to measure willingness to pay for the benefits arising from the revised Bathing Water Directive. CEs assume that the change to be valued can be described in terms of its attributes, or characteristics, and the levels that these take (Louviere et al., 2000; Bennett and Blamey, 2001; Bateman et al., 2002). In a choice experiment, respondents are presented with several scenarios, each described by a number of attributes, including a price attribute, varying at different levels. Respondents are then asked to choose their most preferred scenario, in each set of options they are presented with. How respondents' choices change as attributes are varied enables (marginal) WTP for each attribute to be inferred.

As an attribute-based approach, CEs are particularly useful in multidimensional contexts where trade-offs between the scenario's attributes are of importance, as is the case in this study, where the Bathing Water Directive targets a number of beach characteristics. Furthermore, the exact nature and extent of the revisions to the Directive were uncertain at the time of the study (2002). CEs can be very helpful in the face of uncertainty as they present a range of levels over which the various attributes of interest may vary.

Although there is a considerable literature of stated preference studies about the marine environment, there are very few studies that explicitly consider water quality in the context of EU policy. Table 16.1 summarizes the few European studies available with respect to coastal bathing water. While the conclusions of each study are different depending on the specific study context, the consistent finding that emerges is that individuals attribute a positive value to improvements in water quality. The implication is that poor water quality is undesirable and that the public would be willing to pay positive amounts towards improvements. Typical annual household willingness to pay amounts range from about £10 to £90 per household per year. This suggests significant aggregate benefits when these household values are applied across the population as a whole.

Of note, our study differs from the studies in Table 16.1 in several ways. None of those studies used CEs, or valued specific beach attributes, mostly having applied the contingent valuation method (Mitchell and Carson, 1989; Bateman et al., 2002). Moreover, these other studies were undertaken at beach locations ('on-site') and capture beach users' WTP for discrete changes in water quality (e.g. compliance with a standard). The nature of on-site studies means that they concern only a particular beach or group
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This contrasts with this study’s ‘off-site’ approach, which aims to investigate WTP estimates (use and non-use) for marginal changes on a typical beach in England and Wales.

The basis of our choice experiment was to present respondents with a number of potential future beach scenarios, described by different levels of beach attributes, and to then ask them to state their most preferred option amongst each set. The next sections summarize the main steps involved in the design and implementation of this CE survey.

### Selection of Attributes

The first step in designing a CE study is the choice of attributes that, in this case, are the key characteristics of beaches that are affected by the revision in the Bathing Water Directive, alongside a monetary cost. In order to identify the most important beach attributes consultations were held with government officials and a series of focus groups were carried out with members of the public.

<table>
<thead>
<tr>
<th>Study</th>
<th>Location</th>
<th>Stressor</th>
<th>Quality Change Description</th>
<th>£/Visit</th>
<th>£/Season/Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Georgiou et al. (1998)</td>
<td>UK</td>
<td>Sewage</td>
<td>To EC standard</td>
<td>–</td>
<td>24–90</td>
</tr>
<tr>
<td>Hanley et al. (2001)</td>
<td>Scotland</td>
<td>Sewage</td>
<td>Meet minimum EC standard</td>
<td>–</td>
<td>17–52</td>
</tr>
<tr>
<td>Machado and Mourato (2002)</td>
<td>Portugal</td>
<td>Sewage</td>
<td>Bad to average (EC mandatory)</td>
<td>16</td>
<td>–</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Average to good (EC guideline)</td>
<td>7</td>
<td>–</td>
</tr>
</tbody>
</table>
Is it worth revising the European Bathing Water Directive?

The five attributes selected were (Table 16.2): 1. average water quality; 2. advisory note system; 3. beach cleanliness (litter/dog mess); 4. safety services and amenities, and 5. additional water charges per household per year, required to meet the cost of maintaining each beach scenario. Two different indicators of water quality were used: the 'risk of getting a stomach upset from bathing'; and the 'number of days in the bathing season when it is unsafe to swim'. This gave rise to two versions of the questionnaire (A and B), which were otherwise identical.

The European Commission’s Communication COM(2000)860, on Developing a New Bathing Water Policy, stated that the revised Directive ‘should require the competent authorities to adopt new methods to actively inform the public about bathing water quality, including about all known factors that might influence that water quality. This information should be available at all times at the bathing zone.’ Based on this description, a novel ‘advisory note system’ attribute was defined in the questionnaire as a system of notices posted at beaches that would inform bathers of when it is considered safe or unsafe to bathe (i.e. advising on poor water quality days).

Finally, although the cleanliness of beaches in terms of litter and dog mess are not specifically mentioned in the revisions to the Bathing Water Directive, they were considered important to the public, as was strongly indicated by the focus group discussions. Hence, they were included in the experiment as one of the features of a typical beach (Table 16.2).

Assignment of Levels

The consultation process that aided the selection of attributes was also used to identify their current levels and how these might change under alternative future scenarios. The improved beach levels used in the experimental design are presented in the last column of Table 16.2. All attributes vary at two levels, with the exception of the cost attribute, which has four levels.

Regarding the identification of the baseline levels of the attributes, some complications arose. Specifically, the current water quality at an average beach in England and Wales was not known with certainty. This presented a problem as Machado and Mourato (2002) reported different WTP values for improvements in water quality depending on the baseline: for instance, an improvement from ‘bad to average’ quality was valued more than an improvement from ‘average to good’. To address this problem, a number of baseline water quality values were chosen for the current situation scenario.

In Version A, three different water quality baseline risk levels were used, taking into account the current epidemiological evidence (Kay et al., 1994;
Table 16.2  Beach attributes and levels

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Description</th>
<th>Baseline levels</th>
<th>Improvement levels</th>
</tr>
</thead>
<tbody>
<tr>
<td>Average water quality</td>
<td>• Risk of getting a stomach upset from going in the water (Version A)</td>
<td>2 in 100, 5 in 100, 0.5 in 100, 1 in 100, 7 in 100</td>
<td></td>
</tr>
<tr>
<td></td>
<td>• Number of days when it is unsafe to swim due to poor water quality, out of 150 days in the bathing season (Version B)</td>
<td>10 in 150, 16 in 150</td>
<td>1 in 150, 5 in 150</td>
</tr>
<tr>
<td>Advisory note system</td>
<td>Notice system that advises against swimming on days when water quality is worse than the average and hence the risk of getting a stomach upset is higher than the average</td>
<td>None</td>
<td>None, in place</td>
</tr>
<tr>
<td>Litter/dog mess</td>
<td>Litter and dog mess on the beach</td>
<td>Some</td>
<td>None, some</td>
</tr>
<tr>
<td>Safety and amenities</td>
<td>Quality of the toilets and showering facilities, life-guards and life-saving equipment at the beach</td>
<td>Average</td>
<td>Average, good</td>
</tr>
<tr>
<td>Additional water charges per year</td>
<td>Extra cost to all households in Britain for improving British beaches. Investments would have to be made to improve water quality and beach management, which would mean an increase in water charges that all households across the country would have to pay, per household per year, specifically to provide these services</td>
<td>£0</td>
<td>£3, £11, £15, £25</td>
</tr>
</tbody>
</table>

Note: ‘Stomach upset’ was defined as follows: ‘The possible symptoms of stomach upset are, in order of increasing severity: mild diarrhoea, indigestion with fever, nausea with fever, vomiting, diarrhoea. It is thought that the majority of cases of stomach upset that result from swimming in polluted seawater are at the milder end of the spectrum of symptoms.’
Fleisher et al., 1998) and water quality compliance rates of British beaches in 2001. It was estimated that the probabilities of getting a stomach upset from bathing in poor, average and good water quality were (respectively) 7, 5 and 2 out of every 100 bathers. The baseline starting points in Version B were 16 and 10 advisory note days, that is, days on which swimming is unsafe due to higher risk of stomach upsets for which advisory notes are issued (out of 150 in the bathing season), for poor and average bathing water quality respectively. Collectively, these baseline values represented the range of possible water quality levels across beaches in Britain. These variations resulted in five versions of the questionnaire, as shown in Table 16.3.

Note that the inclusion of several ‘current situation’ scenarios for the water quality attribute permits a test of whether the WTP estimates demonstrate a type of ‘starting point’ bias, that is, to see if respondents value a change in water quality differently depending on how high the baseline is considered to be.3

Experimental Design

Statistical design theory was used to combine the levels of the attributes into a number of alternative beach improvement scenarios to be presented to respondents. Combination of the five attributes at their various levels created a total of 64 (2 × 2 × 2 × 2 × 4) possible scenario combinations. It is obviously too difficult for a respondent to make an informed choice from such a large number of beach scenarios. A fractional factorial design was therefore used to reduce the number of scenarios, while still maintaining the possibility of estimating ‘main effects’, that is, the effects of the attributes on respondents’ choices, which typically explain 80 per cent of all the variation in choices made (Louviere et al., 2000). The fractional factorial design selected eight scenarios, which were then grouped into choice sets to be presented to respondents.

Each choice set contained the current situation for a typical beach and two beach improvement scenarios, from the eight generated by the statistical design. The current situation was included in each choice set to give respondents the opportunity to choose the ‘no change’ option, where they would not incur additional water charges and could hence state a zero WTP for the improvements in question. A sample choice set is presented in Figure 16.1. Each respondent was presented with eight of these choice sets.

The experimental procedure just described was repeated for each of the five questionnaire versions described in Table 16.3 (A1, A2, A3 and B1, B2). Hence, there were five sets of eight choice triplets, which were then assigned to different subsamples.
Table 16.3  Water quality levels in different versions of the questionnaire

<table>
<thead>
<tr>
<th>Version A</th>
<th>Risk of getting a stomach upset from going in the water</th>
<th>Version B</th>
<th>Number of days it is unsafe to swim (out of 150 days in the bathing season)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Current situation</td>
<td>Scenario A</td>
<td>Scenario B</td>
<td>Current situation</td>
</tr>
<tr>
<td>A1</td>
<td>7/100</td>
<td>1/100</td>
<td>0.5/100</td>
</tr>
<tr>
<td>A2</td>
<td>5/100</td>
<td>1/100</td>
<td>0.5/100</td>
</tr>
<tr>
<td>A3</td>
<td>2/100</td>
<td>1/100</td>
<td>0.5/100</td>
</tr>
</tbody>
</table>
Survey Implementation

Focus groups, peer review and field pilot surveys were used to develop and pre-test the CE questionnaire. A series of focus groups and two pilot studies were carried out between January and February 2002, in London and Norwich. The main CE survey took place in March 2002 at 49 different sites in England and Wales, including both coastal and inland locations. The questionnaire was administered face-to-face and comprised four sections: a section on beach use and attitudes towards various beach characteristics and risks; the CE valuation section; debriefing questions relating to responses to the valuation section; and socio-economic characteristics.

**Figure 16.1 Choice set example**

<table>
<thead>
<tr>
<th>Current Situation</th>
<th>Scenario A</th>
<th>Scenario B</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Average water quality</strong></td>
<td><img src="current_water_quality" alt="Image" /></td>
<td>7 in 100</td>
</tr>
<tr>
<td><strong>Advisory notice system</strong></td>
<td><img src="advisory_system" alt="Image" /></td>
<td>None</td>
</tr>
<tr>
<td><strong>Litter/dog mess</strong></td>
<td><img src="litter_dog_mess" alt="Image" /></td>
<td>Some</td>
</tr>
<tr>
<td><strong>Safety and amenities</strong></td>
<td><img src="safety_amenities" alt="Image" /></td>
<td>Average</td>
</tr>
<tr>
<td><strong>Additional water charges</strong></td>
<td><img src="water_charges" alt="Image" /></td>
<td>£0</td>
</tr>
<tr>
<td><strong>I would prefer:</strong></td>
<td><img src="checkbox" alt="Checkbox" /></td>
<td><img src="checkbox" alt="Checkbox" /></td>
</tr>
</tbody>
</table>
The population of interest for the study was that of England and Wales. This includes both those adversely and beneficially affected by revisions to the Directive, and those who may hold use and/or non-use values. A random stratified sample was selected from the English and Welsh population, designed to be representative on the basis of gender, age and socio-economic group. In a CE survey, the required sample size is dependent on the number of attributes chosen, the number of levels to investigate and other design factors. Based on the CE design described above, which involved five versions of the questionnaire and five attributes at two to four levels each, a sample size of 800 respondents was considered to be the minimum needed to support the design. In the event, the sample consisted of 809 respondents.

RESULTS AND DISCUSSION

Overall, the survey sample was considered representative of the population of England and Wales. Furthermore, no significant differences in survey characteristics were identified between each of the five subsamples that were administered different questionnaire versions, hence differences in results across questionnaire versions should not be attributable to differences in sample characteristics.

Beach Use and Attitudes

The vast majority (97 per cent) of respondents had visited a beach in Britain sometime in their life. Ninety per cent had visited in the last five years and 76 per cent had visited in 2001. Local residents were responsible for the largest proportion of beach visits (42 per cent), 24 per cent were attributable to day trippers, whilst those on short and long breaks made up 20 per cent of trips.

The average number of beach visits in 2001 was about 11 per person. Removing the ‘outlier’ observations in the upper portion of the distribution (those people who visited every day of the year) gives a trimmed mean value of 5.7 visits per person per year and a median of three visits per person per year. Assuming the trimmed mean to be more representative, multiplying this value by the population of England and Wales (52.9 million) gives a total number of visits to the beach in 2001 of 301 million.

Walking, relaxing, sun-bathing and picnicking were cited as the most commonly practised activities on the beach. A high proportion of respondents professed never to go in the sea: 57 per cent of respondents and their families never go swimming and 78 per cent never participate in
Is it worth revising the European Bathing Water Directive?

water sports, a result supported by the focus groups' view that many people find it too cold in Britain to go swimming. This finding suggests that the main benefits of the Directive revisions might not be derived from direct contact with water of improved quality.

Respondents cited cleanliness of the beach, water that looks clear of foam and litter and clean toilets/changing facilities as the most important features of a beach. This was followed by information provided about tides, good access and parking. Notably, although over half of respondents (51 per cent) cited water quality as important, this attribute was only ranked the sixth most important beach attribute out of 13 listed.

Finally, when presented with a description of the proposed advisory note system, the majority of respondents (51 per cent) thought it would be 'very useful'. Only 4 per cent thought it would 'not be useful at all'. A third of respondents said they would prefer to see a simple symbol indicating 'do not bathe' whilst just over half (51 per cent) requested more detailed information. The majority of respondents (78 per cent) said they would still visit the beach despite the warning not to swim, which allays fears that an advisory note system could have a significant adverse effect on beach visitation. Nonetheless, the 18 per cent of respondents who said they would not stay at the beach as a result of the sign could still represent a notable economic loss.

Health Risks

Based on responses to questions about contact with water (e.g. number of times water sport activities are undertaken, number of times the head is submerged under water) two rates of potential illness were estimated: a predicted rate of illness and a self-assessed rate of illness (Georgiou, 2004).

The predicted rate of illness was estimated using data on the average number of swims gathered from the survey in conjunction with epidemiological data on infection risks. It was assumed that when the head was not submerged in the water during a swim, the bather was not considered to have been exposed to enteric pathogens (Kay et al., 1994), therefore in order to calculate the annual number of people in England and Wales likely to fall ill from swimming, it was necessary to count only those swims for which the head was submerged. On this basis, and applying a mean number of swims/dips per person per year of 1.6, it was estimated that there could be 1.3 million cases of stomach upset per year in England and Wales associated with swimming alone, or 2.84 million for all water-based activities.4,5

The self-assessed rate of sickness was estimated using the responses to a question on the number of stomach upsets suffered, which respondents
believed to be caused by poor bathing water quality. The total number of such cases of stomach upset was 34, and the average duration of illness was 4.1 days. This figure is similar to that reported by Fleisher et al. (1998). Using these data the number of stomach upsets per year in England and Wales was estimated as 2.2 million. The convergence of the two types of estimates for the number of cases of stomach upset (1.3–2.2 million cases) shows that there is some consistency between the self-assessments of bathing water-related illness and predictions from the epidemiological literature.

Choice Experiment Results

Assessing the relative importance of different beach characteristics and respondents’ willingness to pay requires that responses to the choice experiment questions be analysed using an indirect utility function $U_{ij}$:

$$U_{ij} = V_{ij} + \varepsilon_{ij} = b_1 \text{WATER}_{ij} + b_2 \text{ADVISORY}_{ij} + b_3 \text{LITTER}_{ij} + b_4 \text{AMENITY}_{ij} + b_5 \text{PRICE}_{ij} + \varepsilon_{ij}$$

(16.1)

The function explains the satisfaction that an individual $i$ receives from a typical beach $j$ as a function of a linear index of its attributes ($V$) and an error term that captures unobservable factors ($\varepsilon$). The attributes are described in terms of water quality (WATER), presence of an advisory note system (ADVISORY), beach cleanliness (LITTER), quality of amenities and safety provisions (AMENITY) and the cost to the household of achieving a certain level of beach quality (PRICE). The coefficients $b_1, b_2, b_3, b_4$ and $b_5$ describe the influence of the various attributes on $U_{ij}$.

The model was estimated using a nested logit (NL) specification (Louviere et al., 2000). This implied recasting the choice problem as a hierarchical nested structure, where choices were analysed in two stages. In the first stage, the model predicts whether or not respondents are willing to pay anything towards improved beaches; in the second stage, the model predicts which of the two beach improvement scenarios they prefer.

In a two-tier choice structure, the probability of choosing a particular alternative $k$ out of $n$ second stage scenarios, conditional on having selected a particular alternative $j$ out of $m$ first stage scenarios, is expressed in equation (16.2), where $X$ is the vector of scenario attributes. The logarithm of the denominator of this expression is known as the inclusive value ($I_k|_l$), because it summarizes the information about the alternatives included in this lower nest. Inserting this inclusive value as an explanatory variable in the first stage of the decision tree yields the expression for the unconditional probability of choosing scenario $j$ out of the $m$ first stage scenarios, given in equation (16.3):
Is it worth revising the European Bathing Water Directive?

\[
P(k \mid j) = \frac{\exp(b_{kj} X_{k})}{\sum_{n} \exp(b_{nj} X_{n})} = \frac{\exp(b_{kj} X_{k})}{\exp I_{kj}} \quad (16.2)
\]

\[
P(j) = \frac{\exp(c_j X_{j} + \rho I_{kj})}{\sum_{m} \exp(c_m X_{m} + \rho I_{k,m})} \quad (16.3)
\]

Three variations of the NL model were estimated. The first model explains respondents’ choices between beach scenarios solely as a function of beach attributes (i.e. equation 16.1). The second model uses an alternative specific constant (ASC) for the third beach scenario in each choice set, which always corresponded to the most expensive and largest improvement. This constant captures any variation in choices that is not explained by either the beach attributes or respondent-specific socio-economic variables. The third model includes socio-economic and attitude regressors specific to individual respondents: a dummy variable indicating whether respondents were in the socio-economic segment AB/C1 (CLASS); a dummy variable showing if respondents engage often in swimming or water sports at British beaches (OFTEN); and a dummy variable indicating if respondents thought that water quality was a very important factor in choosing to visit a British beach (WQIMP). This specification allows the introduction of respondent heterogeneity into the model. Individual-specific variables, however, cannot be introduced into the model on their own, as they would drop from the estimation since they do not vary across choices. In this case, these variables were interacted with the ASC.

Estimation results are presented in Table 16.4, for both versions of the questionnaire (A and B). For all models the beach attribute coefficients are significant at the 1 per cent level and have the expected signs across all survey versions and model specifications. The results suggest that higher levels of water pollution are undesirable, as is the presence of litter/dog mess; while increased amenity and safety quality and the presence of an advisory note system have a positive impact on utility. Finally, as expected, an increase in water charges has a negative impact on utility.

Results of the NL models with interactions also indicate that wealthier respondents (those in socio-economic class AB/C1), those who are often in contact with water (via swimming or water sports) and those that consider water quality to be a very important determinant of their choice of a beach (in Version A) are more likely to choose the most expensive and largest beach improvement scenario. These results are consistent with expectations.
### Table 16.4 Nested logit model results

<table>
<thead>
<tr>
<th></th>
<th>Attributes only</th>
<th>Attributes + ASC</th>
<th>Attributes + ASC + individual-specific variables</th>
<th>Attributes only</th>
<th>Attributes + ASC</th>
<th>Attributes + ASC + individual-specific variables</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Coeff. (s.e.)</td>
<td>P-value</td>
<td>Coeff. (s.e.)</td>
<td>P-value</td>
<td>Coeff. (s.e.)</td>
<td>P-value</td>
</tr>
<tr>
<td>ASC</td>
<td>—</td>
<td>—</td>
<td>—0.715 (0.074)</td>
<td>0.000</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>WATER</td>
<td>—0.198 (0.072)</td>
<td>0.000</td>
<td>—0.666 (0.066)</td>
<td>0.007</td>
<td>—0.52 (0.052)</td>
<td>0.017</td>
</tr>
<tr>
<td>ADVISORY</td>
<td>0.034 (0.024)</td>
<td>0.000</td>
<td>0.062 (0.057)</td>
<td>0.000</td>
<td>0.510 (0.076)</td>
<td>0.000</td>
</tr>
<tr>
<td>LITTER</td>
<td>—0.654 (0.074)</td>
<td>0.000</td>
<td>—0.544 (0.075)</td>
<td>0.000</td>
<td>—0.510 (0.076)</td>
<td>0.000</td>
</tr>
<tr>
<td>AMENITY</td>
<td>0.240 (0.062)</td>
<td>0.000</td>
<td>0.354 (0.061)</td>
<td>0.000</td>
<td>0.341 (0.060)</td>
<td>0.000</td>
</tr>
<tr>
<td>PRICE</td>
<td>—0.097 (0.006)</td>
<td>0.000</td>
<td>—0.049 (0.006)</td>
<td>0.000</td>
<td>—0.047 (0.000)</td>
<td>0.000</td>
</tr>
<tr>
<td>CLASS*ASC</td>
<td>—</td>
<td>—</td>
<td>—0.276 (0.093)</td>
<td>0.000</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>OFTEN*ASC</td>
<td>—</td>
<td>—</td>
<td>—0.727 (0.134)</td>
<td>0.000</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>WQIMP*ASC</td>
<td>—</td>
<td>—</td>
<td>0.195 (0.092)</td>
<td>0.035</td>
<td>—</td>
<td>—</td>
</tr>
</tbody>
</table>

Log-Likelihood: 
- Version A: -3111.89
- Version B: -2009.95

Pseudo R²: 
- Version A: 0.13
- Version B: 0.15

N: 
- Version A: 383
- Version B: 249

Log-Likelihood: 
- Version A: -3069.31
- Version B: -1977.57

Pseudo R²: 
- Version A: 0.14
- Version B: 0.15

N: 
- Version A: 383
- Version B: 249

N: 
- Version A: 383
- Version B: 249
Using the coefficients estimated in the models in Table 16.4, point estimates of the willingness to pay for a change in the beach attributes can be calculated. These are the marginal rates of substitution between the beach attribute of interest and the price attribute. It is straightforward to show that for the linear utility index as described above, these implicit prices are given by:

\[
WTP = \frac{-b_{\text{attribute}}}{b_{\text{price}}} \tag{16.4}
\]

The coefficient \(b_{\text{price}}\) gives the marginal utility of income and is the coefficient of the cost attribute. Table 16.5 shows the estimated range of mean WTP values for questionnaire versions A and B. The ranges arise from the three different model variations tested. The nested logit model with the ASC and individual-specific variables provided the best fit, and the higher WTP results. Further details about these results can be found in Eftec (2002).

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Definition</th>
<th>Version A</th>
<th>Version B</th>
</tr>
</thead>
<tbody>
<tr>
<td>WATER</td>
<td>Value of decreasing the chance of getting a stomach upset by 1 in 100</td>
<td>1.10–2.00</td>
<td>—</td>
</tr>
<tr>
<td>(Version A)</td>
<td>(over the range: 0.5/100–7/100)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>WATER</td>
<td>Value of reducing ‘unsafe to swim’, bathing day by 1 per year (over the</td>
<td>—</td>
<td>0.90–1.10</td>
</tr>
<tr>
<td>(Version B)</td>
<td>range: 1–16 days)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>ADVISORY</td>
<td>Value of having an advisory note system in place, advising on poor water</td>
<td>7.20–13.70</td>
<td>5.60–12.10</td>
</tr>
<tr>
<td></td>
<td>quality days</td>
<td></td>
<td></td>
</tr>
<tr>
<td>LITTER</td>
<td>Value of avoiding the presence of some litter/dog mess</td>
<td>6.80–11.10</td>
<td>6.00–10.00</td>
</tr>
<tr>
<td>AMENITY</td>
<td>Value of improving the standard of amenities (toilets/showers) and safety</td>
<td>2.50–7.30</td>
<td>3.40–6.20</td>
</tr>
<tr>
<td></td>
<td>(life-guards and equipment) from average to good</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The estimated values in Table 16.5 indicate that, for example, respondents were willing to pay between £1.10 and £2.00 per year for a 1 in 100 reduction in the chance of contracting stomach illness from bathing water at a typical beach (over the range of risk reduction from 0.5 in 100 to 7 in 100). This value range is very similar to the £0.90 to £1.10 that respondents were prepared to pay to avoid one day of poor water quality. Given that we do
not know the health risk relationship between a 1 in 100 chance of suffering a stomach upset and an unsafe bathing day no further judgments about this outcome can be made. But, reassuringly, both versions A and B produce statistically similar estimates for all common variables.

Notably, the estimated benefits from improvements in water quality are lower than the mean WTP values estimated for either an advisory note system, elimination of beach litter or improved standard of amenities. This result is consistent with attitudinal findings reported previously, which suggested that water quality is not the most important beach attribute for the British public. The value of introducing an advisory note system commanded the highest values amongst all beach attributes in the experiment, ranging from £5.60 to £13.70. This result is also supported qualitatively by attitudinal survey responses that indicated a very positive view of this system. And, as these are household yearly values for all beaches, even these larger figures seem to be well within the bounds of reality.

Analysis of respondents' motivations show that a third of those prepared to pay for beach improvements were considering their own and their families' use of beaches. A further 34 per cent cited altruistic motives, 18 per cent were concerned about the welfare of future generations and 7 per cent were concerned about protecting marine wildlife. As expected, these answers reflect a mixture of use and non-use values.

As was noted above, the survey was designed with five versions specifically to test for a starting point bias (Table 16.3). Analysis of the three subsamples for Version A tested whether the value of a reduction of a 1 in 100 chance of suffering stomach upset differed significantly depending on the baseline risk level, that is, 7 per cent, 5 per cent or 2 per cent. For Version B the analysis tested whether the value of a reduction of one unsafe swimming day differed depending on the number of unsafe swimming days in the baseline, that is, 16 or 10. The results showed there was no strong systematic evidence of this type of starting point bias as none of the WTP estimates for subsamples was found to be statistically different at the 1 per cent level.

Finally, the willingness to pay values for water quality improvements estimated here are lower than those found in previous studies (Table 16.1). However, the current study measured marginal changes, as opposed to the discrete changes measured in previous studies. Furthermore, this study sampled the national population rather than just beach users, as in other studies. Beach users are expected to hold higher preferences for beach quality than the national population as a whole.

**Aggregation**

Calculation of the aggregate benefits of the revised Directive enables a comparison between the overall cost incurred in implementing the Directive
Is it worth revising the European Bathing Water Directive?

and the total benefit that is derived. On the basis of this cost–benefit comparison it is possible to judge whether the revisions are a worthwhile undertaking in England and Wales. The aggregate benefits are calculated using the most conservative WTP (lower bound) estimates from the NL model. Table 16.6 presents aggregate benefits as annual values as well as the present value of these benefits over a 25-year time horizon, using 3.5 per cent and 6 per cent discount rates.

Assuming that the risk of stomach upset from bathing at beaches in the 2001 bathing season was 4.3 per cent per swim (calculated using the methodology of WHO, 2001) then, were the Commission’s suggested mandatory standard of 200 intestinal enterococci at 95th percentile compliance adopted, this would lead to only a 2 per cent risk per swim. This corresponds to a 2.3 per cent reduction in risk per swim. Therefore, using a 6 per cent discount rate, the present value of this improvement is about £776 million over 25 years. This compares to a present value of £1.7 billion for the introduction of an advisory note system, £1.8 billion for the avoidance of dog mess and litter and £767 million for the improvement of amenities from average to good. These figures are highlighted in italic in Table 16.6.

Table 16.6 Aggregate benefits of beach improvements

<table>
<thead>
<tr>
<th>Attribute</th>
<th>£/household/yr</th>
<th>£ million/yr*</th>
<th>PV**, £ million (r = 3.5%)</th>
<th>PV**, £ million (r = 6%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Change in risk of suffering a stomach upset (% reduction in risk)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1%</td>
<td>1.10</td>
<td>26.4</td>
<td>435</td>
<td>337</td>
</tr>
<tr>
<td>2%</td>
<td>2.20</td>
<td>52.8</td>
<td>870</td>
<td>675</td>
</tr>
<tr>
<td>2.5%</td>
<td>2.53</td>
<td>60.72</td>
<td>1001</td>
<td>776</td>
</tr>
<tr>
<td>2.3%</td>
<td>2.75</td>
<td>66</td>
<td>1088</td>
<td>844</td>
</tr>
<tr>
<td>3%</td>
<td>3.30</td>
<td>79.2</td>
<td>1305</td>
<td>1012</td>
</tr>
<tr>
<td>4%</td>
<td>4.40</td>
<td>105.6</td>
<td>1740</td>
<td>1350</td>
</tr>
<tr>
<td>Change in number of safe bathing days (increase in safe days)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1</td>
<td>0.90</td>
<td>21.6</td>
<td>356</td>
<td>276</td>
</tr>
<tr>
<td>5</td>
<td>4.50</td>
<td>108</td>
<td>1780</td>
<td>1381</td>
</tr>
<tr>
<td>9</td>
<td>8.10</td>
<td>194.4</td>
<td>3204</td>
<td>2485</td>
</tr>
<tr>
<td>11</td>
<td>9.90</td>
<td>237.6</td>
<td>3916</td>
<td>3037</td>
</tr>
<tr>
<td>15</td>
<td>13.5</td>
<td>324</td>
<td>5340</td>
<td>4142</td>
</tr>
<tr>
<td>Advisory note system</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Avoiding the presence of some litter/dog mess</td>
<td>6.00</td>
<td>144</td>
<td>2373</td>
<td>1841</td>
</tr>
<tr>
<td>Improving the standard of amenities from average to good</td>
<td>2.50</td>
<td>60</td>
<td>989</td>
<td>767</td>
</tr>
</tbody>
</table>

Notes:
* 24 million households in England and Wales.
** In calculating the present values it is assumed that the benefits of improvements accrue from the end of the first year during which the improvements are introduced.
CONCLUSIONS

The proposed changes to the Bathing Water Directive are inter alia focused on improving the quality of bathing water, providing special areas designated for water sports and improving management across all beaches. The results of this study suggest that residents of England and Wales are willing to pay between £1.10 and £2.00 per household per year for a 1 per cent reduction in the risk of suffering a stomach upset from bathing in polluted waters. Interestingly, this range is similar to respondents’ WTP to avoid one day of poor water quality (£0.90–£1.10). For a 2.3 per cent reduction in the risk of suffering a stomach upset, which is the benefit associated with complying with revisions to the Bathing Water Directive, the overall benefit to England and Wales is estimated to be £60.7 million per annum.

Accepting the current limitations for both the cost and benefit figures, a comparison of the potential costs and benefits of improving bathing water quality alone generates significant negative net benefits (£2.5 billion costs versus £0.78 billion benefits). Initially, this would seem to present a pessimistic assessment of the water quality changes associated with revisions to the Directive: the costs of achieving the standards set by the Directive outweigh the benefits that accrue by a ratio greater than three to one. However, as noted earlier, the valuation of water quality improvements is undertaken on the basis of health risks associated with the Directive, and even then only in relation to gastrointestinal illness. If all benefits that may be associated with the water quality improvements are taken into account, such as reductions in other illnesses and ecological and aesthetic improvements, the outcome of a CBA assessment could become more favourable.

Furthermore, if water quality improvements are defined in terms of the number of safe swimming days, rather than probability reductions in gastrointestinal illness, then only a small change in the number of safe days is required in order for the cost–benefit ratio to roughly equal unity. Referring back to Table 16.6, an increase in the number of safe swimming days by nine days yields a total benefit of £194 million per year, which in present value terms over 25 years equates to £2.5 billion, at a 6 per cent discount rate. Difficulty in linking the probabilistic assessment of health risk to an estimate of the number of safe swimming days means it is not possible to assess the extent to which nine extra safe swimming days corresponds to the 2.3 per cent reduction in the risk of suffering a stomach upset associated in complying with revisions the Directive.

But importantly, the findings of this study also suggest that improving bathing water quality at British beaches is not the most valued change in beach attributes for the English and Welsh populations, as a result of revisions in the Bathing Water Directive: it ranks third together with
improving amenities and safety measures. The cleanliness of beaches and the level of information about water pollution provided at beaches are considered more valuable, and the expected benefits of improving either of these attributes are greater by an order of magnitude than the expected benefits of improving water quality and providing better amenities. In terms of the introduction of an advisory note system, respondents were willing to pay between £5.60 and £13.70 per household per year, which equates to a total benefit of approximately £134.4 million per annum. The greatest benefit was seen to be derived from improvements in beach cleanliness, with respondents willing to pay between £6.00 and £11.00 per household per year, equating to £144 million per year for England and Wales.

Hence, the results of this study provide a valuable insight into the potential benefits of the revised Directive. The estimated values of the various beach characteristics suggest the focus of the revisions should be on the management of the beaches in terms of providing more information to beach users on the state of bathing water, and in providing cleaner beaches, rather than solely focusing on improving bathing water quality.

NOTES

1. Valuation of Benefits to England and Wales of a Revised Bathing Water Quality Directive and other Beach Characteristics Using the Choice Experiment Methodology (Eftec, 2002), report for DEFRA. Bathing water is defined in the Directive as fresh or sea water in which bathing is explicitly authorized, or is not prohibited, and is traditionally practised by a large number of bathers. However, for the purpose of this study, only sea bathing waters were considered.

2. The risk of suffering gastrointestinal illness represents just one aspect of the health risk from bathing in coastal waters. Other risks to health, such as those relating to ear or throat infections, may also be reduced as a result of revisions to the Directive. However, these were not considered in our study due to insufficient epidemiological evidence.

3. While ‘starting point’ bias usually refers to the price attribute, in this instance it is applied to a change in risk.

4. These numbers are estimated by the formula: number of cases of stomach upsets per year = number of swims where head submerged per head of population × population × risk of suffering stomach upset. For swim/dips, there were a stated 460 instances where bathers’ heads were often submerged. The exposure rate is calculated by dividing this figure by the sample population (809), and the risk of suffering a stomach upset is an estimated 4.3 per cent chance per swim. Hence (460/809) × 52.9 million × 0.043 = 1.3 million. For all water sport activities, the corresponding calculation is: (1011/809) × 52.9 million × 0.043 = 2.84 million.

5. However, it is likely that the predicted total number of cases of stomach upset in England and Wales are overestimates. It is possible that frequent swimmers could develop some immunity to gastroenteritis. If this is the case then the total number of illnesses will be smaller, as repeat swims above a certain number would no longer increase the risk of being ill. However, since there is no epidemiological evidence to this effect, this issue is not incorporated into the analysis undertaken here.

6. Number of cases of stomach upsets per year = number of cases per person per year × population of England and Wales: (34/809) × 52.9 million = 2.2 million.
REFERENCES


17. Hedonic price analysis of road traffic noise nuisance

Brett Day, Ian Bateman and Iain Lake

INTRODUCTION

The market in which housing is traded, the property market, differs from that of many other differentiated goods in that it is fundamentally spatial in nature. The property market is not a global or even national market. Rather, many localized property markets may exist simultaneously, even within a single urban conurbation. Unlike other goods that are not constrained spatially, producers (i.e. property owners) cannot transport their products from one location to another. Indeed, spatial constraints in property markets ensure that, in the short run at least, the supply of properties to each market is extremely inelastic. Consequently, we would not expect market-clearing prices to equalize across property markets. Indeed, the equilibrium hedonic price schedule for any particular housing market will be unique to that market reflecting the specific conditions of supply and demand that exist at that locality (see Day, 2001 for a more detailed explanation).

Since property prices in two different markets may be determined by very different hedonic price functions, a primary concern for hedonic researchers is to ensure that data are drawn from a single property market. Using data from the city of Glasgow in Scotland this chapter seeks to identify property submarkets using statistical techniques. Separate hedonic functions are estimated for each submarket and statistical tests used to establish the uniqueness (or otherwise) of each function.

The chapter is organized as follows. The next section provides a brief theoretical overview of hedonic analysis. We describe the hedonic price function, implicit prices for housing attributes and the mechanism by which these prices are determined in the property market. In particular, attention is drawn to the fundamentally spatial nature of property markets. It is argued that property markets are likely to segment spatially and that identifying these market segments is fundamental in the quest for unbiased estimates of the implicit price of traffic noise avoidance. The third section details
Amenity and water quality

the data used in the analysis. GIS allows for the compilation of a rich and diverse set of covariates describing the characteristics of properties in a property market. Unfortunately, the quantity of information provided by the application of GIS complicates the estimation of the hedonic price function. In particular, multicollinearity is rife in the covariate data, with many variables measuring slightly different dimensions of the same basic characteristic. To overcome this difficulty, the fourth section describes the application of factor analysis to the covariate data. Factor analysis provides a way in which the multitude of variables available to the analyst can be concentrated into a smaller number of factors that identify the major dimensions of difference and similarity between properties.

The fifth section returns to the issue of market segmentation and illustrates the use of a statistical technique known as cluster analysis. This technique is used to gather observations into groups of properties displaying similar characteristics, not only in terms of their physical attributes but also in terms of their spatial location. Further statistical tests are applied to these groups of properties in order to identify market segments possessing independent hedonic price functions.

Finally, attention is paid to the estimation of the hedonic price function in each market segment, again acknowledging the spatial nature of the data. In particular, we test for the existence of spatial autocorrelation, that is, the hypothesis that the residuals of the hedonic price regression are correlated for properties located near to each other. This would be the case, for example, if properties in close proximity hold similar values for property characteristics omitted from the analysis. Unsurprisingly, the data show strong evidence of spatial autocorrelation. As a consequence, the final hedonic regressions are estimated using a general method of moments estimator that accounts for spatial autocorrelation.

HEDONIC PRICE THEORY

Housing is an example of a differentiated good. Such goods consist of a diversity of products that, while differing in a variety of characteristics, are so closely related in consumers’ minds that they are considered as being one commodity. Many other goods, including breakfast cereals, cars, computers and beach holidays also fit this description.

Market forces determine that different varieties of the product command different prices and that these prices depend on the individual products’ exact characteristics. For example, properties that have more bedrooms will tend to command a higher price in the market than properties that have fewer bedrooms. Furthermore, the set of prices in the market define
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a competitive equilibrium. That is, in general, the market will settle on a set of prices for the numerous varieties of the differentiated good that reconcile supply with demand and clear the market. This set of prices can be described by a hedonic price function:

\[ P = P(z) \] (17.1)

where \( P \) is the market price of a property and \( z = (z_1, z_2, \ldots, z_K) \) is a vector of values describing the quantities of \( K \) characteristics of a property’s structure, environs and location, what we shall term the property’s attributes. The partial derivative of the hedonic price function with respect to one of these attributes, say \( z_i \):

\[ p_z(z_i; z_{-i}) = \frac{\partial P(z)}{\partial z_i} \quad (i = 1 \text{ to } K) \] (17.2)

where \( z_{-i} \) is the vector of all other property attributes, is the implicit price function. This function describes the additional amount that must be paid by any household to move to a property with a marginally higher level of characteristic \( z_i \), other things being equal. In the research presented here, the primary objective is to identify the implicit price of road traffic noise.

The market in which housing is traded, the property market, differs from that of many other differentiated goods in that it is fundamentally spatial in nature. The property market is not a global or even national market, far from it. Rather, many localized property markets exist simultaneously. The market is said to be segmented.

In fact it is usual to think of the products in a property market as being the set of properties existing in a particular area. The consumers of these products are the households that wish to live in the area. They determine the level of demand in the market. The producers are the landlords that own the properties. They determine the level of supply of properties to the market.

Unlike other goods that are not constrained spatially, producers (i.e. landlords) cannot transport their products from one location to another. Indeed, spatial constraints in the property markets ensure that, in the short run at least, the supply of properties to each market is extremely inelastic. Consequently, we would not expect market-clearing prices to equalize across property markets.

The equilibrium hedonic price schedule settled on in a particular property market will reflect many factors on both the demand and supply sides of the market. For example, we would expect a property market in which
households are generally better off to be characterized by generally higher levels of willingness to pay for property attributes. In such a market, the implicit prices of property attributes such as ‘peace and quiet’ (i.e. the negative of a property’s exposure to traffic noise) will tend to command higher implicit prices than in an identical market in which households are less wealthy. Likewise, on the supply side, the availability of housing attributes will influence the equilibrium hedonic price schedule. Consider, for example, the price paid for waterfront properties in London and Stockholm. Whilst in both cities such properties command considerable premia, the relatively low availability of ‘Thameside’ properties in London means that they command highly inflated prices compared with those in Stockholm, a city built upon a series of islands.

As a general result, the equilibrium hedonic price schedule for any particular housing market will be unique to that market reflecting the specific conditions of supply and demand that exist at that locality. There is no theoretical reason to expect implicit prices from hedonic analyses of different property markets to return the same value. Indeed, given the heterogeneous supply and demand conditions in different property markets we would expect them to return different values.

A primary concern in hedonic analysis, therefore, must be to ensure that the data used to estimate a hedonic price function pertains to houses from a single property market. If this is not the case we risk seriously biasing our analysis. Estimates of implicit prices coming from such an analysis may bear little resemblance to the true implicit prices ruling in the individual property markets.

DATA DESCRIPTION

In empirical applications researchers estimate the hedonic price function of equation (17.1) by collecting data on the selling price of houses in a particular property market and regressing these on the attributes of those properties (i.e. the $z_j$). The data used in this study were collected from the south and north-west of the Scottish city of Glasgow. Using publicly available records, the addresses and prices of around 3500 properties sold in the study area during 1986 were collated.

As a first step, the location of each property was determined. This was achieved using a database provided by the Ordnance Survey (OS) that provides a unique grid reference for each postal address in the UK (OS ADDRESS-POINT®) (Martin et al., 1994). The study area and the locations of all the properties in the sample are illustrated in Figure 17.1.
The houses in a property market are extremely heterogeneous structures. They differ in terms of number of rooms, decor, age, plumbing, central heating, garden size and a myriad of other structural attributes. In the estimation of hedonic price functions researchers would hope to control for the influence of these attributes on property prices. However, as every good estate agent will eagerly inform, another fundamental determinant of property prices is ‘location, location, location!’ Indeed, a property’s market price will probably contain an element reflecting its location with respect to amenities such as town centres, shops, schools and parks. Further, it would not be altogether surprising to find that property prices reflect the socio-economic characteristics of the neighbourhood in which the property is located. Moreover, and fundamental to this research programme, we might

Figure 17.1  Location of properties in the sample
expect environmental characteristics of a property’s location (e.g. exposure to traffic noise) to be capitalized into the market price of the house.

One of the prime motivations of this research was to demonstrate the potential of GIS for compiling data for the estimation of hedonic price functions. Indeed, in this study the vast majority of data on property attributes came from desk-based interrogation of GIS databases. To structure our discussion, consider Table 17.1, which provides a categorization of property attributes.

**Table 17.1 Categories and examples of attributes of housing**

<table>
<thead>
<tr>
<th>Attribute Category</th>
<th>Examples of Attributes in this Category</th>
</tr>
</thead>
<tbody>
<tr>
<td>Structural</td>
<td>Number of rooms; presence of garage; size of garden; presence of central heating; etc.</td>
</tr>
<tr>
<td>Accessibility</td>
<td>Distance to: bus stop; town centre; school; shopping centre; etc.</td>
</tr>
<tr>
<td>Neighbourhood</td>
<td>Average age; race distribution; crime rate; quality of surrounding schools; etc</td>
</tr>
<tr>
<td>Environmental</td>
<td>Noise levels; air pollution levels; quality of views from the property; etc</td>
</tr>
</tbody>
</table>

**Structural Attributes**

The OS provides digital maps of the UK recording the locations of ground features, such as buildings, roads and fences.\(^2\) Given the grid reference of each property in the data set, it was possible to employ GIS to link each property sale observation to features on the ground. Indeed, attributes detailing each property’s ground area, size of garden and general shape were calculated in this way. Further, it was possible to identify the general type of each property from this data. Properties were classified as detached, semi-detached, four blocks, terraced, tenements, flats, subdivided houses or a catch-all category ‘others’.

In matter of fact, in this study the GIS data were supplemented through a brief visual inspection of properties included in the sample. This allowed identification of other structural attributes including, the age of the property, the material from which the property was constructed, the number of storeys and, for flats and tenements, the floor on which the particular property was located. Descriptions of the various structural attributes are provided in Table 17.2.

While the GIS greatly simplifies the collection of certain structural attributes for large hedonic property price data sets, it is noticeable that a
number of key structural attributes have not been collated. For example, the data do not provide details of the number of rooms of different functions in the property, nor does it indicate whether the property has a garage, central heating or double-glazing. Further, no information is available on the state of repair of property utilities such as plumbing and electricity or, for that matter, the quality of the internal decor. Since all of these features are likely to be major determinants of property price, their omission should rightly be considered a weakness of the present data set.

Table 17.2  Structural attributes collected for properties in the data set

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Floor Area</td>
<td>Calculated as the ground area of the property building on the digital map multiplied by the number of floors of the property determined through visual inspection (m²)</td>
</tr>
<tr>
<td>Garden Size</td>
<td>Calculated as the property’s total plot size minus the ground area of the building (m²)</td>
</tr>
<tr>
<td>Shape</td>
<td>Calculated as the ratio of floor perimeter to the square root of area (Blamire and Barnsley, 1996). The greater the ratio, the more complex the property shape</td>
</tr>
<tr>
<td>Storeys</td>
<td>Number of storeys</td>
</tr>
<tr>
<td>Property Type</td>
<td>Series of dummy variables indicating whether the property could be categorized as detached, semi-detached, four block, terrace, tenement, flat, subdivided house or ‘other’</td>
</tr>
<tr>
<td>Building Material</td>
<td>Dummy variable indicating whether the property was built of locally quarried stone</td>
</tr>
<tr>
<td>Age</td>
<td>Series of dummy variables indicating whether the property was built pre-1919, between 1919 and 1945 or after 1945</td>
</tr>
<tr>
<td>Number of Properties in Building Floor</td>
<td>For flats and tenements, a variable indicating the number of other properties contained within the same building as the target property</td>
</tr>
<tr>
<td></td>
<td>For flats and tenements, a series of dummy variables indicating whether the property is located in the basement, ground floor, first floor, second floor or on the third or higher floor</td>
</tr>
</tbody>
</table>

Accessibility Attributes

Since a property’s accessibility attributes are inherently spatial, GIS introduce incredible flexibility and precision into their estimation. For example, it is
possible to use GIS to calculate car travel times to important amenities that reflect the actual distance travelled on the road network taking account of road speeds along various road types. In the same way, walking distances from a property to local amenities can be calculated precisely using the network of pedestrian routes (Lake et al., 1998). Descriptions of the accessibility attributes used in the analysis are provided in Table 17.3.

Table 17.3  Accessibility attributes collected for properties in the data set

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Car City Centre</td>
<td>Travel time by car to Glasgow City centre (mins)</td>
</tr>
<tr>
<td>Walk Rail</td>
<td>Walking distance to nearest railway station (metres)</td>
</tr>
<tr>
<td>Walk Shop</td>
<td>Walking distance to nearest local shop (metres)</td>
</tr>
<tr>
<td>Walk School</td>
<td>Walking distance to nearest school (metres)</td>
</tr>
<tr>
<td>Walk Park</td>
<td>Walking distance to nearest municipal park (metres)</td>
</tr>
</tbody>
</table>

Neighbourhood Attributes

The census provides a myriad of information on the types of properties and socio-economic characteristics of the population living in a certain area. As a result census data are ideal for constructing indicators of the attributes of the neighbourhood in which a property is located. Again GIS is a fast and efficient means of matching properties to census data at different spatial scales.

At the most specific level, neighbourhood attributes can be constructed from the smallest unit of the census, the Output Area (OA). Such attributes will reflect the direct neighbourhood of the property. More broadly, census data can be averaged over larger spatial scales such as a postcode district to provide indicators of the attributes of the wider neighbourhood. Descriptions of some of the neighbourhood attributes collected for the hedonic analysis are provided in Table 17.4.

Notice that the OA characteristics include a large number of variables indicating the structural attributes of properties (e.g. percentage of properties with one, two, three, four, five, six, seven or more rooms, percentage of properties with no central heating, percentage of properties without exclusive access to a WC, bath or shower). Since on average in this study an OA consisted of 56 households, these neighbourhood attributes provide reasonable proxies for some of the structural attributes for which data were not available.
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Table 17.4 Neighbourhood attributes collected for properties in the data set

<table>
<thead>
<tr>
<th>Spatial Scale</th>
<th>Attribute Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Output Area</td>
<td>% of double income families with no children; % of elderly people living alone; % of ethnic people; % of Irish people; % of multiple earning households; % of households with no access to a car; % of households with no central heating; % of residents not owning their home; % of residents with no exclusive use of WC, bath or shower; % of lone parent households; % of households with &gt; one person per room; % of children in households with no earners; % of houses with one room; % of houses with two rooms; % of houses with three rooms; % of houses with four rooms; % of houses with five rooms; % of houses with six rooms; % of houses with seven or more rooms; % of two-car households; % of people &lt; five; % unemployment; % of vacant and second homes; % of residents 16–34 with children &lt; 16; % of residents aged 35–54 with children &lt; 16</td>
</tr>
<tr>
<td>Postcode District</td>
<td>% of houses with more than seven rooms; % of residents born in new commonwealth; % of two-car households; Number of children per child care worker; % of children aged under five; number of children in institutions per 1000 children; % of workers in the construction sector; % of households with &gt; one person per room; % of double income families with no children; % of people working in the ‘free’ economy; % of full-time workers working &gt; 40 hrs per week; % of people with a degree or higher degree; % of people in occupations with gross wages &gt; £23 705; % of homeless people; Ratio of people with limiting long-term illness to no. of health care workers; % of people working in the information economy; % of Irish people; % of houses lacking basic amenities; % of households with working head in social classes IV or V; % of lone pensioner households; % of workers in the manufacturing sector; % of households with working head in social classes I to III; % of residents with a different address one year before census; % of multiple earning households; % of recently moving families; % of recently moving pensioners; % of households</td>
</tr>
</tbody>
</table>
Environmental Attributes

In this study, the primary concern was the effect of road traffic noise on property prices. Unfortunately, though hardly surprisingly, specific data on the traffic noise exposure of each property were not immediately available. Once again, however, GIS provides a powerful tool for deriving estimates of this environmental attribute.

As a first step, GIS techniques were used to extrapolate point measures of traffic volumes at monitored sites around Glasgow City to all roads in the study area. On the basis of these traffic volumes, the level of noise being emitted from each road was calculated according to formulae specified in Department of Transport (1988), which account for the percentage of heavy vehicles, the speed of the vehicles, the gradient of the road and the road surface. Subsequently the noise exposure at each property was calculated using GIS techniques that accounted for the horizontal and vertical distance between the property and the road, the ground surface separating property from road, sound reflections off neighbouring buildings and the presence of other major roads in the vicinity. Clearly, in the absence of specific data, GIS provide the only means by which reasonably accurate measures of an environmental attribute like exposure to traffic noise can be estimated for large hedonic property price data sets. Further details of the model used to estimate traffic noise are available in Bateman et al. (2001).
A somewhat simpler procedure was employed to determine each property’s exposure to noise pollution from aircraft. A digital map outlining noise contours surrounding Glasgow City Airport was linked to the digital map of property locations. A simple GIS enquiry could then be used to calculate the noise exposure level at each property.

A second set of environmental attributes were collated using GIS – those describing the quality of views from a property. Using digital maps indicating land use and accounting for the topography and location of the property, GIS allowed the researchers to estimate the quantity of land of different types visible from the front and back of the property. The environmental attributes used in the analysis are described in Table 17.5.

Table 17.5   Environmental attributes collected for properties in the data set

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Traffic Noise</td>
<td>Exposure to traffic noise at front of property (dBs)</td>
</tr>
<tr>
<td>Aircraft Noise</td>
<td>Exposure to aircraft noise (dBs)</td>
</tr>
<tr>
<td>Front View</td>
<td>General measure of open land visible from the front of the property (m²)</td>
</tr>
<tr>
<td>Front Park View</td>
<td>Area of park visible from the front of the property (aggregated with an inverse distance-decay)</td>
</tr>
<tr>
<td>Front Industrial View</td>
<td>Area of industrial land use visible from the front of the property (aggregated with an inverse distance-decay)</td>
</tr>
<tr>
<td>Front Railway View</td>
<td>Area of railway line visible from the front of the property (aggregated with an inverse distance-decay)</td>
</tr>
<tr>
<td>Front Water View</td>
<td>Area of water visible from the front of the property (aggregated with an inverse distance decay)</td>
</tr>
<tr>
<td>Back View</td>
<td>General measure of open land visible from the back of the property (m²)</td>
</tr>
<tr>
<td>Back Park View</td>
<td>Area of park visible from the back of the property (aggregated with an inverse distance-decay)</td>
</tr>
<tr>
<td>Back Industrial View</td>
<td>Area of industrial land use visible from the back of the property (aggregated with an inverse distance-decay)</td>
</tr>
<tr>
<td>Back Railway View</td>
<td>Area of railway line visible from the back of the property (aggregated with an inverse distance-decay)</td>
</tr>
<tr>
<td>Back Water View</td>
<td>Area of water visible from the back of the property (aggregated with an inverse distance-decay)</td>
</tr>
</tbody>
</table>
Hedonic price studies must account for many property attributes. It is a matter of some concern to analysts that omitting important attributes from the hedonic price regression may bias estimates of implicit prices. For example, Harrison and Rubinfeld (1978), in their hedonic pricing study of air quality, computed regressions with and without accessibility attributes. Their results indicated that the implicit price of air pollution changed significantly when accessibility variables were deleted, which implied that without accessibility the parameter on air pollution reflected both disadvantages of greater pollution and advantages of greater accessibility. As illustrated here, the ability of GIS to calculate large quantities of spatial data rapidly and accurately is a considerable technical advance in the compilation of property attributes for hedonic analysis.

FACTOR ANALYSIS OF NEIGHBOURHOOD ATTRIBUTES

As illustrated by the myriad of attributes listed in Table 17.4, the use of GIS to interrogate census data provides a rich source of information on neighbourhood attributes. Indeed, here the problem for hedonic analysis is not one of lack of data on attributes but one of over-abundance. Not surprisingly, many of the attributes listed in Table 17.4 are highly collinear. For example, at the output area level, the percentage of households not owning cars exhibits high positive correlation with the percentage of households that do not own their own property (correlation coefficient of 0.70) and high negative correlation with the percentage of households that own two cars (correlation coefficient of −0.76).

Whilst each of these neighbourhood attributes might have a bearing on property prices, the presence of such collinearity creates a problem for researchers. As is well known, parameters estimated on highly collinear regressors are difficult to interpret. Parameter estimates may have implausible magnitude or, in the worst case, the wrong sign. Interpretation is further confounded by the fact that individual parameters may exhibit high standard errors and consequently low significance levels. Moreover, it is not clear that each of these neighbourhood attributes will be independently capitalized into the property market. More likely, households in a market will consider more general indications of the neighbourhood of a property, the wealth of the area, its ethnic makeup, the stage of life of its inhabitants etc.

As a result, we propose condensing the excess of neighbourhood attributes into a more manageable set of indices. Each index picks out a major dimension of difference or similarity between property neighbourhoods. For example one index might indicate the wealth of a neighbourhood, effectively
combining the myriad attributes that are indicators of wealth/poverty into one dimension. Subsequently, property neighbourhoods can be scored along each dimension. In our example, poor neighbourhoods would generate low scores on the wealth dimension, whilst affluent neighbourhoods would generate high scores. The procedure by which dimensions are identified and property neighbourhoods are scored along these dimensions is known as factor analysis.

We do not intend presenting the intricacies of factor analysis here (for a highly accessible text on the subject see Lindeman et al., 1980). In essence, the procedure seeks to identify major dimensions of association between variables (in our case the attributes of neighbourhoods) such that a smaller set of variables can be defined that approximate the variation shown in the original data.

Since we are interested in patterns of association, it is not surprising that the first step in a factor analysis is to calculate the correlation matrix of the $M$ variables under study. Each row (or column) of this matrix can be thought of as representing a point in $M$-dimensional space. We use $M$ axes to locate the points in this space, where each axis represents the degree of correlation with one of the $M$ variables and ranges from –1 to 1. Thus the position of the $n$th point indicates the nature of the correlation between attribute $m$ and all the other attributes.

If there were no correlation between the attributes, each of the $M$ points would be located on its own axis. Alternatively, when the attributes are correlated, as is the case here, the rows of the correlation matrix form a cloud of points in the space $(-1, 1)^M$. Two attributes showing strong positive correlation will have points located close to each other in this $M$-space. Likewise, an attribute showing strong negative correlation with these attributes will have a point located near to the mirror image of their points on the opposite side of the origin. Thus, attributes that measure slightly different aspects of one underlying dimension will have points that tend to align themselves along an axis running through the origin.

The first step in a factor analysis is to define an alternative set of axes through this space that capture these patterns of alignment. This is achieved by decomposing the correlation matrix into its eigenvalues and associated eigenvectors. As is well known, the $M$ eigenvectors represent just such a set of alternative orthogonal axes.

Let us consider the eigenvector with the highest associated eigenvalue. It transpires that of all the possible axes that could be drawn through the space $(-1, 1)^M$, the axis defined by the first eigenvector picks out the dimension capturing the most variability in the location of the points. That is, of all possible axes, the first eigenvector distinguishes the most significant alignment of points.
To illustrate, in the data on neighbourhood attributes we know that many of the attributes measure slightly different dimensions of the wealth of the inhabitants of that area. For example, for each neighbourhood we have details of the levels of unemployment, the levels of home ownership, the levels of car ownership, and the percentage of households with earnings above the national average. In general, relatively poor neighbourhoods will have high unemployment, but low levels of high earners and similarly low levels of car and home ownership. Conversely, relatively affluent neighbourhoods will contain many high earners coupled with high levels of car and home ownership, but low unemployment. Indeed, if we could plot the correlations in these attribute levels on a four-dimensional graph it would not be surprising to find that they lie approximately along a straight line passing through the origin. This axis is the first eigenvector. It summarizes the association between the four attributes into one dimension and this dimension can be interpreted as measuring the wealth of the neighbourhood.

Of course it is unlikely that this one dimension will account for all of the variability in the data. Indeed, the eigenvector with the second highest eigenvalue defines a second axis, orthogonal to the first, best approximating the remaining variation in the points. In the same manner, eigenvectors with successively smaller eigenvalues can be used to better and better approximate the location of the $M$ points. The eigenvectors define the ‘factors’ of factor analysis.

The 3544 properties in this study came from 1027 different output areas (OAs) and for each OA our data set contained details of some 25 attributes. The properties were further grouped into 38 postcode districts (PDs) for which information on 45 attributes was available. Tables 17.6 and 17.7 detail the first eight factors for the OA and PD neighbourhood attributes respectively. The second column in these tables provides the eigenvalue of each factor. The third column indicates the percentage of the variation in the location of the $M$ points explained exclusively by that factor. If the attributes were not correlated then each factor would correspond to an original axis and explain $1/M$ of the variation. If the attributes were all perfectly correlated then they would all be aligned along one axis and this axis would explain 100 per cent of the variation. The fourth column provides the cumulative sum of this explained variation.

For both sets of attributes the first factor alone explains around 40 per cent of the variation. This indicates that at both neighbourhood scales many of the attributes are highly correlated (positively or negatively) with a single underlying factor. We shall attempt to give an interpretation as to what these factors describe shortly. Notice that successive factors explain progressively less of the remaining variation. Here we arbitrarily select a cut off point and declare that factors explaining greater than 5 per cent of the
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One of the arts of factor analysis is the interpretation of factors. Interpretation of factors is the process of describing the underlying dimension of similarity or difference between the neighbourhood attributes captured by a factor. To explain how this is achieved, observe that if the axis defined by a particular factor is closely aligned with an original attribute axis, then that attribute is important in determining the factor. Conversely if the factor axis is orthogonal to an original axis, the attribute described variation in the data should be retained in the analysis. As such we select six factors at the OA neighbourhood scale that collectively describe some 88 per cent of the variation present in the original 25 variables. Similarly we select four factors at the PD scale that collectively describe 70 per cent of the variation in the original 45 variables.

Table 17.6 Variation explained by the first ten factors of the Output Area neighbourhood attributes

<table>
<thead>
<tr>
<th>Factor</th>
<th>Eigenvalue</th>
<th>Variation Explained by Factor</th>
<th>Cumulative Explained Variation</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>6.73</td>
<td>0.38</td>
<td>0.38</td>
</tr>
<tr>
<td>2</td>
<td>2.92</td>
<td>0.17</td>
<td>0.55</td>
</tr>
<tr>
<td>3</td>
<td>2.03</td>
<td>0.12</td>
<td>0.67</td>
</tr>
<tr>
<td>4</td>
<td>1.55</td>
<td>0.09</td>
<td>0.76</td>
</tr>
<tr>
<td>5</td>
<td>1.14</td>
<td>0.07</td>
<td>0.82</td>
</tr>
<tr>
<td>6</td>
<td>1.00</td>
<td>0.06</td>
<td>0.88</td>
</tr>
<tr>
<td>7</td>
<td>0.74</td>
<td>0.04</td>
<td>0.92</td>
</tr>
<tr>
<td>8</td>
<td>0.65</td>
<td>0.04</td>
<td>0.96</td>
</tr>
</tbody>
</table>

Table 17.7 Variation explained by the first ten factors of the Postcode District neighbourhood attributes

<table>
<thead>
<tr>
<th>Factor</th>
<th>Eigenvalue</th>
<th>Variation Explained by Factor</th>
<th>Cumulative Explained Variation</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>18.19</td>
<td>0.40</td>
<td>0.40</td>
</tr>
<tr>
<td>2</td>
<td>6.71</td>
<td>0.14</td>
<td>0.55</td>
</tr>
<tr>
<td>3</td>
<td>3.77</td>
<td>0.08</td>
<td>0.64</td>
</tr>
<tr>
<td>4</td>
<td>3.05</td>
<td>0.06</td>
<td>0.70</td>
</tr>
<tr>
<td>5</td>
<td>2.25</td>
<td>0.04</td>
<td>0.75</td>
</tr>
<tr>
<td>6</td>
<td>1.59</td>
<td>0.03</td>
<td>0.79</td>
</tr>
<tr>
<td>7</td>
<td>1.42</td>
<td>0.03</td>
<td>0.82</td>
</tr>
<tr>
<td>8</td>
<td>1.30</td>
<td>0.03</td>
<td>0.85</td>
</tr>
</tbody>
</table>
by that axis plays no part in determining that factor. The degree to which individual attributes contribute to a factor is measured by the factors loadings. A large positive loading indicates that high values of the original attribute are associated with high values of the factor. Similarly a large negative loading indicates that high values of the original attribute are associated with low values of the factor.

Tables 17.8 and 17.9 list the attributes associated with each of the identified factors and provide interpretations of the dimension described by these factors.

The final step in a factor analysis is to define a score for each neighbourhood for each factor. Using the factor loadings a regression-like equation is calculated, the parameters of which indicate how greatly each attribute contributes to each factor. Given the attributes of each neighbourhood, the equation can be used to determine how highly a neighbourhood scores on each factor. In effect, neighbourhoods that exhibit high values for attributes that load positively on a factor receive high scores for that factor whilst neighbourhoods that exhibit high values for attributes that load negatively on that factor receive low scores.

The factors can be used as proxies for the original attributes in regression analysis. As has been demonstrated the factors capture a good proportion of the variation shown in the original neighbourhood attributes. Moreover, the nature of their construction ensures that the factor scores are orthogonal overcoming the problem of collinearity in the original set of attributes.

MARKET SEGMENTATION

Most hedonic analyses make the implicit assumption that an urban area represents a single property market. Data are collected from the whole urban area and a single hedonic price function is estimated to describe the equilibrium prices. Maintaining this assumption we estimate a hedonic price function for the 3544 observations in the Glasgow data set. In the absence of an economic model indicating the functional form of the hedonic price function we adopt the usual assumption and regress the logarithm of the property sale price on a linear combination of the regressors described in the third and fourth sections of this chapter. The estimated model takes the traditional form:

\[ y = X\beta + \epsilon \]  

(17.3)

where \( y \) is an \([N \times 1]\) vector of observations on the dependent variable, \( X \) is an \([N \times K]\) matrix of explanatory variables, \( \beta \) is a corresponding \([K \times 1]\)
### Table 17.8 Interpretation of Output Area factors

<table>
<thead>
<tr>
<th>Factor</th>
<th>Attributes with High Positive Loading</th>
<th>Attributes with High Negative Loading</th>
<th>Interpretation</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Five- and six-room properties; households with two cars; multiple earning households; households with children</td>
<td>Two- and three-room properties; no central heating; households without a car; non-home-owning households, unemployment</td>
<td>This factor loads positively on general indicators of affluence and negatively on general indicators of poverty. It is interpreted as indicating the general wealth of the neighbourhood</td>
</tr>
<tr>
<td>2</td>
<td>Ethnic minorities; households with young children; unemployment; households with &gt; one person per room</td>
<td>Elderly people living alone</td>
<td>This factor loads heavily on the presence of ethnic minorities and picks out attributes that reflect characteristics of such communities. It is interpreted as indicating the ethnicity of the neighbourhood</td>
</tr>
<tr>
<td>3</td>
<td>One-room properties; no exclusive use of WC, bath or shower; unemployment; non-home-owning households</td>
<td>Multiple earning households; households with children</td>
<td>This factor loads very heavily on one attribute; one-bedroom properties. Other attributes indicate relative poverty and a dearth of families. The factor appears to pick out neighbourhoods characterized by bedsits possibly inhabited by migrant workers. We describe this as the bedsit factor</td>
</tr>
<tr>
<td>4</td>
<td>Two-room properties</td>
<td>Four- or five-room properties</td>
<td>This factor is more difficult to interpret. Since it loads very heavily on two-room properties it would seem to indicate neighbourhoods with small properties probably flats or tenements. We describe this as the tenement factor</td>
</tr>
<tr>
<td>5</td>
<td>Dual income households with no children; multiple earning households</td>
<td>Non-home-owning households; unemployment; households without a car; households with children</td>
<td>This factor appears to indicate the presence of relatively affluent, home-owning households, characterized by having two incomes and no children. We describe this as the DINKY (dual income no kids) factor</td>
</tr>
<tr>
<td>6</td>
<td>Six- and seven-room properties; multiple earning households; households with children</td>
<td>Two- and three-room properties; properties without central heating; households without a car; non home-owning households</td>
<td>This factor loads heavily on large properties, containing affluent households with children. We describe this as the affluent suburbia factor</td>
</tr>
</tbody>
</table>
Table 17.9 Interpretation of Postcode District factors

<table>
<thead>
<tr>
<th>Factor</th>
<th>Attributes with High Positive Loading</th>
<th>Attributes with High Negative Loading</th>
<th>Interpretation</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Unemployment; youth unemployment; middle-aged unemployment; children with unemployed parents; households with no car; non-home-owning households; single parents; lower social classes</td>
<td>Multiple earning households; dual income households with no children; working in informational economy</td>
<td>This factor loads heavily on attributes indicating the economic well-being of the inhabitants of a neighbourhood. In particular it picks out differences in the levels of employment and we describe the factor as the unemployment factor</td>
</tr>
<tr>
<td>2</td>
<td>Recently moved to neighbourhood; degree level education; workers in service sector; higher social classes</td>
<td>Households with children; workers in manufacturing or construction sectors; workers in the 'free' economy</td>
<td>This factor picks out attributes describing neighbourhoods characterized by transient inhabitants that are educated, reasonably affluent service sector workers with no children. We describe this factor as the young upwardly-mobile factor</td>
</tr>
<tr>
<td>3</td>
<td>Ethnic minorities; households with young children; youth unemployment; households with &gt; one person per room</td>
<td>Pensioners living alone; households with working parents and young children</td>
<td>This factor loads heavily on the presence of ethnic minorities and picks out attributes that reflect characteristics of such communities. We describe this factor as the ethnicity factor</td>
</tr>
<tr>
<td>4</td>
<td>High social class; workers in non-manual jobs; workers in service sector or information economy; full-time employed working &gt; 40 hrs per week; large properties; degree level education, households with two cars</td>
<td>Lower social class; workers in manufacturing or construction sectors; 17-year-old school leavers; households with no car</td>
<td>This factor loads positively on general indicators of affluence and negatively on general indicators of poverty. We describe this as the wealth factor</td>
</tr>
</tbody>
</table>
vector of unknown parameters and $\varepsilon$ is an $[N \times 1]$ vector of random error terms with expected value 0 and variance-covariance matrix $\sigma^2 I$.

Model (17.3) was estimated using ordinary least squares regression and the results are reported in the second column of Table 17.10. For the sake of brevity we do not provide a full interpretation of the results of this regression here. However, we note that the structural characteristics of the property are highly significant and influence price in a predictable way; larger properties with bigger gardens fetch higher prices. Detached houses fetch higher prices than semi-detached, semi-detached than terraced, terraced than subdivided houses, subdivided houses than four blocks, four blocks than flats and flats than tenements. Older properties fetch higher prices, as do flats on the first floor as opposed to ground floor or basement. The neighbourhood factors also perform well. At the OA neighbourhood scale, the wealthier a neighbourhood and the higher it scores on the DINKY and suburbia factors, the higher the price of property. Conversely, the more a neighbourhood is characterized by tenements and inhabitants from ethnic minorities, the lower the price of property. At the PD scale, the wealth of the neighbourhood increases the price of a property whilst the level of unemployment in the area deflates property prices.

The accessibility variables are not so convincing. Only the variable measuring the distance to walk to the nearest shop is significant. Unfortunately, this parameter has an unexpected sign suggesting that property prices increase with increasing distance from a shop. Likewise, the variables describing the view from a property appear to have little influence on its selling price. Reassuringly, property prices increase with increases in the quantity of open land visible from the front of the property but this is only significant at a 10 per cent level of significance. Finally, and most importantly in this study, the parameter on exposure to traffic noise is negative and highly significant ($t$-statistic $= -3.293$). Given the functional form, the parameter on traffic noise can be interpreted as the percentage change in the selling price of a house brought about by a unit change in traffic noise. In this case then, the analysis suggests that the price of a property will fall by 0.24 per cent for each decibel increase in traffic noise.4

As argued above, however, this figure may be misleading if the urban area (in this case Glasgow City) does not constitute a single property market. A number of researchers have investigated the existence of submarkets in urban areas (e.g. Straszheim, 1973; Schnare and Struyk, 1976; Ball and Kirwan, 1977; Goodman, 1978; Sonstelie and Portney, 1980; Michaels and Smith, 1990; Allen et al., 1995). These studies have applied different rules by which properties in an urban area are allotted to a particular submarket. Criteria include, locational or political boundaries, characteristics of households

---

*p* is the price of the property, $z_{i}$ is a vector of structural characteristics of the property (e.g. area, number of bedrooms), $f_{o}$ is a vector of factors measuring the location of the property, and $f_{n}$ is a vector of factors measuring the neighbourhood characteristics. The parameter $\beta_{s}$ is the coefficient of the structural characteristics, $\beta_{o}$ is the coefficient of the location factors, and $\beta_{n}$ is the coefficient of the neighbourhood factors. The error term $\varepsilon_{i}$ is assumed to be normally distributed with expected value 0 and variance-covariance matrix $\sigma^2 I$.
### Table 17.10  Hedonic price functions for property markets in Glasgow (OLS)

<table>
<thead>
<tr>
<th>Independent Variable</th>
<th>Full Sample</th>
<th>Submarkets</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>One</td>
<td>Two</td>
</tr>
<tr>
<td>Intercept</td>
<td>9.784 (0.088)</td>
<td>9.646 (0.178)</td>
</tr>
<tr>
<td>Log Floor Area</td>
<td>0.135 (0.012)</td>
<td>0.089 (0.205)</td>
</tr>
<tr>
<td>Garden Size</td>
<td>0.183 (0.023)</td>
<td>0.044 (0.133)</td>
</tr>
<tr>
<td>Shape</td>
<td>0.013 (0.002)</td>
<td>0.008 (0.004)</td>
</tr>
<tr>
<td>Storeys</td>
<td>–0.107 (0.026)</td>
<td>–</td>
</tr>
<tr>
<td>Detached</td>
<td>B (0.060)</td>
<td>–</td>
</tr>
<tr>
<td>Semi-Detached</td>
<td>–0.057 (0.030)</td>
<td>–</td>
</tr>
<tr>
<td>Terraced</td>
<td>–0.091 (0.033)</td>
<td>–</td>
</tr>
<tr>
<td>Subdivided House</td>
<td>–0.431 (0.060)</td>
<td>–</td>
</tr>
<tr>
<td>Four Block</td>
<td>–0.470 (0.050)</td>
<td>–</td>
</tr>
<tr>
<td>Flat</td>
<td>–0.549 (0.050)</td>
<td>0.178 (0.072)</td>
</tr>
<tr>
<td>Tenement</td>
<td>–0.575 (0.049)</td>
<td>B (0.093)</td>
</tr>
<tr>
<td>Other</td>
<td>–0.473 (0.067)</td>
<td>0.145 (0.038)</td>
</tr>
<tr>
<td>Building Material</td>
<td>0.040 (0.024)</td>
<td>0.038 (0.057)</td>
</tr>
<tr>
<td>Age (pre-1919)</td>
<td>0.062 (0.034)</td>
<td>0.084 (0.076)</td>
</tr>
<tr>
<td>Age (1919–45)</td>
<td>0.065 (0.031)</td>
<td>–0.064 (0.085)</td>
</tr>
<tr>
<td>No. of Properties in</td>
<td>–0.006 (0.001)</td>
<td>0.00004 (0.004)</td>
</tr>
<tr>
<td>Building</td>
<td>(0.001)</td>
<td>(0.004)</td>
</tr>
</tbody>
</table>
### Hedonic price analysis of road traffic noise nuisance

<table>
<thead>
<tr>
<th>Independent Variable</th>
<th>Full Sample</th>
<th>Submarkets</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>One</td>
<td>Two</td>
</tr>
<tr>
<td>Basement Flat</td>
<td>–0.126</td>
<td>0.050</td>
</tr>
<tr>
<td></td>
<td>(0.037)^a</td>
<td>(0.2)</td>
</tr>
<tr>
<td>Ground Floor flat</td>
<td>B</td>
<td>B</td>
</tr>
<tr>
<td>1st Floor Flat</td>
<td>0.038</td>
<td>0.036</td>
</tr>
<tr>
<td></td>
<td>(0.011)^a</td>
<td>(0.021)^c</td>
</tr>
<tr>
<td>2nd Floor Flat</td>
<td>0.013</td>
<td>–0.011</td>
</tr>
<tr>
<td></td>
<td>(0.012)</td>
<td>(0.021)</td>
</tr>
<tr>
<td>3rd Floor Flat</td>
<td>–0.006</td>
<td>–0.033</td>
</tr>
<tr>
<td></td>
<td>(0.012)</td>
<td>(0.022)</td>
</tr>
<tr>
<td>OA Wealth Factor</td>
<td>0.136</td>
<td>0.166</td>
</tr>
<tr>
<td></td>
<td>(0.007)^a</td>
<td>(0.025)^a</td>
</tr>
<tr>
<td>OA Ethnicity Factor</td>
<td>–0.056</td>
<td>–0.058</td>
</tr>
<tr>
<td></td>
<td>(0.005)^a</td>
<td>(0.014)^a</td>
</tr>
<tr>
<td>OA Bedsit Factor</td>
<td>0.005</td>
<td>0.020</td>
</tr>
<tr>
<td></td>
<td>(0.005)</td>
<td>(0.016)</td>
</tr>
<tr>
<td>OA Tenement Factor</td>
<td>–0.060</td>
<td>–0.080</td>
</tr>
<tr>
<td></td>
<td>(0.004)^a</td>
<td>(0.008)^a</td>
</tr>
<tr>
<td>OA Suburbia Factor</td>
<td>0.048</td>
<td>0.052</td>
</tr>
<tr>
<td></td>
<td>(0.004)^a</td>
<td>(0.009)^a</td>
</tr>
<tr>
<td>PC Unemployment Factor</td>
<td>–0.033</td>
<td>–0.040</td>
</tr>
<tr>
<td></td>
<td>(0.006)^a</td>
<td>(0.016)^b</td>
</tr>
<tr>
<td>PC YUPPY Factor</td>
<td>0.053</td>
<td>0.023</td>
</tr>
<tr>
<td></td>
<td>(0.006)^a</td>
<td>(0.012)^c</td>
</tr>
<tr>
<td>PC Ethnicity Factor</td>
<td>0.004</td>
<td>–0.033</td>
</tr>
<tr>
<td></td>
<td>(0.006)</td>
<td>(0.02)^c</td>
</tr>
<tr>
<td>PC Wealth Factor</td>
<td>0.076</td>
<td>0.079</td>
</tr>
<tr>
<td></td>
<td>(0.005)^a</td>
<td>(0.016)^a</td>
</tr>
<tr>
<td>Walk Rail</td>
<td>0.016</td>
<td>0.0190</td>
</tr>
<tr>
<td></td>
<td>(0.013)</td>
<td>(0.025)</td>
</tr>
<tr>
<td>Walk Park</td>
<td>–0.007</td>
<td>–0.075</td>
</tr>
<tr>
<td></td>
<td>(0.028)</td>
<td>(0.042)^f</td>
</tr>
<tr>
<td>Walk Shop</td>
<td>0.076</td>
<td>0.092</td>
</tr>
<tr>
<td></td>
<td>(0.028)^a</td>
<td>(0.076)</td>
</tr>
<tr>
<td>Walk School</td>
<td>0.021</td>
<td>0.091</td>
</tr>
<tr>
<td></td>
<td>(0.018)</td>
<td>(0.05)^c</td>
</tr>
</tbody>
</table>
### Table 17.10 (continued)

<table>
<thead>
<tr>
<th>Independent Variable</th>
<th>Full Sample</th>
<th>Submarkets</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>One</td>
<td>Two</td>
</tr>
<tr>
<td>Car City Centre</td>
<td>0.0001</td>
<td>-0.015</td>
</tr>
<tr>
<td></td>
<td>(0.003)</td>
<td>(0.007)b</td>
</tr>
<tr>
<td>Front View</td>
<td>0.009</td>
<td>0.011</td>
</tr>
<tr>
<td></td>
<td>(0.005)c</td>
<td>(0.01)</td>
</tr>
<tr>
<td>Back View</td>
<td>-0.007</td>
<td>-0.005</td>
</tr>
<tr>
<td></td>
<td>(0.006)</td>
<td>(0.01)</td>
</tr>
<tr>
<td>Front Park View</td>
<td>-0.028</td>
<td>1.058</td>
</tr>
<tr>
<td></td>
<td>(0.103)</td>
<td>(0.448)b</td>
</tr>
<tr>
<td>Back Park View</td>
<td>0.091</td>
<td>0.440</td>
</tr>
<tr>
<td></td>
<td>(0.022)</td>
<td>(0.471)</td>
</tr>
<tr>
<td>Front Industrial View</td>
<td>-0.123</td>
<td>0.218</td>
</tr>
<tr>
<td></td>
<td>(0.150)</td>
<td>(0.330)</td>
</tr>
<tr>
<td>Back Industrial View</td>
<td>0.210</td>
<td>0.203</td>
</tr>
<tr>
<td></td>
<td>(0.136)</td>
<td>(0.320)</td>
</tr>
<tr>
<td>Front Railway View</td>
<td>0.630</td>
<td>1.326</td>
</tr>
<tr>
<td></td>
<td>(0.688)</td>
<td>(1.042)</td>
</tr>
<tr>
<td>Back Railway View</td>
<td>2.530</td>
<td>0.813</td>
</tr>
<tr>
<td></td>
<td>(1.041)b</td>
<td>(2.147)</td>
</tr>
<tr>
<td>Front Water View</td>
<td>-0.045</td>
<td>3.972</td>
</tr>
<tr>
<td></td>
<td>(1.030)</td>
<td>(2.221)c</td>
</tr>
<tr>
<td></td>
<td>(1.053)a</td>
<td>(2.317)b</td>
</tr>
<tr>
<td>Aircraft Noise</td>
<td>-0.0009</td>
<td>1.779</td>
</tr>
<tr>
<td></td>
<td>(0.001)</td>
<td>(2.317)c</td>
</tr>
<tr>
<td>Traffic Noise</td>
<td>-0.0024</td>
<td>-0.0023</td>
</tr>
<tr>
<td></td>
<td>(0.0007)b</td>
<td>(0.0015)</td>
</tr>
<tr>
<td>$\sigma^2$ (OLS)</td>
<td>0.0430</td>
<td>0.0394</td>
</tr>
<tr>
<td>Observations</td>
<td>3544</td>
<td>859</td>
</tr>
<tr>
<td>$R^2$</td>
<td>0.778</td>
<td>0.655</td>
</tr>
<tr>
<td>Adj $R^2$</td>
<td>0.775</td>
<td>0.638</td>
</tr>
</tbody>
</table>

**Notes:**
- $B =$ Baseline category.
- $^a$ Significant at the 1% level.
- $^b$ Significant at the 5% level.
- $^c$ Significant at the 10% level.
(e.g. income and race), property types and classifications based upon the judgement of estate agents. Here we suggest an approach that makes no a priori assumptions concerning the criteria defining submarkets, rather the data themselves are used to suggest the pattern of market segmentation.

The procedure suggested here is as follows. Properties are grouped into clusters based on their similarity along a multitude of dimensions: locational, structural and socio-economic. Hedonic functions are estimated for each cluster of properties. Using tests suggested by previous researchers each hedonic function is compared with the other hedonic functions. If, for two clusters of properties, we cannot reject the hypothesis of equality of parameters, then the clusters are merged. Ultimately, the properties are partitioned into a small number of clusters each displaying a unique hedonic price function and these clusters are interpreted as submarkets.

The process by which properties are grouped into clusters is known as cluster analysis. Cluster analysis divides a data set into groups (clusters) of observations that are similar to each other. There are two basic approaches to cluster analysis, partitioning methods and hierarchical methods. With both methods, the researcher determines the characteristics that are to be used to cluster the observations. Here properties were characterized by their grid reference (longitude and latitude), their proximity to the city centre (car travel time), their selling price, their structural characteristics (dimensions, property type and property age) and the characteristics of the neighbourhood (OA and PD factors). Each observation then can be plotted in $M$-space according to how highly it scores on each of these $M$ characteristics. Clearly, observations holding similar values for the different characteristics will be located close to each other in this $M$-space.

With partitioning methods the researcher decides upon the number of clusters a priori. Let us denote this number of clusters $k$. The partitioning algorithm seeks to find $k$ locations in $M$-space, known as medoids, such that the sum of the distances between each observation and its nearest medoid is minimized. Once the $k$ medoids have been determined the observations are partitioned into clusters by assigning each observation to its nearest medoid.

Hierarchical methods work in a somewhat different manner. With a bottom-up approach, each observation is initially considered as a small cluster by itself. As a first step the two observations lying closest together are merged into a new cluster. At each subsequent step, the two nearest clusters are combined to form one larger cluster. Clusters are merged until one large cluster remains containing all the observations. Alternatively, hierarchical algorithms exist that start with one large cluster and split this into two smaller clusters and continue splitting until each observation forms a cluster of its own. Either way, the final result is a hierarchy of association.
appearing much like an inverted tree. The researcher can plot this hierarchy and determine which branches of the hierarchy should be treated as separate clusters. The advantage of hierarchical methods is that they do not impose any a priori assumptions on the pattern of association in the observations. The drawback with these methods, however, is that they are computationally burdensome with large data sets.

Here we propose a hybrid method. In the first step, the 3544 observations were clustered into 100 groups using a partitioning method. In the second step, the average values for the characteristics of each cluster were calculated and these were analysed using a hierarchical method. The hierarchical plot of this analysis is presented in Figure 17.2. The labels at the end of each branch refer to the 100 groups generated in the first step. From the hierarchical plot we have identified eight distinct clusters of observations.

Whilst, the cluster analysis identifies groups of properties that are similar locationally, structurally and in their neighbourhood characteristics, it does not indicate whether these clusters represent separate submarkets. Consequently, separate hedonic price functions were estimated for each cluster of properties using OLS. Following Allen et al. (1995) we compare these regression equations using a Chow test. The Chow test identifies whether there is a significant difference between a pair of regression equations under the null hypothesis that the two models are equivalent. The test statistic is given by:

\[
F = \left( \frac{\text{SSR}_c - (\text{SSR}_1 + \text{SSR}_2)}{\text{SSR}_1 + \text{SSR}_2} \right) \times \left( \frac{(N_1 + N_2) - (K_1 + K_2)}{\min(K_1, K_2)} \right)
\]

where \(\text{SSR}_1, \text{SSR}_2\) and \(\text{SSR}_c\) are the sum of squared residuals for the individual models and the combined model and \(N_1, N_2\) and \(K_1, K_2\) are the number of observations and number of parameters in the individual models respectively. The test statistic, \(F\), has an \(F\) distribution with \(\min(K_1, K_2), (N_1 + N_2) - (K_1 + K_2)\) degrees of freedom. Table 17.11 reports the results of this test for the eight clusters of properties. Clearly, in a number of cases, it is not possible to reject the hypothesis of equivalence of hedonic price functions (test scores italicized). We selected the combination of clusters showing the greatest likeness (judged by the Chow test statistic) and combined them into one larger cluster of properties. In this case, cluster 2 and 8 produced the lowest test statistic and hence were merged for the second round. The test statistics were recalculated using the six remaining clusters and the new combined cluster. Again, it was not possible to reject the hypothesis of equivalent hedonic price functions for a number of combinations of clusters. Again the pair of clusters showing the most similarity were merged.
Figure 17.2  Submarket determination using hierarchical cluster analysis
This process was repeated until all the test statistics were significant at a 1 per cent level of confidence or greater.

Table 17.11  Chow test $F$-statistics for differences between hedonic price functions of property clusters (italicized scores show cases where hedonic price functions are equivalent)

<table>
<thead>
<tr>
<th>Cluster</th>
<th>Obs</th>
<th>Clusters</th>
<th>Cluster</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>1 2 3 4 5 6 7</td>
</tr>
<tr>
<td>1</td>
<td>859</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>1586</td>
<td>3.722\textsuperscript{a}</td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>338</td>
<td>2.487\textsuperscript{a} 3.219\textsuperscript{a}</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>291</td>
<td>1.847\textsuperscript{a} 2.637\textsuperscript{a} 2.014\textsuperscript{a}</td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>100</td>
<td>2.437\textsuperscript{a} 2.336\textsuperscript{a} 0.982 0.667</td>
<td></td>
</tr>
<tr>
<td>6</td>
<td>228</td>
<td>2.311\textsuperscript{a} 1.894\textsuperscript{a} 1.408\textsuperscript{c} 1.163 1.446\textsuperscript{b}</td>
<td></td>
</tr>
<tr>
<td>7</td>
<td>32</td>
<td>1.832\textsuperscript{a} 2.489\textsuperscript{a} 1.335 1.894\textsuperscript{a} 1.776\textsuperscript{b} 2.933\textsuperscript{a}</td>
<td></td>
</tr>
<tr>
<td>8</td>
<td>110</td>
<td>0.677 0.655 0.944 0.662 1.657\textsuperscript{b} 1.123 4.334\textsuperscript{a}</td>
<td></td>
</tr>
</tbody>
</table>

\textbf{Note:}\n\textsuperscript{a} Significant at the 1\% level.\n\textsuperscript{b} Significant at the 5\% level.\n\textsuperscript{c} Significant at the 10\% level.

The results of the final round of tests are provided in Table 17.12. Here the eight original clusters have been reduced to four clusters. According to the Chow test, the hedonic price function estimated for each of these clusters of properties is significantly different from the hedonic price function of each of the other clusters of properties. These hedonic price functions are reported in the final four columns of Table 17.10.

Table 17.12  Chow test $F$-statistics for differences between hedonic price functions of property submarkets

<table>
<thead>
<tr>
<th>Submarket</th>
<th>Obs</th>
<th>Clusters \textsuperscript{Submarket}</th>
<th>Submarket</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>1 2 3</td>
</tr>
<tr>
<td>1</td>
<td>859</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>1696</td>
<td>2, 8</td>
<td>3.704\textsuperscript{a}</td>
</tr>
<tr>
<td>3</td>
<td>370</td>
<td>3, 7</td>
<td>2.606\textsuperscript{a} 3.346\textsuperscript{a}</td>
</tr>
<tr>
<td>4</td>
<td>619</td>
<td>4, 5, 6</td>
<td>3.492\textsuperscript{a} 4.183\textsuperscript{a} 2.782\textsuperscript{a}</td>
</tr>
</tbody>
</table>

\textbf{Note:}\n\textsuperscript{a} Significant at the 1\% level.
An important assumption in applying the Chow test as Allen et al. (1995) have prescribed is that the disturbance variance is the same across regression models. A quick glance across the estimates of $\sigma^2$ in Table 17.10 indicates that, in this case, this is unlikely to be true. The disturbance variance from the hedonic price regression for the third cluster of properties is considerably larger than that for the other clusters. When comparing the other models with that estimated for the third property cluster, we must assume that the combined model is heteroscedastic. In these circumstances, it is likely that we shall overestimate the significance of differences in the parameter estimates of the models.

A number of alternative tests suggest themselves. Here we employ a simple Wald test to confirm the results of Table 17.12. Provided the samples are reasonably large, the Wald test will be valid whether or not the disturbance variances are the same. Let us denote the sets of parameters from the two models being compared as $\beta_1$ and $\beta_2$ and their accompanying variance matrices $V_1$ and $V_2$. Now, under the null hypothesis that the two clusters of properties are independent samples drawn from the same property market we can conclude that $\beta_1$ and $\beta_2$ are normally distributed estimators of the same population parameters, $\beta$. Thus, the expected value of $\beta_1 - \beta_2$ will be zero and the variance of the difference in the parameter estimates will be $V_1 + V_2$ (since the clusters are independent samples). The Wald statistic is given by:

$$W = \left( \beta_1 - \beta_2 \right) \left( V_1 + V_2 \right)^{-1} \left( \beta_1 - \beta_2 \right)$$

which has a chi-squared distribution with $k$ degrees of freedom, where $k$ is the number of parameters in common between the two models. The results of the Wald test are presented in Table 17.13.

Table 17.13 Wald test chi-squared statistics for differences between the hedonic price functions of property submarkets

<table>
<thead>
<tr>
<th>Submarket</th>
<th>Obs</th>
<th>Clusters in Submarket</th>
<th>Submarket 1</th>
<th>Submarket 2</th>
<th>Submarket 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>859</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>1696</td>
<td>2, 8</td>
<td>151.4$^a$</td>
<td></td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>370</td>
<td>3, 7</td>
<td>91.5$^a$</td>
<td>116.7$^a$</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>619</td>
<td>4, 5, 6</td>
<td>277.8$^a$</td>
<td>333.4$^a$</td>
<td>180.7$^a$</td>
</tr>
</tbody>
</table>

Note: $^a$ Significant at the 1% level.
Amenity and water quality

All the test statistics are highly significant confirming the conclusions of Table 17.12 and supporting the assumption that the four clusters represent properties being traded in distinct property submarkets of the Glasgow urban area.

To better understand the character of these four submarkets, Table 17.14 provides a list of the mean values of some important characteristics of the properties in each submarket. Further, Figure 17.3 plots the location of the properties in the different submarkets. Notice immediately that whilst the properties in the different submarkets show a good deal of spatial separation, there is also considerable overlap. The submarkets defined by the procedure described here do not result in geographically distinct segmentation of the property market.

Table 17.14 Mean values of selected characteristics of submarkets

<table>
<thead>
<tr>
<th>Variable</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
</tr>
</thead>
<tbody>
<tr>
<td>Log Floor Area</td>
<td>99.64</td>
<td>109.844</td>
<td>196.43</td>
<td>189.221</td>
</tr>
<tr>
<td>Garden Size</td>
<td>3.688</td>
<td>26.611</td>
<td>199.936</td>
<td>301.355</td>
</tr>
<tr>
<td>Detached</td>
<td>.</td>
<td>.</td>
<td>.</td>
<td>0.162</td>
</tr>
<tr>
<td>Semi-detached</td>
<td>.</td>
<td>.</td>
<td>.</td>
<td>0.368</td>
</tr>
<tr>
<td>Terraced</td>
<td>.</td>
<td>.</td>
<td>.</td>
<td>0.449</td>
</tr>
<tr>
<td>Subdivided House</td>
<td>.</td>
<td>.</td>
<td>0.086</td>
<td>.</td>
</tr>
<tr>
<td>Four Block</td>
<td>.</td>
<td>0.065</td>
<td>.</td>
<td>.</td>
</tr>
<tr>
<td>Flat</td>
<td>0.010</td>
<td>0.085</td>
<td>0.592</td>
<td>0.021</td>
</tr>
<tr>
<td>Tenement</td>
<td>0.983</td>
<td>0.848</td>
<td>0.295</td>
<td>.</td>
</tr>
<tr>
<td>Other</td>
<td>0.007</td>
<td>0.002</td>
<td>0.027</td>
<td>.</td>
</tr>
<tr>
<td>OA Wealth Factor</td>
<td>-0.517</td>
<td>-0.413</td>
<td>0.259</td>
<td>1.093</td>
</tr>
<tr>
<td>OA Ethnicity Factor</td>
<td>-0.209</td>
<td>0.308</td>
<td>-0.411</td>
<td>-0.134</td>
</tr>
<tr>
<td>OA DINKY Factor</td>
<td>0.048</td>
<td>0.109</td>
<td>0.285</td>
<td>0.162</td>
</tr>
<tr>
<td>OA Suburbia Factor</td>
<td>-0.508</td>
<td>-0.389</td>
<td>0.182</td>
<td>0.743</td>
</tr>
<tr>
<td>PC Unemployment Factor</td>
<td>0.163</td>
<td>-0.729</td>
<td>-0.163</td>
<td>-0.934</td>
</tr>
<tr>
<td>PC YUPPY Factor</td>
<td>0.450</td>
<td>0.222</td>
<td>1.248</td>
<td>-0.471</td>
</tr>
<tr>
<td>PC Ethnicity Factor</td>
<td>-0.358</td>
<td>0.579</td>
<td>-0.339</td>
<td>-0.121</td>
</tr>
<tr>
<td>PC Wealth Factor</td>
<td>0.364</td>
<td>-0.42</td>
<td>1.212</td>
<td>.388</td>
</tr>
<tr>
<td>Traffic Noise</td>
<td>65.403</td>
<td>65.261</td>
<td>64.225</td>
<td>62.715</td>
</tr>
</tbody>
</table>

Submarkets 1 and 2 are characterized by relatively small and cheap properties. The properties in these two submarkets are mostly located in
tenement blocks, with a smattering of flats and other property types. Not surprisingly, these submarkets are typified by neighbourhoods that score very poorly on the wealth factor and suburbia factor. At first glance little distinguishes these two submarkets. Notice from Figure 17.3, however, that there is considerable geographical separation between submarket 1 and submarket 2. Submarket 1 is concentrated in the north-west of the city whilst submarket 2 is mostly concentrated in the southern part of the city.

Moreover, one characteristic appears to define the difference between these two submarkets: the ethnicity of their inhabitants. In comparison with all the other submarkets, submarket 2 scores very highly on the factors indicating the ethnic makeup of the property neighbourhoods. It would appear that two submarkets exist in Glasgow for similar types of properties.

*Figure 17.3* Location of properties in different submarkets
fetching similar prices. These submarkets are geographically distinct and are inhabited by two different ethnic groups, submarket 1 by ethnically Scottish residents, submarket 2 by members of the ethnic minorities.

Submarket 3 exhibits a greater diversity of property types than the first two submarkets. Properties in this submarket comprise a variety of flats, tenements and subdivided houses. The properties are larger, have larger gardens and command a considerably higher price than those in the first two submarkets. The neighbourhoods containing properties in submarket 3 are relatively wealthy with low unemployment. Whilst submarket 3 scores relatively highly on the suburbia factor, this is not a submarket typified by green-velged, leafy avenues. Rather the clue to the identity of this submarket lies in the very high scores on the DINKY and YUPPY factors. Submarket 3 appears to represent a market consisting of urban dwelling young professionals living mainly in flats to the north of the city centre.

Submarket 4 is the most distinct of the four markets. The properties in this submarket are, in the main, detached, semi-detached or terraced houses. Not surprisingly, they are considerably larger than properties in the other submarkets, have larger gardens and command much higher prices. This is the affluent suburbia submarket. Neighbourhoods containing properties in these submarkets are wealthy and score highly on the suburbia factor indicating the presence of large properties inhabited by high (frequently dual-) earning families. Table 17.15 summarizes the interpretations of the submarkets.

An important consequence of the existence of property submarkets in Glasgow is that estimates of implicit prices derived from the parameters of the pooled data hedonic price function are likely to be biased. To compare the parameters across submarkets we employ the Tiao–Goldberger (Tiao and Goldberger, 1962) test as suggested by Michaels and Smith (1990).

The Tiao-Goldberger test considers the null hypothesis that the coefficients of a particular regressor take the same value in each of the models. In effect, the test works by comparing each parameter estimate to the weighted sum of parameter estimates across all four models. The sum of the squared differences between each parameter and this average value form the core of the test statistic. Following the notation of Michaels and Smith (1990), the Tiao–Goldberger test statistic is given by:

\[
F^\text{TG} = \frac{\sum_{j=1}^{L} \left( b_{ij} \right)^{2} / p_{j} \sum_{j=1}^{L} (T_{ij} - K_{ij})}{\sum_{j=1}^{L} \text{SSR}_{j} / (L - 1)}
\]

(17.6)
where

$$\bar{b}_j = \frac{\sum_{i=1}^{L} \left( b_{ji} / p_j \right)}{\sum_{i=1}^{L} (1/p_j)}$$

and $L$ is the number of models; $b_{ji}$ is the OLS estimate of the $i$th parameter in the $j$th independent model; $P_{ji}$ is the diagonal element of the $i$th parameter of $(X'X)_j^{-1}$ in the $j$th model; SSR$_j$ is the sum of squared residuals for the $j$th model; $T_j$ is the number of observations used to estimate the $j$th model and $K_j$ is the number of parameters in the $j$th model. The test statistic has an $F$ distribution with $(L-1), \sum_{j=1}^{L} (T_j - K_j)$ degrees of freedom.

\begin{table}[h]
\centering
\begin{tabular}{ll}
\hline
Submarket & Brief Description \\
\hline
Number & Name & \\
\hline
1 & White Tenements & Tenements and some flats, located to the north-west of Glasgow City centre, in relatively poor ethnically Scottish neighbourhoods with high unemployment \\
2 & Ethnic Minority Tenements & Tenements and some flats, located to the south of Glasgow City centre, in relatively poor neighbourhoods distinguished by high concentrations of ethnic minorities \\
3 & Young Urban Professionals & Mostly flats, tenements and subdivided houses located in relatively affluent areas to the north of the city centre, in neighbourhoods characterized by relatively young urban dwelling professionals \\
4 & Affluent Suburbia & Detached, semi-detached and terraced houses mostly located on the fringes of the city in suburban areas. Properties are large, expensive and with large gardens and are located in relatively wealthy neighbourhoods \\
\hline
\end{tabular}
\caption{Descriptions and interpretations of the submarkets in Glasgow}
\end{table}
Table 17.16 reports the parameter estimates and the Tiao–Goldberger test statistics for a number of parameters that were estimated across all four submarkets.

Table 17.16  

Tiao–Goldberger test F-statistics for selected parameters from the four hedonic price functions of property submarkets

<table>
<thead>
<tr>
<th>Parameter</th>
<th>White Tenements</th>
<th>Ethnic Minority Tenements</th>
<th>Young Professionals</th>
<th>Affluent Suburbia</th>
<th>$F_{TG}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>9.646&lt;sup&gt;a&lt;/sup&gt;</td>
<td>9.477&lt;sup&gt;a&lt;/sup&gt;</td>
<td>8.902&lt;sup&gt;a&lt;/sup&gt;</td>
<td>9.152&lt;sup&gt;a&lt;/sup&gt;</td>
<td>2.616&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>Log Floor Area</td>
<td>0.089</td>
<td>0.156&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.142&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.228&lt;sup&gt;a&lt;/sup&gt;</td>
<td>3.467&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>Garden Size</td>
<td>0.044&lt;sup&gt;b&lt;/sup&gt;</td>
<td>0.549&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.121&lt;sup&gt;b&lt;/sup&gt;</td>
<td>0.164&lt;sup&gt;a&lt;/sup&gt;</td>
<td>3.448&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>OA Wealth Factor</td>
<td>0.166&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.098&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.100&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.127&lt;sup&gt;a&lt;/sup&gt;</td>
<td>2.274&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>OA Ethnicity Factor</td>
<td>-0.058&lt;sup&gt;a&lt;/sup&gt;</td>
<td>-0.040&lt;sup&gt;a&lt;/sup&gt;</td>
<td>-0.065&lt;sup&gt;b&lt;/sup&gt;</td>
<td>-0.100&lt;sup&gt;a&lt;/sup&gt;</td>
<td>3.299&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>OA Tenement Factor</td>
<td>-0.080&lt;sup&gt;a&lt;/sup&gt;</td>
<td>-0.050&lt;sup&gt;a&lt;/sup&gt;</td>
<td>-0.031</td>
<td>-0.054&lt;sup&gt;b&lt;/sup&gt;</td>
<td>3.551&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>OA DINKY Factor</td>
<td>0.052&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.056&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.0004</td>
<td>0.013</td>
<td>6.168&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>OA Suburbia Factor</td>
<td>0.060&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.066&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.044&lt;sup&gt;b&lt;/sup&gt;</td>
<td>0.076&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.601</td>
</tr>
<tr>
<td>PC Unemployment</td>
<td>-0.040&lt;sup&gt;b&lt;/sup&gt;</td>
<td>-0.044&lt;sup&gt;a&lt;/sup&gt;</td>
<td>-0.125&lt;sup&gt;b&lt;/sup&gt;</td>
<td>-0.026</td>
<td>1.534</td>
</tr>
<tr>
<td>PC YUPPY Factor</td>
<td>0.023&lt;sup&gt;c&lt;/sup&gt;</td>
<td>0.025&lt;sup&gt;b&lt;/sup&gt;</td>
<td>0.014</td>
<td>0.100&lt;sup&gt;a&lt;/sup&gt;</td>
<td>6.13&lt;sup&gt;a&lt;/sup&gt;</td>
</tr>
<tr>
<td>PC Ethnicity Factor</td>
<td>-0.033</td>
<td>0.011</td>
<td>0.019</td>
<td>0.055&lt;sup&gt;a&lt;/sup&gt;</td>
<td>3.449&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>PC Wealth Factor</td>
<td>0.079&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.086&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.078&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.059&lt;sup&gt;a&lt;/sup&gt;</td>
<td>1.214</td>
</tr>
<tr>
<td>Walk Rail</td>
<td>0.0190</td>
<td>0.057&lt;sup&gt;b&lt;/sup&gt;</td>
<td>0.124&lt;sup&gt;b&lt;/sup&gt;</td>
<td>-0.023</td>
<td>2.908&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>Walk Park</td>
<td>-0.075&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.037</td>
<td>-0.040</td>
<td>-0.075&lt;sup&gt;b&lt;/sup&gt;</td>
<td>3.023&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>Walk School</td>
<td>0.092</td>
<td>-0.006</td>
<td>-0.007</td>
<td>0.148&lt;sup&gt;8&lt;/sup&gt;</td>
<td>1.834</td>
</tr>
<tr>
<td>Car City Centre</td>
<td>0.091</td>
<td>0.033</td>
<td>-0.019</td>
<td>0.024&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.742</td>
</tr>
<tr>
<td>Front View</td>
<td>0.011</td>
<td>0.006</td>
<td>0.046</td>
<td>0.054&lt;sup&gt;a&lt;/sup&gt;</td>
<td>2.614&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>Back View</td>
<td>-0.005</td>
<td>0.004</td>
<td>-0.079&lt;sup&gt;a&lt;/sup&gt;</td>
<td>-0.041&lt;sup&gt;b&lt;/sup&gt;</td>
<td>6.699&lt;sup&gt;a&lt;/sup&gt;</td>
</tr>
<tr>
<td>Traffic Noise</td>
<td>-0.0023</td>
<td>-0.0046&lt;sup&gt;a&lt;/sup&gt;</td>
<td>-0.0057&lt;sup&gt;b&lt;/sup&gt;</td>
<td>0.0038&lt;sup&gt;b&lt;/sup&gt;</td>
<td>6.361&lt;sup&gt;a&lt;/sup&gt;</td>
</tr>
</tbody>
</table>

Notes:

<sup>a</sup> Significant at the 1% level.
<sup>b</sup> Significant at the 5% level.
<sup>c</sup> Significant at the 10% level.

For many of the variables listed in Table 17.16 we can reject the hypothesis of stability of parameter estimates across submarkets. For example, observe the parameter estimates on the log of floor area. The Tiao–Goldberger statistic is significant at a 5 per cent level of confidence and we conclude that the parameters across the four submarkets are different. On closer inspection it is clear that the implicit price of floor area is greatest in submarket 4 (Affluent Suburbia); households purchasing properties in this...
submarket pay more for relatively larger properties than households in the other submarkets. We must interpret this result with caution. As described in the second section, the market clearing implicit prices are the result of both demand and supply conditions in that submarket. Thus, if we were to assume that the supply of properties of different floor areas were the same in all submarkets, then it would be reasonable to conclude that there was greater demand for larger properties in submarket 4. Alternatively, if we were to assume that demand for larger properties was constant over all submarkets, we would be forced to conclude that the availability of larger properties was restricted in submarket 4. Intuition suggests that the differences in the implicit price of floor area across submarkets are more likely to be driven by demand-side factors, though clearly the truth will lie somewhere between the two extremes.

Of most interest in this research are the differences in the implicit price of traffic noise. From Table 17.16 we observe that the Tiao–Goldberger statistic indicates that the hypothesis of equality of parameter estimates can be rejected at the 1 per cent level of confidence. The price of properties are most adversely affected by increasing exposure to traffic noise in submarket 3 (Young Urban Professionals), followed by submarket 2 (Ethnic Minority Tenements), then submarket 1 (White Tenements). Contradicting expectations, exposure to road traffic noise appears to increase the price of properties in submarket 4 (Affluent Suburbia) and this is significant at a 5 per cent level of confidence. To understand better the supply conditions, Figure 17.4 provides histograms depicting the traffic noise exposure of properties (in the sample) traded in the four submarkets.

Notice in particular the relative abundance of low noise exposure properties in submarket 4. One possible explanation of the greater price paid for traffic noise avoidance in submarkets 1 to 3 might be related to the fact that quiet properties are relatively more difficult to come by in these submarkets.

Of course this does not explain the unexpected sign on the traffic noise variable in the hedonic price function for submarket 4. We conclude that, in this submarket at least, traffic noise must be proxying for some other variable that positively influences traffic prices. We do not follow up on this contention here.

To ensure that the noise parameter estimate for submarket 4 was not the sole cause of the high Tiao–Goldberger test statistic score in Table 17.16, we reapply the test using only data from submarkets 1 to 3. Once again, the test statistic is large ($F^{TG} = 5.178$) leading us to reject the hypothesis of equality of the noise parameter estimates across these three submarkets at a 1 per cent level of confidence.
So far, our statistical analysis of the hedonic price functions in the separate submarkets has ignored the spatial organization of the data. In effect, we have assumed that the observations of property sales are independent such that we can glean no information on the selling price of a property from the selling price of other properties. Of course, this is hardly likely to be the case. Properties that are located near to each other in space are also likely to share common environmental, accessibility, neighbourhood and perhaps even structural characteristics. Even once we account for the values of known covariates, omitted variables are likely to induce spatial dependence among the errors.

If hedonic residuals are spatially correlated, the parameter estimates from an OLS regression will be inefficient and will produce biased estimates of the standard errors of the parameter estimates. In the case where the residuals are positively spatially correlated, as is to be expected with hedonic property price regressions, OLS will underestimate the population residual variance and the resulting t-statistics will be biased upwards. Whilst OLS parameter estimates remain unbiased, ignoring spatial autocorrelation may lead to

Figure 17.4 Distribution of properties with different exposures to traffic noise in the four submarkets

SPATIAL HEDONIC REGRESSION

This figure shows the distribution of properties with different exposures to traffic noise in the four submarkets. The histograms depict the density of properties across different levels of traffic noise, with each submarket having a distinct pattern. Submarket 1 exhibits a peak at lower noise levels, while Submarket 2 shows a higher density at moderate noise levels. Submarket 3 has a broad distribution across noise levels, and Submarket 4 displays a peak at high noise levels.

These diagrams illustrate the spatial variation in property values due to traffic noise, highlighting the importance of considering spatial effects in hedonic price models.
Hedonic price analysis of road traffic noise nuisance

erroneously high significance being attached to the influence of property attributes on selling prices.

Over recent years, the existence of spatial autocorrelation has received a great deal of attention in the hedonic literature (e.g. Can, 1992; Dubin, 1992; Pace and Gilley, 1997; Basu and Thibodeau, 1998; Bell and Bockstael, 1997). In the main, researchers have focused on the spatial error dependence model that can be expressed as;

\[ y = X\beta + \epsilon \]  

(17.7a)

where

\[ \epsilon = \lambda W\epsilon + u \]  

(17.7b)

where \( y, X \) and \( \beta \) are defined as in (3) but \( \epsilon \) is now an \([N \times 1]\) vector of random error terms with mean 0 and a non-spherical variance-covariance matrix \( \sigma^2(I - \lambda W)^{-1}(I - \lambda W')^{-1} \). The nature of the spatial autocorrelation is defined by equation (17.7b). Here \( W \) is an \([N \times N]\) weighting matrix, \( \lambda \) is the error dependence parameter to be estimated and \( u \) is the usual \([N \times 1]\) vector of random error terms with expected value 0 and variance-covariance matrix \( \sigma^2I \).

The error in the spatial error dependence model, therefore, is made up of two parts: a purely random element and an element containing a weighted sum of the errors on nearby properties. The association between one property and another is contained in the weighting matrix, \( W \). The diagonal elements of the weighting matrix are zero since, clearly, the error for an observation cannot be used to explain itself. The off-diagonal elements of the matrix represent the potential spatial dependence between observations. Thus if the \( ij \)th element of the weighting matrix, \( w_{ij} \), is zero, we are assuming that there is no correlation in the errors of the \( i \)th and \( j \)th observations. Conversely if \( w_{ij} \) takes on a non-zero value we are assuming that there is correlation in the errors of these two observations.

The researcher must stipulate the nature of dependence between observations by defining the weights matrix in advance of estimation. Here we experimented with a variety of weights matrices but the final specification of equation (17.7) used a weights matrix in which it was assumed that properties separated by more than 100 metres were unrelated. Moreover the non-zero elements of the weights matrix were defined as the inverse of the squared distance between properties. This format for the weights matrix allows for the error terms to be more closely correlated with the error terms of close neighbours than with the error terms of more distant neighbours.

Following normal procedure, \( W \) was row standardised such that each row’s elements were made to sum to one. When \( W \) is row standardized, the product \( W\epsilon \) equals \( \sum w_{ij}\epsilon_j \), and has an intuitive interpretation: it is simply a
vector of weighted averages of the errors of neighbouring observations. As Bell and Bockstael (2000) point out, row standardization is undertaken to simplify estimation of the model. There is usually no underlying economic story supporting the procedure. Moreover, the spatial dependence parameter $\lambda$ estimated on a row standardized weights matrix must be interpreted with caution. In particular, $\lambda$ in this case is not directly equivalent to an autocorrelation coefficient.

The characteristics of the weights matrices constructed for the property sales observations in the four submarkets are detailed in columns two to four of Table 17.17. Even with a relatively restrictive 100 metre cut-off, the majority of properties are associated with other properties in the same submarket. In submarket 2, for example, only 46 properties out of the 1696 observations were further than 100 metres from another property in the sample. On average in this submarket, each property was located within 100 metres of 16 other properties in the sample. Notice that the number of associations in submarket 4 is somewhat lower than in the other submarkets and that this is not entirely explained by sample size. One explanation of this observation is that properties in suburbia are more greatly dispersed than those in the other submarkets.

The final columns of Table 17.17 report two tests of spatial dependence based on the Lagrange multiplier principle that can be calculated using OLS residuals and the spatial weights matrices. The first, suggested by Burridge (1980), tests the null hypothesis that $\lambda = 0$. That is, the hypothesis that there is no spatial dependence between error terms. The test statistics reported in column 5 are chi-squared distributed with one degree of freedom. The null hypothesis is rejected with a high degree of confidence in all four submarkets, supporting the contention that the OLS residuals exhibit spatial autocorrelation.

An alternative model of spatial dependence stipulates a dependent variable that is functionally related to the value of the dependent variable of neighbouring observations. In this case our model would include spatially lagged values of the dependent variable rather than the error term. This model is variously known as substantive spatial dependence, structural dependence or spatial autoregression. A test of this model (suggested by Anselin, 1988) is provided in the final column of Table 17.17. In all but one submarket we cannot reject the null hypothesis of no substantive spatial dependence. Again the data would seem to suggest that the correct model is that presented in equation (17.7a and b).

The spatial dependence model in equation (17.7a and b) can be estimated using maximum likelihood (ML) techniques. However, for large samples this may be computationally prohibitive. Instead we follow Bell and Bockstael (2000) and use the generalized moments (GM) estimator developed by
Table 17.17  Characteristics of the spatial weights matrices and tests of spatial dependence

<table>
<thead>
<tr>
<th>Submarket</th>
<th>Characteristics of the Spatial Weights Matrices</th>
<th>Tests of Spatial Dependence</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Obs. Unassociated obs. Average associations per obs.</td>
<td>LM test for spatial error dependence</td>
</tr>
<tr>
<td>1. White Tenements</td>
<td>859 28 13.84</td>
<td>20.50\textsuperscript{a}</td>
</tr>
<tr>
<td>2. Ethnic Minority Tenements</td>
<td>1696 46 16.02</td>
<td>65.67\textsuperscript{a}</td>
</tr>
<tr>
<td>3. Urban Young Professionals</td>
<td>370 49 5.35</td>
<td>3.98\textsuperscript{b}</td>
</tr>
<tr>
<td>4. Affluent Suburbia</td>
<td>619 116 2.18</td>
<td>18.56\textsuperscript{a}</td>
</tr>
</tbody>
</table>

Notes:
\(a\) Significant at the 1% level.
\(b\) Significant at the 5% level.
Kelejian and Prucha (1999). As Bell and Bockstael (2000) describe, whilst this estimator may not be as efficient as the ML estimator it possesses two advantages. First, the calculation of the estimator is fairly straightforward even with extremely large samples. And second, the GM estimator is consistent even when the error terms $u$ are not normal. The parameter estimates from the GM estimator of model (17.7) applied to the data for the four property submarkets in Glasgow are listed in Table 17.18.

### Table 17.18
**Hedonic price functions for property markets in Glasgow estimated using GM estimator accounting for spatial autocorrelation**

<table>
<thead>
<tr>
<th>Independent Variable</th>
<th>Submarkets</th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>One</td>
<td>Two</td>
<td>Three</td>
<td>Four</td>
</tr>
<tr>
<td>Intercept</td>
<td>9.776 (0.186)$^a$</td>
<td>9.588 (0.143)$^a$</td>
<td>8.883 (0.267)$^a$</td>
<td>9.168 (0.172)$^a$</td>
</tr>
<tr>
<td>Log Floor Area</td>
<td>0.079 (0.024)$^a$</td>
<td>0.140 (0.020)$^a$</td>
<td>0.120 (0.038)$^a$</td>
<td>0.236 (0.033)$^a$</td>
</tr>
<tr>
<td>Garden Size</td>
<td>-0.020 (0.204)</td>
<td>0.541 (0.135)$^a$</td>
<td>0.122 (0.043)$^a$</td>
<td>0.131 (0.041)$^a$</td>
</tr>
<tr>
<td>Shape</td>
<td>0.008 (0.004)$^b$</td>
<td>0.008 (0.003)$^b$</td>
<td>0.009 (0.005)$^c$</td>
<td>0.002 (0.003)$^a$</td>
</tr>
<tr>
<td>Storeys</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-0.157 (0.031)$^a$</td>
</tr>
<tr>
<td>Detached</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>B</td>
</tr>
<tr>
<td>Semi-detached</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>0.067 (0.021)$^a$</td>
</tr>
<tr>
<td>Terraced</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-0.142 (0.038)$^a$</td>
</tr>
<tr>
<td>Subdivided House</td>
<td>-</td>
<td>-</td>
<td>-0.034</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>(0.062)</td>
</tr>
<tr>
<td>Four Block</td>
<td>-</td>
<td>-0.012 (0.044)</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Flat</td>
<td>0.201 (0.069)$^a$</td>
<td>-0.037 (0.022)$^c$</td>
<td>-0.034 (0.033)</td>
<td>-0.797 (0.140)$^a$</td>
</tr>
<tr>
<td>Tenement</td>
<td>B</td>
<td>B</td>
<td>B</td>
<td>B</td>
</tr>
<tr>
<td>Other</td>
<td>0.122 (0.086)</td>
<td>-0.256 (0.111)$^b$</td>
<td>0.031 (-0.083)</td>
<td>-</td>
</tr>
<tr>
<td>Building Material</td>
<td>0.027 (0.064)</td>
<td>-0.029 (0.099)</td>
<td>0.233 (0.169)</td>
<td>0.100 (0.030)$^a$</td>
</tr>
<tr>
<td>Age (pre-1919)</td>
<td>0.077 (0.085)</td>
<td>-0.051 (0.113)</td>
<td>0.170 (0.177)</td>
<td>0.005 (0.047)</td>
</tr>
</tbody>
</table>
### Hedonic price analysis of road traffic noise nuisance

**Independent Variable** | **Submarkets**
--- | ---
Age (1919–45) | -0.063 | -0.027 | 0.298 | -0.021<br>(0.094) | (0.082) | (0.199) | (0.043)
No. of Properties in Building | 0.001 | -0.012 | -0.010 | 0.003<br>(0.004) | (0.004) | (0.005) | (0.002)
Basement Flat | -0.023 | -0.089 | -0.162 | -<br>(0.191) | (0.065) | (0.051)<br>a
Ground Floor Flat | B | B | B | B<br>First Floor Flat | 0.036 | 0.045 | -0.007 | 0.136<br>(0.02) | (0.014) | (0.037) | (0.146)
Second Floor Flat | -0.006 | 0.026 | 0.013 | -0.059<br>(0.02) | (0.014) | (0.036) | (0.141)
Third Floor Flat | -0.024 | 0.018 | -0.065 | -<br>(0.021) | (0.015) | (0.047)
OA Wealth Factor | 0.173 | 0.105 | 0.103 | 0.124<br>(0.026)<br>a | (0.016)<br>a | (0.018)<br>a | (0.013)<br>a
OA Ethnicity Factor | -0.046 | -0.036 | -0.062 | -0.102<br>(0.015)<br>a | (0.007)<br>a | (0.026)<br>a | (0.018)<br>a
OA Bedsit Factor | 0.010 | -0.002 | -0.004 | -0.034<br>(0.017) | (0.011) | (0.012) | (0.014)<br>b
OA Tenement Factor | -0.075 | -0.046 | -0.036 | -0.051<br>(0.009)<br>a | (0.006)<br>a | (0.02) | (0.017)<br>a
OA DINKY Factor | 0.051 | 0.053 | -0.004 | 0.022<br>(0.009)<br>a | (0.007)<br>a | (0.017) | (0.011)<br>b
OA Suburbia Factor | 0.065 | 0.067 | 0.056 | 0.070<br>(0.015)<br>a | (0.010)<br>a | (0.020)<br>a | (0.014)<br>a
PC Unemployment Factor | -0.039 | -0.044 | -0.122 | -0.025<br>(0.018)<br>b | (0.015)<br>a | (0.043)<br>a | (0.020)
PC YUPPY Factor | 0.025 | 0.032 | 0.016 | 0.092<br>(0.014)<br>c | (0.012)<br>a | (0.021) | (0.016)<br>a
PC Ethnicity Factor | -0.039 | 0.010 | 0.032 | 0.055<br>(0.023)<br>c | (0.009) | (0.033) | (0.019)<br>a
PC Wealth Factor | 0.085 | 0.091 | 0.086 | 0.065<br>(0.017)<br>a | (0.012)<br>a | (0.026)<br>a | (0.01)<br>a
Walk Rail | 0.008 | 0.05 | 0.137 | -0.037<br>(0.029) | (0.031) | (0.054)<br>b | (0.027)
Walk Park | -0.082 | 0.037 | -0.050 | -0.078<br>(0.048)<br>c | (0.03) | (0.071) | (0.037)<br>b
Walk Shop | 0.102 | - | -0.014 | 0.14<br>(0.086) | (0.053) | (0.097) | (0.052)<br>a
Walk School | 0.088 | 0.04 | -0.023 | 0.02<br>(0.057) | (0.038) | (0.059) | (0.038)
As expected, the parameter estimates do not significantly differ from those reported in Table 17.10. Again we focus on the implicit price of traffic noise avoidance:

As expected, the parameter estimates do not significantly differ from those reported in Table 17.10. Again we focus on the implicit price of traffic noise avoidance:
In submarket 1 the parameter becomes more negative, falling from a value of –0.0023 in the OLS model to –0.0030 in the spatial dependence model. Where the OLS result proved statistically insignificant, we can reject the hypothesis that the parameter in the spatial dependence model is actually zero with over 93 per cent confidence.

There is little change in the estimates for submarket 2 between the two models. The parameter is –0.0046 in the OLS model and –0.0047 in the spatial dependence model. Both are highly statistically significant.

In submarket 3 the actual parameter value changes little between the two models (–0.0057 in the OLS model, –0.0058 in the spatial dependence model). However, the standard error in the spatial dependence model is somewhat smaller. Indeed, in the spatial dependence model we can reject the hypothesis of a zero value for the noise parameter with over 99 per cent confidence.

In submarket 4, the parameter on traffic noise took a value of 0.0038 in the OLS model. The unexpectedly positive value for this parameter was significant at the 95 per cent level of confidence. Reassuringly, the parameter value falls to a value of 0.0031 in the spatial dependence model and is no longer significant at the 95 per cent confidence level.

It would seem, that in comparison with the OLS model, the spatial dependence model returns estimates of the parameter on traffic noise that are more in line with prior expectations.

**CONCLUSIONS**

Property markets are essentially spatial in nature and this fact should not be ignored in hedonic analyses of property prices. This case study has illustrated some important analytical techniques that can be used in hedonic analysis of property markets that take account of spatial considerations. First and foremost, GIS proves to be an extremely powerful tool for compiling data for the estimation of hedonic price functions. Researchers can collect data sets rich in information on the structural, accessibility, neighbourhood and environmental characteristics of properties from the comfort of their own desk.

Further, theory suggests that the property market will not be a homogeneous entity. Rather, it will be characterized by segmentation, with the implicit prices of property characteristics differing across market segments according to the conditions of supply and demand for characteristics prevailing in each market segment. In the past researchers have imposed a number of criteria (geographical, structural or socio-economic) in order to define these
submarkets. Here we propose an alternative approach based on cluster analysis that allows the data themselves to dictate the pattern of market segmentation. The results of this analysis suggest property submarkets with intuitively appealing interpretations. These submarkets are not defined by one criteria but a combination of spatial, structural and socio-economic characteristics.

Finally, we illustrate the application of an estimation technique that explicitly allows for the spatial relation between properties in the sample. In general, the results of the hedonic analysis in the separate submarkets concord with prior expectations. In particular, the implicit price for the avoidance of traffic noise is negative and significant in three out four submarkets. Moreover, these implicit prices are shown to be statistically different supporting the contention that separate hedonic price schedules rule in the different submarkets.

NOTES

* Based on The Effect of Road Traffic on Residential Property Values: A Literature Review and Hedonic Pricing Study (Bateman et al., 2001), report for the Scottish Executive.

1. Fortunately, the selling price of properties in Scotland is public information. Property sales are recorded in the Register of Sasines. Amongst other information the register records the exact postal address of the property and the price at which the property was sold.

2. In this case the digital map used was OS Land-Line.Plus, which records ground features with a spatial accuracy of 40 centimetres (OS, 1996).

3. Frequently researchers will rotate the factor axes to improve the ease with which factors can be interpreted. That is, the subspace defined by the factor axes is not changed, but the orientation of the axes themselves are rotated such that they best align with original axes describing the attributes.

4. This measure is known as the Noise Depreciation Sensitivity Index (NDSI) and is the measure that dominates hedonic price studies into the impact of noise on property prices. Typical values range between 0.10 and 1.30. For a recent review see Bateman et al. (2001).

5. All cluster analysis was carried out using the excellent ‘Cluster Package’ written in the R programming language.

6. Note that not all models contained the same number of parameters. For example, clusters 1, 2, 3 and 4 contained no detached properties such that this indicator variable was not included in the hedonic price functions for these clusters.

7. Parameters unique to one of the models being tested were dropped from the calculation of the statistic.

8. Computational details can be found in Anselin and Hudak (1992).

9. For an excellent and accessible introduction to this and other models of spatial dependence see Anselin (1993).

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PART III

Valuation and Sectoral Green Accounting
INTRODUCTION

The main driver for creating environmental accounts is the recognition that the current national accounting system does not reflect the full costs and benefits to society of economic activities and therefore, is an inadequate indicator of well-being or true economic progress. Given the primary importance of traditional accounting indicators such as gross domestic product (GDP) and net domestic product (NDP) in public policy-making, adjustments of these measures for environmental outcomes of economic activities are a step towards a better understanding of the sustainability (or otherwise) of economic development.

Agricultural land occupies most of the landscape in the UK, covering approximately 74 per cent of the total land area (DEFRA e-digest, 2003). As such, agriculture plays a large role in determining the character and quality of the UK environment. Over the centuries agricultural practices have helped to create and manage landscaped and wildlife habitats and continue to contribute to the maintenance of biodiversity in some contexts. On the other hand, agricultural activity has often led to environmental degradation such as through soil erosion and water pollution. Understanding the balance of the sector’s both positive and negative contributions to the environment, and placing these in the wider context of agriculture’s overall contribution to the UK economy and national well-being is a key aim of an adjusted set of accounts.

_Agriculture in the United Kingdom_ (DEFRA et al., 2004), published annually, presents a set of accounts for the agricultural sector and contains a chapter dedicated to the quantification of the environmental impacts of agriculture. These impacts come mainly in the form of headline indicators, such as tonnes of CO₂ emitted, farmland bird counts and measures of river water quality. In 2003 and 2004, the environmental chapter included some...
monetization of these impacts, based on research contained in Hartridge and Pearce (2001), Pretty et al. (2000) and the Environment Agency (2002). The work presented here is a continuation of these efforts by government to integrate environmental costs and benefits into the accounting framework. It expands on previous efforts in a number of ways, including 1. providing an up-to-date view on the appropriate data sources for both measuring impacts and employing valuation study results; 2. examining the positive as well as negative environmental contributions of the sector; and 3. tailoring an accounting framework to the agricultural sector. Although currently only in its infancy, once fully developed, a monetized environmental account for agriculture would be capable of providing:

- an economic measure of the sustainability of agriculture and a truer measure of the quality of life;
- an indication of the extent to which agriculture is a net contributor to the nation’s well-being as well as how it affects the welfare generated by other sectors;
- information that can be used for priority setting within agricultural policy; and
- inputs to cost–benefit analysis for agricultural and related environmental policies.

There are a number of interesting issues that arise in the development of an accounting framework as applied to a single sector in isolation, and these are presented in the next section of this chapter. Following that, the summary results from the study are presented according to seven impact categories: water, air, soil, landscape, habitat and species, waste and nuisance. The data and calculations behind these results are briefly explored, and the chapter concludes a summary of the issues raised by our findings as well as some recommendations for further research.

OVERVIEW OF SELECTED ACCOUNTING ISSUES

Sustainability and Well-being

At the national level, much of the interest in green accounting has focused on constructing comprehensive measures of income, saving and wealth. Popular debate has surrounded green alternatives to GDP or 'green GDP'. However, it is arguably estimates of genuine or adjusted net saving rate that have achieved prominence in much of the academic and policy debate about how a nation’s development prospects can be scrutinized: that is, for example,
Towards green sectoral accounts for UK agriculture

is a country on or off a development path that is sustainable (in the sense of consumption, or more broadly well-being being at least non-declining)?

What the genuine savings rate indicates is whether an economy is, on balance, adding to or liquidating its wealth (see, for example, Hamilton and Atkinson, forthcoming). If the latter then this is a signal of unsustainability (i.e. declining aggregate wealth) consistent with more popular notions of ‘not eating into one’s capital’ or ‘not selling the family silver’.

While such notions apply relatively easily to the national economy, it is less clear what precisely sustainability or unsustainability means for an economic sector such as agriculture. However, one relatively simple interpretation is that, just as conventional value-added in a sector indicates the contribution of that sector to value-added in the aggregate economy, environmentally-adjusted estimates of value-added indicate a truer picture of the contribution of that sector to national income or output. Such a measure might be construed as a better measure of the sector’s contribution to the ‘quality of life’. Similarly, environmentally-adjusted estimates of the sector’s net accumulation of assets give a truer reflection of that sector’s contribution to aggregate investment. Put this way, one function of green accounting in the agricultural sector might be to signal the contribution of the sector to sustainability more generally.3

As we will see later in this chapter, this contribution might take at least two broad forms. First, there are concerns about whether current levels of agricultural production are being financed by farming practices that, other things being equal, diminish future income-generating potential within the sector, for example by leading to erosion of soil productivity. Second, there are wider concerns about the (net) impact of agriculture and farming practices on society’s well-being and the impacts on other sectors. These external impacts too fall within the ambit of a notion of sustainability in the agricultural sector in the sense of partly determining the (net) contribution of the sector towards the sustainability of the wider economy.

Functions of Agriculture in the Economy

The process of building environmental accounts for agriculture should start by recognizing the three main roles of environmental assets as serving a:

• resource function, whereby the environment provides the raw materials that are transformed by the economy to produce goods and services;
• sink function whereby pollution generated by production and consumption is assimilated by the environmental media of air, water or land; and
Valuation and sectoral green accounting

- service function, which provides both survival functions and amenity functions, such as recreation.

The resource functions (or raw materials) are generally provided as market goods that are paid for in the economy and, are therefore, already included in the accounts; whereas, the sink and service functions are not marketed, not priced and, are therefore, absent from the current set of accounts. Thus environmental accounts for agriculture should aim to value the sink and service functions provided by the environment. In its simplest form the value of these functions is the product of society’s willingness to pay for a unit of these functions and their current level of provision.

For the accounting framework, environmental service and sink functions impacted upon by agriculture are considered systematically according to seven key impact headings: water, air, soil, landscape, habitats and species, waste, and nuisance. Agricultural activities either maintain or enhance environmental assets (where assets are water, air or land-based) for example through maintenance of landscapes, or serve to reduce them, for example by reducing air quality. Impact categories such as waste and nuisance are not linked to any one asset, but can impact upon a range of asset functions.

Accounting Adjustments: Assets, Income and Transfers

A key distinction to make when building environmental accounts for agriculture is between those assets that are under agricultural control – or can be attributed to agriculture – and those that are not. Land-based assets managed by agriculture can be thought of as assets of the agricultural sector. This is particularly the case in the UK where landscapes, habitats and species are often intimately linked to past and current agricultural activity. For these assets that are owned or managed by agriculture there are two adjustments relevant to the construction of an environmental account for this sector.

First, the existing stock of land-based assets provide a stream of environmental services in any accounting period. These flows might consist of the provision of landscape amenities and so on, the value of which typically will not be associated with any transaction that plausibly appears in conventional accounts for agriculture. However, such service flows may well represent an important contribution of agriculture to national well-being. Thus, it follows that the positive environmental services that flow from land-based assets should be attributed as an additional (but non-market) output of the sector.

Second, if the stock of land-based assets is changing – perhaps as a result of changing land-use – then the provision of future environmental services...
may also change. For example, if land is lost to housing development then the provision of say landscape amenities might diminish. This change in wealth could be measured as the loss of future (market and non-market) services from that lost stock of land (although a gain in wealth from the change is presumably enjoyed elsewhere in the economy).

Other environmental assets, such as water and air, generally are not managed or owned by the agricultural sector. Yet, the enjoyment that people receive from consuming their services might be affected by economic activity in the agricultural sector. Degradation of clean air and water (assets not managed by the agriculture sector) caused by agriculture typically can be construed as the change in the quality of an environmental resource stock. Thus, we can talk about changes in the stock of clean water, of clean air or of habitats in ‘good ecological status’. In the case of water, for example, it is the ability of the water environment to produce amenity services that is eroded through water pollution and abstraction that are considered as changes (or depreciations) in water assets. To the extent that these losses are attributable to activity in the agricultural sector, this should show up as a negative item or debit in an environmental account. However, if the sector is restoring any assets (perhaps by sequestering carbon through the planting of trees and so on) then this would be a positive item in the accounts.

This is a case of accounting reflecting a normative judgement that the (net) damage caused by the agricultural sector in the rest of the UK (e.g. pollution of waterways enjoyed by recreational users) and elsewhere in the world (e.g. releases of greenhouse gases) represent at least nominal claims against the income generated by the sector. This judgement seems justified if what is sought from an environmental account is an indication of the full social costs and benefits of agriculture.

Another distinction to make concerns to whom the costs and benefits of environmental outcomes accrue. Whether positive or negative, impacts of agriculture on the environment can affect either 1. society in general, resulting in an increase/decrease in social welfare or 2. other sectors, resulting in gains/losses to the productivity of those sectors. In practical terms, the former is typically measured with non-market data, such as that gathered by stated preference surveys; and the latter is most easily measured with market data: that is, prices. Where environmental impacts affect the performance of the agricultural sector itself, these are assumed to be evident already in the sectoral accounts, for example through reduced productivity.

Note that losses and gains to other sectors are ‘transfers’ in that accounting for these impacts does not alter the national accounts, but rather the measured allocation of value-added between economic sectors. From the perspective of measuring ‘true’ income in the agricultural sector and in a productive sector whose output is adversely affected, for example by
agricultural pollution, it is arguable that 1. the ‘true’ income in agriculture should be lower reflecting the fact that farming lowers output in another productive sector; and 2. the ‘true’ income of the affected sector should be correspondingly higher. This re- attribution of pollution costs between polluter and victim leaves national income unaffected.

Table 18.1 presents the set of adjustments that would need to be calculated in order to integrate the environmental impacts of agriculture into the current set of accounts. These adjustments could be made to the balance sheet, the production accounts or any other account to give a truer picture of income and wealth in the sector. The text in italics indicates that those adjustments are not relevant for agricultural accounts either because they are not attributable to the sector or they are included within other adjustments in the table.

**Accounting for Subsidies and Taxes**

One purpose of a set of environmental accounts is to record the value-added by the sector in question and to add in or net out any positive and negative non-market impacts (or externalities). The correct procedure is for the accounts to record these impacts regardless of the fact that they may be ‘internalized’ through policy measures. Consider the example of waste that goes to landfill. Such waste should be debited with the external costs of transportation of the waste to a landfill site and the relevant site externalities (disamenity, greenhouse gas emissions etc). In the UK, waste sent to landfill is subject to a landfill tax. Does the existence of such a policy instrument remove the need to account for the adverse impacts of waste that still ends up in landfills? The simple answer, from an accounting perspective, is arguably no. That is, a primary rationale for environmental accounts is to account more comprehensively for changes in the quantity and quality of natural assets regardless of the policy measures in place.

**Establishing the Baseline**

A particular challenge in building a framework for environmental accounts is the treatment of the counterfactual or the baseline. The framework here assumes a baseline of no agriculture. This translates into no agricultural activity or zero water or air emissions from agriculture or zero provision of landscapes, habitats and species associated with agriculture.

This baseline assumption might appear at odds with reality. For policy purposes ‘zero agriculture’ is not credible since it implies that we would have to stop eating beef and milking livestock or else import all agricultural produce from abroad. Assuming a world without agriculture also begs the
Towards green sectoral accounts for UK agriculture

Table 18.1 Adjustments to the agricultural accounts

<table>
<thead>
<tr>
<th>Adjustments for Welfare Impacts on Society</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>I. Water</strong></td>
<td></td>
</tr>
<tr>
<td><em>Flow not attributable to agriculture</em></td>
<td>Stock change (quantity and quality)</td>
</tr>
<tr>
<td><em>Value of water pollution arising from agricultural production</em></td>
<td></td>
</tr>
<tr>
<td><em>Flow not attributable to agriculture</em></td>
<td>Value of agricultural water abstraction</td>
</tr>
<tr>
<td><strong>II. Air</strong></td>
<td></td>
</tr>
<tr>
<td><em>Flow not attributable to agriculture</em></td>
<td>Value of air pollution arising from agricultural production</td>
</tr>
<tr>
<td><strong>III. Soil</strong></td>
<td></td>
</tr>
<tr>
<td><em>Impact of (net) soil erosion on-farm on current yields is already accounted for</em></td>
<td>Value of (net) soil erosion on-farm on future yields</td>
</tr>
<tr>
<td><strong>IV. Landscape</strong></td>
<td></td>
</tr>
<tr>
<td>Value of landscape amenity services by the current provision of landscapes (within the agricultural sector)</td>
<td>Value of (net) change in landscape amenities</td>
</tr>
<tr>
<td><strong>V. Habitats and species</strong></td>
<td></td>
</tr>
<tr>
<td>Value of habitat and species protection services provided by current land-use (within the agricultural sector)</td>
<td>Value of (net) change in habitats and species</td>
</tr>
<tr>
<td><strong>VI. Waste</strong></td>
<td></td>
</tr>
<tr>
<td>Value of waste pollution and disamenity arising from agricultural production</td>
<td></td>
</tr>
<tr>
<td><strong>VII. Nuisance</strong></td>
<td></td>
</tr>
<tr>
<td>Value of noise and odour disamenity arising from agricultural production</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Adjustments for Impacts on Other Sectors (Productivity Gain or Loss)</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>I. Water</strong></td>
<td></td>
</tr>
<tr>
<td>(-- Cost of clean up for water pollution)</td>
<td></td>
</tr>
<tr>
<td>(-- Costs of flooding)</td>
<td></td>
</tr>
<tr>
<td><strong>II. Air</strong></td>
<td>Included in value of stock change above</td>
</tr>
<tr>
<td><strong>III. Soil</strong></td>
<td></td>
</tr>
<tr>
<td><em>Impact of (net) soil erosion on-farm on current yields is already in existing accounts</em></td>
<td>(-- Cost of off-site soil erosion (cost of dredging streams, etc))</td>
</tr>
<tr>
<td><strong>IV. Landscape</strong></td>
<td>(+) e.g. Value of landscape to tourism</td>
</tr>
<tr>
<td><strong>V. Habitats and species</strong></td>
<td>(+) e.g. Value of habitats and species to tourism</td>
</tr>
<tr>
<td><strong>VI. Waste</strong></td>
<td>None</td>
</tr>
<tr>
<td><strong>VII. Nuisance</strong></td>
<td>(-- e.g. Cost of dealing with nuisance complaints)</td>
</tr>
</tbody>
</table>
question: if there was no agriculture what would be the alternative land-use? Clearly, this is an insoluble question – that is, presumably some land would revert to a rather different natural or semi-natural state while other land might be developed in some way, but what would happen, on average, would not be known with any precision. Much would depend ultimately on ‘unknowns’ such as future public policy. However, this is immaterial to the accounting exercise. National accounts are intended to provide a ‘snapshot’ of the world as it is now. This suggests that analytical anxiety about the future in the context of large-scale future reform of the agricultural sector, while important from a policy perspective, is not the domain of national or environmental accounts.

Accounting procedures work on the basis of ‘with and without’ the level of agricultural activity that prevails. This does not mean that zero agricultural activity is desirable, nor does it tell us anything about the desirable level of agricultural activity. Accounts simply tell us what positive and negative impacts arise from the prevailing level of agricultural activity, and, ideally, how those impacts will change when activity changes. Note also that zero agricultural activity is exactly the same baseline as that used by the conventional sectoral accounts. We measure UK agricultural output as the total market value of output (due allowance being made for subsidies), and that total market value has a baseline of zero agricultural activity. In other words, total market value approximates what the UK would have to pay for imports if there was zero agricultural activity in the UK. Since the implied baseline for measures of marketed output is zero, the same baseline is required for the environmental impacts.

This baseline assumption means that all of the positive impacts of agriculture (or environmental services provided by it) in a given accounting period should be quantified. The same holds for all of the negative impacts of agriculture in a given period. For the purposes of this framework, the monetary expressions of environmental impacts are the product of the unit economic value of the impact and the physical quantity of the impact. It is in the calculation of environmental impacts, specifically in the application of valuation data, that the treatment of the baseline becomes transparent.

EMPIRICAL APPLICATION TO UK AGRICULTURE

Methodology

Economic costs and benefits of an environmental impact are estimated using data on individuals’ (or households’) preferences for that impact. Preferences, in turn, are measured by people’s willingness to pay (WTP)
to maintain the current quality and quantity of environmental assets, to avoid a negative environmental impact or to secure a positive one. Similarly, preferences can be measured by people’s willingness to accept compensation (WTA) to tolerate a negative environmental impact or to forgo a positive one. Most of the studies relevant here use the WTP measure.

The first source of economic data on people's preferences is the market data for those environmental impacts that are traded or reflected in actual markets. These ‘marketed’ services of the environment include supply of drinking water, formal recreation or tourism and so on. In fact such market data (costs and prices) are akin to WTP or WTA of individuals in that market prices reflect the WTP of buyers and WTA of sellers. Note that in most cases, the market price paid for a good is only a lower-bound estimate of an individual’s WTP for that good and will not capture any consumer surplus (the difference between the price paid for a good and the maximum willingness to pay for that good).

The second source of economic data is non-market data. The two main approaches to quantifying the non-market values are revealed preference techniques and stated preference techniques. These techniques aim to measure individuals’ preferences directly and capture consumer surplus as well as market price-based expenditure when the latter exists. Revealed preference techniques use existing markets as surrogates for estimating the economic values for the environment. For example housing market data are used to estimate the value to households of cleaner air and reductions in traffic noise, by holding all other factors constant and observing changes in property value due to changes in environmental characteristics. Stated preference techniques create hypothetical, or simulated, markets by way of surveys to elicit individuals’ preferences for environmental changes. For instance, an example might be asking households to state their willingness to pay increased water bills for improvements in local river water quality. Generally speaking, market data are employed for impacts on other sectors, whilst non-market data are employed to estimate welfare losses to society. Some examples of the two data types are provided in Table 18.2.

Benefits transfer is the approach employed in accounting studies and the current chapter is no exception. This essentially involves borrowing estimates of non-market values from previous studies and applying them to a new, but similar, context. This study employs two benefits transfer approaches, mean WTP transfer and WTP function transfer (for landscape and habitats impacts). WTP values typically need to be adjusted into current monetary values (using a price inflator) and also converted to appropriate units (e.g. transforming WTP per household per year for a stated environmental change into WTP per hectare per year, WTP per kilometre of river per year, per bird species per year and so on).
Valuation and sectoral green accounting

Table 18.2  Market and non-market economic data for valuing environmental impacts

<table>
<thead>
<tr>
<th>Market Economic Data</th>
<th>Non-market Economic Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Costs to water companies of removing pesticides from water</td>
<td>• Welfare losses from degradation of environmental assets</td>
</tr>
<tr>
<td>• Monitoring costs (e.g. to the Environment Agency) [not used in this study]</td>
<td>• Welfare losses from damages to human health</td>
</tr>
<tr>
<td>• NHS costs for treating human health impacts [included in the costs of air pollution]</td>
<td>• Welfare losses from degradation of animal and plant species</td>
</tr>
<tr>
<td>• Costs of restocking fish stocks lost due to pollution incidents</td>
<td>• Welfare gains from environmental services provided by the current stock of environmental assets [included in this study for land-based assets under agricultural control]</td>
</tr>
</tbody>
</table>

The calculations presented below according to impact category provide monetary estimates of the environmental impacts of agriculture (subject to data availability) that can be used in an income accounting exercise. They do not cover wealth accounting (accounting for the value of the stock of assets) since it is more meaningful to quantify the services provided or impacted upon by agriculture than to quantify the value of the entire set of environmental assets, most of which are beyond the province of agriculture and hence are not attributable to agricultural accounts. In practical terms, this means that the calculations presented here are in £ per year terms (representing service flows or reductions in income).

Thus, the time period for the calculations is one year, in line with standard practice. In order to produce a yearly account of such impacts, one essential component of the environmental accounts is annual data for the physical impacts (such as the state of water quality in rivers each year or the number of hectares of land providing landscape benefits). This raises a data availability issue, as some physical data relating to various aspects of – and impacts from – agriculture are not collected or collated on an annual basis.

For most impacts, the entire UK population is taken as the relevant population (a more sophisticated approach is used in the valuation of landscape benefits that relate WTP to regional populations and characteristics). Exemptions include the larger (global or European) populations impacted by global and regional air pollution from the UK. These population estimates are embedded in the unit economic costs used here.
In line with the objectives of the study, country-level breakdowns of the calculations are presented in the results table (Table 18.3). This breakdown has not been possible for all impacts due to the impact data not being collected or presented at the country level. Where country level impact data are available, it is sometimes the case that economic data for the UK as a whole or from a particular location are applied uniformly to all countries.

As with all exercises that entail economic valuation, and, indeed, other empirical or statistical procedures, sensitivity analysis should be conducted in order to reflect areas of uncertainty. Therefore, where possible, from the available literature, a range of economic estimates have been used to calculate lower and upper bound values as well as a central one. On the flip side, expert judgement has been employed to try and narrow these ranges as much as is possible, and all summary data are presented in using midpoints or central estimates.

**Aggregate Results**

Table 18.3 presents estimates of the total costs and benefits of the environmental impacts of agriculture in the UK. The table is complete only so far as the available physical and economic data allow. Where ‘n/a’ appears this indicates missing country-level data, and bold type indicates that the data cover more than one country (for example, it is often the case that only UK-level data are available). Notes to the table indicate some overlap in values calculated for habitats and species. Elsewhere in the table distinctions are made between coastal and inland water pollution, local and global air pollution and between sectors upon which water clean up costs are imposed (i.e. water companies for drinking water clean-up, and the government – through the Environment Agency – for restocking of fish following pollution incidents). Note that monetary values presented in the table have been rounded up or down, which might mean that the totals of the rounded estimates do not equal the rounded totals.

Following is a brief presentation of the calculations for individual impact categories, following the format of Table 18.3. For most calculations a central estimate is presented, which is derived from a range of values presented in the literature. No discussion is provided for the nuisance category as economic data were not available to complete the analysis of this impact category. However, the quantitative data available show that this impact is not likely to be significant at the national level. A more general point worth noting, is that we do not use the data in Table 18.3 to adjust agricultural value-added – for reasons that we discuss in more detail below. However, this calculation would be relatively straightforward. Positive items would boost
Table 18.3  Estimated monetary adjustments to agricultural accounts (£million 2003, central estimates)

<table>
<thead>
<tr>
<th>Impact Category</th>
<th>Accounting Adjustment</th>
<th>England (E)</th>
<th>Wales (W)</th>
<th>Scotland (Sc)</th>
<th>N. Ireland (NI)</th>
<th>UK</th>
</tr>
</thead>
<tbody>
<tr>
<td>Adjustments for Welfare Impacts on Society:</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>I. Water</td>
<td>Value of water pollution arising from agricultural production</td>
<td>Inland</td>
<td>£48</td>
<td>£1</td>
<td>£14</td>
<td>£7</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Coastal</td>
<td>£3</td>
<td>n/a</td>
<td>n/a</td>
<td>E &amp; W only</td>
</tr>
<tr>
<td></td>
<td>Value of agricultural water abstraction</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>II. Air</td>
<td>Value of air pollution arising from agricultural production</td>
<td>Global</td>
<td>£543</td>
<td>£109</td>
<td>£143</td>
<td>£95</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Regional/Local</td>
<td>£43</td>
<td>£7</td>
<td>£10</td>
<td>£7</td>
</tr>
<tr>
<td>III. Soil</td>
<td>Value of (net) soil erosion on-farm on future yields</td>
<td></td>
<td></td>
<td>n/e</td>
<td></td>
<td></td>
</tr>
<tr>
<td>IV. Landscape*</td>
<td>Value of landscape amenity services by the current provision of landscapes (within the agricultural sector)</td>
<td></td>
<td></td>
<td>£124</td>
<td>£321</td>
<td>£45</td>
</tr>
<tr>
<td>V. Habitats and Species*</td>
<td>Value of habitat and species protection services provided by current land-use (within the agricultural sector)</td>
<td>Habitats</td>
<td>£225</td>
<td>n/a</td>
<td>n/a</td>
<td>E only</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Species</td>
<td></td>
<td>n/a</td>
<td>n/a</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>+£307</td>
<td></td>
<td></td>
</tr>
<tr>
<td>VI. Waste</td>
<td>Value of waste pollution and disamenity arising from agricultural production</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>£15</td>
</tr>
<tr>
<td>VII. Nuisance</td>
<td>Value of noise and odour disamenity arising from agricultural production^2</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>n/e</td>
</tr>
</tbody>
</table>

Adjustments for Impacts on Other Sectors:

<table>
<thead>
<tr>
<th>Impact Category</th>
<th>Accounting Adjustment</th>
<th>England (E)</th>
<th>Wales (W)</th>
<th>Scotland (Sc)</th>
<th>N. Ireland (NI)</th>
<th>UK</th>
</tr>
</thead>
<tbody>
<tr>
<td>I. Water</td>
<td>Cost of water pollution clean up costs</td>
<td>Gov’t</td>
<td>£0.3</td>
<td>£0.1</td>
<td>n/a</td>
<td>E, W &amp; Sc only</td>
</tr>
<tr>
<td></td>
<td>Costs of flooding</td>
<td>Water Company</td>
<td></td>
<td></td>
<td></td>
<td>£181</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>£153</td>
<td></td>
</tr>
<tr>
<td>II. Air</td>
<td>Included in measure above</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>III. Soil</td>
<td>Cost of off-site soil erosion</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>£9</td>
</tr>
<tr>
<td>IV. Landscape</td>
<td>(+) e.g. Value of landscape to tourism</td>
<td></td>
<td></td>
<td></td>
<td>n/e</td>
<td>n/e</td>
</tr>
<tr>
<td></td>
<td>(+) e.g. Value of habitats and species to tourism</td>
<td></td>
<td></td>
<td></td>
<td>n/e</td>
<td>n/e</td>
</tr>
<tr>
<td>VI. Waste</td>
<td>None</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>VII. Nuisance</td>
<td>(-) e.g. cost of dealing with nuisance complaints</td>
<td></td>
<td></td>
<td></td>
<td>n/e</td>
<td>n/e</td>
</tr>
</tbody>
</table>

Notes:
* These two categories overlap, which means that some element of habitat and species will be captured in the landscape valuations, and some element of landscape value will be captured in the habitat valuations.
n/e: Not estimated due to lack of physical and/or economic data.
any measure of value-added and, by contrast, negative items shrink value-added. As a point of reference for judging the relative size of magnitudes in Table 18.3 in 2003 UK gross value-added in agriculture was about £7950 million with net value-added at £5340 million (with the difference between these two values due to depreciation of produced assets in the sector).

**Water: Adjustments for Welfare Impacts on Society**

**Water quality in rivers**
Farming is one of four major sources of water pollution in the UK. A distinction can be made between diffuse pollution, arising from the spreading of nutrients on the land, for example, and point sources that include run off from livestock buildings. Key areas of concern in relation to water quality are nitrate pollution in surface and groundwater, phosphorus levels in surface water, contamination by pesticides and harmful effects of soil sediments and mineral salts. A decline in water quality can result in impaired drinking water quality, resulting in potential health implications, as well as environmental problems.

For the purposes of calculating welfare impacts to society from depreciation on inland (or surface) waters the results from Georgiou et al. (2000) were used. Applying the willingness to pay estimates from this study to all UK rivers provides a measure of the annual welfare impact on society from loss of amenity associated with rivers classified as ‘fair’ or ‘poor’ (or the depreciation of water as a natural asset) of £5560 per kilometre of river. However, not all of the pollution causing this impact is from agriculture and hence not all of this cost can be attributed to it. Environment Agency (2002) suggests that 70 per cent of nitrogen entering surface waters is from agriculture (on the basis of WRc, 1999). Hence this proportion is attributed to chemical water quality in this report, where chemical water quality is total ammonia, biochemical oxygen demand and dissolved oxygen. For the UK, this results in a cost of around £71 million per year (≈ £5560/kilometre × 18 132 kilometres of river considered ‘fair’ or ‘poor’ × 0.7).

**Coastal water quality**
Environment Agency (2002) estimates that 5 per cent of faecal pathogens in bathing waters is accounted for by agricultural sources. The welfare cost of this impact is calculated using eftec (2002), which estimated WTP for avoiding stomach upsets associated with bathing in faecally contaminated bathing water on British beaches to be £61.8 million when aggregated across England and Wales. This results in a cost of around £3 million per year (≈ £61.8 million/year × 0.05).
Water availability
Agricultural production affects the quantity of water available for other human consumption and natural processes in a variety of ways. The most direct way is through abstraction of water and the diversion of watercourses for irrigation. The use of water for agriculture can conflict with other human and ecosystem needs. For example, over-abstraction is leading to the decline of aquatic habitats in some areas. Agricultural water use does not represent a large proportion of water abstracted being about 1 per cent of the total in water use in the late 1990s. The total abstraction of water by agriculture, including for spray irrigation, was 365 megalitres per day in 2001 (DEFRA e-digest, 2003).

eftec (2003) gives a value of the benefit to society from reducing abstraction and leaving more water in the environment. This benefit estimate, when applied to the amount of water abstracted by agriculture (367 ML/day, both abstraction and spray irrigation), gives the cost of abstraction (or avoided benefits) to society. In order to match the units of economic data (£ per cubic metre per day) to physical data (ML), the economic cost estimate was multiplied by 365 (days in year) and by 1000 (number of cubic metres in a megalitre). For England and Wales, for which abstraction data (excluding spray irrigation) are available, this means a cost of around £36 million per year (= £0.27 per cubic metre per day × 367 × 365 × 1000).

Water: Adjustments for Impacts on Other Sectors
Water quality
Two impacts under this heading can be expressed in monetary units: the cost of restocking fish populations in rivers after pollution incidents from point sources and the cost of treating drinking water for nitrates, phosphates and pesticides.

Point sources of pollution are associated with housed livestock and other facilities such as slurry stores, silage-making clamps, pesticide stores, crop treatment processes and so on. Dairy farms are one of the main sources of surface water pollution, often associated with the run off of slurry, yard washings, silage effluent and milk in some cases. Serious pollution incidents often involve sharp increases in biological oxygen demand (BOD) and fish kills, causing considerable damage to aquatic ecosystems as well as contributing to nutrient enrichment in most cases. The welfare impacts of reduced river water quality from point sources would be subsumed in the calculations above. Here the financial or market costs are considered.

Environmental data show that there were 13 Category 1 and 137 Category 2 pollution incidents in England and Wales in 2002 (eftec and IEEP, 2004, Table 3.2, p. 35) and 56 ‘significant’ pollution incidents in Scotland.
Pretty et al. (2000), it is assumed that only Category 1 and 2 incidents (as defined by the Environment Agency) are severe enough to result in fish kills and that ‘significant’ incidents in Scotland are deemed to be sufficiently similar to Categories 1–3 in England and Wales. The cost of restocking fish was reported in Pretty et al. (2000) and these values updated to 2003 prices are used here. For England and Wales, this results in a cost (central estimate) of about £0.05 million per year for Category 1 (= £3772 per incident × 13 incidents) and £0.22 million per year for Category 2 (= £1886 per incident × 118 incidents) – a total of £0.27 million. The Category 2 costs are used for ‘significant’ pollution incidents in Scotland, which gives a cost of £0.1 million. These are attributed to government to indicate that the Environment Agency is paying for the damages, and not the private sector.

Market cost data are also used to estimate the costs of removing nitrate, phosphates and pesticides from drinking water. Following Pretty et al. (2000), capital and operating costs of water treatment per year for the period 1992 to 1997 are used in conjunction with estimates of agriculture's contribution to nitrates, phosphates and pesticides in the water supply. Since more up-to-date figures were not available from the water sector at the time of the study, they are used here as updated to 2003 prices. The share of agriculture’s contribution to emissions of nitrates (70 per cent), phosphates (43 per cent) and pesticides (89 per cent) in drinking water are assumed to apply to the whole country (Environment Agency, 1998). For the UK, this means a total cost of about £181 million per year (= £22.5 million × 0.7 + £80.2 million × 0.43 + £147.4 million × 0.89).

These results point to significant potential gains in terms of both improvements to social welfare (see above) and economic gains to other sectors if water pollution from agriculture can be further reduced.

Water availability (flooding)
There is much concern about the potential link between agricultural practices, run off processes and flooding. However, the nature of this link remains uncertain. The Environment Agency (2002) attempts to draw a link based on its own flood event data and finds that 25 per cent of all flooding events in the 1980s and 1990s were hillslope events. On the basis that 57 per cent of hillslope events were caused by erosion and deposition, the report concludes that at least 14 per cent of all flood events, and flood event costs, should be attributed to agriculture. The Environment Agency purports that this estimate is likely to be conservative due to under-reporting of local agriculture-related flooding events and also lack of data linking agriculture's contribution to fluvial flooding.

The Foresight. Future Flooding report (Evans et al., 2004) provides estimates for the present day annual economic damage to the UK from
flooded of £1.09 billion (in 2003). If the Environment Agency approach is adopted, then the cost of flooding that could be attributed to agriculture would be about £153 million per year (= £1.09 billion \times 0.14).

Table 18.3 shows that the largest cost of agriculture in terms of water quality and quantity is its share in the cost of flooding. This is also the least certain of the impact categories depending entirely on assumptions about agriculture’s contribution to the problem, which merits further research.

**Air**

Agriculture’s contribution to the level of atmospheric pollution is two-fold. Depending on the type and scale of production methods, land managed for agriculture can act as a sink for pollutants already in the atmosphere or emissions from manures and the soil itself can increase pollution levels. The main gas fluxes (in volume terms) from agriculture are ammonia, carbon dioxide, methane and nitrous oxide. However, the levels of specific volatile organic compounds such as hexachlorobenzene and methyl bromide are also affected. The processes that result in agriculture influencing the flux of gases and the effects these additions and removals have are both highly varied and compound-specific.

Physical data on airborne emissions from agriculture are provided by Netcen for this study. These data are also reported in Agriculture in the United Kingdom (DEFRA et al., 2004) and are used in the ONS Environmental Accounts. The data are regularly updated and reliable, and also provide a good match with the economic data, which estimate the damages associated with a tonne of each air pollutant. These are matched with the unit damage cost for the pollutants covered, namely, £400 per tonne of methane and £5588 per tonne of nitrous oxide (from Eyre et al., 1997), between £87 and £270 per tonne of ammonia (Holland et al., 1999), £190 to £897 per tonne of nitrogen oxide, £744–£2296 per tonne of sulphur dioxide, £1.4 per tonne carbon monoxide (AEA Technology, 2004), £70 per tonne of carbon (Clarkson and Deyes, 2002; see, for a critique, Pearce, 2003) and £1367 per tonne for non-methane volatile organic compounds (Pearce and Howarth, 1998). The impact of those airborne pollutants covered in this analysis on the welfare of the society results in a total cost (central estimate) of £956 million per year (equal to the sum of global and regional/local emissions, where the global emissions represent 93 per cent of the total). While some of these costs are due to productivity losses (e.g. damage to materials or medical costs incurred due to impacts on health), these were not possible to disaggregate within the scope of this study.

The range of air pollution costs presented here is high compared with £585 million estimated by Hartridge and Pearce (2001) and £393 million
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estimated by Pretty et al. (2000). This increase is partly due to updated unit damage cost estimates and partly due to a larger set of pollutants and impacts being valued.

As Table 18.3 shows the majority of the cost of air pollution reflects the global damages of greenhouse gas emissions. While £889 million for global air pollution represents a sizeable cost, it is worth keeping in mind that agriculture accounts for only 7 per cent of the UK’s greenhouse gas emissions. Finally, while these costs appear large, they are still likely to be underestimates, as a number of the impacts of air pollution are not included in the economic data applied here due to a lack of research. These impacts include impacts on ecosystems, damage to cultural heritage, effects of ozone on materials and additional health effects.

Soil

Soil is a valuable but vulnerable resource. It takes hundreds of years or more to develop, but mismanagement can result in its loss and degradation, sometimes quite rapidly with the greatest deterioration taking place on the most vulnerable land. Many farming operations involve working the soil, hence altering its chemical composition and structure. Exposing it to the elements and inappropriate management can lead to both on- and off-farm problems. On-farm, many of the impacts of soil erosion are as a consequence of particles becoming progressively coarser grained. The loss of fine grains results in reduced water storage capacity of the soil, loss of nutrients or fertility and deterioration of soil structure (Skinner et al., 1997). Off-farm impacts of soil erosion include damage to property and accidents (if deposited on other land or roads) and flooding (if deposited in rivers, ditches and drains). In addition, there are also negative impacts on aquatic habitats.

As reported in Pretty et al. (2000) and Environment Agency (2002), Evans (1996) estimates on the basis of data from local authorities, that annual damage in the UK from off-site soil erosion is approximately £9.2 million (updated to 2003 prices). This figure includes damage to roads, footpaths, property and most significantly channel degradation from soil erosion of all causes. Using the data from the Environment Agency (2002) which suggests that 95 per cent of all off-site soil erosion is caused by agriculture, we can attribute 95 per cent of that cost to agriculture. This calculation shows a cost of £8.8 million per year (= 0.95 × £9.2 million). Clearly this is an average estimate, as erosion levels will vary considerably between years.

The on-farm impacts of soil erosion on the current yields are already reflected within income in the existing sector accounts, while the impacts of soil erosion on future yields are not captured in the existing accounts
and are not possible to quantify here due to a paucity of economic data and research in this area.

**Landscape, Habitats and Species**

The stock of land held by agriculture represents the majority of land in the UK (Haines-Young et al., 2000). This environmental asset provides, among other flows of services, landscape amenity and habitat and species provision. Agriculture plays a key role in shaping the quality of the national ‘stock’ of landscape. Indeed agricultural landscapes are the visible outcome of the interactions between agriculture, natural resources and the environment and encompass amenity, cultural and recreational values. Certain agricultural practices are essential for the survival of designated UK habitats and species, while other practices can be damaging. In accounting for the services provided by the land managed or impacted by (and hence attributable to) agriculture in any one year, four sets of physical data are used:

- **stock of broad habitat types** considered to be managed by agriculture (e.g. arable lands, heathland, grassland, etc), differentiated by:
  - land designated as Site of Special Scientific Interest (SSSI) (data available for England only) and
  - land not designated as such (non-SSSI) (data available for England only);
- **stock of linear features** managed by agriculture (such as hedgerows, stone walls);
- **arm woodland**;
- **populations of keystone species** related to agriculture (in this case, farmland bird population data are used).

The use of the above reflects the availability of physical and economic data, and the potential to match the two, which requires common units. The Environmental Land Features (ELF) model (IERM and SAC, 2001) provides economic value estimates per hectare that can be applied to broad habitat classification type, allowing the landscape value of most of the land area under agricultural management to be accounted for, when matched with data from the ‘Countryside Survey’ 2000 (Haines-Young et al., 2000) on broad habitat type coverage. The ELF model provides a validity tested benefits transfer model based on the majority of the existing literature on UK landscapes and habitats. Per hectare values range from £21 to £135 per hectare, depending on habitat type (linear features such as neutral grassland, bog, dwarf shrub heath, acid grassland, fen, marsh and swamp, calcareous grassland and farmed woodlands). The total annual value of landscape
services from all of these habitat types combined is about £488 million per year (≈ £21/hectare × 2.2 million hectares + £84/million hectares × 69.7 million hectares + £92/hectares × 201.4 million hectares + £23/million hectares × 32.2 million hectares + £84/hectares × 108.7 million hectares + £92/hectares × 55.3 million hectares + £84/hectares × 2.4 million hectares + £135 million hectares × 69.4 million hectares).

Separate economic research on the value of SSSIs (Willis, 1990) provides monetary data for ‘special’ sites within this stock of land (£718 per hectare). The data provided on the state of SSSIs in England by English Nature (2003) allow these economic value estimates to be employed to provide a fuller picture of environmental value and agriculture’s contribution in England. The use of Willis (1990) is interpreted as providing an estimate for the provision of biodiversity services (or habitats and species services in the language used in the accounting framework).

When applying economic valuation data to land-based amenities, it is usually the case that all services generated by one type of land (or one specific site) are valued as a bundle of goods – so that WTP per hectare reflects all these aspects of the environment. Thus, results from the ELF model, when applied to broad habitat types will typically include valuations for both use (recreational) and non-use (existence value) of the flora and fauna found on those sites, as well as for the landscape amenity.

In the interpretation of physical data from the English Nature report, it has been assumed that the proportion of SSSIs under agricultural management that are in unfavourable condition provide no biodiversity services, but provide landscape services, and thus these hectares are valued using the ELF study estimates described above. The biodiversity services provided by those in favourable condition are attributed to agriculture. These total £225 million to England and Wales in 2003. Note that while accounting for habitats in this way will result in a positive adjustment to the income accounts, any losses in hectares of SSSIs or degradation in quality of SSSIs will become evident when comparing the income from these assets in the current accounting period with those in the previous accounting period (conversely, any gains will clearly be evident in this way as well). While this provides a manageable approach and a useful starting point for the framework environmental accounts, a better handle on the true extent to which agriculture is the provider of all of these benefits is required for these calculations to better reflect reality.

However, the data on woodlands under farm management are taken from farm woodland scheme data due to the difficulty of establishing the extent to which all woodlands under the broad habitat classification were on agricultural land. It is likely that the data on woodlands is an
underestimate of total area of woodland managed as an inherent part of agricultural enterprises.

Economic data provided by Foster et al. (1998) are used to calculate the economic importance of farmland birds, and agriculture’s role in providing habitats for them. The health of farmland bird populations is an important indicator of overall farmland biodiversity and hence the economic quantification of this indicator is a useful addition to the accounts. RSPB’s farmland bird index (Gregory et al., 2003) employs the 1970 bird population level as a baseline to measure current bird populations. In 2003, these were at 60 per cent of 1970 levels (an index of 60). From Foster et al. the annual welfare value of avoiding the decline to zero of nine farmland species is £246 million per year. This is adjusted to provide a value to society ‘per species per population %’ of £5.12 million/year. For 2003 this gives an indicative value of biodiversity services from agriculture of about £307 million (= £5.12 million/year × 60).

As with the other adjustments calculated here, the value of farmland birds will appear as a positive income adjustment to the accounts. This reflects the baseline of ‘no agriculture’ and the role that agricultural systems have in providing for these birds. However, if the accounting exercise were repeated for the year 1970 (when the farmland bird population was at 100) then the comparison of the service flows provided by this adjustment in the two periods would reveal a steep drop in economic value of this asset (and by inference a loss in wealth and income to the sector).

The scale of the benefit accrued from agricultural practices that maintain landscapes, habitats and species valued by society can be compared with the payments made to farmers to provide these services. While the data only allow a crude comparison, total payments to agri-environment schemes totalled £387 million in 2002 (DEFRA et al., 2004), which is far outweighed by the magnitude of benefits recorded in Table 18.3. Currently the data on the physical environment are the most limiting factor in achieving time series comparison of such costs and benefits.

Gaps in the economic literature have meant that it was not possible to account for the visual (dis)amenity of intensive agricultural land-use (following the broad habitat types, this refers to improved grassland, and non-SSSI-designated arable land).

Waste

Agriculture generates a considerable array of waste products ranging from typical waste materials such as packaging to agriculture-specific wastes such as pesticide washings. The wastes produced in the greatest quantities by agriculture are organic, for example, silage effluent, dirty
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water, milk, blood, vegetable washings, animal carcasses and crop waste cuttings, with agriculture producing an estimated 90 per cent of all organic waste (Environment Agency, 2002). In addition to organic wastes, there are also inorganic wastes amounting to approx 0.5 million tonnes per year for England and Wales (Environment Agency, 2001). These include agrochemical washings and concentrates, and wastes relating to animal health waste such as unused medicines, syringes and building wastes.

It was only possible to employ data on general waste from agriculture, in line with what could reasonably be used in the accounts and applied to economic data. A Framework for Environmental Accounts for Agriculture (eftec and IEEP, 2004) presents data on general waste arisings, which were 1 million tonnes in 1999. An estimated cost of £15 per tonne of waste is used in this report (derived from COWI, 2000). This results in a cost for agriculture of £15 million per year (= £15 per tonne × 1 million tonnes).

DISCUSSION AND CONCLUSIONS

A full set of environmental accounts for agriculture, complete with an understanding of the economic value of the environmental impacts would be an invaluable resource for the UK. Already, the data presented in this chapter provide a useful reference point. Overall, however, we are only at the beginning of a longer process and much additional data and coordination are required to see the full benefits of a monetized environmental account. In this chapter, we have shown that these adjustments may well be substantial. Some of these items are positive, as in the case of the provision of current flows of environmental services from land assets managed within the sector, while others are negative. On balance, it appears that in 2003 the value of negative items outweighed those that were positive. Similarly, it appears that the contribution of the sector to the net accumulation of environmental assets could well be negative. There are substantial caveats to this assessment. For example, it is based on an incomplete evaluation due to data limitations and gaps. Moreover, many of those items for which data exist, such as the social cost of carbon, are highly uncertain.

This, in turn, raises broader issues. The valuation of the environmental impacts of agricultural activity, within an accounting framework, is a highly useful means of thinking about the contribution of the sector to well-being or sustainability in general. Nevertheless, an interpretation of the uncertainties inherent in at least some of the data generated in this respect as well as data gaps that we have identified is that it may be premature to use these data to adjust national accounting aggregates. Put another way,
prudence is arguably warranted if an objective is to move beyond academic applications to use in the public policy process.

A further reason for this conclusion is that it arguably makes little sense to provide a ‘green’ account for agriculture that rests on distorted (i.e. subsidized) agricultural product prices. In addition to accounting for the value of environmental impacts, a ‘green’ account for agriculture (ideally) should embrace shadow pricing of produced outputs. However, while this clearly would be important for enhancing the policy usefulness of the accounts, the latter raises many complex issues (e.g. Hartridge and Pearce, 2001) and, hence, we have not sought to use our findings to adjust agricultural value-added.

The benefits of a more fully developed analysis will not be limited to agriculture as other sectors and government departments are sure to see the benefits of efforts made to collect more environmental and economic data to improve our understanding of these complex agriculture–environment–economy interactions. The following provides a brief analysis of some of the gaps in our understanding and some suggestions for future work.

There are numerous impacts for which scientific and economic data are lacking. These gaps on the whole arise as a result of the following reasons:

- There may be no scientific or economic data for an environmental impact.
- There may be physical data about an environmental impact but the share of agriculture as one of the sources of that impact is not known.
- There may be no economic valuation data for an environmental impact, even where agriculture’s contribution is understood.

These data gaps are pervasive throughout all impact categories, and also have implications for the ability of the data to provide annual estimates, geographical dis-aggregation of estimates, and so on. In order to improve the data availability in future, designated contact points within each major institution that collects and collates data are required. They would be able to assist with the identification of the most up-to-date and appropriate data sets and coordinate to collect data more regularly and consistently to avoid current inconsistencies and any future duplication of effort.

Including in the ‘Countryside Survey’ questions on whether the different habitats surveyed were under agricultural management would provide a more accurate view of agriculture’s stewardship role in providing landscape, habitat and species benefits. Ideally, such efforts would attempt
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to measure qualitative change for these habitats in terms of biodiversity (habitat and species) that could be linked to land and landscape types. These measurements could learn from OECD work on indicators (OECD, 2001), notably the Natural Capital Index.

More research is required to establish the nature of the relationships between agricultural practice, non-farming factors and the population dynamics of different species, in order to augment current data on keystone species, such as farmland birds. New work in establishing transferable economic values for changes in river water quality should employ the Environment Agency’s GQA index as the measure of quality as it is a good accounting measure and groups different environmental effects together. Ideally a benefits transfer model similar to that developed for the ELF model (IERM and SAC, 2001) for landscapes could be developed for the water environment, which would allow sensitivity for different regions to be incorporated.

Additional economic research linking agriculture’s contribution to overall water quality, and its responsibility for the resultant economic costs, is also required to improve calculations of accounting aggregates. Considering that the merit of this research for future Periodic Review processes for water companies, as well as for meeting Water Framework Directive requirements, it should be a priority.

More research is required to fill the gap in knowledge about the economic value of ecological impacts on the marine environment such as changes in marine flora and fauna, and specifically agriculture’s contribution to these changes.

Research into the potential social value of intensive agrarian landscapes would help complete the picture in this category of impact. The current literature concentrates on extensive agriculture such as the semi-natural habitats and the designated areas. Further research in this area could make use of computer-generated visual aids to allow a differentiation between different landscapes, including future possible landscapes. Such work could augment the ELF benefits transfer model employed here and benefit from visualization lab technology for informing the public’s preferences.

The UK government is clearly keen to advance green accounting, and work is furthest ahead in the agricultural sector. Enthusiasm for the task should not be diminished by the data challenges faced. The work presented in this chapter demonstrates that a sensible framework can be constructed to accommodate the complexities of the relationship between agriculture and the environment. What remains is to build up the base of knowledge and thereby populate this framework with data that will add strength to the exercise over time. The lack of data also points to another reason for constructing accounts: that they can eventually tell us what we do not know.
NOTES

1. This chapter is based on research carried out for the UK Department for Environment, Food and Rural Affairs (DEFRA), Department of Agriculture and Rural Development (Northern Ireland) (DARDNI), Scottish Executive and the Welsh Assembly in response to a call for a study to ‘develop methodologies enabling the government to produce an account for agriculture that includes an adjustment for the environmental impacts of the sector’. The views expressed in this chapter are not necessarily shared by these bodies.

2. Pioneering studies in this field include Adger and Whitby (1993) and Whitby and Adger (1996). For example, the latter estimates the value of asset changes in the UK agricultural sector in order to discuss the contribution of agriculture to net wealth creation.

3. There remains an unresolved debate between weak or strong sustainability. The former emphasizes changes in the real value of wealth in the aggregate while the latter (typically) also emphasizes the conservation of critical natural capital (i.e. critically important resources for which there are essentially no substitutes). On the one hand, the approach in this chapter would be characterized by many as being based on the weak sustainability principle. On the other hand, if WTP values used to determine changes in environmental and resource stocks reasonably accurately reflect substitution possibilities between different assets then this debate about which camp our findings fall into is rendered largely superfluous.

4. Category 1 pollution incidents are the most severe resulting in persistent and extensive effects on quality, major damage to the ecosystem, closure of a potable abstraction, major impact on property, major impact upon amenity value, major damage to agriculture and/or commerce and serious impact upon man. Category 2 is significant but less severe. Category 3 relatively minor.

5. Using this value follows official (DEFRA) guidance about the social cost of carbon. At £70/tC this is at the high end of findings of such studies (Pearce, 2003).

REFERENCES


DEFRA (Department for Environment, Food and Rural Affairs), Scottish Executive Environment and Rural Affairs Department, Department of Agriculture and Rural Development Northern Ireland and Welsh Assembly Government’s Department of Environment, Planning and Countryside (2004), Agriculture in the United Kingdom, London: The Stationery Office.
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INTRODUCTION

The implementation of the European Water Framework Directive (WFD) has increased policy and decision-maker demand for integrated hydro-economic information at the level of river basins. In order to meet this increasing demand, a new integrated hydro-economic accounting system has been developed in the Netherlands, the National Accounting Matrix including Water Accounts for River Basins (NAMWARiB). NAMWARiB provides information about the interactions between the physical water system and the economy at national and river basin scale. This involves not only linking water-related environmental data to economic data, but also presenting these data at the relevant spatial scale through appropriate up- and downscaling procedures. It is this issue of matching available data across various spatial scales that has proven to be one of the major challenges in the compilation of the new integrated river basin information system. According to the European Task Force on Water Satellite Accounts (2002, p. 4), the implications of the WFD include more focus on the geographical boundaries of the data, that is, water bodies and river basin districts. Also the new Handbook on Integrated Environmental and Economic Water Accounting, drafted by the UN acknowledges this: ‘it is important that the spatial reference is the same for hydrological and economic data’. However, ‘river basins for which hydrological data is usually available, do not generally coincide with administrative regions for which economic information is collected’ (United Nations Statistics Division, 2006, p. 17). In fact, no consistent water economic information system exists yet, which allows for the consistent disaggregation and aggregation between the national and river basin level. Most efforts in the area of integrated water accounting are based on national statistics.

The main objective of this chapter is to present and discuss a new up- and downscaling procedure developed in NAMWARiB and the challenges faced
when using this procedure. The next section briefly presents the structure of NAMWARriB. The third section presents the up- and downscaling procedure used to translate the available environmental and economic data related to water use to the level of river basins. The fourth section presents some examples of integrated indicators at river basin level and section 5 concludes.

SYSTEM OF INTEGRATED WATER ACCOUNTS

In the beginning of the 1990s, the National Accounting Matrix (NAM), the official account of the economic transactions in a particular year in a country, was extended in the Netherlands with a ‘satellite account’ to the National Accounting Matrix including Environmental Accounts (NAMEA). NAMEA includes the environmental pressures related to the production of goods and services and the consumption of households (de Haan et al., 1993; Keuning, 1993; de Haan and Keuning, 1996). NAMWARiB uses the same basic structure as the NAMEA, but was set up specifically for water. Within this structure, each column represents the supply of a good or service, whereas the rows describe the demand for those goods and services. The monetary flows are in exactly the opposite direction: columns represent receipts and rows represent expenditures. The total of the columns equal the total of the rows. Together rows and corresponding columns make up an account for a specific good or service, reflecting where it comes from and where it goes.

Basically, the structure of NAMWARiB consists of three parts (Figure 19.1):

- an economic account (the first ten accounts, all in millions of euros per year);
- a water extraction and discharge account (account numbers 11 and 13 in millions of cubic metres per year);
- an emission account (account numbers 12 and 14 in kilograms per year).

Whereas the conventional economic accounts 1–10 are all in millions of euros, the water extraction and discharge and emission accounts are expressed in physical units. NAMWARiB describes the emissions of 78 substances to the aquatic environment originating from economic activities. The emission balance consists of two accounts, account number 12 and account number 14. The columns of account number 12 present the emissions by consumers and producers, and the import of transboundary pollution, while the rows of account number 12 describe the destination of the substances. Substances can
be absorbed by producers (during the production process), environmental services (wastewater treatment), exported or contribute to one of or more of the environmental themes distinguished in NAMWARiB: eutrophication, wastewater and dispersion of heavy metals. The environmental themes link account number 12, which describes water pollution, to account number 14, which describes the contribution of the various emitted substances to the various environmental themes. Account number 14 presents the actual pressure on the environment caused by a particular substance in the water environment. Excess amounts of nitrogen (N) and phosphorous (P) contribute to eutrophication, while emissions of heavy metals contribute to the dispersion of heavy metals in the water environment.

The use of water by different economic activities is described in two accounts: water flows (account number 11) and changes in water stocks (account number 13). The water flow accounts, that is, water extraction and discharge, are expressed in millions of cubic metres (m$^3$) per year. Account number 11 describes the extraction of three types of water sources: groundwater, surface water and tap water. For groundwater a distinction is furthermore made between fresh and brackish groundwater and for surface water between fresh and salt water. The rows describe water consumption by households, different branches of industry and other sources including water losses as a result of evaporation. Water consumption is furthermore broken down into water consumption for cooling water purposes and other purposes. Total freshwater use equals the use of drinking water, fresh groundwater and fresh surface water minus freshwater used for cooling purposes (cooling water is only extracted temporarily and recycled again into surface water). Account number 13 describes the changes in stocks and is further specified into groundwater and fresh surface water. The rows present the extraction from these sources and the columns reflect additions to groundwater and surface water resources through replenishment by

<table>
<thead>
<tr>
<th>Account Number</th>
<th>1–10</th>
<th>11</th>
<th>12</th>
<th>13</th>
<th>14</th>
</tr>
</thead>
<tbody>
<tr>
<td>1–10</td>
<td>NAM; million euros</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>11</td>
<td>Water balance; million m$^3$</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>12</td>
<td>Emission balance; kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>13</td>
<td>Water balance; million m$^3$</td>
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<td>14</td>
<td>Emission balance; kg</td>
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</tbody>
</table>

Figure 19.1 The basic structure of NAMWARiB
rivers or rainfall. For a more detailed description of NAMWARiB and its usefulness in actual policy and decision-making, the interested reader is referred to Brouwer et al. (2005).

AGGREGATION OF ECONOMIC AND ENVIRONMENTAL DATA TO HYDROLOGICAL UNITS

The Netherlands consists of four main river basin districts: Rhine, Meuse, Scheldt and Ems. For the implementation of the WFD, the largest basin the Rhine (covering approximately 70 per cent of the Netherlands) is split up into four different sub-regions: Rhine-North, Rhine-West, Rhine-East and Rhine-Centre (Figure 19.2) in order to enable meaningful analysis.

The three different types of data in the economic account, emission account and water extraction and discharge account are available at different accounting levels, as shown in Table 19.1. The most important challenge during the development of NAMWARiB was to find out how the data at these different accounting levels could be made comparable and aggregated in a systematic and consistent way, that is, where regional totals add up to national totals.

Table 19.1  Accounting levels at which economic and water-related data are available

<table>
<thead>
<tr>
<th>Economic Accounting Levels</th>
<th>Water Accounting Levels</th>
</tr>
</thead>
<tbody>
<tr>
<td>National</td>
<td>National</td>
</tr>
<tr>
<td>12 provinces</td>
<td>Four main river basins</td>
</tr>
<tr>
<td>40 COROPs</td>
<td>Eight regional directorates&lt;sup&gt;a&lt;/sup&gt;</td>
</tr>
<tr>
<td>129 economic geographical units</td>
<td>17 sub-river basins</td>
</tr>
<tr>
<td>&gt; 500 municipalities</td>
<td>50 water boards</td>
</tr>
<tr>
<td>&gt; 5000 postcodes</td>
<td>80 PAWN districts&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td></td>
<td>&gt;1000 water discharge units</td>
</tr>
</tbody>
</table>

Notes:
<sup>a</sup> Regional directorates are responsible for the management of the state-owned water system in their specific region. For example, the river basin of the Rhine is controlled by two such regional offices and the river basin of the Meuse by three regional offices.

<sup>b</sup> Districts that were developed in the 1980s for Policy Analysis of Water Management (PAWN).

Most emission data are available at the level of individual plants, including their x and y coordinates. The emission data were first allocated...
to economic branches of industry and subsequently to river basins based on their geographical location and with the help of a geographical information system (GIS). In some cases a problem arises if the location of an economic activity does not correspond with the location of the emission source. For instance, a factory is located in one river basin, but its wastewater is transported to and discharged in another river basin.

The economic and water flow data are available by branches of industry and were allocated to river basins with the help of regional economic accounts. The regional economic accounts give a quantitative description of the economic processes in the various regions in the country in such
a way that it is linked to the national accounts. The regional accounting system focuses on the production processes in each business unit in the various regions (Statistics Netherlands, 1999). In the Netherlands, regional accounts are composed at the level of 40 so-called COROP areas (see Figure 19.2). COROP areas are the official regional economic units distinguished by Statistics Netherlands.

The economic and water flow data at the level of the 40 COROPs (production volume, use of intermediary products, value added and labour) are aggregated to the seven river basins in a number of steps (Figure 19.3). In a first step, data for COROPs that are situated entirely in one river basin are allocated directly to this river basin. This is the case for 23 of the 40 COROPs in total. For the remaining 17 COROPs, data are allocated in subsequent steps on the basis of the distribution of employees in the specific branches of industry in these 17 COROPs in a GIS. The economic data are allocated to two or more river basin districts with the help of the estimated percentage of employees working in a specific river basin. These percentages are estimated by identifying:

1. the specific branches of industry in the remaining 17 COROPs;
2. the total number of employees working in these branches of industry;
3. the municipalities in which the business units in these branches of industry are located;
4. which of these municipalities fall entirely in one specific river basin, and which municipalities overlap with other river basins.

After the specific branches of industry have been identified in these 17 COROPs, these branches of industry and the number of employees working in these branches of industry are linked to the municipalities in which the underlying business units are found. These municipalities are linked again to the specific river basin districts in which they fall. Business units and their number of employees in municipalities falling entirely inside a specific river basin district are allocated directly to that specific district (Step 2 in Figure 19.3). For those municipalities located partly in one and partly in another river basin district, the identified business units are linked in a next step to the postal codes within these municipalities. Also these postal codes are allocated to river basin districts. Business units falling entirely in postal code areas within one specific river basin district, are allocated directly to that district (Step 3 in Figure 19.3). For those remaining postal codes found in two or more river basins, business units and their employees are allocated to a specific river basin on the basis of the area of the postal code falling in that river basin district (Step 4 in Figure 19.3).
Most of the economic data can be allocated in this way directly at the level of COROPs. On average, 65 per cent of the employees in each branch of industry are found in COROPs falling entirely in one specific river basin district. Twenty seven per cent of the economic data per branch of industry are allocated at the level of municipalities and 3 per cent at postcode level. Five per cent of all data is allocated by considering the area within postcode areas.

**EXAMPLES OF INTEGRATED RIVER BASIN INDICATORS**

The WFD requires that river basins across Europe are described in both physical and economic terms. According to Article 5 in the WFD, the
economic characterization of river basins should include an assessment of the economic significance of current water use and future water use up to 2015 (WATECO, 2002). The economic significance of water use in the different river basins can be measured in NAMWARiB in two different ways. Economic significance is measured through production values and value added generated in river basins per sector (expressed in euros). Water use is measured through water extraction (expressed in cubic metres) by economic activities and the emission of polluting substances (expressed in kilograms) per sector. Water use can moreover be measured through wastewater discharge per sector in each river basin (expressed in inhabitant equivalents).

Furthermore, based on time series analysis possible trends can be identified. In NAMWARiB, trends in economic driving forces can be linked to pressures such as water consumption, wastewater production and the emission of polluting substances (nutrients, metals etc). An example is given in Figure 19.4 where nutrients and metals refer to the environmental themes eutrophication and dispersion of metals respectively. At the national level, real economic growth (in terms of GDP in constant prices) over the period 1996–2001 was 18 per cent (on average 3 per cent per year). Total wastewater production remained more or less the same over that time period, whereas the emission of nutrients decreased by approximately 15 per cent and the emission of metals by about 10 per cent. Figure 19.4 hence suggests that economic activities use the water environment in a more efficient way.

![Figure 19.4 Economic growth, wastewater production, emission of nutrients and metals in the Netherlands over the period 1996–2001 (1996 = 100)](image_url)
An important advantage of NAMWARiB is that it allows detailed trend analysis of specific substances per sector at river basin level. The trends identified at national level, are, for instance, also found in Rhine-West (Figure 19.5), but in the Meuse river basin metals show a more fluctuating development, increasing over the period 1998–2000 (Figure 19.6).

These diagrams have to be interpreted with the necessary care. They provide, for instance, no hard evidence of a direct link between production and water use such as wastewater production or emission of pollutants. Nevertheless, these types of indicators are helpful in assessing the success (or failure) of environmental (sector) policy, as they provide important insight in the environmental efficiency of economic activities (i.e. the relationship between production output and the use of the environment or environmental inputs). They provide a basis for trend analysis. Based upon the observed development of economic activities within sectors and corresponding water use over the past five or ten years, one can extrapolate this development into the future. Using the average growth rates in Figure 19.4 and assuming that the observed trend of a more efficient water use will continue into the future, economic driving forces (production volumes) and corresponding water pressures can be calculated.
DISCUSSION AND CONCLUSIONS

In this chapter, we presented the up- and downscaling procedure for environmental and economic data in an integrated system of water accounts. The most important challenges during the development of the integrated water accounting system were 1. linking available environmental and economic data in a consistent and coherent way, and 2. disaggregating these data to the relevant scale of river basins. In-depth research was needed to assess the compatibility of different types of data at different levels of detail and confidence. Some of the main problems and uncertainties encountered include the use of different statistics from different sources, classifications, monitoring scales, sampling and aggregation procedures.

The advantage of a more refined scaling procedure presented in this chapter can be demonstrated by comparing its results with more simple ways of downscaling available national data to geo-hydrological units based on area size or administrative units. For example, allocating economic data from the National Accounts to the river basin districts based upon the district’s relative area size – not an uncommon procedure – and taking the more refined approach presented here as the point of reference, the resulting errors can be as high as 240 per cent (Table 19.2).

Figure 19.6  Economic growth, wastewater production, emission of nutrients and metals in the Meuse river basin over the period 1996–2000 (1996 = 100)
Table 19.2 Gross regional product (GRP) per river basin in 2000 based on different allocation methods and corresponding differences in GRP compared with NAMWARiB

<table>
<thead>
<tr>
<th>River Basin</th>
<th>Allocation Method Based On NAMWARiB (€ 10³)</th>
<th>Area size (€ 10³)</th>
<th>Difference (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ems</td>
<td>10 618</td>
<td>23 438</td>
<td>120.7</td>
</tr>
<tr>
<td>Scheldt</td>
<td>9 491</td>
<td>28 186</td>
<td>197.0</td>
</tr>
<tr>
<td>Meuse</td>
<td>75 982</td>
<td>68 127</td>
<td>-10.3</td>
</tr>
<tr>
<td>Rhine-Central</td>
<td>27 806</td>
<td>52 319</td>
<td>88.2</td>
</tr>
<tr>
<td>Rhine-East</td>
<td>38 483</td>
<td>60 143</td>
<td>56.3</td>
</tr>
<tr>
<td>Rhine-North</td>
<td>19 581</td>
<td>67 195</td>
<td>243.2</td>
</tr>
<tr>
<td>Rhine-West</td>
<td>185 008</td>
<td>71 644</td>
<td>-61.3</td>
</tr>
<tr>
<td>Total</td>
<td>371 053</td>
<td>371 053</td>
<td>0a</td>
</tr>
</tbody>
</table>

Notes:
a GRP aggregated across all basins (= GDP) is the same irrespective of the allocation method used. Differences occur when allocating economic values across basins based on different methods.

An important source of uncertainty when allocating economic data to river basins is the location of water-related economic activities. The allocation of economic data to river basins is based on the location of the administrative headquarters, and not necessarily the location of the sub-branches. For how many companies this is actually the case and to what extent this distorts the allocation of economic data across river basins is not known. Related to this is the allocation of the emission data and the location of pollution sources. In some cases, the source of pollution and the treatment of the pollution are not located in the same place. In NAMWARiB pollution data are allocated to the district where the household or company is located that produces the wastewater. This may result in a bias in the representation of the pressures exerted across different basins. An example is the largest wastewater treatment facility in the Scheldt basin with a total treatment capacity of more than 400 000 inhabitant equivalents (equal to more than one-third of the total treatment capacity in the entire basin). This treatment facility receives approximately 85 per cent of all its wastewater from households and companies located in the neighbouring Meuse basin, but discharges all of the treated wastewater in the Scheldt basin.

Also water used for drinking water production and distribution may originate from a different river basin district than where it is actually used.
Valuation and sectoral green accounting

consumed. NAMWARiB allocates all water use to the basin in which it is consumed, which does not always necessarily correspond with the basin where it is extracted. For example, in the Scheldt basin approximately 55 per cent of all water used for drinking water production has its origin in the Meuse basin. Nevertheless, all water extracted is allocated to the Scheldt, resulting in a considerable bias of the actual pressure exerted on the limited available fresh groundwater resources in this basin for drinking water purposes.

Notwithstanding these uncertainties and potential biases, by linking water and substance flows to economic flows and doing this systematically for a number of years, insight is gained into the (nature of the) relationship between our physical water system and the economy. The integration of physical and economic information allows for the construction of integrated indicators. For instance, water use by various economic sectors can be related to the economic interests involved. It is this integration of water and economy at river basin level, which makes NAMWARiB an important information tool to support policy and decision-making in the field of integrated water management as advocated by the WFD. By linking information about the physical pressures exerted on the water system by economic agents and the associated economic interests, NAMWARiB enables policy-makers and water managers at national and river basin level to assess the necessary measures to reduce these pressures and meet the environmental objectives in the WFD in an integrated way. NAMWARiB offers opportunities to analyse the trade-offs between environmental goals and the economic interests involved at the relevant level of analysis, that is, river basins.

NOTES

1. At river basin level we only show the development of the indicators over the period 1996–2000 as two of the four indicators (value added and wastewater) were not yet available for the year 2001 at the time of writing this chapter.

2. It has to be pointed out that a longer time period than the past five years is preferred when trying to detect trends in economic driving forces and associated pressures. This will be the case in the next few years when NAMWARiB will be extended to also include more recent years (2001–03) and possibly a few more years in the past as well (1990–95).

REFERENCES


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Titles of publications are in italics.

Abbreviations used in the index include:
CE (choice experiment)
CV (contingent valuation)
WTP (willingness to pay)

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