Valuation of Air Pollution Effects on Ecosystems: A Scoping Study

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Valuation of Air Pollution Effects on Ecosystems: A Scoping Study

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Executive Summary

1. Valuing the benefits of ecosystem recovery in monetary terms is not straightforward because many ecosystem services are not traded in markets and the science that is required to underpin a valuation exercise is insufficiently robust.

2. The overall aim of this study is to assess current prospects for valuing ecosystem recovery in the context of air pollution and to give guidance regarding future research needs, in terms of economic valuation and scientific understanding. Specific objectives are to:

   - review the literature on ecosystem valuation in the context of air pollution
   - describe the main challenges confronting valuing ecosystem recovery
   - assess future research needs and establish priorities

3. The approach required a review of existing literature supplemented by a series of group meetings involving leading experts in the fields of ecosystem science and valuation. The literature review concentrated on peer-reviewed articles that fell into one of the following categories:

(i) UK valuation studies directly applicable to ecosystem recovery from air pollution
(ii) Non-UK valuation studies on air pollution and ecosystems that might be relevant to the UK context
(iii) Valuation studies not in the context of air pollution, but of relevance in methodological terms.

4 Two panels were convened to enrich the review by providing an assessment of the 'state of the art.' Members of the Economics Panel were Prof. Nick Hanley (University of Glasgow), Dr Douglas Macmillan (University of Aberdeen), Guy Garrod and Dr Ricardo Scarpa, (both University of Newcastle). Science Panel members were Prof. Alan Jenkins (CEH-Wallingford), Dr Ron Harriman (Freshwaters Fisheries Laboratory), Dr Bob Ferrier and Dr Robin Pakeman (both MLURI).

Ecosystem Valuation Literature

5 Over 50 ecosystem valuation studies were reviewed. Studies from North America and Scandinavia dominate the international literature on valuing ecosystem effects of air pollution. In part this reflects the importance attached to air pollution in these regions, but also the stronger policy and academic interest in valuation methods. Many of these studies are concerned with impacts on forest growth and timber, and estimating the cost of damage to recreational fishing.

6 There have been several UK freshwater valuation studies in the context of air pollution dealing with fish stocks and biodiversity, but none for woodlands and grassland/heathland ecosystems.

7 The reliability of the science used varies considerably. While some studies have attempted to link with state of the art scientific models, many more have been only weakly linked to science, often adopting dose-response functions that were either inappropriate to the context or entirely hypothetical.

8 Overall the research emphasis revealed by the review reflects academic interest in certain valuation methodologies (e.g. Travel Cost approaches) and, to a lesser degree, the reliability of the underpinning science, rather than the value of the resource affected.

Research Challenges: The Science

9 Information is available on a $5km^2$ grid for determining Critical Loads at a national scale but at a site-specific level, deposition and soil data are poor. There is a need to improve spatial resolution in order to identify impacts on highly valued, but localised, conservation resources such as Sites of Special Scientific Interest (SSSIs).

10 Although it is possible to predict recovery in freshwater chemistry from reductions in sulphur (and to a lesser extent nitrogen) deposition, there is a strong requirement to enhance the chemical:biological link especially with regard to biodiversity.

11 Dynamic models such as MAGIC offer the best current methodology to account for recovery in both terrestrial and aquatic ecosystems. This type of model can be coupled to biological models for fish populations and for diatom assemblages.

12 Ecosystem recovery from air pollution is characterised by considerable scientific uncertainty, with the degree of uncertainty increasing as the time span increases. Uncertainty in predictions is generally badly represented, especially in the Critical Load (CL) approach. Dynamic models can produce numerical representation of uncertainty, but the challenge is to translate this into something that is meaningful to a non-specialist.

13 Confounding factors may have a big influence on the pattern and timing of ecosystem recovery processes. Although new European initiatives such as the RECOVER: 2010 project are specifically focused on identifying the nature and consequences of confounding factors during recovery, more work is required.

14 Many scientific indicators of ecosystem change are ill-suited for benefit identification and measurement purposes. For example, information regarding the toxic effects of heavy metals on root growth is useful if it can be translated into more direct impacts such as timber production or tree health. In the case of biodiversity, changes in charismatic species such as otters are more likely to be understood and appreciated by people than changes to more obscure species or indices.

Research Challenges: Economics

15 The potential to use market price approaches is limited in the UK ecosystem context, as few marketed goods are affected. For those that are, such as timber and grazing, there are no reliable dose-response functions available.

16 Non-use values are expected to be a major component of the total economic value of ecosystem recovery in the UK. Stated preference (SP) approaches are the only way to measure non-use values in monetary terms. Some of the main valuation issues involved are the complex, unfamiliar, and long-term nature of the environmental change, problems with double-counting, and the extent to which hypothetical willingness to pay (WTP) elicited would reflect actual WTP.

17 Recent innovations in environmental valuation techniques offer some interesting possibilities for enhancing reliability of benefit estimates. One development that has particular relevance to ecosystem recovery is the use of deliberative valuation fora. The 'Market Stall' method, for example, is potentially more suited to a complex scientific issue such as ecosystem recovery than personal interviews, as it provides people with more time and information to consider their WTP.

18 For policy purposes, benefit estimates have to be scaled-up to the relevant level of aggregation for policy discussions and negotiations. Some of the main challenges are identifying and sampling the relevant population of beneficiaries, and the incompatibility of economic models (which are often based on national data) and scientific models (which are often catchment- based).

19 The long-term nature of ecosystem recovery presents difficulties for temporal aggregation. Typically this is achieved by discounting, but as recovery may take hundreds of years the choice of discount rate will be crucial. Furthermore there is the problem of accounting for the preferences of future generations within the Cost-Benefit framework.

Freshwaters Case-Study

20 Freshwater was selected as the case study subject because the science of recovery is comparatively better understood than for other ecosystems. Also, there have been several UK valuation studies of freshwater resources in the context of air pollution.

21 The principal aims of the case-study were to review and assess the current potential to value recovery in the freshwater ecosystem; to identify specific research needs (both economic and scientific) that would allow valuation to take place, and identify the limits to valuation.

22 Four studies were identified from the literature review as having some direct policy relevance to freshwater recovery in the context of air pollution. However, the benefit transfer potential of these studies was limited due to problems in methodology or with the scope of the valuation exercise.

23 In prioritising research needs we have considered a range of criteria including cost and feasibility, the reliability and potential magnitude of benefits generated by ecosystem recovery, and the scope for benefit transfer.

24 The priority for valuation research is to estimate both the use and non-use values associated with biodiversity recovery. Contingent valuation (CV) using deliberative valuation fora , is probably most suited to the task, but there is scope for applying choice experiments (CE) with recreational users. In both cases, given the scientific complexities and uncertainties surrounding ecosystem recovery, considerable effort would have to be invested in preparatory work such as focus groups and scenario portrayal.

25 By building on previous work it should be possible to generate reliable estimates of the benefits of recovery in UK salmon fisheries by applying the hedonic pricing (HP) method. Benefits are likely to be small relative to non-use values but they would be perceived as 'real' economic market benefits as opposed to 'hypothetical' and would provide evidence of direct economic benefits to specific groups (salmon anglers, riparian owners etc).

26 Additional scientific research that would help underpin valuation includes: (i) an enhanced link between chemistry and biology, with the focus on biodiversity and ecosystem function; (ii) dynamic modelling; and (iii) defining measures of ecosystem change that could be easily understood by the general public. One suggestion would be to develop an 'ecological ladder' to describe recovery in terms of 'steps' that are defined by recognised indicators such as the presence/absence of individual species and related water quality measures.

General Conclusions and Recommendations

27 Valuing ecosystem recovery in the context of air pollution is a difficult task due to scientific uncertainty about the impact of abatement on ecosystem recovery and the challenges of valuing long term and complex environmental changes.

28 The 1999 IGCB Report noted that it would not be possible to conduct an economic analysis of ecosystem benefits from reductions in air pollution. In our view this is an overly pessimistic assessment, at least for freshwaters, given new developments in the dynamic modelling of 'effects' and advances in economic valuation methodology. We believe that appropriately planned valuation research conducted in collaboration with scientific experts would yield reliable benefit estimates that would be useful for policy purposes.

29 Scientific understanding and predictive modelling is most advanced for freshwater ecosystems, with models such as MAGIC capable of predicting long term ecosystem response. However, our understanding of the effects of air pollution on grassland/heathland and forest ecosystems is currently insufficient to allow us to undertake a valuation exercise.

We recommend that research is undertaken to estimate the national benefits of recovery in the freshwater ecosystem from further reductions in air pollution

30 The main valuation priority is to estimate the use and non-use values associated with biodiversity recovery using SP approaches. CV is most suited to the task of valuing biodiversity changes, with scope for applying CEs to recreational users if attributes such as 'fish catch' can be identified and quantified.

Priorities are as follows:

1. **A large scale (national) CV to estimate non-use values**
2. **A small scale CE to establish the magnitude of user benefits based on several case-studies (but designed to allow scaling up to national level estimates).**
3. **A scaling-up exercise for salmon fisheries recovery based on earlier work by Macmillan and Ferrier (1994) in Galloway.**

31 Estimating ecosystem values is likely to provide a serious challenge to existing survey-based SP methods such as CV and CE due to the complex nature of ecological recovery processes.

We therefore recommend that:

- **Considerable effort is invested in preparatory work such as focus groups and scenario portrayal.**
- **Group-based 'deliberative' valuation methods such as the Market Stall are preferred to interview-based surveys as the former provides participants more time to gather and assimilate information and evaluate their willingness to pay.**
- **Descriptive devices such as an 'ecological ladder' need to be developed to describe recovery over time in a simpler way, with each 'rung' of the ladder defined in terms of carefully selected ecological criteria such as the presence/absence of key species and the status of water quality measures.**
- **When conducting a WTP study using CV or CE it will be necessary to directly describe further ecosystem deterioration under the status quo. Existing evidence, suggests that WTP for an abatement programme will be considerably enhanced if this avoided damage is made explicit to respondents (the 'endowment effect').**

32 In order to enhance the scientific base for valuing freshwaters we require more investment in scientific research to:

- identify the extent of damage to areas of high nature conservation interest
- model time delays and potential hysteresis during the recovery phase
- account for land use practices and other confounding effects such as climate, pests, and diseases.
- enhance the link between chemistry and biology, especially biodiversity and ecosystem function

Limits to Valuation

33 While new methodological advances in terms of valuation and increased scientific knowledge are improving our capacity to value ecosystem recovery, some fundamental difficulties remain. First, economics is limited in the sense that ecosystem change only matters when impacts on human welfare can be recognised and measured scientifically. Second, some aspects of ecosystem recovery may be so complex and/or uncertain that valuation scenarios for SP studies may be too simplistic or even misleading. Third, the impact of pollution on the foundation services of ecosystems may not be fully understood by scientists, let alone valued by economists.

To help overcome these limitations we suggest the following:

- **Scientists and economists need to work together more closely in integrated projects.**
- **Introduce 'expert opinion' more overtly into the valuation process. For example by using expert witnesses in group-based valuation fora or by providing expert 'interpretations' of complex scientific data in the information set given to respondents.**
- **Where valuation is not considered possible alternative decision-making techniques such as Cost-Effectiveness Analysis and Multi-Criteria Analysis that do not require monetary valuation should play a role. (However neither of these approaches give guidance whether it is optimal to proceed with abatement or not and hence do not perform the same role in policy appraisal).**
- **The uncertainties concerning monetary valuation should not be understated and benefit estimates must be considered to be conservative, lower bound estimates of total ecosystem value.**

Introduction

The overall aim is to establish the extent to which there is scope to demonstrate the benefits to UK ecosystems from reductions in air pollution, specifically in terms of valuation of these benefits.

Valuing the benefits of ecosystem recovery in monetary terms is not straightforward and a report by the Interdepartmental Group on Costs and Benefits (IGCB) concluded that it was currently not possible to conduct an economic analysis of benefits arising from air pollution reductions (DETR 1999). The main problems identified were that many ecosystem goods and services affected, such as biodiversity, are not directly traded in markets and that the underpinning science is insufficiently robust.

In this study we use the available literature and group discussions with leading experts in the field of economic valuation and environmental science to examine the potential for ecosystem valuation in the light of recent advances in methodology and understanding.

Specific objectives are to:

- review the existing literature on ecosystem valuation in the context of air pollution
- describe the main challenges confronting the valuation of ecosystem recovery
- consider the links between the current scientific work on air pollution effects on UK ecosystems and the potential for economic analysis of these effects
- assess future research needs and establish priorities for valuing ecosystem recovery

The effects of air pollution on ecosystems are pervasive but for the purposes of this study it was decided to focus attention on three ecosystem types: *Forests, Freshwaters and Heathland/Grasslands*. These ecosystems were chosen because of their significance in a UK context and because observable environmental changes in the function of these ecosystems have been identified and linked to air pollution. The main air pollutants considered are sulphur dioxide (SO_2), oxides of nitrogen (NOx), ammonia (NH^{3+}), volatile organic compounds (VOCs) and heavy metals.

The approach involved both a review of existing literature and a series of consultative meetings with experts regarding the scientific and economic challenges of valuing ecosystem recovery in monetary terms.

The literature review involved an extensive search of databases on the World Wide Web and on CD Rom sources. Previous reviews of the ecosystem valuation literature were also consulted (CSERGE, 1992; ERL, 1991; and Wilson and Carpenter, 1999). The review concentrated on peer-reviewed articles concerned with monetary valuation[1] that fell into one of the following categories:

[1] The review did not consider non-monetary approaches such as Multi-Criteria Analysis, although these are discussed in Chapter 6.

(i) UK valuation studies that are directly applicable to ecosystem recovery from air pollution;
(ii) Non-UK valuation studies on air pollution and ecosystems that might be relevant to the UK context
(iii) Valuation studies not in the context of air pollution, but of relevance in methodological terms.

Two expert panels were convened to enrich the review by providing a 'state of the art' assessment in terms of the science of ecosystem recovery and appropriate economic methods. Each panel met twice. The first 'Economics' panel meeting focused on the development of a valuation framework and the requirements the framework would place on scientific knowledge and research. Members of the Economics Panel were Prof. Nick Hanley (University of Glasgow), Dr Douglas Macmillan (University of Aberdeen), Guy Garrod and Dr Ricardo Scarpa, (both University of Newcastle). Each member was selected on the basis of their previous experience in developing and applying ecosystem valuation methods.

The initial meeting of the Science Panel focused on the extent to which existing scientific knowledge and modelling capabilities matched the demands of economic valuation. Panel members were Prof. Alan Jenkins (CEH-Wallingford), Dr Ron Harriman (Freshwaters Fisheries Laboratory), Dr Bob Ferrier and Dr Robin Pakeman (both MLURI). This panel was chosen for their understanding of dynamic and Critical Loads modelling, and for their scientific knowledge of the long-term impacts of air pollution on terrestrial and freshwater ecosystems. Jonathan Foot, a scientist from the JNCC also attended the first Panel meeting on behalf of the Steering Group[2].

The second Panel meeting was a joint meeting involving both panels. The aim was to assess the potential for valuing recovery in the freshwater ecosystem and to recommend future research priorities. The choice of freshwaters as a case study reflected the higher level of scientific knowledge and valuation experience available for freshwaters relative to other ecosystems.

The remainder of the report is structured as follows. Chapter 1 provides an introduction to the principles and practice of ecosystem valuation and Chapter 2 reviews the existing literature on ecosystem valuation in the context of air pollution. Chapter 3 describes a valuation framework for ecosystem recovery and highlights the main scientific and economic challenges associated with implementing the framework in the context of air pollution. Chapter 4 applies the framework to freshwaters as a case study and conclusions and recommendations are provided in Chapter 5.

[2] Brief resumés of each expert are given in Appendix 1.

Chapter 1 **Ecosystem Valuation**

CHAPTER SUMMARY

This chapter provides a general introduction to ecosystem valuation. The range of goods and services provided by ecosystems are defined in economic terms. The concepts of use and non-use values, and 'total economic value' are introduced. The main approaches to valuing ecosystem recovery, and their respective advantages and disadvantages are described. The basic approaches reviewed are

- Market price approaches (e.g. Productivity Method)
- Revealed preference approaches (e.g. Travel Cost Method)
- Stated preference approaches (e.g. Contingent Valuation Method)
- Imputed preference (e.g. Replacement Cost Method)

1 Ecosystem Valuation

Valuing ecosystem recovery from air pollution is technically difficult and controversial. Difficulties arise because of scientific uncertainty about effects and their link to human welfare and because many ecosystem services affected are not marketed and hence have no pre-assigned market value. The controversy stems from an ethical debate about man's relationship with nature and whether it is appropriate or 'right' to value nature in monetary terms.

This report is primarily concerned with the technical aspects of the debate. In this section we describe the range of goods and services provided by ecosystems and give examples of the damage done to these resources by air pollution. We also describe relevant approaches for valuing ecosystem recovery.

1.1 What are ecosystems?

Ecosystems are a 'spatially explicit unit of the earth that includes all the organisms, along with all components of the abiotic environment within its boundaries' (Likens, 1992). Within this broad definition ecosystems can be further defined by their characteristic assemblages of organisms – such as forest or desert ecosystems. In this study we have identified three such ecosystems for study – Forests, Heathland/Grassland and Freshwaters.

However, one has to bear in mind that this typology is somewhat arbitrary in the context of this study because many services affected by air pollution are jointly provided by several ecosystems. For example, changes in heathland biodiversity may impact on neighbouring forest or freshwater ecosystems, while changes in forest soil chemistry will affect freshwater quality within a catchment.

1.2 Ecosystem functions, services and values

In the literature economists often make the distinction between ecosystem function, and the goods and services ecosystems provide. Functions can be defined as the physical, chemical, and biological processes or attributes that contribute to the self-maintenance of an ecosystem. These include the provision of wildlife habitat, carbon cycling, or the trapping of nutrients (sometimes referred to as foundation services).

Ecosystem goods and services have directly beneficial outcomes for humans (frontier services). Some examples are food and timber production, the provision of clean water or of scenic views. In order for an ecosystem to provide services to humans, some interaction with, or at least some appreciation by, humans is required. Thus, functions of ecosystems are value-neutral, while the goods and services produced have value to society.

Ecosystems provide a wide array of the goods and services, some of which are distinctive to the ecosystem. However, even with relatively simple or well understood ecosystems it can be difficult to determine the relationship between foundation and frontier services – in other words to relate changes in the fundamental health of the ecosystem to something meaningful in human welfare terms. For example, the climate change debate is characterised by uncertainty about its ramifications for human activity. Where knowledge of these links to welfare is incomplete then there is a risk that ecosystems will be undervalued.

From a neo-classical economic perspective the aim is to value ecosystem goods and services in monetary terms. This is achieved by estimating the amount people are willing to pay to preserve or enhance these services. In this way, economic value differs from scientific values based on 'objective' measures such as species richness or other diversity indices, or indeed value judgements expressed by experts or specialists[3] as they are (i) based on a census of individual preferences across society and (ii) constrained by income.

Typically, ecosystem values are classified into 'use' and 'non-use' values (Box 1.1). Use values for ecosystems include fishing for trout and salmon (both commercial and recreational), and timber production. Use values also encompass activities such as watching a TV programme about nature (a non-consumptive use value) or inputs that help to produce something that people use directly (an indirect use value). For example, lower organisms on the aquatic food chain provide indirect use values to recreational anglers as they help sustain fish populations.

Non-use value relates to the notion that individuals derive utility from environmental resources outwith expected personal use. In the context of air pollution it is likely that people care about ecosystem recovery, even though they would never visit or use the ecosystem in any direct way. Altruism underpins non-use values and is manifested in concern for other humans (vicarious consumption), future generations (bequest value) or purely toward the environment itself (pure existence value). Bequest value is particularly important in the context of ecosystem recovery from air pollution as the benefits are only likely to be realised many years into the future.

Since the non-consumptive services which give rise to non-use value are available to all without possibility of exclusion, and the enjoyment to one person does not interfere with that of others, it is normal to consider non-use values to be pure public goods (McConnell, 1983). Although Krutilla's (1967) seminal paper discussed the possibility of 'existence' value in the context of uniqueness and irreversibility, it is now widely accepted that non-use values can be held for a range of environmental resources[4].

Including non-use value within a total economic value framework has attracted criticism from some economists. The debate has tended to focus on measurement issues and the treatment of altruism in cost-benefit analysis. While some economists believe values based on altruism in individual's utility functions will lead to double-counting (Bergstrom 1982, Milgrom, 1993), the prevailing view is that altruistic behaviour is valid if it is focused on the utility of others with respect to the environmental change concerned (Jones-Lee, 1992; Johansson, 1992), rather than their 'well-being' per se.

[3] See Nunes *et al*., (2001) for a description of ecological indicators of value.

[4] Non-use value as defined above is a utilitarian concept, that is people derive some value from the existence of environmental resources. There is a distinct, but not altogether separate notion that resources (e.g. wildlife, the planet earth) have a value, an intrinsic value, quite unrelated to humans. In reality these values are very similar, but the distinction is philosophical and arises from how individuals view the world.

1.3 Measuring Ecosystem Values

For some ecosystem goods like timber, which are bought and sold, valuation is based on their market price. For services which are not marketed, including non-use values, alternative methods are required for estimating how much money people (consumers) are willing to pay (WTP) to obtain the service, or how much people would be willing to accept (WTA) in compensation in order to give it up.

Valuation research has grown considerably in the last 20 years and many new methods for valuing ecosystem services are now available. Although innovative variations abound in the literature, there are four general approaches to estimating monetary values for ecosystem services: market prices; revealed preference, stated preference and imputed preference. Within each of these approaches a number of alternative methods have been developed and these are summarised below (and listed in Table 1.1).

1.3.1 Market Prices

Some ecosystem goods affected by air pollution, such as fish or timber, are valued in the market place. The main approach is the **Market Price** (MP) method whereby the economic value of ecosystem recovery can be determined by changes in consumer and producer surplus in the relevant market. (Assuming that the market price actually reflects the opportunity cost of the good and is not distorted by market failure or by some form of government intervention). Opportunities to apply this method to ecosystem recovery in the UK would be limited to commercial timber production and certain types of fishing (e.g. netted salmon).

For example, reductions in ozone levels could lead to enhanced forest growth and greater timber yields. If productivity gains are substantial, timber price may be affected by the enhanced supply of timber on the market and these changes would have to be modelled. However, in cases where productivity changes are relatively small and hence have little or no impact on price, a cruder approach which simply multiplies the estimated change in quantity by the prevailing market price, can be adopted.

Some ecosystem services, such as clean water, are used as inputs in the production of a marketed good, and their value may be measured by their contribution to the profits made from this good. This is referred to as the **Productivity** (P) method. Barbier (1994) used the Productivity method to value wetlands in Nigeria for crop and fuelwood production, and fish catch.

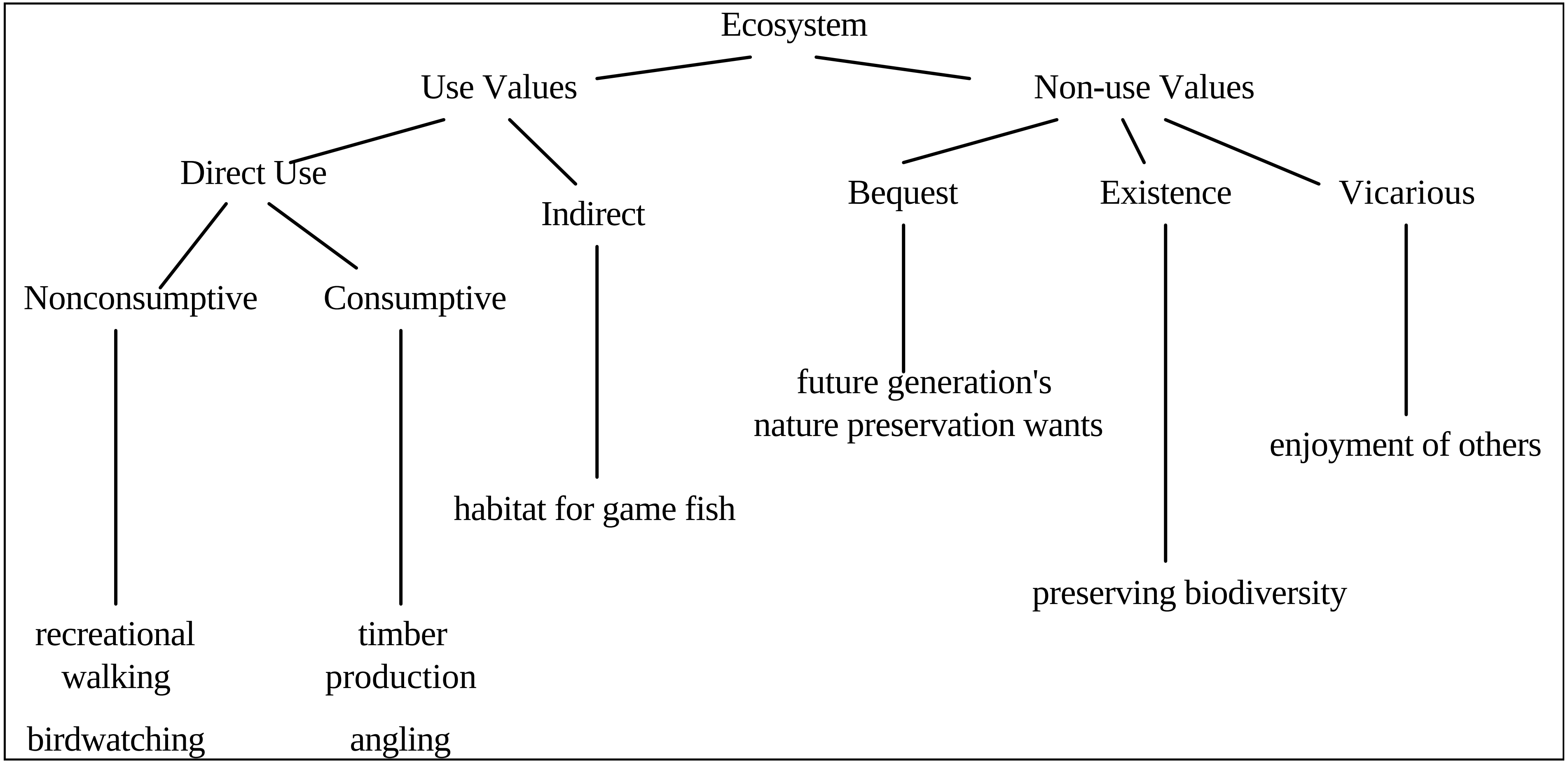
1.3.2 Revealed Preference

The value placed on some ecosystem services which are not directly marketed can be revealed from how much people are willing to pay for goods or services that are marketed. For example, a person could pay a higher price for a home with a nice woodland view, or will spend more money and time travelling to a quality location for bird watching. Two techniques, which exploit variation in the price of marketed goods to determine WTP for changes in ecosystem quality are the **Hedonic Price** (HP) method and the **Travel Cost** (TC) method.

Table 1.1 Benefit Valuation Methods

Category	*Method*	*Abbrev.*
Market Prices	Market Price	MP
	Productivity	P
Revealed Preference	Hedonic Pricing	HP
	Travel Cost	TC
Stated Preference	Contingent Valuation	CV
	Choice Experiments	CE
Imputed Preference	Damage Cost Avoided	DCA
	Replacement Cost	RC

Box 1.1 Ecosystem values potentially affected by air pollution



HP is most commonly applied to labour or capital markets such as residential housing. The basic premise of the HP is that the market price is partly related to its environmental characteristics. In the case of housing, the price could reflect the value of local environmental attributes such as air and water quality, neighbourhood noise, as well as proximity to green space and recreational sites. In relation to ecosystem recovery from air pollution the scope for applying the HP approach is probably best suited to salmon fisheries and woodland properties, which are both marketed. In the case of salmon the price of an individual beat[5] will reflect the characteristics of that beat some of which, such as catch, will have been affected by pollution. For woodlands, prospective owners may pay less for woodlands which show signs of defoliation or dieback caused by air pollution.

The HP approach identifies the WTP of an ecosystem service by exploring variation in the price and the attribute of interest using statistical techniques and is very data demanding. One of the main limitations in relation to valuing ecosystem recovery would be the availability of sufficient market data on the sale price of either salmon beats or woodlands.

The TC Method is used to estimate economic use values associated with ecosystems that are used for recreation. The method can be used to estimate the economic benefits or costs resulting from changes in access costs for a recreational site; elimination of an existing recreational site; addition of a new recreational site; and changes in environmental quality at a recreational site. The basic premise of the travel cost method is that the time and travel cost expenses that people incur to visit a site represent the "price" of access to the site. Thus, peoples' willingness to pay to visit the site can be estimated based on the number of trips that they make at different travel costs. This is analogous to estimating peoples' willingness to pay for a marketed good based on the quantity demanded at different prices. For example, recovery in fish stocks or the scenic quality of woodlands may stimulate demand (i.e. encourage people to make more fishing trips).

1.3.3 Stated Preference

Many ecosystem services are not traded in markets, and are not closely related to any marketed goods hence people cannot "reveal" what they are willing to pay for them through their market purchases or actions. In the words of the influential NOAA panel, there are 'neither obvious or subtle behavioral trails that can provide information' about non-use values' (NOAA, 1993). In these cases, surveys can be used to ask people directly what they are willing to pay, based on a hypothetical scenario in **Contingent Valuation (CV**). Alternatively, people can be asked to make tradeoffs among different alternatives, from which their willingness to pay can be statistically inferred from **Choice Experiments (CE**).

CV is used to estimate economic values for all kinds of ecosystem and environmental services. It can be used to estimate both use and non-use values, and it is the most widely used method for estimating non-use values. CV involves directly asking people, in a survey, how much they would be willing to pay for specific environmental services. In some cases, people are asked for the amount of compensation they would be willing to accept to give up specific environmental services. The approach is "contingent" because people are asked to

[5] Salmon rivers are divided up into 'beats' for management purposes and the rights to fish for salmon are often sold on an individual beat basis.

state their willingness to pay, contingent on a specific hypothetical scenario and description of the environmental service.

CE is similar to CV, in that it can be used to estimate economic values for virtually any ecosystem or environmental service, and can be used to estimate non-use as well as use values. Like CV, it is a hypothetical method, but asks people to make choices based on a hypothetical scenario rather than ask people to state their monetary valuation directly. Instead, values are inferred from the hypothetical choices or tradeoffs that people make. CE asks the respondent to state a preference between one group of environmental services or characteristics, at a given price or cost to the individual, and another group of environmental characteristics at a different price or cost. Because it focuses on tradeoffs among scenarios with different characteristics, CE is especially suited to policy decisions where a set of possible actions might result in different impacts on natural resources or environmental services. For example, improved water quality in a lake will improve the quality of several services provided by the lake, such as drinking water supply, fishing, swimming, and biodiversity. The method can be used to rank options which have different impacts on lake quality with regard to these individual uses.

As neither method depends on links to actual expenditure or behaviour, SP approaches are the only ways to assign money values to non-use values of ecosystems. While this is a distinct advantage in relation to ecosystem recovery from air pollution, it also means that the resulting values can be controversial and difficult to verify. Stated Preference approaches have been criticised by many economists for providing biased and unreliable measures of non-use value (Diamond *et al.*, 1993; Milgrom, 1993). For example, Kahneman and Knetsch, (1992) have argued that CV does not actually measure economic preferences, but rather generates values which reflect cultural symbolism and social ideology. These criticisms are discussed in more detail later in this report.

1.3.4 Imputed Preference

In many studies the value of some ecosystem services is imputed from assessing the level of investment expenditure that would be required to offset the loss of those ecosystem services. Two of the main approaches are **Damage Cost Avoided** (DCA), and **Replacement Cost** (RC).

DCA uses either the value of resources protected, or the cost of actions taken to avoid damages, as a measure of the benefits provided by an ecosystem. For example, if a wetland protects adjacent property from flooding, the flood protection benefits may be estimated by the damages avoided if the flooding does not occur, or by the expenditures property owners make to protect their property from flooding. RC assumes that the cost of replacing an ecosystem or its services is an estimate of the value of the ecosystem or its services. For example, the flood protection services of a wetland might be replaced by a retaining wall or embankment. Andreasson-Gren (1991) compared the costs of nitrogen abatement via wetlands restoration with the costs of conventional technologies in the Baltic sea.

These approaches can end up with incorrect estimates of WTP. In the case of damage avoided (DCA), people may be willing to pay to avoid the inconvenience or distress of flood damage, while replacement cost approaches (RC) it is cost that is measured and not the value of the services or goods provided. (Expenditure may be higher or lower than WTP). A further limitation to the RC approach is that the proposed interventions may not be a perfect substitute for the lost ecosystem service. For example, many ecosystem services may have an existence value that is not replaceable. Also, given the scientific uncertainty surrounding how ecosystems are affected by air pollution there is every chance that replacement technologies may not adequately replicate all of the functions provided. For example extensive liming operations in Scandinavia have allowed fish populations to recover but have not re-established a pre-acidification assemblage of invertebrate and macrophytic communities.

Chapter 2 Review of Ecosystem Valuation Literature

CHAPTER SUMMARY

- This chapter reviews previous studies most relevant to valuing ecosystem recovery in the UK in the context of air pollution. We also examine new developments in valuation methodology and assess the extent to which they enhance prospects for generating reliable benefit estimates for ecosystem recovery.

- There have been relatively few ecosystem valuation studies carried out in the UK in the context of air pollution, with the exception of studies concerned with valuing fish stocks and biodiversity.

- Studies from North America and Scandinavia dominate the international literature on valuing ecosystem effects of air pollution. In part this reflects the importance attached to air pollution in these regions, but also the stronger policy and academic interest in valuation methods. Many of these studies are concerned with impacts on forest growth and timber, and estimating the cost of damage to informal recreational fishing.

- Many of the studies reviewed are more than a decade old and could no longer be considered to be 'state of the art' in terms of economic methodology.

- The reliability of the scientific base used varies considerably. While some studies have attempted to link with state of the art scientific models, many more have had been only weakly linked to science, often adopting dose-response functions that were either inappropriate to the context or entirely hypothetical.

- Overall the valuation research reflects academic interest in certain valuation methodologies (e.g. Travel Cost approaches) and, to a lesser degree, the reliability of the underpinning science, rather than the magnitude of the benefits of recovery.

- New developments such as combining revealed and stated preference approaches, and deliberative valuation fora such as the 'Market Stall' approach may offer some potential advantages in the context of valuing ecosystem recovery.

- The potential of deriving reliable benefit estimates for ecosystem recovery in the UK from a benefit transfer exercise is limited due to variation in environmental conditions and methods used. The greatest scope for benefit transfer exists for UK freshwaters where there have been several recent valuation studies on biodiversity and angling.

Chapter 2 Review of Ecosystem Valuation Literature

Over the last 20 years there have been a number of ecosystem valuation studies carried out in the context of air pollution in Europe and North America. In this chapter we review these studies in the following way. First, the main UK ecosystem valuation studies are summarised. The review then extends to non-UK studies also targeted at air pollution effects on ecosystems. Finally, other studies which describe methodological developments that show some promise with regard to valuing ecosystem recovery in the context of air pollution are highlighted.

A tabular summary of the main valuation studies carried out on ecosystems in the context of air pollution is presented in Table 2.1. Each study is categorised according to the nature of the benefits being valued, the policy context, the methodology used, and the country.

2.1 UK Studies

In this section the main UK studies relating to forests, freshwater and heathland/grassland ecosystems are summarised.

2.1.1 Forestry

There have been no authoritative studies concerning the cost of lost timber production from air pollution in the UK. The absence of any major valuation exercise for forestry in the UK is surprising. In part, it probably reflects the absence of a reliable dose-response model for timber. Considerable scientific uncertainty surrounds the link between air pollution and timber production, with many forest health surveys failing to detect any chronic deterioration in forest condition in the UK (NEGTAP, 2001). One of the biggest problems has been identifying the influence of other factors such as pest and climate stress on tree health. Currently, the most robust evidence for damage relates to ground level ozone, where tree growth has been affected and can cause visible leaf injury.

Most valuation studies that refer to the UK have been focused on providing European-wide estimates of damage, and provide little detailed assessment for the UK. Typically, many of the dose-response relationships used are not suited to UK conditions and hence there must be some doubt regarding reliability. (These studies are reviewed in Section 2.2). There has been no attempt to value the damage to non-market timber goods or services as a result of air pollution.

2.1.2 Freshwater

Scientific understanding of the impacts of air pollution in the UK is perhaps most advanced for freshwaters and there have been several attempts to estimate the benefits of recovery of freshwater services affected by air pollution.

Macmillan *et al*. (1996) estimated the total economic value placed on biodiversity recovery in the semi-natural uplands of Scotland from acidification using CV. Approximately 1000 households were sampled by mail-shot and the study investigated WTP for six scenarios which offered, in a split sample, alternative future ecosystem recovery levels (following abatement) and damage levels (under the *status quo*). Time-scale and uncertainty regarding future recovery were also investigated, with ecosystem change depicted by 'species boxes' that pictorially represented changes in the relative abundance of biodiversity. Average household WTP, elicited using a dichotomous choice format, for abatement of acid rain ranged from £247 to £351 per household depending on the scenario. WTP was (statistically) significantly influenced by the level of future damage but not by future recovery level. A weakness of the research, with respect to policy appraisal, was that the link between future emissions reduction and biodiversity levels was not underpinned by a reliable scientific model but by the informed opinion of scientific experts.

ECOTEC (1993) used similar pictorial representations of species affected by acidification in a study of the non-market benefits of reduced SO_2 emissions for the UK. Using in-person interviews the sampling frame included 1606 non-users (general public) and 587 users (anglers). Annual household WTP for non-users (additional water rates) was £26 and £40 for anglers. These values are much lower than the estimates by Macmillan *et al.*, (1996) above, but this may be partly due to differences in methodology. The ECOTEC study excluded very high WTP amounts and used an open-ended payment format which typically produces lower estimates of WTP than the dichotomous choice format used by Macmillan *et al*. However, one cannot rule out the possibility that the Scottish population value recovery more highly than other parts of the UK – perhaps due to their closer proximity to damaged areas.

The UK rod and line (r&l) salmon fishery is a private resource traded in the market place, hence market data for individual beats would allow the links between fish catch, fishery value, and water quality to be explored. Macmillan and Ferrier (1994) used the HP method to predict the economic benefits of recovery in the r&l salmon fishery in Galloway. Three alternative deposition scenarios for SO_2 (constant 1988 levels; a 60% reduction from 1980 levels by 2003; and a 90% reduction from 1980 levels by 2008) were linked to changes in water chemistry, fish population status and fish catch over a 50-year time scale using MAGIC, a process-based catchment model for acidification. The impact of increased catch on salmon values were then linked to the value of the fishery using a Hedonic Price relationship for the UK salmon fishery developed by Radford *et al*., (1991). The results were consistent with the assumption of diminishing marginal returns. Under Scenario 1 (*status quo*) the market value of the fishery was predicted to decline gradually from £12.6 million in 1988 to £11.7 million in 2033 in response to declining catch. Under both scenario 2 and 3 SO_2 emission reductions initiate a relatively modest recovery in market value: under scenario 2 the market value of the fishery rose to £13.6 million, and under scenario 3, £14.0 million after 50 years.

Milner and Varrallo (1990), also used Radford's survey of salmon r&l fisheries to estimate the cost of acidification in Welsh fisheries. Damage estimates were in the region of £1 to 5 million. Their analysis relied on a simple presence/absence relationship between water chemistry and fish catch verified by the results of an angler questionnaire and fish population survey. The study did not consider the timing and extent of recovery.

2.1.3 Grassland/Heathland

There has been no attempt to value air pollution effects on grassland/heathland ecosystems. As in the case of forestry, the absence of reliable scientific research to underpin valuation is a limiting factor. Although some effects of air pollution on these vegetation types are documented, the problem of identifying the role of air pollution from other influences such as grazing management and erosion remains.

Possible effects include increase in fertility of acid grasslands due to N deposition, a shift towards vegetation types characteristic of moorland grass from heath/bog flora, and reduced frost hardiness of *Calluna vulgaris* due to ozone. The impacts of these changes on welfare have not been valued and are not fully understood. The impact on grazing potential is likely to be negligible, but nature conservation may be affected. However these benefits would be difficult to quantify as air pollution effects are confounded with changes in grazing management and increased trampling by walkers.

2.2 Non-UK studies of air pollution and ecosystems

Studies from North America and Scandinavia dominate the international literature on valuing ecosystem effects of air pollution. In part this reflects the importance attached to air pollution in these regions, but also the stronger policy and academic interest in valuation methods. Many of these studies are concerned with impacts on forest growth and timber, and estimating the cost of damage to recreational fishing. This emphasis relates less to the overall magnitude of the damage costs, than to the ready applicability of appropriate valuation methodologies (e.g. Production Function and Travel Cost approaches) and to a lesser degree, the reliability of the underpinning science.

2.2.1 Forestry

There are considerable differences in the approaches taken to estimating forest damage and in the conclusions drawn by researchers between Europe and the US. Generally US studies, using more sophisticated economic and scientific models to estimate timber losses, have concluded that air pollution effects on timber growth are not substantial (with the exception of ground-level ozone). In contrast European studies have found damage, in terms of harvestable volume, to be significant. However, European studies tend to be more broad-brush than the US studies. For example, one (relatively weak) scientific relationship between forest damage and air pollution is frequently applied to all European countries, irrespective of tree species and climate.

Gregory *et al*., (1996) attempt to estimate the value of damage to European forests from a coal-fired power station in the East Midlands using a simple MP approach based on timber volume lost and current timber price. The power plant was estimated to have reduced timber production by 200,000 tonnes, across Europe valued at £4.6 million. Although costs for the UK forest area are not reported, the study estimates that under 10% of the UK forest area is affected based on Critical Load exceedance. Costs of transport and harvesting, as well as the negative externalities associated with transport such as noise and pollution were not taken into account; hence damage costs are gross estimates only. Also, the dose-response relationship was based on a crude relationship between critical load exceedance and a damage function derived from evidence of forest dieback in the 'Black

Triangle' of East Germany, Southern Poland, and the Czech Republic. In this region air pollution is extreme compared to other areas and the suitability of transferring this dose-response relationship must be doubtful.

IIASA (1991) estimated losses to UK forests in terms of both timber and non-timber outputs to be in the region of $1173 million per year. However, the methodology used is questionable as non-timber losses were somewhat arbitrarily estimated by multiplying timber losses by a factor of 2.7. Also no account was taken of potential price effects caused by changes in supply and demand.

In the US researchers have had access to more complex models of the timber sector and have been able to model potential changes in both consumer and producer surplus. NAPAP (1991) predicted potential economic losses associated with air pollution effects on timber yield in the southern US. The approach predicted timber losses using hypothetical damage functions linked to a regional forest inventory model, with welfare effects (i.e. changes in consumer and producer surplus) estimated using an economic model of the US timber sector.

For the 1999 EPA report on progress toward meeting the targets of the Clean Air Amendment Act (CAAA), a tree productivity model (PnET II) was used to estimate the combined effects of several environmental stresses, including ground-level ozone, on net primary forest production (EPA, 1999). Model output was then linked to TAMM, a timber model, to provide benefit estimates. (TAMM combines information on inventory and growth effects that then feed though to future harvests and market responses). Discounted annual benefits in the period 1990 to 2010 were estimated to be in the region of $1.9 billion. Additional benefits in terms of carbon storage were not quantified.

Callaway *et al.,* (1986) used hypothetical forest growth functions to predict reductions of 10% 15% and 20% in forest productivity for hardwoods and softwoods in the eastern US. A US forest sector model, which accounted for price changes and limited input substitution was used to predict welfare effects. A similar approach was used by Crocker and Regans (1985) to estimate the benefits of a 100% reduction in acid deposition in the same region.

Comparatively few studies have investigated the impact of air pollution on non-timber goods and services from forests. The EPA report (1999) estimated that the CAAA would lead to additional annual benefits from forest recreation as a result of improved visibility of $2.9 billion. This figure is not entirely reliable as it is based on a benefit transfer exercise which extrapolates a WTP function devised by Chestnut and Rowe (1986) from a regional to a national context.

The impact of air pollution on visibility in recreation areas has been the focus of several US studies studied. Rowe *et al*., (1980) and Hylland and Strand (1983) used CV to estimate WTP for different levels of scenic quality. Crocker (1985) also used CV to value the visual and health effects of defoliation and disease but in the context of insect attack in US forests. Schulze *et al*., (1983) estimated the cost of air pollution in terms of reduced visibility of the Grand Canyon to be in the region of $3.5 billion per year. In some of these studies, the authors report that respondents had difficulty disassociating the visibility effect of air pollution with perceived health impacts: benefit estimates associated with visibility improvement could therefore be over-estimated. In the context of ecosystem valuation, improved visibility is likely to enhance the benefits that viewing ecosystems (from a distance) bring hence the value is likely to be very dependant on the quality of the landscape.

There have been no published studies that have specifically valued air pollution effects on forest biodiversity.

2.2.2 Freshwater

Valuation research in the context of air pollution and freshwaters has focused almost exclusively on damage to fish populations, particularly popular angling species such as trout and salmon.

Early studies include those of Violette (1985), and Mullen and Menz (1985) which estimated changes in consumer surplus attributable to reductions in catch rates and fishable acreage using the Travel Cost method. Mullen and Menz (1985) valued recreational fishing lost as a result of acid deposition in the Adirondack mountains in the north-east of the US. Based on 'angler days' lost, annual costs of air pollution were estimated to be approximately $1.6 million. A simple binary damage function (fish/no fish) was used, with lakes falling into the damaged category assumed to have no value for angling. If substitution is allowed between angling sites (i.e. fishermen move to other unaffected sites nearby rather than stop fishing altogether) then damage costs fell from $1.96 million to $1.66 million.

Criticisms of this study include the simple damage function used and the failure to model demand for angling as a function of more than just fish catch. For example, Forster (1984) argues that many other factors influence the decision to fish such as habit, connections to the site, and particularly attributes which enhance relaxation such as tranquillity and landscape. Despite these limitations Crocker and Regans (1985) went on to use these results in a scaling-up exercise for the eastern US.

Morey and Shaw (1992) using a more sophisticated individual TC model which included a wider selection of trip variables to analyse the impact of marginal reductions in air pollution on the value of angling in New York state. A key finding was that only anglers who considered catching trout as central to their recreational experience had a positive WTP for enhanced catches arising from pollution abatement. Consequently the overall benefits of abatement were small, with a 25% increase in catch valued at around only $3 per angler.

Using the link between air pollution abatement and fish population health generated by the MAGIC model for different species, the EPA report (1999) estimate the benefits associated with the implementation of the CAAA to vary from $12 to $49 million depending on the pH threshold for fish survival assumed. The costs were measured in terms of lost fishing days which were valued using Travel Cost-derived estimates of the value of a fishing day. Apart from the problems normally associated with using this method (see section 1.3.2) the main criticism of this approach is the simplistic binary damage response function (fish/no fish) used which did not allow for intermediate damage levels. Damage estimates for eutrophication were substantially higher ($82-$88 million).

Although the TC approach has been the most popular choice among economists to measure the benefits of recovery in fish populations, some other approaches have been used. For example, Hough *et al.*, (1982) used the Productivity Method to estimate costs of acidification to commercial fisheries in Canada. Driscoll and Menz (1983) imputed the damage caused to fish stocks on the basis of the cost of 'restoring' water quality by liming. Epp and Al-Ani (1979) devised a Hedonic Price model to investigate how stream pH affected property values in rural Pennsylvania and found that a one unit increase in pH increased property value by $1439.

CV has also found application in the context of fisheries and air pollution. Strand (1981) and subsequently Navrud (1989) estimated WTP to protect all fish stocks from acidification in Norway. The Navrud study produced estimates of WTP (based on an open-ended format) in the range 335-387 NOK per annum for emission levels that corresponded to different levels of fish population status. Information was provided about the level of damage and the number of lakes damaged for each emission scenario based on a simple dose-response relationship between water chemistry and 3 distinct population levels (extinct, reduced, and no damage). Although the fish population model had good predictive power for the extinct and no damage levels, the relationship between water chemistry and the reduced damage category was less reliable. The study was repeated by Navrud using a dichotomous choice format with derived mean WTP estimates double those obtained in the earlier study (Navrud, 2001b).

Johansson and Kristrom (1988) used CV to estimate WTP for a programme that would almost completely eliminate sulphur emissions in Sweden. Average WTP per respondent was 4500 SEK per annum. Health impacts were also included hence no separate WTP estimates for ecosystems were provided.

Water quality ladders have been a popular approach to estimating marginal changes in water quality as they describe changes in a number of attributes affected by pollution. For example, Mitchell and Carson (1993) assessed the benefits of the Clean Water Act in 1993 using a ladder that included distinct improvements ('unsuitable for activities' to 'boatable' to 'fishable' to 'swimable'). In Canada, ARA (1982) used CV to estimate WTP to avoid further deterioration in freshwater ecosystem quality from air pollution. A 'quality ladder' was used to describe possible future quality levels, with each 'rung' on the ladder corresponding to changes in a set of ecological indicators such as fish population status, wildlife diversity, and vegetation. Benefit estimates ranged from $260-300 per household depending on the levels of damage.

The environmental attributes affected by pollution have been more explicitly investigated using choice experiments. For example, Johnson and Desvouges (1997) used the Contingent Rating approach to estimate WTP for reduced pollution from electricity generation in the US. Attributes included restrictions on fish consumption bans, human health impacts, employment, and damage to sugar maple.

2.2.3 Grassland/Heathland

No specific reference to valuing effects on this ecosystem could be found.

Table 2.1a: Freshwater Ecosystems

Benefits	Study	Context	Valuation Method[6]	Country
Commodities				
Recreational Fisheries	Macmillan & Ferrier	Acidification & salmon catch	HP	Scotland
	Milner and Varallo	Acidification & salmon catch	MP	Wales
	Mitchell & Carson	water pollution & recreation	CV	USA
	Navrud (2001b)	acidification & fish status	CV	Norway
	Navrud (1989)	acidification & fish status	CV	Norway
	Mullen & Menz	acidification & fish catch	TC	USA
	Morey & Shaw	acidification & fish catch	TC	USA
	Johnson	power plants & environment	CE	USA
	&Desvouges	acidification & fish catch	TC	USA
	Violette	acidification & fish catch	CV	Canada
	Hough et al	liming of acidified waters	RC	USA
	Driscoll & Menz	acidification & fish status	CV	Norway
	Strand			
Water Quality & Yield	Epp & Al-Ani	water quality & property prices	HP	USA
Biodiversity Conservation	Macmillan et al.	Acidification & biodiversity	CV	Scotland
	ECOTEC	acidification & biodiversity	CV	UK
	Hoehn et al.	wetlands	CE	USA
Cultural & historic	Rowe et al.	power plants & visibility	CV	USA
Other				

[6] Valuation Method:

MP	Market Price	PM	Productivity Method
CV	Contingent Valuation	TC	Travel Cost
HP	Hedonic Price	CE	Choice Experiment

Table 2.1b: Forestry Ecosystems

Benefits	Study	Context	Valuation Method[7]	Country
Commodities (timber)	IIASA Gregory et al Callaway et al Crocker & Regans EPA	acidification & tree growth acidification & tree growth acidification & tree growth acidification & tree growth air pollution & tree growth	MP MP MP MP MP	European European USA USA USA
Forest Recreation	Kenyon et al Chestnut & Rowe Rowe et al Schulze et al	Native woodland restoration Scenic quality Scenic quality Scenic quality	CV CV CV CV	Scotland USA USA USA
Water Quality & Yield				
Biodiversity Conservation				
Cultural & historic				
Other				

[7] Valuation Method:

MP	Market Price	PM	Productivity Method
CV	Contingent Valuation	TC	Travel Cost
HP	Hedonic Price	CE	Choice Experiment

2.3 Promising methodologies

This section looks at recent (post 1990) advances in environmental valuation methods, from the perspective of their potential usefulness in estimating the benefits of ecosystem recovery from reduced air pollution. The section cover expressed and revealed preference methods, joint approaches, and benefit transfer.

2.3.1 Stated Preference

With regard to *contingent valuation,* advances have occurred in both how questions are asked and how they are analysed. With regard to the former, the increasing use of the double-bounded dichotomous choice approach (DBDC), associated first with Hanemann, is much apparent. The key attraction of the DBDC approach is that each respondent gives two answers (from the choice no-no, no-yes, yes-no, and yes-yes) to two payment amounts. This increases the information the researcher gains on preferences. However, although the double-bounded format increases efficiency allowing for smaller sample sizes than a single bounded model (Hanemann and Kanninen 1999), it has also been criticised for introducing bias through response incentive and learning effects (McLeod and Bergland 1999). This latter paper is also of interest as it illustrates the increasing interest in using non-parametric approaches for handling discrete-choice CV data.

There is increasing interest in incorporating validity tests in CV studies. One way of doing this is to include a scope test in the design: that is, to test whether WTP is sensitive to variations in the level of environmental quality/quantity being "bought" Carson (1997) has argued that only poorly designed CV studies, or CV studies concerned with unfamiliar goods, fail the scope test. As ecosystem valuation may be unfamiliar to many people, considerable problems may arise in developing a reliable test for scope. Small sample sizes can also be a cause of failure to pass a scope test.

Another way of testing for validity is to calibrate hypothetical CV payments by comparing them with real payments in experimental markets. Many papers still show that real WTP is less than (and often much less than) hypothetical WTP (e.g. Frykblom, 1997). However, this seems to depend on the design of the payment mechanism, the choice of bid vehicle (Carson *et al*, 1999) and the nature of what is being valued. For example, Macmillan *et al* (2001) found that hypothetical donations to a community land trust were lower than actual donations. Although a range of payment methods have been tested in the literature no clear evidence has emerged regarding which method generates WTP estimates that are closer to actual WTP.

Researchers have also been interested in comparing CV bids obtained from standard survey formats with CV-type responses when people have more time to think about and discuss the proposed environmental change (Kenyon *et al*, 2001). Such evidence as is available shows that time to think/discuss leads respondents to revise their bids, often as a result of discussions with family and friends. For example, Macmillan *et al*, (2001) found that mean household WTP from the 'Market Stall' approach, was significantly lower than mean WTP from a conventional interview survey. Furthermore, significant scope effects were found only for the MS sample and variability in individual household WTP was better explained by relevant socio-economic variables. (See Box 2.1 for more details about the Market Stall).

Box 2.1 The Market Stall (MS) Approach

The Market Stall is a group-based deliberative fora for stated preference methods. Typically between 5 and 10 participants are invited to attend two meetings held approximately a week apart.

The first meeting (MS1) is primarily concerned with the presentation of relevant information, described in an 'Information Folder', about the proposed project, and a detailed explanation of the contingent market and payment vehicle. Participants are given the opportunity to discuss any aspect of the project and to question the moderator. A 'Question and Answer Sheet' at the back of the folder is also provided to help clarify issues such as the choice of payment vehicle and the scope of the environmental good. The meeting concludes with the WTP question.

During the week-long interval participants are asked to complete a daily diary in which to record their thoughts and questions about the environmental project and any relevant activities such as watching nature programmes or visiting bird reserves.

At the second meeting participants are given the opportunity to ask further questions and to discuss any unresolved issues concerning the project. After the WTP question is repeated a de-briefing exercise is carried out to establish the extent to which participants understood the process used to establish their valuation of the project.

In comparison with the interview approach, the Market Stall approach provides a very different decision-making environment. In particular it attempts to address three important limitations of conventional interviews:

(i) It provides participants with more time and information to determine their WTP

(ii) Participants can benefit from an informal setting where in-depth discussions with the moderator and other group members can take place

(iii) The week-long interval between the two meetings provides the opportunity for participants to re-evaluate their WTP following further thought, information searching, and crucially for household economic decisions, discussions with family members and/or friends.

In addition more detailed deliberations can be facilitated to provide the decision-maker with a richer and more complete picture of the environmental issue than CV surveys. A recent application of the MS approach to wildlife conservation can be found in Macmillan *et al*, (2001) Finally, researchers have tried to find ways of producing better answers to the question of *who* benefits from environmental improvements. This is important as it defines the population over which benefits are aggregated. In some instances, deciding whose benefits to count is fairly straightforward, but in others this decision is difficult. This is especially the case with non-use benefits, since in principle they could accrue to anyone. For some types of environmental goods, there does seem to be a relationship between distance from the site and the magnitude of non-use benefits. Distance decay functions can be estimated which allow for the appropriate benefiting population to be selected, but distance decay relationships are unlikely to be found in all instances where questions of aggregation are important.

With regard to other expressed WTP methods, an increasing interest has developed in using various types of choice modelling. A full review of these approaches is given in Hanley *et al.* 2001 and DETR (2002). Choice modelling approaches include choice experiments, contingent ranking (Foster and Mourato, 2000) and contingent rating. They have advantages over CV in terms of (i) the ability to estimate values for different attributes of environmental goods more simply and (ii) the fact that each respondent gives many responses, hence sampling is more efficient. Recent environmental applications include studies on water quality improvements, landscape change and forest characteristics. Two problems with such approaches are whether they are incentive-compatible (Carson *et al*, 1999); and the effects of the number of choices offered on models of preferences. Unfortunately not much evidence exists on how close choice modelling estimates of WTP are to "real" WTP.

2.3.2 Revealed willingness to pay approaches

The most relevant innovations in revealed preference approaches would appear to lie in the area of recreation demand modelling. One drawback with the simple travel cost model was the difficulty of incorporating changes in site quality in a satisfactory way (for instance, changes in water quality in fishing streams). Another problem was the treatment of substitute sites. The random utility site choice model (RUSC) has become widely used over the last 10 years as a means of addressing these problems. The RUSC approach models recreationalists' site choices across a group of substitute sites (say, all rivers in a region) as a function of the characteristics of these sites. If water quality falls in river A, then the model allows probabilistic predictions of changes in visits to all sites, and the effect on consumers' surplus per visit. In this way, welfare measures of changes in site quality may be obtained. Nested RUSC models are also widely used, where different decision nodes are represented by different levels of nesting (Kling and Thomson, 1996).

However, two problems exist with the RUSC approach. The first is the old problem of the appropriate value of leisure time: should this be included as a cost, and if so, at what value? A recent discussion of the issues here is Feather and Shaw (1999). The second problem is how to handle participation decisions: that is, how many trips are demanded, not just where they are taken. An area-wide improvement in water quality could quite well increase total trips and count models have developed for predicting such participation rate changes (e.g. Haab and McConnell, 1996). These are usually Poisson or Negative Binomial single-equation approaches that predict how many trips will be taken. Clearly, it would be desirable to combine the "how many trips" issue with the "where are they taken?" issue, since both may change when water quality changes. Various approaches have been used here, such as combining count models with RUSC models in a simultaneous equation approach; and systems of count models (a survey is given by Parsons *et al*., 1998). However, no completely satisfactory alternative has yet been found.

2.3.3 Combined stated and revealed WTP approaches

Environmental economics has recently realised the advantages of combining stated (SP) and revealed preference (RP) approaches, something which has been done for some time in the fields of transport and market research. There are several possible reasons for combining revealed and stated preference techniques: (i) as a check on convergent validity: SP and RP data from the same sample can be compared to see whether, for instance, they reveal the same underlying model of preferences; (ii) as a means of more efficient sampling: In most (but not all) combined approaches, each individual in the sample provides more than one observation; and (iii) to combine the desirable features of the two approaches: We might want to ground SP estimates in actual behaviour, but extend the range of goods and services of interest beyond that currently observed.

Three main approaches exist for combining SP and RP data. These are Random Utility Models combining SP/RP data, and two versions of the contingent behaviour approach: price changes in a Poisson panel model, and environmental quality changes.

2.3.3.1 Random Utility Models combining SP and RP data

Joint estimation of choice models using stated and revealed preference data is widely used in transport applications, although there remain technical difficulties. The basis for the approach is that while people make hypothetical responses to choice tasks in an SP interview, (and therefore their answers may not correspond to what they would actually do), RP data are based on real choices made, and may therefore be more reliable.

Adamowicz *et al* (1997) pioneered an approach to environmental valuation that pooled SP and RP recreational site choice data in a random utility framework. The advantages of this approach are:

- one can specify attribute levels outside of the range of observed values (e.g. higher water quality, better fish catches);
- stated and revealed preference answers can be compared;
- SP responses can be calibrated on RP behaviour; and
- differences in the underlying scale factors can be allowed for

2.3.3.2 Contingent behaviour panel data models of price changes

This method has been applied to the study of the demand for recreation by Englin and Cameron (1996). Their insight was to recognise that some of the weaknesses of traditional travel cost models could be addressed by using a *panel data* approach. Panel data is data where each individual in the sample provides a number of observations. It is widely employed in labour economics, where data on hours worked by *n* workers over *m* months may exist giving a (*n* x *m*) data set, with each worker generating *m* observations.

In travel cost models, data are collected by interviewing recreationalists on site or by mail shot. However, it would be very expensive to repeat the survey for the same group of individuals many times to collect panel data similar to the workers example. In a travel cost study, each person gives two vital pieces of information: how many trips they made to a site or group of sites and the cost to them of visiting the site. If each respondent was asked how they would change their behaviour if these costs rose or fell by some precise amount, then this would generate extra observations for each individual (e.g. we could ask "how many fewer trips would you make next year if your costs were 30% higher than they are at present?"). This process thus provides a data set where for each person there is one observation on existing trips as a function of actual costs (RP data) and a series of observations on predicted trips for a range of hypothetical prices (SP data).

2.3.3.3 Contingent behaviour models of environmental quality changes

This approach is very similar to that outlined in the previous section, except that instead of asking people how their demand for the environmental good would change if its price changed, the interest is in how their demand would change if environmental quality alters. Both pooled and panel models can be used, and the advantages are similar to those set out in the preceding section. Principally, scenarios that lie outside of the range of currently (or historically) observed levels for environmental quality can be used, and the differences in revealed and stated behaviour tested for. An example of this approach is given by Eisworth *et al* (2000).

2.3.4 Benefits transfer

Benefits transfer is not a valuation method, but rather a way of extending the usefulness of original valuation studies. Benefits transfer means using figures obtained in one circumstance to predict values in a different context; for example, using an estimate of the value of improved water quality on river A to measure similar improvements at river B. Benefits transfer is desirable due to the high cost of undertaking original valuation studies. Benefit transfer can be attempted by a unit value transfer approach (where WTP is adjusted to local circumstances such as income differences) or a 'function' transfer, which involves the development of a predictive function or model that can be used to predict WTP in a new context. Variables in the model might include income, environmental membership and descriptors of the environmental change.

Benefits transfer techniques have shown some potential in policy appraisal but need to be improved. Recent work on benefits transfer has come up with rather mixed results, and it is difficult to give a simple answer to the question "is benefits transfer reliable?". For example, Navrud (1994) carried out a test on benefit transfer by comparing original and transferred values for recreational sites affected by hydropower schemes in different locations and found that WTP differed by up to 400%. Ready *et al*., (1999) carried out a benefit transfer exercise in the context of WTP to avoid episodes of ill-health caused by air and water pollution across five European countries with similar mixed results.

Certainly, improvements have been made in some specific contexts, such as forest recreation, coastal water quality improvements, and wetlands, through the use of meta-analysis models. Meta-analysis involves statistical modelling of WTP data obtained from a range of valuation studies in order to predict WTP based on common attributes in the database. Apart from the usual problems associated with statistical analysis of this kind (*e.g.* multi-collinearity and mis-specification) models tend to perform poorly in terms of predictive power. For example, Brouwer et al., (1999) applied meta-analyis to over 30 wetland valuation studies in North America and Europe, but their model only accounted for 37% of variation in WTP. Furthermore the variables that had the most significant explanatory power in terms of WTP were context related (e.g. country of study) or methodological (e.g. payment method) rather than related to policy attributes such as wetland quality.

Chapter 3 Ecosystem Valuation: The challenges

CHAPTER SUMMARY

- This chapter describes a framework for valuing ecosystems in the context of air pollution. The Framework consists of 3 stages: Scientific Linkages, Valuation and Analysis.

- Using evidence from the literature review and the Expert Panels we describe the nature of the challenges confronting ecosystem valuation research, ranging from scientific issues of scale and accuracy to economic concerns about double-counting and embedding.

Scientific linkages

- Knowledge and understanding of scientific effects is most advanced for freshwaters, with considerably less known about forest and other terrestrial ecosystem effects.

- Information is available at the 5km level for determining Critical Loads at a national scale but at a site-specific level, deposition and soil data are poor. Often this means that the damage status of highly valued, but localised, conservation resources such as Sites of Special Scientific Interest (SSSIs) is not known.

- Scientific predictions about the recovery of freshwaters from sulphur deposition are good but less so for nitrogen deposition. Predictions are more difficult for soils due to buffering.

- Dynamic models such as MAGIC currently offer the best methodology to predict recovery over time in both terrestrial and aquatic ecosystems.

- There is a strong requirement to enhance the chemical-biological link in scientific models, particularly with respect to biodiversity and ecosystem function

- Not enough is known about the process of recovery, especially where local extinctions may prevent re-colonisation.

- Confounding factors such as climate and land-use change will have a big influence on the recovery process. New European initiatives such as the RECOVER:2010 project are specifically focusing on the nature and consequences of confounding factors during recovery.

- Predicting ecosystem recovery is characterised by considerable scientific uncertainty, with the degree of uncertainty increasing as the time span increases. Uncertainty in predictions is generally badly represented, especially in the CL approach. Dynamic models can produce numerical representation of uncertainty, but the challenge is to translate this into something that can be understood by the non-specialist.

Valuation

- Many scientific indicators of ecosystem change are ill-suited for benefit identification and measurement purposes. For example, the toxic effects of heavy metals on root growth are difficult to translate to impacts on timber production or tree health. In the case of wildlife, changes in charismatic species such as otters are more likely to be understood and appreciated by people than changes to more obscure species or indicators (e.g. diatom assemblages).

- The potential to use market price approaches is limited in the UK context, as few marketed goods are affected. For those that are, such as timber and grazing, there are no reliable dose-response functions.

- Non-use values are expected to be a major component of the total economic value of ecosystem recovery in the UK. SP approaches are the only way to measure non-use values in monetary terms. Some of the main challenges are describing the complex and unfamiliar nature of the environmental change to the general public, the problem of double-counting, and whether hypothetical WTP is an accurate reflection of actual WTP.

- Recent innovations in environmental valuation techniques offer many advantages for the valuation of the environmental costs and benefits. Clearly, these innovations are useful in a wide range of policy contexts: for example, the greater statistical precision of WTP estimates from contingent valuation is valuable in many instances. One development that has particular relevance to ecosystem recovery is the use of deliberative valuation fora such as the ‘Market Stall’ which allow people more time and information to consider their WTP.

Policy Analysis

- For policy purposes, benefit estimates have to be scaled-up to the relevant level of aggregation and must be analysed in a way that lends itself to both economic appraisal and policy discussions and negotiations. Some of the main issues that are likely to be of concern include scaling-up benefit estimates from case-studies, identifying beneficiaries, and the treatment of uncertainty and error in both scientific and economic models

3 Ecosystem Valuation – The Research Challenges

Estimating the economic benefits arising from ecosystem recovery requires an understanding of how marginal changes in emission levels will affect our welfare. This chapter describes a framework for ecosystem valuation that charts out the stages that require to be taken in order to link emissions to welfare. The framework consists of three distinct phases: Scientific Linkages, Valuation and finally Analysis (see Figure 3.1).

Air pollution and ecosystem recovery is recognised as one of the more difficult areas for valuation. Using evidence from the literature review and the Expert Panels, we also describe our capability to implement this framework based on current knowledge and models, and highlight some of the main challenges facing research.

3.1 Scientific linkages

In order to link changes in air pollution levels to environmental effects, and ultimately economic impacts, complex modelling of the physical relationship between the emission, transportation and chemical reactions in the atmosphere of air pollutants and their impact on the environment are required.

3.1.1 Emissions - transport – deposition & concentration

The link between emission sources and atmospheric transportation, deposition and concentration can only be established using long-range pollutant transport models. The EMEP Eulerian model has replaced the Lagrangian model for work under the CLRTAP and operates at a spatial resolution of 50 km^2. The UK emissions/depositions are modelled at a $5km^2$ scale using HARM for NO_X and SO_2 and FRAME for ammonium. Considerable uncertainty must obviously surround modelling of this nature due to the range of factors influencing deposition patterns and concentration. Transport models for SO_2 and acid deposition are the most robust, with the largest uncertainty associated with ammonia, VOCs and heavy metals.

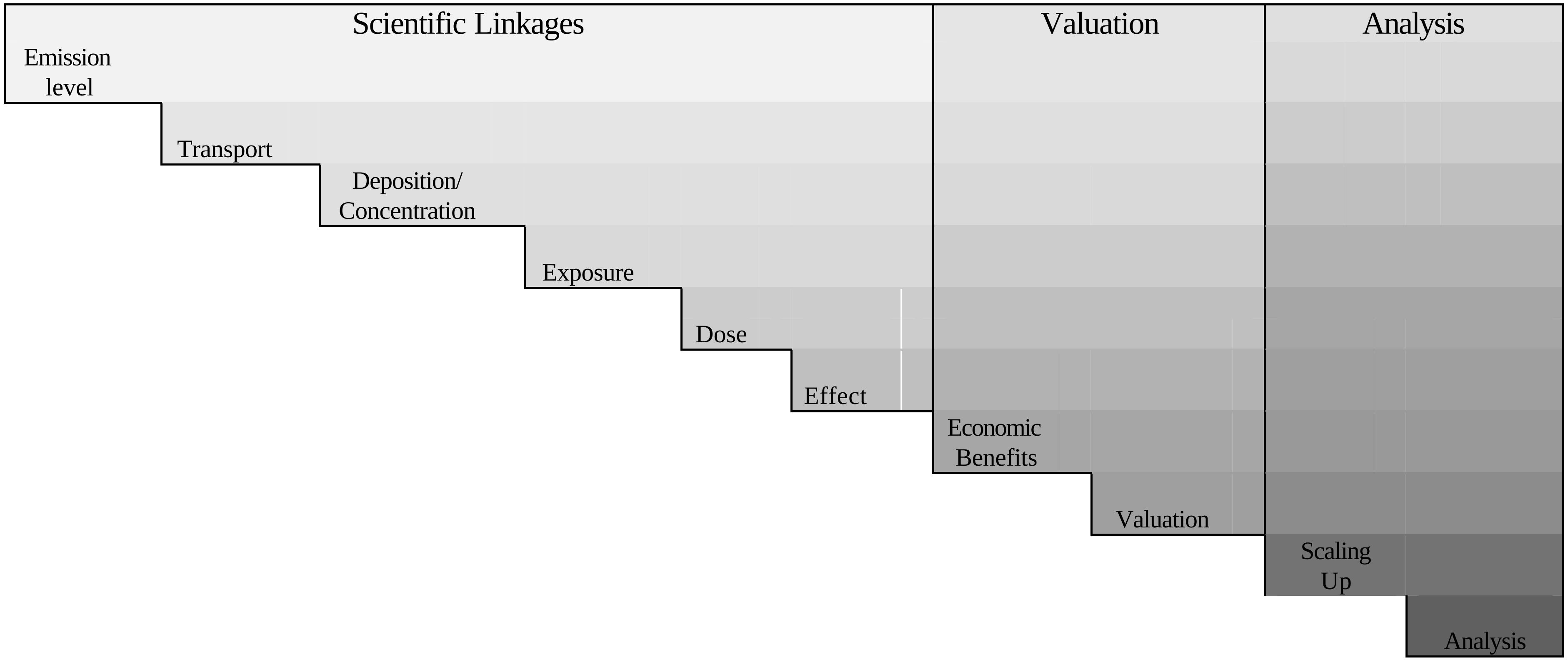
<u>Challenges</u>

- **Spatial scale**
  In general emission and transport modelling is appropriate, however, the distribution of deposition across the landscape is key. Available information at 5km level may be appropriate for determination of Critical Loads at national scale and regional modelling, but at a site specific level, data (modelled and observational) are poor. This is particularly relevant to recovery in highly valued, but localised, conservation resources such as National Nature Reserves (NNR) or Sites of Special scientific Interest (SSSIs).

3.1.2 Exposure-Dose-Effect

This stage involves predicting the biological or chemical response in the ecosystem as a result of exposure to air pollution. Establishing a reliable and appropriate link between exposure to air pollution and an environmental effect has proven difficult as ecosystem effects are pervasive rather than locally acute or dramatic (although some short term impacts have occurred such as fish kills). The buffering capacity of the soil

Figure 3.1 Valuation Framework



has meant that many effects are likely only to emerge over long time spans.

To help understand the long-term impacts of air pollution on ecosystems scientists have examined environmental status of lakes on a spatial and temporal basis. For example, the effects of acidification on freshwaters have been inferred from comparisons of biota along a pH gradient (Flower and Battarbee, 1983), and from comparison of present day flora with historical surveys (Jones *et al*., 1986). Experimental work, has also been used extensively in researching the impact of air pollution on commercial agricultural crops.

Figures 3.2a-c summarise the main potential scientific effects of air pollution on forests, heathlands, and freshwaters based on existing experimental work and monitoring. Evidence for these effects in the UK and the ecological criteria used to measure these effects are also described.

Overall understanding of scientific effects is best for freshwaters, and poorest for heathland/grassland. Models to predict effects based on changes in ambient air pollution levels are also more developed for freshwaters, with reliable biological links to diatom and fish populations. Below we set out what we consider to be the most significant demands on science in terms of providing appropriate dose-effect relationships for ecosystem valuation.

Research Challenges

- **Effects Transfer**
  Given a lack of evidence for some effects in the UK some researchers have taken the expedient step of applying a dose-effect model to a new context, often outwith its geographic or environmental parameters. For example, Gregory *et al*., (1996) applied a dose-effect relationship developed for forest species in the 'Black Triangle' of Eastern Germany, Southern Poland, and the Czech Republic) to estimate the damage to all European forests from a coal-fired UK power station. Effects transfer of this nature is often not appropriate due to differences in confounding factors such as climate and other pollutants.

- **Dynamics of ecosystem recovery**
  Ecosystem recovery from the effects of air pollution is a dynamic process, with the rate of recovery dependant on reductions in pollutant load, and the current biological and chemical status of the ecosystem. Unfortunately, current understanding of colonisation dynamics and biological processes of affected biota are generally insufficient to provide expected time-scales and patterns of recovery. Particular challenges include hysteresis, thresholds and episodes, and the influence of confounding factors.

  While pH, sulphate levels and some diatom communities have shown a rapid recovery toward pre-acidification levels in recent years, historical monitoring may also be of limited help in developing predictions about future recovery due to confounding factors such as climate and land use. For example, large-scale

Figure 3.2a: Forest Ecosystems

Potential Effects	Evidence in UK	Ecological Criteria	Relevance to Benefit Valuation
Decreased production through root damage and toxicity to Al **(P)**	No field based evidence (limited to extrapolation from laboratory/ greenhouse manipulation)	Ca/Al ratios; [NH_4] toxicity; BC/Al	Unclear consequences for timber production and tree health.
Increased production from enhanced N deposition **(NW)**	Unclear - influence of grazing changes and management are difficult to separate. Time lags in response	-	Unclear consequences for timber production and tree health
Changes in ground flora **(BNW)**	CS2000 reports; • Mean species richness in broadleaved woodland has declined (could be due to changes in management or succession following the 'Great Storm')	Ellenberg scores Spp. Presence/absence	Difficult to convey these changes in species diversity to public in SP study. Unless effects on key 'target' species can be identified, would have to rely on proxies such as 'expert opinion' or SSSI designation to guide valuation.
Loss of epiphytic lichens **(NW)**	• Ellenberg fertility scores have increased in England and Wales	presence/absence (re-colonisation slower than loss)	
Tree death (exposure to elevated SO_2) **(P,NW)**	M. Ashmore's work - presence/ absence linked to historical deposition		Acute effects less difficult to value but no evidence in UK
Less acute damage (NOx, ozone, frost, pests)**(P,NW)**	No • Weak correlations • Canopy conditions (needle loss, yellowing) arising from range of stresses • 2nd any impacts of reduced canopy cover resulting in eutrophication response of ground flow	- % loss of canopy Altered ground flora (e.g. bramble increase, etc)	Difficult to 'attributise' damage to air pollution and establish link to either timber production or to public preferences concerning forest health.
Below-ground biodiversity	Not much evidence though highly likely to have impact	Loss of ecosystem structure and function	No clear linkage to ecosystem services that can be valued

P - production forests; **NW** - native woodland; **BNW** - broad-leaved native woodland

Figure 3.2b: Freshwaters

Potential Effects	Evidence in UK	Ecological Criteria	Relevance to Benefit Valuation
- potable impacts	- Problems with acidity and Al in localised areas	-	Simple to value using additional treatment costs. Benefits relatively low due to low cost of treatment
- colour, aesthetics	- Waters "clear" during acidification potential increase in DOC during recovery	Little impact (except decreased light penetration)	No obvious impact on benefit valuation - some people may prefer clear water
- change in chemistry	- Strong evidence	Altered ANC, pH, Al impacts on aquatic ecosystems (See below)	Benefits of recovery more appropriately linked to recovery in fish and biodiversity than chemistry
- Direct fish kills	None documented in UK, and difficult to predict	-	-
- fish population decline	Yes, well documented	Potential episodic impacts in spawning streams. Salmon more sensitive than trout	Recovery linked to increased fish catch for both salmon and trout (HP and TC methods). However problems with confounding factors (disease, high seas catch) with salmon.
- Biodiversity of fish populations	Yes but limited	Presence/absence of char, powan, etc	Loss of endemic fish species of considerable concern to public (SP methods)
Change in spp. Composition; • Invertebrates • Macrophytes • Bryophytes	Yes	Loss of key species Abundance (within species) Community composition (diversity)	Concern about changes in food chain, but not in public eye. Better to describe in terms of changes in terms of higher order species (fish, birds) lost (see below) or recovery in SSSIs damaged by pollution.
Bird and animal populations	Yes - for dippers, otters maybe	Key species	Effects on key species less difficult to describe in an SP study than impacts lower in food chain.
Upland freshwater eutrophication (N)	N limitation responses unclear	Eutrophication response/algal blooms	

Figure 3.2c: Grasslands, and Heathlands

Potential Effects	Evidence in the UK	Ecological Criteria	Economic Criteria
Species Composition (N impacts dominate, along with ozone)	Strong for grasslands and heathlands CS 2000; • Significant increase in fertility score for England and Wales for acid grassland • Between 1990 and 1998 shift towards vegetation types characteristic of moorland grass and net movement away from heath/bog • Significant increase in fertility scores for bog broad habitat • Ozone reduced frost hardiness of *Calluna vulgaris* and affected root growth and physiology	Fertility scores increased Change in species assemblage but difficult to separate N from management impacts	Difficult to convey these changes in species diversity to public in SP study with exception of walkers who may recognise changes. Unless key 'target' species effects are identified, (possibly moorland species like red grouse, dottrel) would have to rely on proxies such as 'expert opinion' or SSSI designation to inform valuation
"Stress" (pest attack)	Increased frequency of heather beetle outbreaks on Netherlands - possibly in UK, but have to assume precautionary principle	Loss of "heather"	Link is weak and hard to describe in SP study
Loss of bryophytes	Direct effects on *Rhacomitrium*. Evidence to show that bryophytes are not too sensitive to ozone, except *Sphagnum recurvem*	Loss of vegetation cover and key species	Enhanced erosion and loss of visual amenity could be valued using SP but very localised. Impact on water quality negligible -
Increase in bracken	Evidence from Pennines but confounded with changes in grazing and trampling by walkers	Possible spread of bracken	Loss of grazing could be valued using MP approach but benefits likely to be very low. Difficult to assess any impact on landscape appreciation, but not thought very important
Indirect effects on birds	Circumstantial evidence only		Reduced grouse bags could lower value of grouse moors

coniferous afforestation in Galloway has been implicated in lake acidification. Although European research initiatives such as the RECOVER:2010 project are specifically focused on identifying the nature and consequences of confounding factors during recovery, more research is required.

Hysteresis refers to the delay observed between emission reductions and ecosystem recovery. Hysteresis in the recovery phase is both physiochemical and biological. For example, geochemical controls on sulphur de-sorption from soils will delay recovery of freshwater ecosystems. Similarly, in ecological situations it is easier to lose species than to get them back, and some intervention may be required (see 'Thresholds' below). Experiments with liming in Scandinavia indicate that some species, like trout, respond quickly to improved conditions, but some floral and invertebrate assemblages tend to show a marked delay in response (Weatherly and Ormerod, 1991). Local and regional species pools are important for recovery: isolated sites may recover slower than ones near unaffected areas. Hysteresis emphasises the need to predict timescales for recovery: people with highly positive rates of time preference are likely to value a rapid recovery more than a more prolonged process (see section 3.3.1).

Biological recovery processes may be highly non-linear, and characterised by threshold effects. For example, Montgomery *et al* (1994) describes how the probability of survival of the Northern Spotted Owl, increases from 0.1 to 0.9 as the area of old growth forest habitat increase past a certain area threshold. In the case of acidification, some sensitive species may only re-colonise if water pH exceeds a certain value. These threshold effects and non-linearities do not sit well with marginal analysis of economic theory (and hence Cost Benefit Analysis).

Although dose-effect models should incorporate threshold effects, where they are known to exist, it is clear from the literature that unsubstantiated threshold values have tended to be used as a substitute for dose-effect relationships based on scientific observation. For example, several recreational fishing studies (e.g. Mullen and Menz, 1985) assumed simple threshold values for fish survival based on average lake pH. This crude approach oversimplifies damage effects and could lead to biased estimates as damage is likely to be more linearly related to pH or other water quality variables.

Damage to freshwater organisms can also be mainly associated with peak episodes of acidity (acid flushes) which wipe-out entire populations. Many scientific dose-response models are based on average values of exposure and are not sufficiently sensitive to account for these 'catastrophic' events.

- **Risk and Uncertainty**
  Ecosystem response to reduced air pollution is highly uncertain, with the degree of uncertainty increasing with time span. Although several valuation studies have attempted to quantify uncertainty numerically (e.g. EPA, 1999) existing models, including CLs, generally represent uncertainty badly. There is a need to quantify uncertainty in terms of risk but immediate prospects of doing so are not good. Scenario modelling and sensitivity analysis will continue to play an important role in presenting uncertainty to the

decision-maker, but its portrayal to the general public (in a CV exercise for example) remains problematic. Where risk has been incorporated in CV, WTP for specific programmes has been affected (Johansson 1987; Macmillan *et al.*, 1996).

- **Holistic approach**
  Air pollution can have pervasive effects on ecosystem function. For example, the effects of acid deposition on the ecology of semi-natural environments are multiple and operate at a number of temporal and spatial scales. Some organisms, such as fish and invertebrates can be killed by severe, and relatively transient acidic conditions following rainfall events. If the fish population fails to recover, long term changes in the composition of prey species at lower trophic orders in the aquatic ecosystem will occur. Indirect effects on the food chain are also important. For example, as acidification proceeds, humic matter is precipitated, lakes become clearer and as the light and temperature regime alters, widespread changes in the composition of flora and fauna can occur.

  Ideally ecosystem recovery models should describe ecosystem change in an holistic context, rather than focus on one or two indicator species. Currently there is no satisfactory single measure of biodiversity response for the non-specialist, but modelling changes in different species assemblages is progressing, such as those represented for grassland/heathlands within the MOVE formulation. (Although this is not a dynamic model and hence cannot predict the speed of recovery). Individual species indicators may be more useful when the species involved are ‘keystone’ species for that ecosystem, or are ‘charismatic’ in the sense that they are highly valued by the public in their own right.

- **Critical Loads**
  Critical Loads (CLs) are defined by the UNECE as 'a quantitative estimate of exposure to one or more pollutants below which significant harmful effects on sensitive elements of the environment do not occur according to present knowledge.' CLs have been defined for terrestrial and freshwater ecosystems based on a threshold value above which damage is believed to occur. As CLs vary spatially, they can be represented as a map. Land management and other confounding factors that can cancel or lead to heightened susceptibility to pollutant inputs can also be incorporated.

  By combining CLs with spatial deposition data, the resulting map can identify the location of areas where critical loads are exceeded (Exceedance maps). A number of Critical Load Exceedance maps for different receptors (*e.g.* buildings, freshwaters and soils) have been commissioned by the UK government as a contribution to international negotiations. While CLs have proven to be a helpful negotiating tool for guiding abatement decisions they have several limitations.

  1. The CL approach is based on a threshold above which environmental damaged is anticipated. The degree of damage is not considered.

2. There is no dynamic element within the critical loads approach. Time delays and potential hysteresis during the recovery phase are not assessed.

3. Although CLs provide a measure of ecosystem sensitivity (freshwaters indicated by ANC: forest soils by Ca/Al or BC/Al ratios), these are poorly linked to observed patterns of damage (effects).

4. Land use practices are included in the calculation of terrestrial critical loads, but secondary impacts, such as interactions with climate, pests, diseases and multi-pollutant interactions with ozone are poorly represented.

3.2 Valuation

The valuation stage is concerned with translating recovery, described in scientific terms, to monetary benefit estimates. Two stages are involved. First, ecosystem recovery has to be described in terms that are meaningful to people or are relevant to their welfare in some way. The second stage involves applying appropriate valuation techniques, to convert these impacts to monetary values. In the rest of this section, and drawing on the results of the panel meetings, we identify some of the main research issues concerning valuation.

3.2.1 Benefit identification and measurement

Air pollution effects on ecosystems are complex, hence they may prove difficult to convey to people in non-scientific terms. Although these effects are pervasive, scientific research into effects has tended to focus on measuring specific criteria that are relatively easy to measure and provide a reasonable indication of overall ecosystem status. Examples include chemical criteria such as pH and ANC, or biological criteria such as diatom counts or presence/absence of indicator species.

Issues

- Although some measures can be cost-effective scientific indicators of ecosystem change, they may often be ill-suited to benefit identification and measurement. For example, toxic effects of heavy metals on root growth may be difficult to translate into 'welfare' impacts such as timber production or woodland appearance and health. In the case of biodiversity values, changes in charismatic species such as otters are more likely to be understood and appreciated by people in a stated preference exercise than changes to biodiversity indices or diatom assemblages.

- Some important ecosystem changes are imperceptible to the human eye and it is difficult to describe these changes to people in a simple, but accurate, way. For example, changes in diatom assemblage or pH are not perceived to be as strong a signal of damage as acute pollution events that result in fish kills and dead trees! One way to overcome this within the context of an SP study, would be to emphasise final long term outcomes, which might be fish death, rather than the current intermediate situation (e.g. fish morbidity).

- Science has not yet fully revealed how ecosystems function and the indirect services they ultimately provide human society. Even with relatively simple, or well understood ecosystems, it is often difficult to determine the causal relationship between human actions and ecosystem functions and processes. Where this is a predominant concern, one would have to question the desirability of attempting monetary valuation.

- Where rigorous translation of scientific effects into benefit impacts is not possible, the researcher may have to resort to subjective judgements about impacts. For example, how changes in water chemistry will affect species higher up the trophic order. Expert groups have been used in this situation, but there is potential for disagreement and bias to emerge in areas where speculation of this kind is involved.

3.2.2 Valuation methods

An important decision facing the researcher is to select an appropriate valuation methodology. The choice of method has to take account of applicability in terms of the benefit stream being measured (i.e. use or non-use), the reliability of the scientific base and cost. In this section we consider the potential to apply each valuation approach and some of the main issues that might arise.

3.2.2.1 Market Prices

The Market Price approach is the most straightforward to apply if price, quantity and cost data can be easily obtained for established markets. In the case of air pollution few of the ecosystem services affected are bought and sold with the exception of timber and salmon fishing rights. There is potential to apply the PM to estimate the benefits associated with reduced water treatment costs as a result of reduced acidity in potable water supplies, but the actual costs are unlikely to be significant.

Issues

- As yet there is no adequate dose-effect relationship for linking air pollution to timber growth and yield in the UK, nor for predicting the effects of nitrogen reductions on grazing quality. In order to be reliable, dose-response relationships would have to take account of confounding effects such as management and climate.

- In the case of timber, where harvesting may take place many years following damage, it will be necessary to predict future timber prices. Over the last 30 years, the price of UK timber has fallen in real terms: this trend may or may not continue.

- The price of some goods are influenced directly or indirectly by government policy (such as the Common Agricultural Policy) hence it would be necessary to correct for price distortions using shadow prices. Shadow pricing would also be required where production of the good is associated with non-market externalities. For example, timber transport generates a range of negative externalities such as air pollution and noise.

3.2.2.2. Revealed Preference

The Travel Cost method has been relatively widely used in the US to estimate the costs of acidification to recreational fishing but benefit estimates are relatively low compared to health and agricultural impacts (EPA, 1999). There have been no equivalent TC studies in the UK despite the clear scientific evidence of damage to fish populations in regions such as Galloway and North Wales. However, these areas are relatively remote and inaccessible and it is possible that anglers would not fish these areas more often following recovery.

Salmon fishing is considerably more valuable in the UK than trout fishing, but it is a marketed resource. In the case of the Galloway salmon fishery the HP approach has been used to value recovery (Macmillan and Ferrier, 1994). The scope for applying the HP approach is otherwise restricted as few capital assets such as houses or services such as labour are potentially affected by ecosystem recovery.

Issues

- TC would be costly in terms of data and time to implement, especially where the link between ecosystem recovery, fish populations and angling behaviour are not clear. For example, an increased chance of catching a fish may not have a major influence on the decision to visit a remote upland lake to fish.

- Problems remain regarding the treatment of travel time in TC (is it a cost or a benefit?) and how to handle multi-purpose trips.

- Older versions of TC models found it difficult to satisfactorily deal with site substitution, leading to the mis-estimation of benefits from water quality improvements. New refinements which introduce a random utility approach to travel cost modelling can tackle these issues in a theoretically-consistent manner but these are still under development (see section 2.3.2).

- Another drawback of traditional RP approaches is that it is difficult to estimate the value of changes in environmental quality indicators outside of the range currently observed in the sample. This would be a problem if, for instance, pH levels in fishing streams rise above those currently observed anywhere in a region. By using a combined revealed and stated preference approach economic benefit figures could be estimated for this un-observed region, for instance by asking fishermen about intended changes in their behaviour should a predicted change in water quality occur.

- Typically the HP approach relies on cross-sectional data from a range of properties. However, cross-sectional data do not properly reflect the relationship between price and environmental quality in a dynamic context. Abelson and Markyanda (1985) show how this can lead to benefits being underestimated when the future level of environmental quality changes over time because the model may not allow expectations about future recovery (e.g in salmon catch levels) to influence price.

3.2.2.3 Stated Preference

Non-use values are expected to be a major component of the total economic value of ecosystem recovery in the UK. Recovery in freshwater biodiversity or changes in water quality are likely to be valued by a wide cross-section of people who neither visit polluted areas or use them in any direct way. SP methods are the only way to value non-use values.

Due to the complex and long-tem nature of ecosystem recovery from air pollution, valuation and the application of SP methods is likely to be demanding. Some of the main issues that would arise are discussed below.

<u>Issues</u>

- **Complex and unfamiliar environmental changes**

  During an SP interview respondents are expected to assimilate information about the environmental project, search their memory for other pertinent information, integrate this into a judgement about their WTP based on their preferences and income, and communicate this judgement to the interviewer. For decisions involving unfamiliar and/or complex environmental projects, such as air pollution effects on ecosystem, or methods of payment (e.g. special funds), the SP survey places considerably greater demands on the consumer than most market transactions. In particular, people would seem to have problems understanding the concept of biodiversity (Hanley *et al*., 1995) and understanding how ecosystems function. For example, Hoehn *et al*, (2001), describe how several respondents in a CE study assumed that wetlands kill trees, despite the fact that wooded wetlands were an ecologically valuable habitat in the region.

  One way to overcome this problem is to describe ecosystem change in terms of a range of ecological indicators. A number of CV valuation studies used specially selected indicator species as proxies for wider more pervasive impacts of air pollution on ecosystem quality. For example, Green and Tunstall (1990) used selected ecological indicators to describe water quality changes and Navrud (1989) used fish population status as a proxy for ecosystem health in the context of acidification. Describing ecosystem recovery in terms of top-predators or keystone species that may be familiar to people has obvious advantages in terms of information requirements. It also has some ecological merit in the sense that these species tend to reflect overall ecosystem health. However, there is a concern that over-reliance on indicator species could over-simplify the environmental changes involved and could for example, result in people valuing the species rather than the ecosystem. Valuation workshops and other group-based approaches may be more desirable in this context than conventional surveys, as they give people more information about the project and the opportunity to explain and discuss with participants the nature of the ecosystem change.

- **Valuing the ecosystem or the sum of the parts**

  If individual benefit streams are valued separately and then added up there is a risk of over-estimating the value of recovery. Apart from concerns about income and substitution effects, dividing ecosystems by function (wildlife, landscape recreation, etc.) and valuing them as separate attributes (forest health, distinct from freshwaters) is somewhat arbitrary and could lead to double-counting. For example, Hylland and Strand (1983) found that WTP for improved visibility from reduced air pollution was influenced by the respondent's perceptions about the impact on their health, and Hanley and Ruffell (1993) found that many respondents preferred certain forest landscapes because of their perceived impact on wildlife, rather than their appearance.

- **Embedding**

  Embedding manifests itself in three ways: lack of sensitivity to the scope of environmental change, sequencing bias, and sub-additivity bias. With a range of goods and services we would expect both sequencing and additivity effects to emerge due to income constraints and various substitution possibilities. However, scope tests are an important way of assessing the validity of CV because economic theory predicts that consumers will prefer more of a good to less. Given the degree of uncertainty surrounding future ecosystem recovery scope tests, executed through the presentation of various future recovery and damage scenarios, would not only provide a scope test but also provide policy-makers with a guide to the potential range of benefits associated with recovery.

- **Uncertainty and information**

  Many of the benefits of reduced acidification are subject to scientific uncertainty and this uncertainty can be directly incorporated in valuation exercises (e.g. MacMillan *et al*. 1996). However, an additional source of uncertainty is that connected with people's preferences for unfamiliar environmental goods: people may be unsure what their preferences are and about how much they would be WTP (Kask *et al*., 2000). The latter may be helped by the use of payment ladders that include a range of uncertainly about WTP (e.g. definitely would pay, probably pay etc), while the former could be resolved by giving people more time and information to discuss/reflect on their values, as happens in the deliberative valuation fora such as the "market stall" approach (MacMillan *et al.*,2001) or valuation workshops (Kenyon *et al.*, 2001).

- **Are preferences actually revealed?**

  There is an on-going debate regarding the extent to which stated willingness to pay reliably reflects what people would actually pay. This is part of wider concerns common to all survey techniques, that respondents may actually be answering a different question than the surveyor had intended. One example is where a respondent makes associations about the environmental good that the researcher had not intended: for example, if asked for willingness to pay for improved visibility (through reduced pollution), the respondent may actually answer based on the health risks that he or she presumes are associated with dirty air. A second example, would be where the respondent takes the opportunity to express support for the environment rather than express their WTP for the specified project or benefit from the act of 'giving' (warm-glow effect).

  Strategic behaviour such as free-riding or over-bidding to influence the decision about whether the good will be provided or not is also problematical. Protesting is also common and can occur where respondents who actually value the good, state that they are not willing to pay for it because they object to some aspect of the contingent market (e.g. the mechanism for payment). In an air pollution context where cross boundary effects are common there would appear to be considerable scope for protesting over who should pay and how.

An underpinning assumption of the stated preference approach is that people are capable and willing to make a trade-off between income and ecosystem quality. Some participants may reject this potential trade-off and may refuse to answer the question on moral or philosophical grounds. For example, they may feel that it is wrong to base decisions about the environment on money, or that the environment is 'priceless' in the sense that no trade-off is worthwhile. In SP applications people who hold such 'lexicocraphic' preferences are normally treated as protestors and excluded from further analysis. Consequently their preferences are ignored for the purposes of valuation, even though they value the environment highly. In the case of ecosystem restoration, where lexicographic preferences might be quite prevalent, questions arise as to how to minimise responses of this nature, or how to incorporate these values within the confines of the CBA approach.

- **Bequest values and future generations**
  Ecosystem bequest value, that is the value we place on the benefits that would accrue to future generations as a consequence of ecosystem recovery, is an important motivation for current generations being prepared to pay for reductions in air pollution. However, a fundamental problem for CBA and ecosystem valuation in particular is that the preferences of future generations are not incorporated.

- **Functional form of WTP model**
  In common with RP approaches, benefit estimates in SP models, particularly CE and poly-chotomous choice CV studies, model specification can have a major influence on the magnitude of the benefit estimate. For example, Whitehead (1990) found that WTP for wetlands preservation varied between $2.9million and $19m depending on functional form of the WTP model.

3.2.2.4 Imputed preference

Replacement cost and defensive expenditures have frequently been used to estimate ecosystem benefits. Despite their theoretical shortcomings they are relatively easy to implement and are perceived as reliable because they are based on cost information that can easily be established. In the context of ecosystem recovery from air-pollution in the UK, liming costs would be an obvious focus for such a study.

Issues

- **Costs not an accurate guide to benefits**
  These methods do not consider social preferences for ecosystem services, or individuals' behaviour in the absence of those services. Thus, they should be used as a last resort to value ecosystem services.

- **Inconsistency**
  The methods may be inconsistent because few environmental actions and regulations are based solely on cost-benefit comparisons, particularly at the national level. In some cases, the cost of a protective action may actually exceed the benefits to society.

- **Perfect Substitutes?**
  The goods or services being replaced probably represent only a portion of the full range of services provided by the natural resource. Thus, the benefits of an action to protect or restore the ecological resource would be understated. For example, liming has been widely used to restore lakes in Scandinavia and although fish populations have recovered many other invertebrate and plants communities have not.

3.3 Policy Analysis

For policy purposes, benefit estimates have to be scaled-up to the relevant level of aggregation (normally the national level), and must be analysed in a way that lends itself to both economic appraisal (such as CBA) and to policy discussions and negotiations. For example, in the context of air pollution it may be desirable to produce benefit estimates that are compatible with existing policy drivers such as Critical Loads. In this section we describe some of the main issues that are likely to arise with regard to policy analysis.

3.3.1 Scaling-up

Scaling-up is the term given to adjusting benefit estimates from the case-study or experimental level to the desired policy level. Policy-makers desire benefit estimates at different spatial scales. National estimates tend to be of greatest interest, for example to support international negotiations, but regional estimates may also be required to highlight distributional effects of a proposed policy.

There are several approaches to scaling-up depending on the valuation method used. At its simplest, scaling-up may only involve aggregating mean WTP from a representative sample involved in a CV exercise to the total population. (Although adjustments may be necessary if the sample is biased or not representative in some way). With other approaches, scaling-up can be a more complex technical exercise involving benefit transfer from site-specific models to the entire country based on per-unit measurements, such as value per hectare or value per fish. Temporal aggregation, that is aggregating benefits and costs over time is another aspect of scaling up that is particularly relevant to ecosystem valuation due to the long term nature of recovery. Assumptions about time preference and discount rates are clearly important here.

Issues

- **Boundary effects**
  Ecosystems may be more heterogeneous than markets. Economic boundaries are determined by markets, while scientific boundaries are spatially defined. This creates problems for scaling-up where catchment-based scientific models are combined with national economic data to predict changes in fisheries value for instance. Environmental and political boundaries often do not coincide and this can also create problems for valuation. For example, air pollution control in the UK would benefit ecosystems in several other European countries.

- **Area Affected**

In order to estimate national benefits we need to know the total area affected, either to scale-up catchment estimates or to describe the environmental good to people in an SP exercise. Critical Loads provides a useful guide to the location and extent of damaged areas but is relatively broad brush and does not identify areas of special importance such as SSSIs that might be affected.

- **Population Affected**
  The magnitude of benefit estimates is influenced to a great extent by the population size of the beneficiaries. Non-use values often dominate the total economic value of many ecosystem resources because non-users typically outnumber users by a considerable margin. While sampling for non-use values among the general public can be straightforward if sample size is sufficiently large targeting users can be much more difficult. For example, anglers may be expected to benefit from improved fish population status in a number of freshwater rivers and lakes, yet they may be hard to identify as not all anglers are members of a club, and permit records are often incomplete.

- **Spatial and Attitudinal Aggregation**
  We can anticipate that many of the benefits of biodiversity improvement will be non-use benefits. People who do not go fishing, plant-spotting or bird-watching may still value an improvement in upland biodiversity: just from knowing that more dippers or more otters are around, for instance. It is important to sample non-users that recognises the potential for WTP to vary based on distance from site, or other contextual factors such as attitude to the environment. The Axford enquiry showed how important the issue of the "appropriate population" is for non-use benefits in a policy context (Moran, 1999). Where location is important, 'distance-decay' functions can be developed to identify how WTP varies depending on distance from the site of interest. In the context of air pollution, where many of the affected areas are remote from centres of population, there is a possibility that household WTP may vary strongly by region.

- **Temporal Aggregation**
  A problem of particular relevance to ecosystem recovery is that the relevant population must be decided on both a spatial *and* a temporal scale. Whilst distance decay methods might reveal how many households in a region should be counted as beneficiaries from an improvement in upland biodiversity, the question remains as to *when* benefits will occur, both in terms of when they start to appear and how long they persist for. This is important both in terms of intergenerational equity and for the current generations' preferences for rapid recovery. The conventional assumption is that people prefer benefits sooner rather than later, but the choice of discount rate is contested especially for long-term environmental changes such as ecosystem recovery from air pollution. A larger discount rate gives more weight to the benefits received by the current generation than those received by future generations. Many have argued for a social discount rate for environmental projects that is lower than the market rate, in order to leave more opportunities for future generations. From a more fundamental perspective, the main criticism of CBA and other decision-making tools is our inability to accommodate the preferences of future generations .

- **Price and income effects**

  In the case of marketed goods such as timber, pronounced impacts on production may affect the market price of the timber, or indeed of other production inputs or outputs. Income may also be affected leading to an array of altered consumption and production decisions. Modelling these impacts is very difficult at the macro-level and hence they are often ignored.

3.3.2 Analysis

Analysis should provide the decision-maker scope to apply the results in a practical way to policy, while being consistent with the fundamentals of economic appraisal. In the case of air pollution, economic benefit estimates are expected to play a role in international agreements about abatement, a debate that has thus far been dominated by costs to industry and scientific effects. Issues that need to be considered are described below.

Issues

- **Compatibility with policy drivers**

  European decisions on air pollution are in part driven by Critical Loads. Although the Critical Load approach creates difficulties for economic valuation (see section 3.1.2) there would be some advantage in ensuring that benefit estimates are complimentary to the CL approach. Benefit values for ecosystem recovery could be derived for alternative 'target load' scenarios, if CL grid squares are used to help describe or delineate recovery. For example, Macmillan and Ferrier (1994) used CLs to define the boundary and area of the Galloway salmon fishery affected by acidification. In SP approaches, alternative recovery scenarios could be presented by changes in the area identified as being damaged by pollution (i.e. above the CL threshold).

- **Marginal estimates**

  For the purposes of CBA it is necessary to compare marginal costs with marginal benefits. Valuation studies should therefore attempt to indicate how benefits will change with different levels of abatement. For some valuation methods this may not be difficult, but in the case of CV the survey design would have to accommodate a range of recovery scenarios.

- **Policy benefits also influenced by predicted damage under the status quo**

  In order to assess the net impact of reduced air pollution it is necessary to consider the degree and extent of future damage under the *status quo* (i.e. if no further action is taken) as well as future recovery following abatement. Existing evidence, although sketchy, would suggest that under the *status quo,* species diversity will continue to decline on some affected areas, with local extinction of a range of species, including fish, passerine birds and otters, possible. Evidence from a recent CV study found that WTP for abatement was significantly larger for an abatement programme that avoided 'high damage' compared to a policy that avoided 'low damage' under the status quo – irrespective of future recovery level (Macmillan *et al*., 1996). This implies that information on the status quo is highly relevant to individual valuations.

- **Validity**

Validity of benefit estimates is an important consideration for policy-makers. Some methods are implicitly considered more reliable if they rely on 'real' market data. SP methods have come under closest scrutiny in terms of validity as the data is hypothetical but also because a number of comparative studies have indicated that CV estimates of WTP can be up to ten times greater than actual WTP. To help address this issue a range of validation tests have been developed such as comparing how the survey methodology matches up to current best practice; testing the internal validity of responses by statistically relating WTP to socio-economic and attitudinal variables; comparing SP estimates with benefit estimates generated by other studies and/or approaches (e.g. RP methods); and finally by obtaining feedback from survey participants about the valuation exercise and the tasks they were required to complete.

- **Uncertainty and information**
  Considerable uncertainty surrounds both our understanding of ecosystem recovery and the nature of people's preferences for recovery. The degree of uncertainty and error surrounding benefit estimates has also to be made clear to policy-makers in quantitative terms wherever possible.

Chapter 4 Valuing Recovery of the Freshwater Ecosystem

Chapter Summary

- This chapter reports on the discussions of the final panel meeting on the potential for valuing recovery in UK freshwaters. The freshwater ecosystem was selected as the case study because the science of recovery is comparatively better understood than for other ecosystems and there have also been a number of studies which have attempted to value UK freshwaters in the context of air pollution.

- The principal aims of the case study are to (i) assess the current prospects for generating reliable benefit estimates for valuing recovery in the freshwater ecosystem; (ii) identify specific research needs (both economic and scientific) that would allow valuation to take place and (iii) identify limitations to valuing ecosystem recovery.

- The main benefits of freshwater recovery were identified as increased fish catch (salmon and trout), an enhanced recreational experience for walkers and other outdoor enthusiasts, improved potable water quality, and recovery in biodiversity.

- The main challenges associated with valuing freshwater benefits were identified. These included limited information on the link between biology and water chemistry, the long-time scale involved, and problems with estimating and aggregating non-use values. However, these challenges were not considered to be insurmountable given recent developments in economic methods and environmental modelling.

- Four studies were identified from the literature review as having some potential to contribute to a benefit transfer exercise. Problems in methodology or with the scope of the valuation exercise suggested that the benefit estimates generated by these studies would not satisfy current policy requirements. However, in the case of the UK salmon fishery study it would be possible to apply the scientific-economic model used for Galloway if suitable national data could be gathered.

- New valuation research is therefore considered necessary. In prioritising research needs a range of criteria was considered including the effort required and the potential magnitude of the benefits that might be generated by ecosystem recovery.

- The main valuation priority is to estimate the non-use and use values associated with biodiversity recovery using SP approaches. Given the scientific complexities and uncertainties surrounding ecosystem recovery a CV study using deliberative valuation methods, rather than personal interviews is probably most suited to the task. There is also scope for applying CE as a case-study to recreational users. In both cases, considerable effort would have to be invested in preparatory work such as focus groups and scenario portrayal.

- When conducting a WTP study using CV or CE it will be necessary to directly describe further ecosystem deterioration under the status quo. Existing evidence, although sketchy, would suggest that under the *status quo,* species diversity will continue to decline on some affected areas, with local extinction of a range of species, including fish, passerine birds and otters, possible. As this decline in ecosystem quality represents a 'loss' from the current endowment level there is evidence this avoided damage is made explicit to respondents (the 'endowment effect').

- By building on previous work it should be possible to generate reliable estimates of the benefits of recovery in UK salmon fisheries by applying the HP method. Benefits are likely to be large relative to trout fishing, but reasonably small compared to non-use benefits. However, they would be perceived as 'real' economic market benefits as opposed to 'hypothetical' estimates.

- Additional scientific research that would help underpin valuation would include: (i) an enhanced link between chemistry and biology, with the focus on biodiversity and ecosystem function; (ii) more dynamic modelling; and (iii) descriptive measures of ecosystem change that would be easily understood by the general public. One option would be to develop an 'ecological ladder' which would describe recovery in 'steps' defined by easily-recognised indicators such as the presence/absence of keystone species as well as water quality measures.

- Valuation is unlikely to capture all of the benefits of ecosystem recovery as certain aspects of recovery such as 'ecosystem resilience' are difficult to quantify in monetary terms. Monetary estimates should therefore be considered to be a conservative, lower bound estimates of total ecosystem value.

4 Case Study: Valuing Recovery in the Freshwater Ecosystem

The final panel meeting, held at MLURI, Aberdeen, discussed the potential for valuing recovery in the freshwater ecosystem. Freshwater was selected as the case study because the science of recovery is comparatively better understood for freshwaters than for other ecosystems. In addition there have also been several UK freshwater valuation studies in the context of air pollution that may be suitable for benefit transfer.

The panel meeting included both economists and scientists and the principal aims were to:

- review and assess the current potential to value recovery in the freshwater ecosystem following reductions in air pollution
- identify specific research needs (both economic and scientific) that would allow valuation to take place
- identify limitations to valuing ecosystem recovery

The main topics discussed by the expert panel are presented in Appendix 2 and the key outcomes of the discussions are summarised, in annotated form, in Figures 4.1 and 4.2.

4.1 Valuation Framework for Freshwaters

The Valuation Framework presented in Chapter 4 was used as a template for assessing the potential for valuing recovery in freshwaters. The panel considered the three main components of the framework: Scientific Linkages, Benefit Valuation, and Analysis.

4.1.1 Scientific linkages

The primary impact of air pollution on freshwaters has been the acidification of upland catchments as a result of SO_2 and NO_X deposition. Eutrophication, which has affected some terrestrial ecosystems is of less concern as phosphorous, rather than nitrogen, is the limiting nutrient in freshwaters. Table 5.1 describes some of the main effects of air pollution on Freshwaters.

Although scientific research on freshwaters has been underway in the context of air pollution for almost 20 years, there are important gaps in our scientific knowledge with respect to predicting recovery.

4.1.1.1 Emissions –transport – deposition & concentration

- The 5 km^2 grid used to predict deposition for sulphur can be very inaccurate, with some predictions up to 100% higher or lower than recorded deposition. This is mainly due to differences in altitude, land cover and topography. In the case of NH_{3+}, the situation may be worse.

4.1.1.2 Exposure-dose-effect

- The most reliable information is available for fish and diatom populations. Effects on higher order species such as dippers, ospreys and amphibians, which may be of greater concern to the general public are less well understood and modelled. Scientists should address how their research can be portrayed in a way that would complement valuation studies.

- Exceedance maps produced on 10km^2 grid based on deposition models and Critical Loads have been used to predict areas of the UK where Freshwaters are likely to be 'protected' from air pollution. A widely accepted Critical Load is an ANC value of 20 ueq/l which corresponds to an accepted damage threshold for fish populations. However, due to the coarse spatial resolution used, exceedance maps tend to mask variation in deposition rates, ecosystem resilience and resources at risk. Consequently it is difficult to identify specific resources of high value such as salmon spawning grounds, or SSSIs that may have been damaged using this approach.

- The CL approach is not dynamic, hence it does not account for 'damage delay time' or 'recovery time'. In the case of biological recovery, the delay may be considerable if local extinctions have taken place. Threshold effects are also difficult to establish for many biological indicators. For example, many species of invertebrate can survive across a wide range of ANC values.

- The very long time scale for recovery implies that model predictions will have to take account of confounding influences such as changes to the climate and in land use and management. In the case of salmon, high sea fishing and commercial fish farms can have major impact on stocks that may far outweigh the effects of acidification.

- Recovery in some species will be affected by changes in the distribution and abundance of other species, but these interactions are poorly understood and currently cannot be modelled.

- Dynamic models such as MAGIC have been used to predict effects of air pollution policy on freshwater fish populations both in the UK and USA. Their primary advantages over Critical Loads is that they predict recovery over time and can link to changes in the survival probabilities of key fish species such as trout. Box 4.1 provides more information about the MAGIC model and its potential for underpinning economic valuation.

4.1.2 Valuation

Valuation involves identifying how recovery on freshwater ecosystems will impact on people and their welfare, and applying suitable valuation methods to measure these impacts.

4.1.2.1 Welfare impacts

- The decline in salmon and trout catch is one of the more researched impacts of acidification. However, it is not clear how acidification has affected catch, which is one of the more important attributes determining value. Generally there is only anecdotal evidence, although some empirical relationships have been established

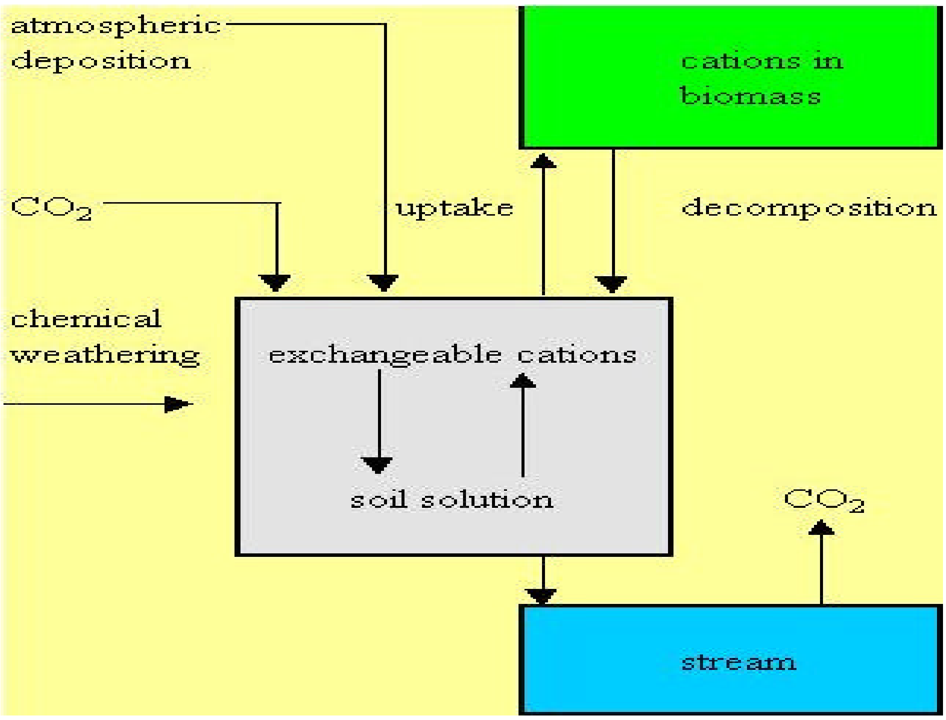
Box 4.1: The MAGIC Model

MAGIC (Model for Acidification of Groundwater In Catchments) is a process-oriented intermediate-complexity dynamic model by which long-term trends in soil and water acidification can be reconstructed and predicted at the catchment scale (Cosby et al., 1985 a,b). **MAGIC** consists of :

1. soil-soil solution equilibria equations in which the chemical composition of soil solution is assumed to be governed by simultaneous reactions involving sulphate adsorption, cation exchange, dissolution and precipitation of aluminium, and dissolution and speciation of inorganic and organic carbon;
2. mass balance equations in which the fluxes of major ions to and from the soil and surface waters are assumed to be governed by atmospheric inputs, mineral weathering, net uptake in biomass, and loss in runoff.

MAGIC produces long-term reconstructions and predictions of soil and streamwater chemistry in response to scenarios of acid deposition and land use. **MAGIC** uses a lumped approach in two ways:

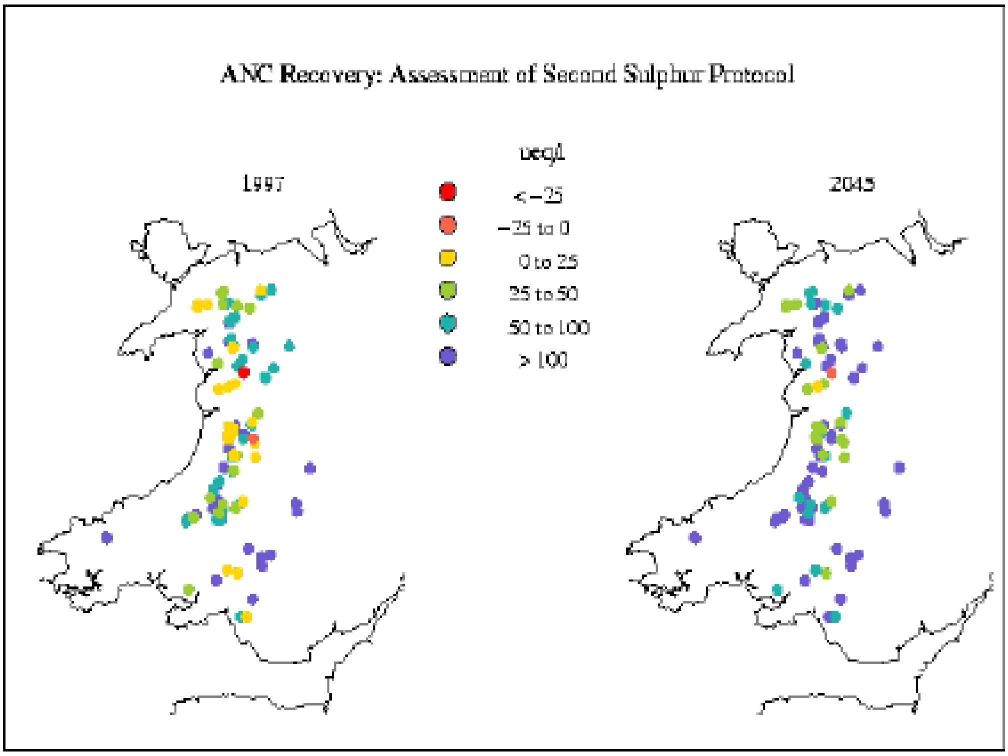
1. a myriad of chemical and biological processes active in catchments are aggregated into a few readily described processes;
2. the spatial heterogeneity of soil properties within the catchment is lumped into one set of soil parameters.



Schematic representation of the flows and stores in MAGIC

Site and Regional Applications
MAGIC has been used on an individual site and regional basis to evaluate the long-term historical and future response of surface water chemistry to changing patterns of anthropogenic deposition. The following example is from a multiple calibration to a population of monitored lake sites throughout Wales. The results indicate that the predicted response within 50 years, relative to present day, is generally a small recovery in surface water Acid Neutralising Capacity (ANC) across the region. However, a clear difference is predicted between recovery at moorland and forested sites; Mean ANC recovery at moorland sites is 21 ueq/l in comparison to 11 ueq/l at forested sites. In addition, all moorland sites are predicted to show an increase in ANC, whereas, despite the emission reductions, 11 of the forested sites are predicted to undergo a further drop in ANC.

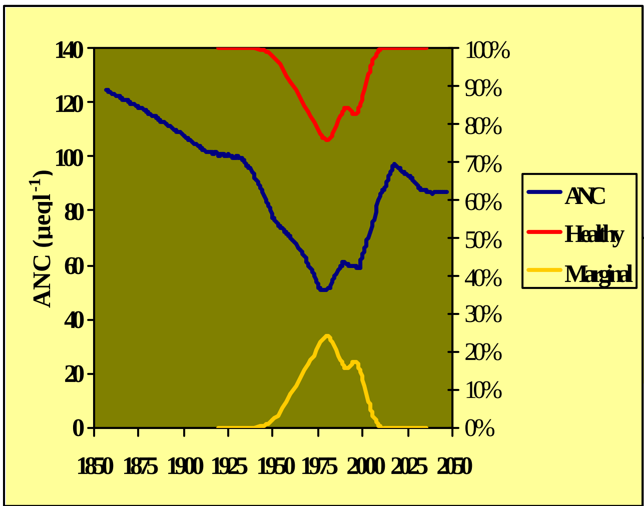
Simulated 1997 ANC and predicted 2045 streamwater ANC for the Welsh

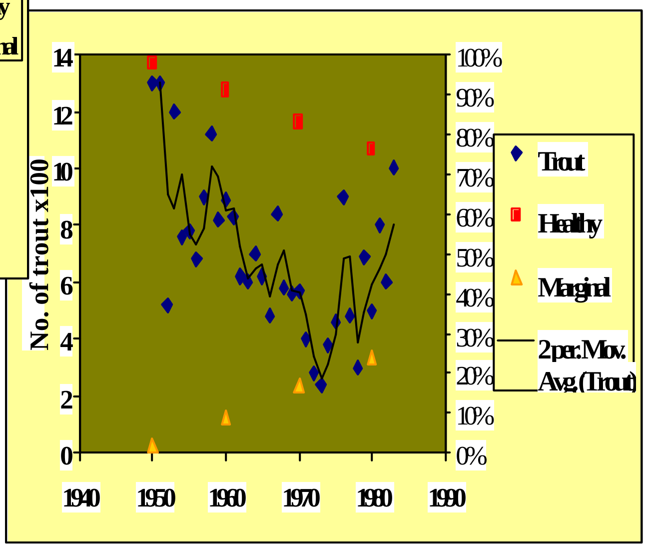


Linking chemistry and biology

The most recent advances with MAGIC have focused on the prediction of biological response following chemical change. Information from the 1000 lakes survey of Norway, has identified chemical indices (in particular ANC) which can be linked to trout population status. This statistical classification of health, marginal, and barren population status has been integrated within the MAGIC model framework and has been used to estimate the economic value of recovery in fish stocks in Galloway (Macmillan and Ferrier, 1994), based on an empirical relationship between fish population status and catch per unit effort

(e.g. Macmillan and Ferrier, 1994). In any case there has also been an increasing trend





toward fishing in stocked lakes and ponds, and it is therefore not clear if freshwater recovery of acidified lakes in fairly remote and inaccessible locations would actually affect welfare.

- Hill-walkers are another potential 'user group' who may benefit from reductions in air pollution. Recovery in certain bird and fish populations could, for example, contribute to the enjoyment of visiting recovering areas. However, the extent to which enjoyment would actually be affected is not clear: the environmental changes that would be initiated by abatement, are likely to be relatively slow and subtle and hence not easily observable, even to frequent and loyal visitors to affected areas. It is likely therefore, that the user benefits associated with recovery are likely to be relatively small.

- Some water catchments supplying potable water to urban areas have been affected by acidification. While low pH by itself does not pose any serious threat to human health, it does enhance the leaching of toxic metals, such as aluminium, manganese and iron, from soils and water pipes. Recovery will help reduce treatment costs associated with these effects. However, it was also noted at the Panel meeting that recovery in freshwaters may have negative impacts as well – most notably the increase in colour caused by higher DOC[8] levels. This may be a cost in welfare terms as treatment may be required to improve the colour of potable supplies, and/or recreationalists prefer clearer waters (to observe aquatic life).

- Acidification has caused pervasive changes in ecosystem function and the biodiversity that it supports. These range from alterations in diatom assemblages to local extinction of individual species such as dippers, trout or frogs. Recovery in these populations may be welcomed by a wide cross-section of the general public because they care about the ecosystem and the biodiversity it supports.

- The panel did not consider recovery would have any discernible benefits in terms of landscape appreciation, or on the non-biological heritage of affected areas. Nor would the production of marketed commodities be enhanced as a result of ecosystem recovery.

4.1.2.2 Valuation Methods

The suitability of the main valuation approaches for valuing ecosystem recovery in freshwaters is summarised below

- **Market Prices**

There would appear to be little potential to apply the Market Price approach as commodity production is not likely to be affected.

- **Revealed WTP**

Potential increases in fish catch and increased enjoyment for walkers as a result of enhanced biodiversity in the uplands suggest that there is potential to apply revealed preference approaches such as the Hedonic Price and Travel Cost models. Macmillan and Ferrier

[8] Dissolved Organic Carbon

(1994) created an Hedonic model to estimate benefits of recovery in the Galloway salmon fishery by linking changes in water chemistry to catch per unit effort, and then to changes in the economic value of the fishery. There would appear to be some potential to extend this approach to the UK as a whole.

With regard to the brown trout fishery in the UK, it could be possible to estimate recreational benefits using TC. Recent innovations could help address problems with substitute sites, but the scope for applying TC may be limited as it is not clear the extent to which relatively small marginal changes in fish population health will affect catch and in turn, angler behaviour. One way to overcome this problem would be to combine Contingent Behaviour questions within a Travel Cost study. *(*For example, by asking: '*if fish stocks increased how many more trips would you take?')*

However, the overall benefits of recovery in terms of fish catch are not likely to be that great as those anglers who consider the size of their catch to be an important reason for fishing are more likely to fish at local ponds and other stocked waters, rather than remote upland locations. Benefits associated with walking and other outdoor pursuits are also anticipated to be small - do we actually expect walkers to take more trips because their chances of seeing a dipper have marginally improved?

RP techniques are also data demanding and there might be problems identifying and sampling a sufficient sample of the relevant user groups. Club membership lists and permits may help here but are unlikely to be comprehensive.

- **Stated WTP**

SP approaches have potential for valuing the benefits of ecosystem recovery provided that the technical concerns elaborated in Chapter 5 are genuinely taken account of in the research design. Both CV and CE approaches could be applied to estimating recovery in biodiversity and other non-use values. User benefits of fishermen and hill-walkers could also be estimated using these approaches.

In the case of user groups, CE may offer interesting insights into which specific attributes of ecosystem change are of interest, such as fish catch or fish size. (During recovery it is expected that average size of fish will decrease, although total catch should increase). Trade-offs between use and non-use benefits and for different recovery scenarios (e.g. partial v pristine) could also be investigated. On the other hand if the priority for policy-makers is to establish a total value for various ecosystem recovery scenarios then CV would be the most suitable technique to use.

Where separate values for different ecosystem functions are desired then CE is clearly a better choice. However, CE can suffer from problems if respondents unintentionally infer other changes from the one intended. CE may have other drawbacks in the context of ecosystem valuation: (i) specifying ecosystem change in terms of only three of four attributes may considerably under-describe the whole picture and hence give a partial valuation; (ii) the selected attributes may not capture the value associated with particular ecological interactions in the ecosystem, some of which may be highly valued and (iii) recovery will tend to affect many ecosystem attributes positively and independent of policy actions, hence there is little scope for realistic trade-offs between attributes. For example, it would be difficult to suggest in one scenario that fish populations double, but their food supply (invertebrate populations) remain unaffected, while in another food supply increases and the fish population stays the same!

One advantage of CV is that is there is no particular requirement to attributise the ecosystem in an arbitrary fashion and this may be particularly important in dealing with holistic changes to biodiversity arising from reductions in air pollution. Trade-off decisions implicit in CE are more suited to recreation and other user benefits, where specific attributes of the recreational experience can be identified as components of overall value.

Considerable attention would have to be given to describing ecosystem recovery in an easy to understand way. Most people have little familiarity with the science of air pollution and its effect on the environment, nor are they likely to comprehend the time-scales and uncertainties involved. Deliberative valuation approaches that involve time for reflection and discussion such as the Market Stall, would appear to offer some significant advantages in this respect over conventional interview surveys.

- **Imputed WTP**

As recovery will affect potable water supplies both negatively (more colouring) and positively (lower liming and filtration treatment) it should be possible to apply the replacement cost approach to estimate the benefits (costs) of recovery. However, this approach has serious methodological limitations and a better approach would be to ask people what they would be willing to pay to improve water quality (possibly as part of a CE that also includes biodiversity non-user benefits).

4.1.3 Scaling-up

Dynamic models for predicting recovery in freshwaters tend to be catchment or regionally specific hence it is difficult to scale-up benefit estimates to the national level. One way around this would be to apply models such as MAGIC to representative areas throughout the UK where the degree of damage from air pollution varies and use the output to 'calibrate' recovery for different Critical Load Exceedances. Although MAGIC has been applied to a number of acidified regions of the UK, further applications would be required to give a more comprehensive picture of recovery in areas such as the northern highlands and the Pennines.

With regard to measuring non-use values using SP approaches, national estimates should be reasonably straightforward to obtain if an appropriate sampling frame is adopted. In addition to normal concerns about representative-ness in terms of income, age, and gender, the sampling frame would have to allow for geographic variability in benefit estimates. For example, several studies have shown that non-use values decline with distance from the described environmental change and it may be necessary to allow for a distance-decay function in the sampling frame. User groups such as walkers and particularly anglers may present further difficulties. For example, it might be difficult to identify anglers who currently fish or might potentially fish in recovering catchments.

4.1.4 Analysis

Several issues were raised during the Panel meeting which are relevant to the analysis.

- Critical Loads clearly play an important role in policy decisions about abatement of air pollution. It would therefore seem appropriate to attempt to generate benefit estimates that relate to alternative target loads for the UK. One way of achieving this would be to derive benefit estimates for each $10km^2$ in the UK. Benefit estimates for alternative abatement scenarios could then be estimated by adding up the value attached to recovery in all the 'protected' squares. (Benefit estimates would of course have to relate to full recovery as assumed by Critical Loads).

- **Policy benefits also influenced by predicted damage under the status quo**
  In order to assess the net impact of reduced air pollution it is necessary to consider the degree and extent of future damage under the *status quo* (i.e. if no further action is taken) as well as future recovery following abatement. Existing evidence, although sketchy, would suggest that under the *status quo,* species diversity will continue to decline on some affected areas, with local extinction of a range of species, including fish, passerine birds and otters, possible. As a further decline in ecosystem quality under the status quo represents a 'loss' from the current endowment level it is possible that WTP for any abatement programme will be considerably enhanced (the 'endowment effect').

- Ecosystem recovery is characterised by uncertainty. Current models have difficulty quantifying this uncertainty yet it cannot be ignored from a policy standpoint. In terms of benefit estimation using SP methods it would be important to incorporate uncertainty in the valuation exercise. This can be done either by presenting respondents with alternative scenarios corresponding to potential recovery outcomes and/or by investigating preferences for risky outcomes (*e.g.* by presenting respondents with probabilities of certain outcomes occurring). Neither approach fully resolves the problem of uncertainty but would give policy-makers an indication of how important uncertainty is.

- Given the very long time scales involved in recovery, the choice of discount rate will clearly have a major impact on the overall benefit estimate.

- Benefit estimates should be calculated by comparing the annualised benefit stream generated by ecosystem recovery (say over the next 50 years) with the benefits (costs) that would accrue under the status quo over the same period.

4.2 Research for valuing recovery in freshwaters

In the afternoon session the Panel considered the need for, and nature of future valuation research for recovery in UK freshwaters. For each of the four main benefit categories (salmon rights, recreation, potable water quality, and biodiversity) the research priority was established by the panel based on the likely magnitude of potential benefits, the transferability of previous studies, and the economic and scientific effort required to implement a valuation study.

Figures 4.1 and 4.2 summarise the findings and provides an overall score reflecting the research priority attached to each benefit category. The rationale for the scoring system is described in Table 4.1.

Table 4.1 Scoring system used in assessing research priority

Criteria	Rationale
1. Magnitude of benefits	Effort should be invested in estimating significant benefit flows arising from air pollution control (low = 1; high =5)
2. Transferability of Previous Studies	If there is scope to transfer estimates or approaches from previous studies then there is less need to commission new research (high transferability = 1; low transferability = 5)
3. Economic Effort	Some benefit flows may be more difficult to estimate reliably than others due to practical problems (e.g. sampling) or theoretical limitations (high effort = 1; low effort =5)
4. Scientific Effort	The scientific base required for valuation may require to be strengthened (high effort = 1; low effort = 5)
5. Research Priority	Overall assessment of priority based on the above criteria. High overall score suggests a higher priority rating.

4.2.1 Salmon fishing rights

4.2.1.1 Magnitude of benefits

The right to fish for salmon is highly valued in many UK rivers, with some prestigious beats bought and sold for sums in excess of six-figures. The total net economic value of salmon fisheries in 1988 was estimated to be £340 million (Radford *et al.*, 1991). but the industry has recently been in decline due to a fall in the number of returning salmon. While acidification has been implicated in this decline in some fisheries (such as Galloway), other factors such as high sea fishing and disease have been identified as primary causes. Further reductions in air pollution could help the salmon fishery to recover, but the magnitude of benefits may be restricted by these confounding factors and because the most prestigious salmon rivers are not seriously affected by air pollution (e.g. the Tweed). The study by

Macmillan and Ferrier (1994) estimated benefits from recovery in the Galloway salmon fishery (one of the most acidified in the UK) in present value terms, to be less than £1million.

4.2.1.2 Transferability of existing studies

Two studies were identified with potential in a benefit transfer exercise.

- **Macmillan and Ferrier 1994.** A Bioeconomic Model for Estimating the Benefits of Acid Rain Abatement to Salmon Fishing: A Case Study in South West Scotland.

  This study generated marginal annual benefit estimates for recovery in the Galloway fishery for the period 1990 to 2040. Changes in water chemistry and fish population health predicted by MAGIC were linked by regression analysis to fish catch per unit effort and economic value using a Hedonic approach. Overall benefit transfer potential is high.

- **Milner and Varallo 1990.** Effects of acidification on fish and fisheries in Wales. This study combined market data from Radford's study with survey information of fish population levels to estimate the damage to Welsh salmon and trout fisheries from acidification. Analysis relied on a simple presence / absence relationship between acidified and non-acidified waters based on various (unspecified) results from angler questionnaires and fish population surveys. The binary damage function is perhaps too *ad hoc* to be used at a national level, and is unsuited to predicting the benefits of reduced air pollution as the model takes no account of either the timing or the extent of recovery. Overall benefit transfer potential is low.

4.2.1.3 Economic Effort

The panel considered the Hedonic model for Galloway to be appropriate in theoretical terms and had the potential to be applied to other areas where MAGIC has been calibrated (e.g North Wales). Economic estimates of salmon beat values are available for the whole of the UK (Radford *et al.*, 1991), but these may require to be updated. However, in order to extrapolate results to other affected areas more case-study information on the link between fish catch and water chemistry would be required (The Galloway estimates relied on a statistical relationship from one loch only).

4.2.1.4 Scientific effort

The main effort involved in generating UK wide estimates would be re-calibrating the MAGIC model for the main fisheries affected. In some cases this has already been done, but there are significant areas where MAGIC has not been applied such as Central Highlands and the Pennines.

4.2.1.5 Research Priority

Although the Hedonic Model could be transferred to other fisheries in the UK the magnitude of benefits that might accrue is expected to be quite low. Hence salmon fisheries is considered to have only a medium priority rating.

4.2.2 Recreation

4.2.2.1 Magnitude of benefits

No UK studies have looked at the impact of air pollution on recreational activities such as non-commercial angling, walking and nature observation, hence it is difficult to assess the potential magnitude of benefits. The Panel believe that tangible increases in the population of rare species such as the dipper and otter would enhance the recreational experience, but were unsure whether such increases would be sufficiently marked to affect enjoyment, particularly given the protracted nature of the re-colonisation process. In the case of trout fishing in the more remote hill lochs and tarns, where an increase in fish catch may take place, the number of anglers benefiting is expected to be low. Evidence from Norway suggests that the value of remote fisheries is much less than better stretches which are fished far more frequently (Navrud, 2001a).

4.2.2.2 Transferability of existing studies

No recreation studies were identified.

4.2.2.3 Economic Effort

Considerable effort would be required to carry out a specific SP or RP study on recreation. Measuring benefits to recreational fisherman using a RP approach would present some methodological challenges as the link to behaviour and changes in water chemistry are not clear. TC type approaches would have to consider the difficult issue of valuing travel time as a cost: for some, the time spent reaching remote fisheries may actually contribute to their overall enjoyment. SP approaches would be a more realistic option. For example, fish catch and the presence of dippers could be included as attributes in a CE. One of the main problems faced by SP and RP approaches is identifying and recruiting a sufficient number of users to areas affected by air pollution. As mentioned earlier most of these areas are relatively remote and recreational activity is less frequent than in other more populated and accessible locations.

4.2.2.4 Scientific effort

In order to investigate WTP for enhanced recreational experience a clear description of how the levels of each attribute varies within each choice set (i.e. how much more likely is the chance of seeing a dipper?). As this kind of information is not known for many species the researcher would therefore have to rely on scientific opinion rather than reliable scientific evidence.

4.2.2.5 Research Priority

Generating benefit estimates for recreation would not be a straightforward task either in terms of sampling or providing relevant scientific information. On the other hand, users represent an important potential group of beneficiaries that cannot be ignored in the valuation effort. As recreationalists are also likely to value non-use aspects of recovery (such as biodiversity conservation) a more cost-effective option might be to include a sub-sample of recreationalists in a wider study involving the general public. Recreation is considered to have a medium priority rating.

4.2.3 Potable water quality

4.2.3.1 Magnitude of benefits

Although air pollution has caused changes in water chemistry in quite large areas of the uplands, there has been little evidence that this has led to any substantial increase in water treatment costs. Liming, which would be the principal remedial treatment is routinely added to water supplies in any case. There have been no UK studies that have looked at WTP for improved potable water quality in the context of air pollution hence the magnitude of the benefits is difficult to determine. However, given the relatively inexpensive nature of treatment the issue is not likely to be relevant to decisions regarding further reductions in air pollution.

4.2.3.2 Transferability of existing studies

No studies were identified.

4.2.3.3 Economic Effort

Two alternative approaches are possible. Treatment costs could be used to impute WTP but this approach has serious methodological weaknesses (see Chapter 3). A CE approach could be used either as part of a wider study but this would require significant effort to identify and sample affected users.

4.2.3.4 Scientific effort

In order to investigate the benefits of improved potable water the location of affected catchments, and the volume and quality of the water yield would have to be estimated. Considerable additional effort would be required to do this and it is not clear whether existing records and/or models could provide the information required.

4.2.3.5 Research Priority

Methodologically both SP and IP approaches would be reasonably straightforward to implement but the absence of any real concern about this issue suggests that this task has a relatively low priority.

4.2.4 Biodiversity

4.2.4.1 Magnitude of benefits

Existing evidence suggest that non-use values are likely to dominate the economic value of ecosystem recovery in the UK. Two previous studies have generated relatively high estimates of mean household WTP (see below).

4.2.4.2 Transferability of existing studies

- **MacMillan, Hanley and Buckland 1996.** A Contingent Valuation Study of Uncertain Environmental Gains.

This study estimated the annual benefits of recovery in the uplands of Scotland (both heathland and freshwater ecosystems) based on a CV study of 1100 Scottish households. A range of alternative damage and recovery scenarios associated with different biodiversity levels (presence and abundance) were described using 'species boxes'. The effect of time to recovery and uncertainty about outcomes were also investigated. Transfer potential is limited by the use of a mailed questionnaire, which is not now recommended, and the sampling frame which was restricted to members of the public in Scotland only. Overall benefit transfer potential is reasonable.

- **ECOTEC 1992. A Cost Benefit Analysis of Reduced Acid Deposition: UK Natural and Semi-Natural Ecosystems**

  This study employed CV and adopted the 'Species box' approach described above to value full recovery in the freshwater ecosystem. WTP was estimated for anglers and other users, as well as for non-users across the UK. However WTP was estimated based on an open-ended question format and alternative recovery and damage scenarios were not investigated (full recovery was assumed throughout the UK). Hence, the results cannot be used to look at marginal increases in abatement. Overall benefit transfer potential is reasonable.

4.2.4.3 Economic Effort

SP approaches are the most suited to estimating the value of recovery in biodiversity. CV and CE have potential and are suited to targeting users and non-users (3/5). However, a large sample size would be required to derive precise estimates for different user groups and to account for geographic variation in WTP. Given the scientific complexities and uncertainties surrounding ecosystem recovery, considerable effort would also have to be invested in focus groups and scenario portrayal. There may also be a role for deliberative valuation fora such as the Market Stall approach, where respondents are given more time and information to make a decision. This type of approach would require more targeted sampling, and would be more expensive per respondent than conventional surveys, but might yield more reliable results.

4.2.4.4 Scientific effort

Scientific information on biodiversity impacts and prospects for recovery would have to presented to respondents. Impacts on some of the more charismatic species are not clearly understood and some effort would be required to establish the best scientific opinion about future recovery of different species.

4.2.4.5 Research Priority

Given the potential magnitude of biodiversity benefits and the relatively moderate effort the valuation exercise would require, this research is assigned the highest priority.

4.3 Recommendations for further research

4.3.1Valuation

- The priority for valuation research is Biodiversity Conservation. Valuation can be achieved using both CV and CE approaches, but the former is more suited due to the complex nature of the good. The potential for applying deliberative valuation approaches such as the Market Stall to help overcome potential problems with the unfamiliar nature of the 'good' should be investigated.

- A CE approach is favoured for recreational anglers and other user groups. Given that the benefits may not be great and the population affected is not clearly established we recommend that a pilot study is initially carried out to identify the magnitude of user benefits. The MAGIC model could be used to establish the link between water chemistry and fish populations for the case study.

- It should be possible to value the benefits to the UK salmon fishery using the existing Hedonic model. However, economic and catch data would have to be updated and new applications of the model to other regions affected by air pollution may be required.

4.3.2 Underpinning Science

- Modelling approaches must be dynamic, i.e. take account of time to recovery

- There is a requirement to identify damaged stock and stock at risk at a higher spatial resolution, particularly where valuable conservation resources are at risk (e.g SSSIs)

- Further bio-geochemical research effort should be directed towards the interaction of deposition and climate (especially Carbon & Nitrogen interactions), and land use and management

- There was a strong requirement to enhance the chemical:biological link. The focus of biological response must be towards issues of biodiversity and ecosystem function. Better relationships between recovery in freshwater chemistry and species that are likely to be of interest to the public and/or are indicator species with respect to ecosystem function need to be developed (e.g. dippers or otters)

- Measures of ecosystem change need to be easily translated to ecosystem change that is easily understood by the general public. For example, by developing an 'ecological ladder' which describes recovery in steps defined by changes in several variables (such as indicator species or water quality standards for drinking). Careful preparatory work in focus groups would be required to identify the most relevant criteria and the most appropriate and meaningful descriptions of these criteria. Box 4.2 provides one example of such a ladder for freshwater ecosystem recovery.

4.4 Transferability to other ecosystems

There was some discussion at the Panel concerning the transferability of the Panel's findings about freshwaters to other affected ecosystems. Some of the main points to emerge were as follows:

4.4.1 Heathland/Grassland

- Reduced air pollution and ecosystem recovery could have a negative effect on grazing due to the positive impact of N on the productivity of vegetation (although overall economic impact small)

- The impact of changes in species composition on the landscape may have to be valued

- No appropriate dynamic models yet available to predict vegetation succession in relation to air pollution. Although some biological effects based models (e.g. CALLUNA) are available, these are based on small-scale experimental work

Box 4.2 Ecosystem Quality Ladder: an example

Quality A: Fully recovered
• Fish population fully recovered and good chance of catching fish • Drinking water quality excellent, with no additional treatment necessary • Otters & dippers have returned permanently • Natural balance in other species achieved

Quality B: Partially Recovered
• Fish present but little chance of catching any • Drinking water quality excellent, with no additional treatment necessary • Otters and dippers only occasionally seen • Some species overabundant, other species absent

Quality C: Current conditions maintained
• Only a few fish survive and no chance of catching any • Drinking water quality moderate, with some additional treatment necessary • Otters and dippers never seen • Many species of insect and plants have died out. Dominated by one or two species adapted to acidic conditions

Quality D: Damage worsens

- Fish no longer survive
- Drinking water quality moderate, with some additional treatment necessary
- Otters and dippers locally extinct
- Many species of insect and plants have died out. Dominated by one or two species adapted to acidic conditions

4.4.2 Forests

- Need to establish a reliable effects based model for UK forests that covers both timber and non-timber impacts and takes account of confounding effects such as disease and climate

- Ozone damage could be a problem but not well modelled at the moment

- Changes in management practice, particularly altered rotation lengths and species choice would have to be taken into account when estimating damages

4.5 Limitations to valuing ecosystem recovery

While new methodological advances in terms of valuation and increased scientific knowledge are improving our capacity to value ecosystem recovery, some fundamental difficulties remain.

First, economics is limited in the sense that ecosystem change only matters when impacts on human welfare can be recognised and measured. Economics therefore depends on, and follows from, scientific advance yet the two disciplines continue to work independently or in ignorance of each other. As Bockstael *et al*., (1995) states: '*ecological models have tended to ignore humans … and economic models have taken stylised versions of 'ecological facts'*

Second, some aspects of ecosystem recovery may be so complex and/or uncertain that valuation scenarios for SP approaches may be over-simplistic or misleading, for example by not mentioning the uncertainties involved.

Third, the impact of pollution on the foundation services of ecosystems may not be fully understood by scientists, let alone valued by economists.

To help overcome these limitations we suggest the following:

- Scientists and economists need to work together more closely in integrated projects. In the past research has been undertaken independently and has meant that important synergies and insights have been overlooked.

- Introduce 'expert opinion' more overtly into the valuation process. For example by using expert witnesses in group-based valuation fora or by providing interpretations of scientific data in the information set given to respondents. For example, Kenyon and Edwards-Jones (1998) found that expert opinion about the nature conservation value of different sites influenced the WTP of the general public for their protection. While this approach could be criticised in the sense that respondents may forsake their own preferences in deference to the experts, it at least reduces the cognitive demands on participants.

- Where valuation is not considered possible alternative decision-making techniques such as Cost-Effectiveness Analysis and Multi-Criteria Analysis that do not require monetary valuation should play a role. Benefits can be measured in quantitative terms such as total area protected from air pollution (e.g. the Target Load approach), in qualitative terms (high, very high etc) or a mixture of both. Some quantitative approaches allow experts to provide 'values' in terms of ecological worth using some form of weighting and scoring. See Macmillan *et al.,* (1998) for an application of an expert scoring system to valuing alternative woodland ecosystem restoration options. However neither of these approaches give guidance whether it is optimal to proceed with abatement or not and hence do not perform the same role in policy appraisal.

- The uncertainties concerning monetary valuation should not be understated and benefit estimates must be considered to be conservative, lower bound estimates of total ecosystem value.

Figure 4.1 Valuation Framework for Freshwater Ecosystems

What are the benefits?	Scientific underpinning	Valuation	Scaling up	Policy Analysis
Commodities: Salmon rights	Science available to link emissions to population status but modelling only available for some case study areas.	RP: Hedonic Model could link catch to value of fisheries. SP: higher catch an attribute of WTP study	Economic model for UK available but recovery of fish stocks difficult to predict nationally. CLs may provide an indication of where damage may be expected but nature of damage and degree not known	Dynamic models can predict recovery over time and could be compatible with CLs. Uncertainty is beginning to be quantified but reliance also on scenario modelling and sensitivity analysis
Recreation: Trout fishing Walking	Link between fish population status and water quality improving but recovery in bird populations and other wildlife poor	RP: Requires good behavioural link for catch and visit to be established SP: use attributes of visit (catch, bird-spotting) as part of a CE to enhanced recreational experience	CLs could be used to identify areas recovering from air pollution but too small scale to identify specific resources affected. May also be difficult identifying population of users.	Behavioural link with users uncertain but dynamic models can predict recovery over time.
Water Quality & Yield	Chemistry well understood and links to water treatment can be established	Imputed WTP – based on treatment cost/ replacement cost for potable supplies.	CLs approach could indicate where affected catchments might be, but further work required to identify water supplies affected throughout the country	Although fairly localised in terms of scope, estimates would be robust estimates
Biodiversity Conservation	Biodiversity impacts pervasive but no models at ecosystem level available. Circumstantial evidence mainly backed up by some scientific monitoring.	SP: non-use values important for many people. Difficult to 'attributise' recovery in CE context, hence CV may be more suitable	CLs give indication of area affected but degree and extent not identified. Sampling would have to consider distance-decay function for UK population	Uncertainty, long tie spans and complexity of environmental changes may test robustness of method. May be scope for more deliberative CV such as 'Market Stall'

Figure 4.2: Priorities for valuing freshwater recovery

Benefits	Scale of benefits 1 (low)… 5 (high)		Transferability of Previous Studies 1 (high) to 5 (low)		Economic effort 1 (high) to 5 (low)		Scientific effort to underpin valuation 1 (high) to 5 (low)		Research Priority 1 (low) to 5 (high)	
Commodities: Salmon rights	Many other factors affecting catch recovery and small number of beneficiaries	**2**	Macmillan & Ferrier Milner & Varrallo	**2**	Could scale-up to Macmillan& Ferrier to UK level to produce fairly robust benefit estimates	**4**	More data to establish reliable link between catch and fish population status in MAGIC. Combine with CLs to obtain national estimates	**3**	Tangible economic benefits possible with modest effort, but magnitude of benefits quite small	**3**
Recreation Trout fishing	Small group of beneficiaries	**3**	None available suitable for UK application		RP possible but weak behavioural link may undermine potential	**2**	More data to establish reliable link between catch and fish population status and participation	**3**	Low potential benefits and considerable effort	**2**
Walking	Larger group of potential beneficiaries but unclear if enjoyment affected	**2**			SP: could include fish catch or bird spotting as attributes in a CE	**4**	More information on wider range of species needed	**4**		**4**
Potable Water Quality	Does not appear to be a significant issue	**1**	None available suitable for UK application		IP methodology has limitations hence an SP approach is preferred	**3**	Would need to identify potable water supplies likely to recover.	**2**	Not a significant issue	**1**
Biodiversity Conservation	Evidence suggest non-use benefits high	**5**	ECOTEC Macmillan et al.	**3**	SP: Both CV and CE could be used although there may be conceptual and methodological problems 'attributising' ecosystem recovery with CE	4	More research required on wider ecological recovery patterns, especially charismatic species	**3**	Large potential benefits with modest economic and scientific effort	**5**

5 General Conclusions and Recommendations

1 Valuing ecosystem recovery in the context of air pollution is a difficult task due to scientific uncertainty about the impact of abatement on ecosystem recovery and the challenges of valuing long term and complex environmental changes.

2 The 1999 IGCB Report noted that it would not be possible to conduct an economic analysis of ecosystem benefits from reductions in air pollution. In our view this is an overly pessimistic assessment, at least for freshwaters, given new developments in the dynamic modelling of 'effects' and advances in economic valuation methodology. We believe that appropriately planned valuation research conducted in collaboration with scientific experts would yield reliable benefit estimates that would be useful for policy purposes.

3 Scientific understanding and predictive modelling is most advanced for freshwater ecosystems, with models such as MAGIC capable of predicting long term ecosystem response. However, our understanding of the effects of air pollution on grassland/heathland and forest ecosystems is currently insufficient to allow us to undertake a valuation exercise.

We recommend that research is undertaken to estimate the national benefits of recovery in the freshwater ecosystem from further reductions in air pollution

4 The main valuation priority is to estimate the use and non-use values associated with biodiversity recovery using SP approaches. CV is most suited to the task of valuing biodiversity changes, with scope for applying CEs to recreational users if attributes such as 'fish catch' can be identified and quantified.

Priorities are as follows:

1. **A large scale (national) CV to estimate non-use values**
2. **A small scale CE to establish the magnitude of user benefits based on several case-studies (but designed to allow scaling up to national level estimates).**
3. **A scaling-up exercise for salmon fisheries recovery based on earlier work by Macmillan and Ferrier (1994) in Galloway.**

5 Estimating ecosystem values is likely to provide a serious challenge to existing survey-based SP methods such as CV and CE due to the complex nature of ecological recovery processes.

We therefore recommend that:

- **Considerable effort is invested in preparatory work such as focus groups and scenario portrayal.**
- **Group-based 'deliberative' valuation methods such as the Market Stall are preferred to interview-based surveys as the former provides participants**

more time to gather and assimilate information and evaluate their willingness to pay.

- **Descriptive devices such as an 'ecological ladder' need to be developed to describe recovery over time in a simpler way, with each 'rung' of the ladder defined in terms of carefully selected ecological criteria such as the presence/absence of key species and the status of water quality measures.**
- **When conducting a WTP study using CV or CE it will be necessary to directly describe further ecosystem deterioration under the status quo. Existing evidence suggests that WTP for an abatement programme will be considerably enhanced if this avoided damage is made explicit to respondents (the 'endowment effect').**

6 In order to enhance the scientific base for valuing freshwaters we require more investment in scientific research to:

- identify the extent of damage to areas of high nature conservation interest
- model time delays and potential hysteresis during the recovery phase
- account for land use practices and other confounding effect such as climate, pests, and diseases.
- enhance the link between chemistry and biology, especially biodiversity and ecosystem function

7 While new methodological advances in terms of valuation and increased scientific knowledge are improving our capacity to value ecosystem recovery, some fundamental difficulties remain. First, economics is limited in the sense that ecosystem change only matters when impacts on human welfare can be recognised and measured scientifically. Second, some aspects of ecosystem recovery may be so complex and/or uncertain that valuation scenarios for SP studies may be too simplistic or even misleading. Third, the impact of pollution on the foundation services of ecosystems may not be fully understood by scientists, let alone valued by economists.

To help overcome these limitations we suggest the following:

- **Scientists and economists need to work together more closely in integrated projects.**
- **Introduce 'expert opinion' more overtly into the valuation process. For example by using expert witnesses in group-based valuation fora or by providing expert 'interpretations' of complex scientific data in the information set given to respondents.**
- **Where valuation is not considered possible alternative decision-making techniques such as Cost-Effectiveness Analysis and Multi-Criteria Analysis that do not require monetary valuation should play a role. (However neither of these approaches give guidance whether it is optimal to proceed with abatement or not and hence do not perform the same role in policy appraisal).**

- **The uncertainties concerning monetary valuation should not be understated and benefit estimates must be considered to be conservative, lower bound estimates of total ecosystem value.**

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Appendix 1: Resumés of Panel Members

ECONOMICS

Dr Douglas Macmillan (University of Aberdeen) has a strategic research interest in the valuation of air pollution impacts on ecosystems. Together with Dr Ferrier, he was involved in a 5-year research programme to value the economic benefits of recovery in semi-natural ecosystems arising from reductions in SO_2 emissions. He has led studies to value eutrophication of freshwaters, woodland ecosystem restoration, biodiversity enrichment, and the conservation of wild geese and has developed innovative approaches to ecosystem valuation such as the CV Market Stall and delphi-based expert techniques. Dr Macmillan has published over 70 academic papers, reports and books.

Relevant Publications

1. Macmillan, D.C., Duff, E.I., and Elston, D., 2000. Estimating non-market benefit environmental costs and benefits of biodiversity projects using CVM. *Environmental and Resource Economics* (in press).

2. Macmillan, D.C., Hanley, N.D. and Buckland, S. 1996. Valuing Biodiversity Losses due to Acid Deposition. *Scottish Journal of Political Economy* 43(5), 519-533.

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the benefits of Environmentally Sensitive Areas in Scotland. *Journal of Agricultural Economics* 49 (1), 1-15.

Professor Nick Hanley (University of Glasgow) is currently working on the new DETR guidelines on stated preference approaches to environmental valuation, and was a member of the expert group for the recent quarrying tax study carried out by London Economics. He also taught on the recent Peebles course on environmental valuation for government economists.

Relevant Publications

1. Nick Hanley, Douglas MacMillan, Robert. E. Wright, Craig Bullock, Ian Simpson, Dave Parsisson and Bob Crabtree "Contingent Valuation versus Choice Experiments: Estimating the Benefits of Environmentally Sensitive Areas in Scotland." Journal of Agricultural Economics, 1-15, January 1988

2. Hanley N, Spash C and Walker L (1995) "Problems in valuing the benefits of biodiversity protection" Environmental and Resource Economics, 5, 249-272.

3. Hanley N, Kirkpatrick H, Oglethorpe D and Simpson I "Paying for public goods from agriculture: an application of the Provider Gets Principle to moorland conservation in Shetland". Land Economics, (February 1998)

4. Hanley N, Wright R and Adamowicz W (1998) "Using choice experiments to value the environment: design issues, current experience and future prospects". Environmental and Resource Economics, 11 (3-4), 413-428.

Mr Guy Garrod (University of Newcastle-upon-Tyne) is one of the leading researchers on non-market valuation. He has worked on a range of projects for the Environment Agency, British Waterways, the Forestry Commission, English Nature, National Park Authorities, the Department of the Environment, the Department of Transport and the Ministry of Agriculture, Fisheries and Food. Many of these studies have been commissioned as part of the project or policy appraisal process and have encompassed the measurement of both the public benefits of policy changes that could effect entire landscapes (e.g. the ESA scheme) and those arising from smaller initiatives that would alter landscape features (e.g. low flows in rivers).

Relevant Publications

1. Garrod, G.D. and Willis, K.G. (1994). Valuing Biodiversity and Nature Conservation at a Local Level. Biodiversity and Conservation 3, 555-565.

2. Garrod, G.D. and Willis, K.G. (1995). Valuing the Benefits of the South Downs Environmentally Sensitive Area. Journal of Agricultural Economics 46, 160-173.

3. Garrod, G.D. and Willis, K.G. (1996). Estimating the Benefits of Environmental Enhancement: A Case Study of the River Darent. Journal of Environmental Planning and Management 39, 189-203.

4. Garrod, G.D. and Willis, K.G. (1997). The Non-Use Benefits of Enhancing Forest Biodiversity: A Contingent Ranking Study. Ecological Economics, 21, 45-61.

Dr Ricardo Scarpa (University of Newcastle-upon-Tyne) has a PhD in economics from the University of Wisconsin - Madison, U.S.A.. Main areas of expertise include econometric analysis of WTP estimates and development of non-market valuation methods.

Relevant Publications

1. Scarpa, R., Chilton S. and Hutchinson G. 2000. Benefits From Forest Recreation: An Empirical Case Study With Flexible Functional Forms For WTP Distributions. *Journal of Forest Economics*. February, 6(1):41-54.

2. Scarpa, R., Chilton S., Hutchinson G. and J. Buongiorno. 2000. Valuing the Recreational Benefits from the Creation of Nature Reserves In Irish Forests. *Ecological Economics*. April, 33(2):237-250.

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4. Romano D., Scarpa, R., Spalatro F., and L. Viganò. 2000. Modeling Determinants of Participation, Number of Trips and Site Choice for Outdoor Recreation in Protected Areas. *Journal of Agricultural Economics*. May, 51(2):224-238.

5. Scarpa, R., J. Buongiorno, J.-S. Hseu, and K. Lee. 2000. Assessing The Non-Timber Value of Forests: A Revealed-Preference, Hedonic Model for Wisconsin Hardwoods. *Journal of Forest Economics*. June, 6(2):83-107.

6. Scarpa, R., W. George Hutchinson, Susan M. Chilton and Joseph Buongiorno. 2000. Importance of Forest Attributes in the Willingness To Pay for Recreation: A Contingent Valuation Study of Irish Forests. *Forest Policy and Economics,* December, 1(3-4): 315-329.

SCIENCE

Dr Bob Ferrier (Macaulay Institute) leads the Catchment Management Research at MLURI. This programme focuses on the hydrological and hydrochemical consequences of environmental change, in particular land use and management. His research interests centre on the potential role of policy and land use change on water resources and the development of decision support systems for sustainable management. He has considerable experience of commissioned research management nationally and internationally.

Relevant Publications

1. Ferrier, R.C. and Edwards, A.C. The sustainability of Scottish waters in the 21st Century. Hydrology and Earth System Sciences, (in press)

2. Ferrier, R.C., Edwards, A.C., Hirst, D., Littlewood, I.G., Watts, C.D. and Morris, R. (2000) Water quality of Scottish rivers: spatial and temporal trends. *Science of the Total Environment* (In press).

3. Ferrier, R.C. (1998) The DYNAMO (Dynamic models to predict and scale up the impact of environmental change on biogeochemical cycling) Project: An Introduction. *Hydrology and Earth Systems Science,* 2, 375-384.

4. Ferrier, R.C., Littlewood, I.G., Hirst, D. and Watts, C.d. (1998) *Review of the Harmonized Monitoring Scheme, Scotland 1974-1994*. Final Report to Scottish Office Environment Protection Unit, 171.

5. Davis, A., Ferrier, R.C., Edwards, A.C., Macmillan, D.C. and Malcolm, A. (1998) *Eutrophication of Loch Davan and remedial management option*. Final Report to SNH (Phase 1), 148.

6. Ferrier, R.C., Malcolm, A., McAlister, E., Edwards, A.C., Morrice, J. and Owen, R. (1996) *Hindcasting of in loch phosphorus concentrations based on land cover classification*. Final Report for SNIFFER, 166.

Professor Alan Jenkins (Institute of Terrestrial Ecology) is head of the Water Quality Division at CEH Wallingford. He has a B.Sc in Physical Geography and a Ph.D which involved modelling sanitary indicator bacteria in reservoir feeder streams. Since then he has developed models to predict surface water acidification and recovery and in this capacity is a member of the National Expert Group on Transboundary Air Pollution. He is currently a Visiting Professor of Hydrochemical Modelling in the Department of Geography at University College London. His research Division at CEH focuses on field experiment, monitoring and modelling of key water pollutants including pesticides, nutrients, acidity and sediments.

Relevant Publications

1. Jenkins, A., Boorman, D.B., Cooper, D.M. and Ferrier, R.C. (1996) Water quality modelling at catchment scale: State of the art and future challenges. In; *Integrated Catchment Management* (Maxwell, T.J. and Byrd. S. (Eds)). MLURI, Aberdeen

2. Jenkins, A., McCartney, M., Sefton, C. and Whitehead, P.G. (1993*) The impacts of climate change on river water quality in the UK*. Report under contract to DOE.

3. Jenkins, A., Ferrier, R.C. and Cosby, B.J. (1997) A dynamic model for assessing the impact of coupled sulphur and nitrogen deposition scenarios on surface water acidification. *Journal of Hydrology* 197, 111 – 127.

4. Jenkins, A. and Gardner, R. (1995) *Land Use, Soil Conservation and Water Resource Management in the Nepal Middle Hills.* Report under contract to ODA.

5. D'Arcy, B., Dils, R., Ellis, J.B., Ferrier, R.C. and Jenkins, A. (Eds) (2000) The Environmental and Economic Impacts of Diffuse Pollution in the UK. Lavenham Press, UK.

Mr Ron Harriman (Freshwater Fisheries Laboratory) leads the Freshwater Environment Team at the Freshwater Fisheries Laboratory, Pitlochry, and is responsible for providing scientific advice to SERAD on the effects of environmental change on the abundance and structure of salmon and trout populations. The research elements of this work range from ecological interactions in upland streams to long-term chemical responses to diffuse sources of pollution in Scottish fresh waters. He is a member of various government advisory groups on acidification and long-term environmental change.

<u>Relevant Publications</u>

1. Harriman,R., and Pugh, K.B., (1994) Water Chemistry. Maitland P.S., Boon P. J and McLuskyD.S. (Eds) *The fresh waters of Scotland: a national resource of international significance. 89-112.* Chichester . Wiley &sons.

2. Harriman, R., Bridcut , E.E., and Anderson, H. (1995). The relationship between salmonid fish densities and critical ANC at exceeded and non-exceeded stream sites in Scotland. *.Water ,Air and Soil Pollution 85: 2455-2460.*

3. Harriman , R. Morrison , B.R.S., Birks , H.J.B., Christie, A.E.G., Collen, P., and Watt, A.W. (1995). Long-term chemical and biological trends in Scottish streams and lochs. *Water ,Air and Soil Pollution.* 85: 701-706.

4. Harriman, R Adamson, E.A., Shelton. R.G.J., and Moffett G. (1997) An assessment of the effectiveness of straw as an algal inhibitor in an upland Scottish loch. *Biocontrol Science and Technology .* 7: 287-296

Dr Robin Pakeman (Macaulay Institute) leads the Plant Ecology and Community Dynamics programme of MLURI. This programme is focussed on understanding the relationships between external drivers, such as grazing, nitrogen deposition and climate change, on species interactions within plant communities. Much of the research in concerned with the development of dynamic models of vegetation change. His own research interests include developing methodologies to restore natural vegetation, understanding species dispersal in fragmented landscapes and using models to predict species spread and community change.

Relevant Publications

1. Hinsley, S.A., Pakeman, R.J., Bellamy, P.E. & Newton, I. (1996) Influences of habitat fragmentation on bird species distributions and regional population sizes. Proceedings of the Royal Society B, 263, 307-313.

2. Pakeman, R.J. & Marrs, R.H. (1996) Modelling the effects of climate change on the growth of bracken (Pteridium aquilinum) in Britain. Journal of Applied Ecology, 33, 561-575.

3. Pakeman, R.J., Hankard, P.K. & Osborn, D. (1998) Plants as biomonitors of atmospheric pollution: their potential for use in pollution regulation. Reviews of Environmental Contamination and Toxicology, 157, 1-23.

4. Pakeman, R.J., Le Duc, M.G. & Marrs, R.H. (2000) Bracken distribution and control methods: their implications for the sustainable management of marginal land in Great Britain. Annals of Botany, 85 Supplement B, 37-46.

Appendix 2

Workshop on the Economic Valuation of Freshwater Recovery from Air Pollution

Joint Meeting of the ‘Science’ and ‘Economics’ Panels for the DETR Project: *‘Valuation of air pollution effects on ecosystems’*

MLURI, June 6th, 2001

Aims of the Workshop

- Review and assess current potential to value recovery in the Freshwater Ecosystem

- Identify specific research needs (economic and scientific)

- Identify limitations to valuing recovery in the Freshwater Ecosystem

Guideline topics for Panel Workshop on Freshwater Ecosystems

Aims of the Meeting

- Outline project and specific aims of this workshop (OH)
- Policy background/drivers (OH)
- Framework (OH)
- Modus operandii

Predicting recovery in freshwaters – the underpinning science

- Go through Freshwaters Table (OH)
- Consider main issues relating to predicting recovery…….

  e.g. take account of confounding factors, time etc

Expected benefits of freshwater recovery

- Put up table of potential benefits from freshwater recovery (OH)
- Are we missing anything?
- What scientific effects are relevant as links to benefit assessment?
  e.g. Relevant to people (charismatic species)
  Relevant to wider ecosystem health – proxy or keystone species
- Can we link to abatement?
  e.g. Can be measured and monitored
  Responsive to air pollution changes

Valuation

- Methodologies to use
- New approaches
- Some of the main issues to be confronted

Scaling-up

- Area of ecosystem affected/likely to recover
- Populations/beneficiaries affected

Analysis

- Policy drivers – can we use Critical Loads in all of this? Links to BAPs and SSSIs
- Discounting over long time periods
- Uncertainty

Review of previous freshwater valuation studies

- Briefly describe the four studies I am aware of (OH)
- Can they be used
- Scope for benefit transfer

Further valuation required

- Where are the gaps
  e.g. Benefits not valued, previous work updated with new methods etc

Need for underpinning scientific research

- New research required or re-packaging old
  e.g. consider time and cost involved
- Population modelling? (e.g. VORTEX, METAPOP)
- Links to policy drivers (BAPs (Species and Habitats), SSSIs etc)

Priorities for valuation research

- Bearing in mind the significance of the gaps identified above, the need for scientific research to support new valuation studies, provide a score from 1 to 5 that indicates your priority for research. % would be potential big benefits, science available, cost not great

Possible limits to ecosystem valuation

- Which benefits can we value satisfactorily and why
- Aspects which we do not know enough about scientifically
- What about non-monetary methods
- Combination of monetary and non-monetary methods

Transferability of approach to other ecosystems

- Anything different for other ecosystems