Valuing the Environmental Benefits of Reduced Acid Deposition in the Semi-Natural Environment

Douglas C. Macmillan

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Department of Economics
Department of Environmental Science

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Declaration

I declare that this thesis has been composed by the candidate and that the work it embodies has been done by the candidate and has not been included in any other thesis. Due acknowledgement is given where appropriate to the contribution of others to the research described in the thesis.

Douglas Craig Macmillan

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ABSTRACT

Acid deposition is a present and future cause of environmental damage in vulnerable areas of Scotland important for nature conservation and salmon fishing. The UK government, in cooperation with other European states, has agreed to substantial reductions in emissions of SO_2, the primary cause of acidification. Although the cost of abatement will be extremely high little effort has been made to value the environmental benefits of ecosystem recovery. This partly reflects the difficulties involved in establishing reliable dose-response functions that can predict long-term ecological change for acidified ecosystems, but also the problem of providing a monetary estimate for biodiversity losses which have no market value.

This study aims to generate reliable estimates of the future economic benefits generated by recovery from acidification in the semi-natural environment of Scotland. The Contingent Valuation Method is applied to value the non-use benefits of abatement under a range of acidification scenarios. Average household willingness to pay (WTP) was £247 and £351 per year when faced with low and high damage , with a present value in excess of £9 and £13 billion respectively. WTP was not influenced by future recovery level or rate or recovery. When faced with risky outcomes respondents were found to be risk averse when both environmental gains and losses are considered.

A hedonic price model, which links market data to changes in water chemistry and fish populations predicted by the MAGIC model, was used to estimate the economic benefits to the rod and line salmon fishery. The present value of the benefits to the Scottish salmon fishery were estimated to be £3.7 million.

Non-use benefits associated with recovery from acidification therefore greatly outweigh user benefits but may be prone to hypothetical bias. Further research into scope effects in CVM studies is required, and the potential to calibrate CVM responses in line with

real economic commitments should be investigated.

PREFACE

The case studies described in this thesis have been published or submitted to, recognised academic journals. Where appropriate they have been fully referenced in the text and are listed below and in the bibliography.

Macmillan, D.C., and Ferrier, R.C. 1994. A bioeconomic model for estimating the benefits of acid rain abatement to salmon fishing: a case study in south-west Scotland. Journal of Environmental Management and Planning 37(2), 131-144

Macmillan, D.C., Hanley, N. and Buckland, S.T. 1996. A contingent valuation study of uncertain environmental gains. Scottish Journal of Political Economy 43 (5), 519-533.

Buckland, S.T., Macmillan, D.C., Duff, E.I. and Hanley, N. 1996. Estimating willingness to pay from dichotomous choice contingent valuation studies. The Statistician (submitted).

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CHAPTER 1 INTRODUCTION AND AIMS

1.1 Introduction

Emissions of sulphur dioxide (SO_2) and other acidifying compounds such as nitrogen oxides (NO_x) from the combustion of fossil fuels, have caused serious damage to the natural and built environment in the industrialised nations of Europe and North America. In urban areas high atmospheric concentrations of sulphate and nitrate particles are known to have been deleterious to human health, plant life and building materials. Dispersal of SO_2 and NO_x in the atmosphere has also been implicated in damage to agricultural and forestry crops, and has resulted in widespread damage to semi-natural ecosystems many hundreds of kilometres from the source of emissions (Harriman *et al.*, 1987; Baker *et al.*, 1986). In Scandinavia, for example, the acidification of lakes and loss of fish populations has been partly blamed on emissions of SO_2 from power stations in the United Kingdom (Wright and Henrickson, 1980).

Although the concentration of SO_2 in urban areas has reduced significantly as a result of changes in industrial processes, and following the introduction of stricter UK pollution control legislation (e.g The Clean Air Acts 1956, 1968; The Control of Pollution Act, 1974) acidification of the natural environment is still occurring. A particular cause for concern is the damage to some of the most natural, and least disturbed ecosystems, which are important for nature conservation. A recent survey of Sites of Special Scientific Interest (SSSI) by English Nature (Rimes, 1992), has revealed that damage from acidification extends to almost one-quarter of the total area classified as an SSSI in the UK (Figure 1.1). The spawning of the Atlantic salmon (*Salmo salar*), an economically important species, and brown trout *(Salmo trutta)* have also been affected. These ecosystems are characterised by slow weathering geology and acidic soils which provide limited capacity for neutralising acid inputs.

The damage caused by acid rain is an example of a negative environmental externality. Externalities arise because the economic cost to the environment (including damage to forests and fisheries) is not accounted for in the production costs of the polluter. Emissions of SO_2 and other acidifying pollutants are therefore oversupplied in comparison to the optimal level that would result from the operation of an efficient market.

Since acidification is a transboundary pollution issue the UK government, together with other European states, have had to act jointly to reduce emissions of SO_2. Under the EC's Large Combustion Plant Directive (88/609/EEC) the UK is committed to reducing emissions of SO_2, which contribute approximately two-thirds of all acidic inputs, by 60% by the year 2003. Further cuts to approximately 80% of 1980 levels are planned under the Second Sulphur Protocol of the United Nation's Economic Commission for Europe (UNECE). The long term aim is to reduce emissions of SO_2 to non-damaging levels

Figure 1.1: SSSI's damaged by acidification in the UK

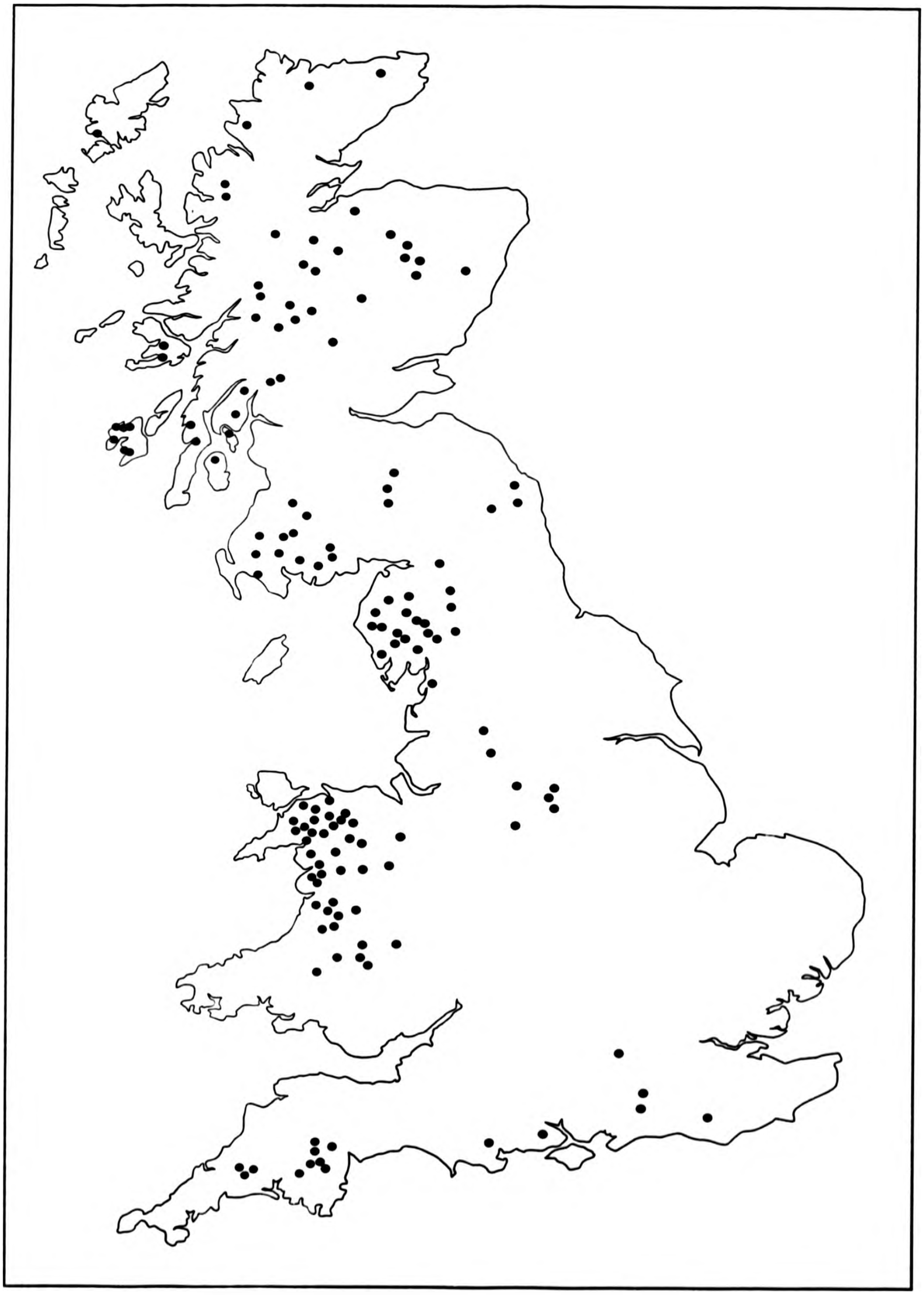
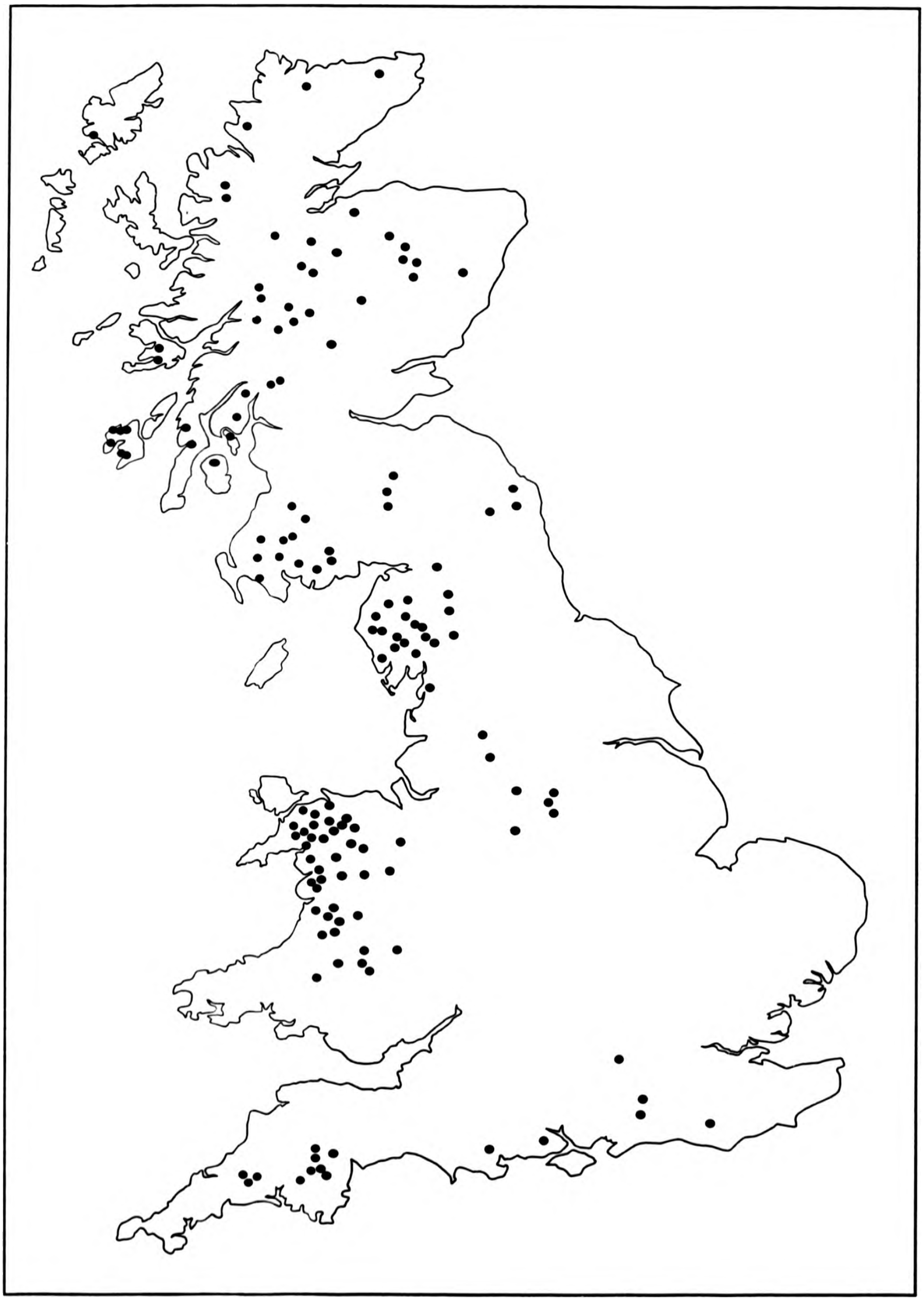


Figure 1.1: SSSI's damaged by acidification in the UK



throughout Europe (Klaassen, 1995).

The cost to the UK economy of meeting the additional cuts in SO_2 emissions under the Second Sulphur Protocol is predicted to be over £600 million per year (Sliggers and Klaassen, 1994)[1]. The benefits of abatement are also expected to be large (ETSU and IER, 1995, ECOTEC 1992) but are difficult to quantify. This partly reflects the difficulties involved in establishing reliable dose-response functions that can incorporate predicted long term ecological change for acidified ecosystems, but also the problem of valuing changes in biodiversity resources which have no market value. Economic analysis of acid rain abatement has therefore not focused on attaining optimal abatement, but rather on abatement strategies which meet abatement targets at least cost either at the national (Shaw and Young, 1986; Rico 1995; London Economics, 1992) or international level (Halkos, 1994).

However, the potential for generating reliable benefit estimates for recovery in the semi-natural environment has increased considerably in recent years. This is, in part, due to the development of sophisticated environmental models which link changes in emission levels to acidification processes in affected areas, but also as a result of the continual refinement of economic methods for valuing non-market environmental benefits. Awareness among policy-makers of the need to value the environmental benefits of SO_2

[1] It is important to note that this and other cost estimates for reductions in SO_2 emissions are typically based on investment in end-of-pipe technology and are therefore likely substantially to overstate actual costs.

abatement has also been growing. The UK government, committed to the long term aim of reducing emissions to non-damaging levels, is keen to justify the vast cost involved (Department of the Environment, 1992b), and the evaluation of environmental benefits in monetary terms is required under the UNECE's Second Sulphur Protocol.

1.2 Research Aim

This study aims to generate reliable estimates of the future economic benefits arising from the elimination of acidification damage in the semi-natural environment of Scotland[2].

Specific objectives are to:

1. identify accurate physical dose-response functions for recovery from acidification in the semi-natural environment.
2. establish linkages between dose-response functions and appropriate economic models in order to estimate the market and non-market benefits of recovery.
3. investigate the influence of uncertainty on individual's preferences for environmental recovery, and provide benefit estimates, together with statistical estimates of reliability, for a range of recovery scenarios.

[2] In this study the semi-natural environment is defined as the area of land that has not been greatly modified by man for urban, agricultural and forestry development.

1.3 Structure of the Thesis

The thesis is divided into eleven chapters. Chapter 2 describes the evolution of abatement policy for SO_2, while Chapter 3 examines the role of economics in abatement policy and the difficulties of developing benefit values for environmental recovery. Prospects for recovery in the semi-natural environment based on the most recent scientific research are outlined in Chapter 4. Chapter 5 describes the economic benefits that can be expected to arise from abatement and reviews appropriate valuation methods. Chapters 6, 7, and 8 describe the design, implementation and results of the Contingent Valuation exercise to value the non-market benefits of recovery. Chapters 9 and 10 describe the development and application of a model which links long term predicted changes in water chemistry and fish populations to the market value of the rod and line salmon fishery. The thesis concludes with a general discussion of the research and its principal conclusions (Chapter 11).

CHAPTER 2 THE EVOLUTION OF ABATEMENT POLICY FOR SULPHUR DIOXIDE POLLUTION

2.1 Introduction

Sulphur dioxide (SO_2), from the combustion of coal and oil, is the principle acidifying pollutant of the semi-natural environment, contributing approximately two-thirds of all acidic inputs[3] (Department of the Environment, 1992a). In this chapter the evolution of UK policy to reduce emissions of SO_2 is described, with special reference to recent international agreements to control acidification in the semi-natural environment.

2.2 Sulphur dioxide as a Health Problem

Prior to the signing of the Large Combustion Plant Directive (LCPD) in the late 1980's, UK legislation had been aimed at reducing air pollution in the major industrial towns and cities. Up until World War II, pollution was largely controlled through the Alkali *Etc*. Works Regulation Act of 1906. Dealing primarily with the control of specific

[3]Minor contributions from nitrogen oxides (NO, NO_2) and nitric acid (HNO_3), formed during the burning of fuel at high temperatures in internal combustion engines and power station boilers also contribute to total acid inputs in remoter areas. In agricultural areas reduced forms of nitrogen such as gaseous ammonia (NH_3) and ammonium (NH_4^+), following biological transformation in the soil or plants, are significant sources of local acidity.

substances and industrial processes which were considered to pose a danger to human health, legislation was ineffective at preventing further deterioration in air quality (Cowling, 1989).

By the 1950's atmospheric pollution from SO_2 and particulates had been linked to detrimental effects on human morbidity and mortality, and to localised damage to plant life (*e.g.* tree growth, loss of moss and lichens), and damage to buildings, including discoloration from layers of thick black soot (Rose, 1990). The spur to government action came with the death of over 4000 Londoners in 1952, due to concentrations of SO_2 many times greater than normal levels. Although brought about by unusually adverse meteorological conditions the incident raised awareness amongst citizens about air pollution and the government was forced to introduce new legislation.

The main feature of the Clean Air Acts of the 1950's was the construction of tall chimneys by large-scale industrial plants. The aim was to disperse pollutants, including SO_2, harmlessly into the atmosphere, where they could be diluted and carried well away from population centres. The Tall Stack approach, as it is known, and other controls enacted through the Control of Pollution Act (1974), and the Alkali Acts, together with the trend away from heavy industry (Barrett *et al.*, 1982), considerably reduced air pollution in urban areas. For example, ground level concentrations of SO_2 decreased dramatically between 1960 and 1979 (Figure 2.1), and there were no repeats of the infamous 1952 incident.

Figure 2.1: Sulphur dioxide concentrations in urban areas (1960 - 1979)

* Adapted from CEE, 1981

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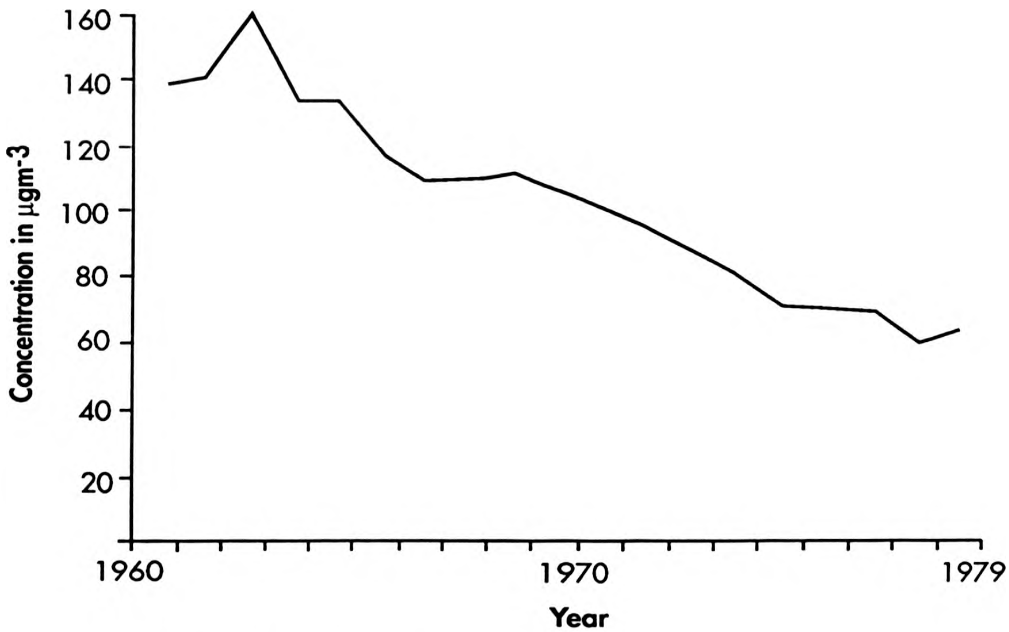


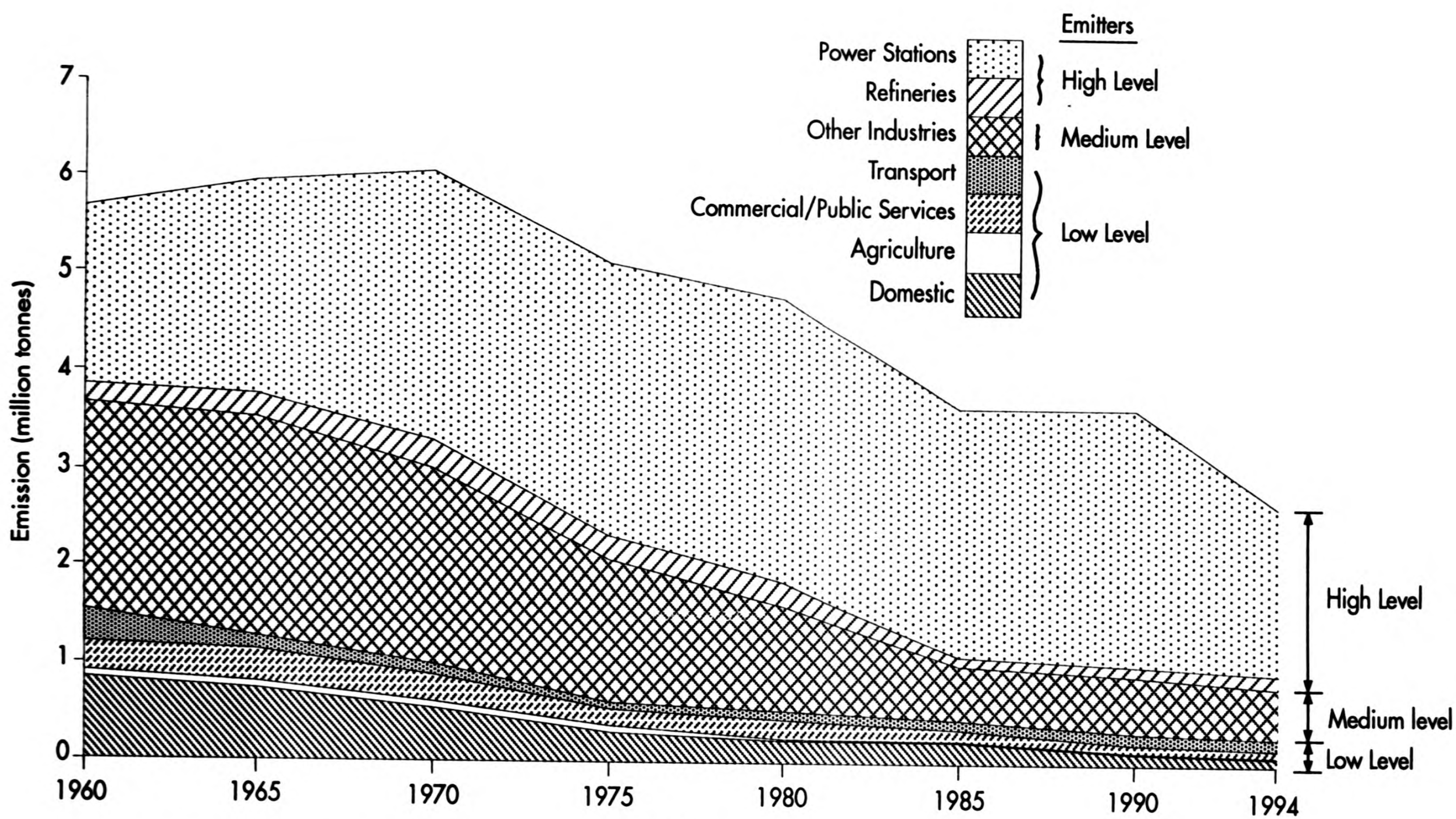
* Adapted from CEE, 1981

However, the Clean Air Acts and subsequent legislation did little to reduce the total level of SO_2 emissions into the atmosphere during this 20 year period. Their major impact was in fact to encourage a switch in emissions from low to high level outlets, such as power stations and oil refineries. Emissions from these sources increased from below 40% in the 1950's to approximately 80% of the total amount of SO_2 in the atmosphere in the late 1980's (Figure 2.2). Although viewed as a success in relation to urban air quality the switch to high level emissions was to have unforeseen consequences for remote, semi-natural environments in the UK and Scandinavia (Rose, 1990).

2.3 Acidification of the Semi-Natural Environment

The discovery that precipitation in large regions of Europe was becoming more acidic followed the establishment in the 1950's of the European Atmospheric Chemistry Network (Likens *et al.*, 1979). However, it was not until 1968 that the possibility of a link between the emissions of SO_2 from industrial areas many miles away and lake acidification in Scandinavia was suggested by the Swedish scientist, Svante Oden (Rose, 1990). This marked the beginnings of a period of intensive research to investigate the causes and consequences of acidification of Scandinavian surface waters and elsewhere.

Figure 2.2: Sulphur dioxide: trends in emissions*



* Sources: CEE 1981, Digest of Environmental Statistics 1996

In 1972 a Norwegian study was initiated which was later to provide substantial evidence that freshwater fish populations, particularly in the south-western region of Norway were being seriously damaged by acidification (Overrein *et al.,* 1980). By the mid-70's it had also been demonstrated that air pollutants could be transported over hundreds and thousands of miles. Growing awareness about this, hitherto little known phenomenon, led to the establishment in 1977 of a European-wide programme known as EMEP, responsible for the monitoring of trans-boundary air pollution across more than 25 countries.

Subsequent monitoring was to reveal a complex relationship between emissions, air circulation, and meteorological conditions which could transport sulphur emissions over hundreds of kilometres and across national boundaries. For example, over one third of the total sulphur deposited in Sweden is estimated to originate from either Germany (23%) or from the UK (14%). The UK, a net exporter of sulphur, contributes to acid deposition in over 30 European countries (EMEP, 1994).

2.4 Early International Agreements

By the late 70's there was a growing awareness among many European countries of the need for international action to control transboundary pollution caused by SO_2. The government of the Federal Republic of Germany (FRG), concerned about the death of trees in central Europe, and under increasing political pressure from 'Die Grunen', and the Scandinavian nations where considerable damage to fish stocks had been observed, led the calls for action (Rose, 1990).

In 1979 the 'Convention on Long Range Transboundary Air Pollution (CLRTAP) was signed by all 35 members of the United Nations Economic Commission for Europe (UNECE). In the face of opposition from the UK, Poland and several other industrial nations, the agreement did not call for any binding commitment to reduce SO_2. Instead the convention requested that countries endeavoured 'to limit and, as far as possible, gradually reduce and prevent air pollution, including long range transboundary air pollution, using the best available technology that is economically feasible'.

Although the UK government signed the agreement it was reluctant to embark on an expensive abatement programme based on insufficient scientific evidence. Despite research on diatom profiles in lake sediments, which suggested that some areas of the UK, notably Galloway and upland Wales had become rapidly acidified during this century, the link with heavy industry and power generation was not universally accepted in government. Changes in land use (afforestation), and management practices (reduction in agricultural liming) were believed by some scientists (and the Department of Energy), to be the most likely cause of freshwater acidification.

For the Scandinavian countries, where acidification of the semi-natural environment was most clearly manifested, and which also received a large proportion of acid inputs from the UK, the vagueness of the 1979 agreement was unsatisfactory. A concrete proposal to reduce sulphur emissions by at least 30% between 1980 and 1993 as a first step in reducing long range pollution was therefore put forward by Norway. In July 1985, with the support of a number of countries including the FRG, Switzerland, and Canada a legally binding protocol which committed participating nations to a 30% cut in SO_2 emissions was signed in Helsinki, Finland (The First UNECE Sulphur Protocol).

The UK, along with other major polluters such as Poland, did not sign the protocol and were excluded from the '30% Club'[4]. With privatisation of the power generating sector on the horizon the government, particularly the Department of Energy, remained sceptical of the need to reduce SO_2 while scientific doubts remained (Rose, 1990). Most importantly the government was also unhappy with the choice of 1980 as the baseline for future emission reduction targets since this would mean that the reductions achieved in the UK prior to this date would be ignored.

However, by the late 1980's the UK was coming under increasing pressure from environmental groups at home, and from foreign governments to act. Much of the pressure came from the Federal Republic of Germany where fitting of flue-gas desulphurisation (FGD) equipment to coal-fired power

[4] The USA also did not sign the agreement, preferring to establish its own legislation on emission levels.

stations had been mandatory (Newbery, 1994). Some of the other members of the '30% Club' including Finland, the Netherlands, and Sweden set out to meet even more ambitious reduction targets. Sweden, for example, declared that their national target for SO_2 was to be an 80% reduction from 1980 levels by 2000 (Acid News (2), 1993).

Around this time the first substantial evidence that acidification was occurring in the upland areas of the UK as a result of industrial emissions was also beginning to emerge from the work carried out by Flower and Batterbee, (1983) using diatom analysis. Trends in the population of acid sensitive diatoms contained in the sediment of upland lakes trends revealed that upland lochs had experienced rapid acidification following the onset of the industrial revolution (Jones *et al.*, 1986).

Other research, carried out under the Surface Water Acidification Programme, indicated that acidification was widespread, affecting areas in Wales, Rannoch Moor (Flower *et al.*, 1986), Galloway (Batterbee *et al.*, 1985), the Trossachs (Harriman and Morrison, 1982), and the Cairngorms (Batterbee, 1989). The Department of the Environment (DoE) became convinced that action was necessary and fought a successful battle within government. In the late 1980's the UK agreed to enter into negotiations with other member states of the European Community to meet substantial, and legally binding, SO_2 reduction targets under the Large Combustion Plant Directive.

2.5 The EU's Large Combustion Plant Directive

The EC Directive (88/609/EEC) on the limitation of pollutants from Large Combustion Plants (LCP) was formally adopted in November 1988[5]. It concentrates on reducing emissions of SO_2 from high level emitters such as power stations and refineries, and covers plants which use over 50MW of energy per year. Controlling emissions of SO_2 to protect remote natural areas, rather than urban areas, represented a marked departure for government intervention to control air pollution.

Under the Directive, emission reductions are to be achieved through:

i) emissions standards for new large combustion plants granted an operating license after July 1987. These standards vary depending on size of plant and type of fuel used (Figures 2.3 and 2.4).

ii) An overall sulphur emission ceiling for each country specified for the years 1993, 1998, and 2000 in terms of future percentage reductions in SO_2 from 1980 levels for each country.

Table 2.1 describes the reductions required of each participating country. Unlike the first sulphur protocol

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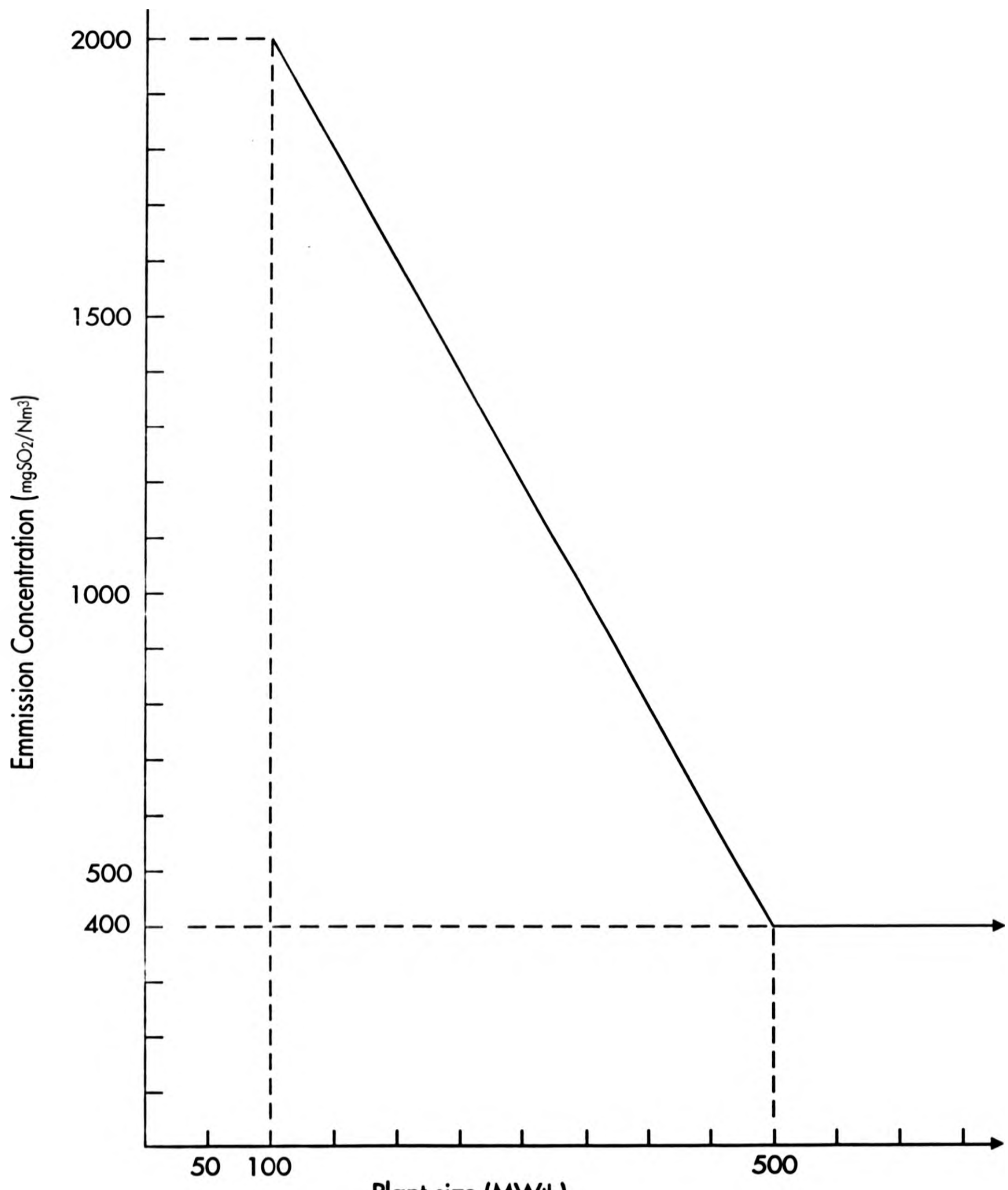
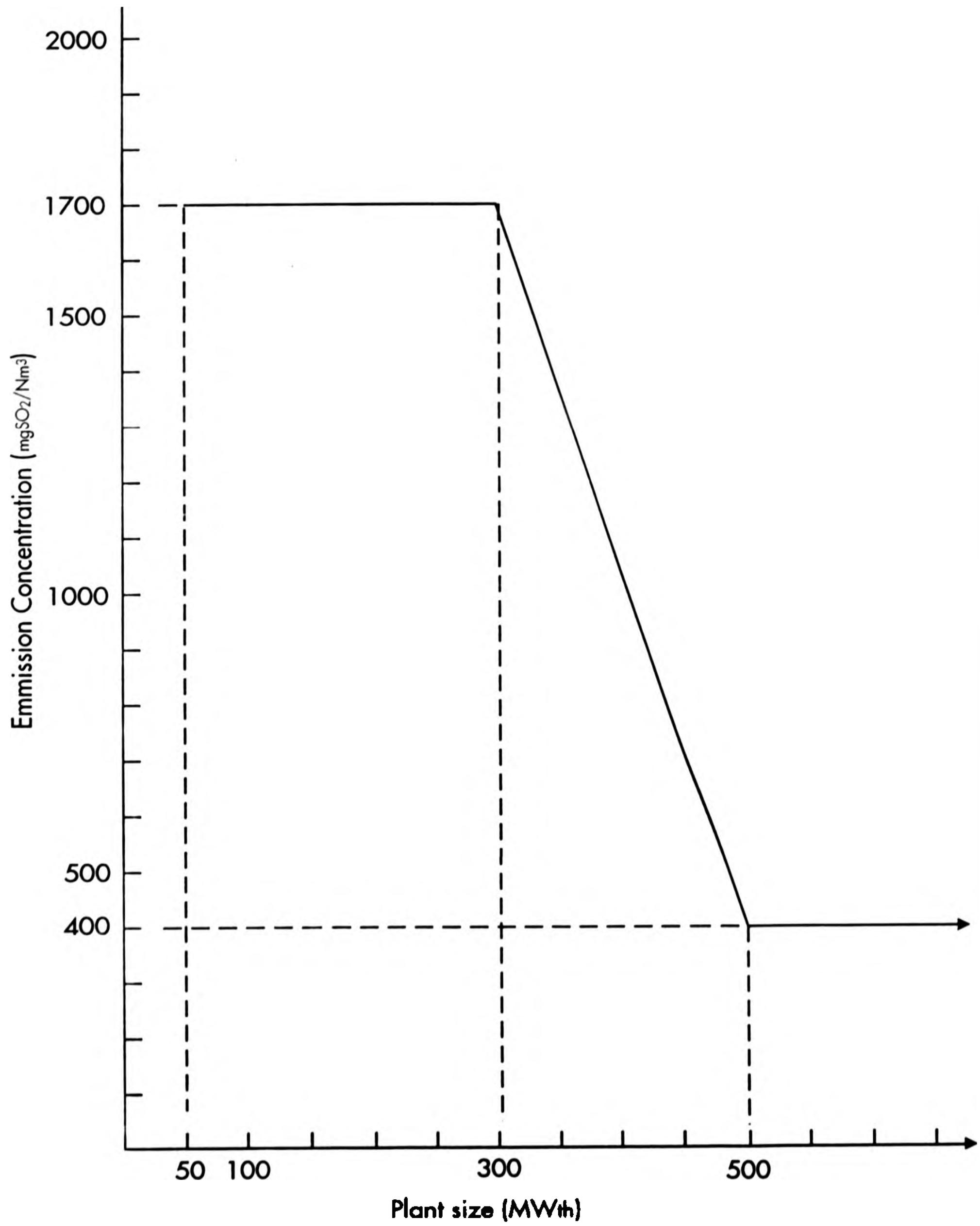
[5]This directive builds on earlier EC legislation (80/779/EEC) which required a binding agreement for explicit standards to be met for ground level concentrations of smoke and sulphur dioxide in the atmosphere necessary to protect human health.

Fig 2.3: EC SO_2 emission standards for solid fuels



* Source: Klaassen (1995)

Fig 2.4: EC SO_2 emission standards for liquid fuels



* Source: Klaassen (1995)

Table 2.1 Ceilings and reductions targets for emissions of SO_2 from existing plants under the LCPD*

Member State	SO_2 emissions by large combustion plants 1980 ktonnes	Emission ceiling			% reduction over 1980 emissions		
		Phase 1 1993	Phase 2 1998	Phase 3 2003	Phase 1 1993	Phase 2 1998	Phase 3 2003
Belgium	530	318	212	159	-40	-60	-70
Denmark	323	213	141	106	-34	-56	-67
Germany	2225	1335	890	668	-40	-60	-70
Greece	303	320	320	320	+6	+6	+6
Spain	2290	2290	1730	1440	0	-24	-37
France	1910	1146	764	573	-40	-60	-70
Ireland	99	124	124	124	+25	+25	+25
Italy	2450	1800	1500	900	-27	-39	-63
Luxembourg	3	1800	1500	1500	-40	-50	-60
Netherlands	299	180	120	90	-40	-60	-70
Portugal	115	232	270	206	+102	+135	+79
United Kingdom	3883	3106	2330	1553	-20	-40	-60

***Source**: London Economics (1992)

where a blanket reduction of 30% was required, the ceilings on sulphur emissions imposed on member states varied according to their individual environmental, economic and energy situations. For example, poorer EU nations, such as Greece, Ireland and Portugal, where economic growth was being encouraged, were allowed more emissions than in 1980 (Klaassen, 1995).

Under the directive the UK is required to reduce SO_2 emissions by 20% in 1993, 40% in 1998, and 60% by 2003. Separate emission ceilings have been set for England and Wales and for Scotland, with quotas established for the major sources of SO_2 (*i.e.* power generators, refineries and other industry). In Scotland SO_2 emissions are to decline from 270 kilo-tonnes in 1980 to 89 kilo-tonnes in 2003, equivalent to a reduction of 67% in SO_2 emissions (Table 2.2).

Unlike new plants, the emission limits for existing plants are not specified, nor are the technologies which should be adopted. Instead emission reductions are to be achieved through the application of Best Available Techniques Not Entailing Excessive Cost (BATNEEC)[6]. Compliance with quota is secured by Her Majesty's Inspectors of Pollution (HMIP)[7] through plant authorizations issued and enforced under the UK Environmental Protection Act 1990. Each plant has been allocated a quota of emissions related to the total expected emissions during a full calender year and these

[6] This approach to pollution control is very similar in principle and operation of 'Best Practicable Means' (BPM) traditionally employed by the British government.

[7] Now the Environment Agency

Table 2.2 **SO_2 emissions quotas/limits for existing combustion plants from 1991-2003 in accordance with the LCPD (in kilotonnes)***

	1980	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003
	Emission Quotas													
National Power Quota		1595	1583	1497	1373	1290	1189	1085	982	920	857	793	727	660
PowerGen Quota		1085	1077	1019	969	878	810	739	669	626	583	540	495	450
England & Wales Sub Total	2776	2680	2660	2516	2342	2168	1999	1824	1651	1546	1440	1333	1222	1110
% Reduction from 1980				9					41					60
Power Stations (Scotland)	142	109	106	104	102	102	102	102	99	88	77	68	62	57
% Reduction from 1980				27					30					60
Powerstations in N Ireland	88	92	86	80	75	69	64	58	53	49	46	42	39	35
% Reduction from 1980				9					40					60
United Kingdom Sub Total	3006	2881	2852	2700	2519	1339	2165	1984	1803	1683	1563	1443	1323	1202
Reduction from 1980				10					40					60
Refineries														
England & Wales Quota	218	86	86	86	85	84	84	83	82	81	80	80	79	78
Scotland Quota	50	14	14	14	14	14	14	14	13	13	13	12	12	12
United Kingdom Sub Total	268	100	100	100	99	98	98	97	95	95	93	92	91	90
% Reduction from 1980				63					65					66
Other Industry														
England & Wales Quota	543	273	257	241	233	225	217	209	201	189	177	164	152	140
Scotland Quota	78	39	37	35	34	32	31	30	29	27	25	24	22	20
United Kingdom Sub Total	621	312	294	276	267	258	248	239	230	216	202	188	174	160
& Reduction from 1980				56					63					74
Total	3895	3293	3246	3076	2885	2645	2511	2320	2128	1993	1858	1723	1588	1452
% Reduction from 1980				21					45					63

*Source: London Economics (1992)

quotas will be steadily reduced each year in order to ensure compliance with the LCPD.

2.6 Second UNECE Sulphur Protocol

The UNECE's Convention on Long Range Transboundary Emissions long term aim is to reduce emission levels to non-damaging levels (i.e. below the critical load) through a series of phased reductions (Klaassen, 1995). The EU's LCPD therefore marked the start rather than the end of UK action to control sulphur emissions in Europe. In 1993 the UK entered negotiations with other European countries over a second UNECE sulphur protocol.

Unlike previous international agreements negotiations for the new protocol were based on a scientific-effects approach. The target was to reduce the gap between present levels of SO_2 emissions, and the 5% 'critical load' level of emissions for environmental receptors by at least 60%. A critical load is that level of pollution *'below which significant harmful effects on sensitive elements of the environment do not occur according to present knowledge'* (CLAG, 1994). The 5% Critical Load refers to that level of deposition which will protect 95% of ecosystems within the target area from acidification.

Each country produced Critical Load maps for soils and freshwater based on a mapping procedure involving allocation of 20 by 20 km squares to Critical Load classes.

Coordination of the Critical Load exercise throughout Europe was the responsibility of the UNECE's Coordination Centre for Effects at the National Institute of Public Health and Environmental Protection in the Netherlands. Using the RAINS model (Regional Acidification Information and Simulation) information on the emission, transport and deposition of SO_2 in Europe, could be combined with the Critical Load maps, to predict the cost and level of protection afforded by a range of abatement strategies[8]. The agreed basis for national ceilings under the new protocol was that level of abatement which achieved the Critical Load target at lowest cost across Europe as a whole[9] (Klaassen, 1995).

In June 1994 the Rt. Hon. John Gummer signed the new UNECE protocol in Oslo on behalf of the UK government. The agreement committed the UK to greater reductions in sulphur emissions than were agreed under the LCPD. Again using 1980 levels as the base-line, the UK has to reduce emissions of SO_2 by 80% by 2010, with interim targets of 50% by 2000 and 70% by 2005 (Table 2.3). In addition to national ceilings, the protocol also incorporated emission standards for new and existing plants, and limits on the sulphur content of fuels. The opportunity to trade SO_2 emission quota between countries was also included within the treaty, as long as trading does not jeopardise the achievement of the overall environmental

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[8]UNECE also use two other models: CASM at the Stockholm Environment Institute at York, and ASAM at Imperial College, London.

[9]The GAP60CRP scenario.

Table 2.3 National ceilings for the UNECE Second Sulphur Protocol*

Countries	1980	CRP	Scenario GAP60CRP kton SO_2
	kton SO_2	kton SO_2	change (%)
Albania	101	138	+31
Austria	390	78	-80
Belarus	740	456	-38
Belgium	828	430	-77
Bulgaria	2050	520	-74
Former CSFR Czech Republic	2257		-72
Slovakia	843		-72
Denmark	448	176	-87
Estonia, Latvia, Lithuania	621	435	-86
Finland	584	116	-80
France	3348	1210	-80
Germany	7494	990	-90
Greece	400	595	+49
Hungary	1632	1094	-68
Ireland	222	240	-41
Italy	3800	1976	-73
Luxembourg	24	10	-58
Netherlands	466	106	-77
Norway	140	70	-76
Poland	4100	2600	-66
Portugal	266	294	+11
Rep of Moldova	330	231	-30
Romania	1800	2592	-41
Russian Federation[a]	7161	4440	-38
Spain	3319	2143	-55
Sweden	519	100	-83
Switzerland	126	60	-52
Turkey	860	2887	+236
Ukraine	3850	1696	-56
United Kingdom	4898	2552	-79
Former Yugoslavia	1300	1576	-22
Croatia	150		-40
Slovenia	230		-45
Rep of Yugosl	920		-13
EUROPE	58017	31981	-59

[a]European part of the Russion Federation within the EMEP area.

*Source: Klaassen (1995)

objective.

2.7 Concluding Remarks

The past 15 years has witnessed a dramatic shift in UK air pollution policy away from urban problems to the control of long range acid deposition. Bowing to international pressure the UK government has agreed to reduce emissions of SO_2 in order to protect sensitive semi-natural environments both at home and abroad. Underlying the international debate has been the trade-off between the costs to industry and the environmental damage caused by acidic pollution. The next chapter examines this issue from an economic perspective.

CHAPTER 3 ECONOMIC ANALYSIS OF SULPHUR DIOXIDE ABATEMENT

3.1 Introduction

To date negotiations over the control of SO_2 have focused on the trade-off between the pressure exerted through the political process by environmental interests and the financial burden that emission reductions would impose on industry. Economic appraisal of abatement policy can extend beyond the estimation of costs alone if the benefits of lower SO_2 emissions can be monetised and included in a cost-benefit analysis of abatement. However, benefit estimation for recovery in the semi-natural environment is not straightforward. In this chapter the economics of pollution abatement are examined, and the difficulties surrounding benefit estimation highlighted.

3.2 Market Failure: Acidification as an Externality

Under a successful market system, optimal allocations of resources follow from the decisions of both consumers and producers intent on maximising their net private benefits. Pareto optimality in the economy occurs when a reallocation of resources which makes someone better off cannot take place without making someone else worse off.

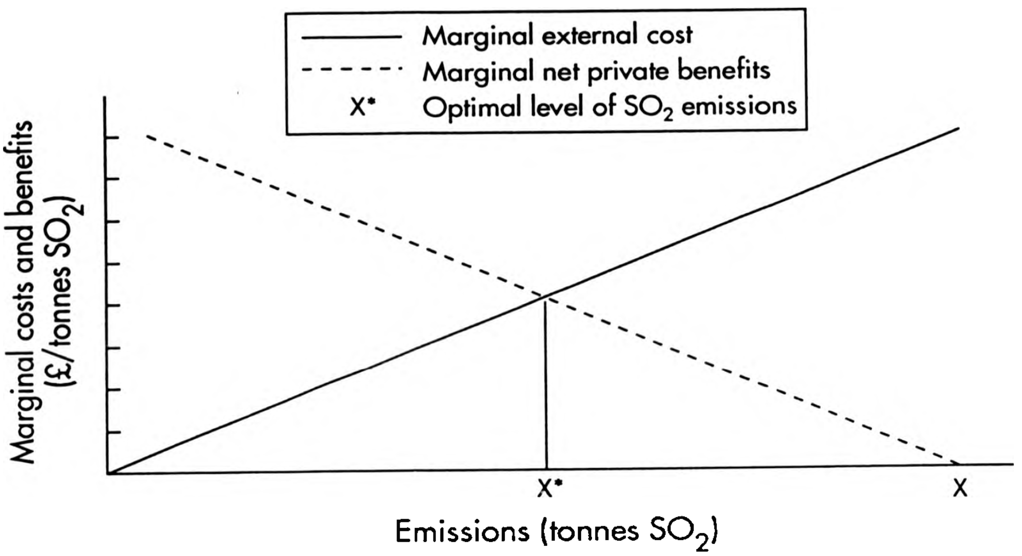
In the context of pollution arising from acidification,

SO_2 emissions will be optimal at the point where the marginal net private benefits of the polluter per tonne of SO_2 emitted, equal the external costs of pollution (x^* in Figure 3.1). It is at this point that the net benefits of SO_2 emissions are maximised (Pearce and Turner, 1990). The social costs relate to the damage done to all environmental resources harmed by acid deposition.

The market must satisfy four main conditions to reach a Pareto optimal allocation: (1) Consumers and producers act competitively by maximising benefits and minimising costs; (2) market prices are known by all consumers and firms; (3) transaction costs are zero so that charging prices does not consume resources (Hanley *et al.*, 1996) and finally, of most significance to acidification, (4) a complete set of markets with well-defined property rights should exist to allow buyers and sellers to exchange assets freely. Tietenberg (1992) describes four general characteristics of a well defined property rights system.

1) Universality. All resources are privately or communally owned, and all entitlements are completely specified and enforceable.
2) Exclusivity. All benefits and costs accrued as a result of owning and using the resource should accrue to the owners, and only to the owner, either directly or indirectly by sale or rental to others.
3) Transferability. All rights should be transferable from one owner to another in a voluntary exchange.

Figure 3.1: Optimal emissions of SO_2



4) Enforceability. Rights should be secure from involuntary seizure or encroachment by others.

Unfortunately non-optimal pollution arises because many environmental resources, such as air and water, are not protected by well-defined property rights. As a result economic agents harmed by acid deposition cannot receive payment for the damage incurred by the pollution. When the consumption or production of an individual or firm shifts another agent's utility or production function, a Pareto-relevant externality exists[10].

As the external costs to agriculture, forestry, and fisheries arising from acidification are not paid for by the polluters, industry has no incentive to reduce SO_2 emissions. Industry will therefore produce SO_2 up until the point where marginal net private benefits of emissions are zero (point x, in Figure 3.1). The divergence between social costs and private costs therefore results in the emission of pollutants in excess of the socially optimal level (Pearce and Turner, 1990).

Coase (1960) showed, in the absence of transaction costs, that market failure associated with externalities can be overcome if well defined property rights can be assigned to the affected non-market assets. If property rights are endowed with the parties harmed by pollution they would then be free to negotiate compensation for the costs incurred.

[10]Since acidification occurs in areas remote from the source of pollution and will cause damage sometime into the future, acid deposition can be more strictly defined as a transferable externality (Hanley *et al.*, 1996).

Pollution output will decline as these costs are internalised in the producer's cost function and a socially optimal level of emissions would be achieved. Conversely, when the property rights are assigned to the polluters, compensation would be paid them by the 'victims' for lowering production of SO_2. The socially optimal level of emissions will be achieved when the marginal costs of pollution damage equals the marginal level of compensation required. If these entitlements are freely exchanged, the role of government is restricted to allocating the initial allocation and subsequent enforcement of property rights (Hanley *et al.*, 1996).

Although some 'victim pays' bargains to control acid deposition have been agreed between Scandinavian countries and the Baltic states mixed strategies involving regulatory control have been preferred. This partly reflects the difficulty associated with establishing Coase-type agreements. For example, even if the government can establish well defined property rights the transaction cost associated with bringing affected parties together to agree compensation could be large. In the case of acid deposition, this would necessitate bargains with a wide range of agents spread over many countries (*e.g.* Scandinavia). Although the failure to bargain may indicate that efficiency is served by avoiding high transaction costs (Pearce and Turner, 1990), there are other problems. In the case of acidification where damage will occur many years into the future, and will affect as yet unborn generations, some relevant transactions are impossible unless current generations bargain on their behalf.

Acidification damage presents a number of other barriers to successful bargains. Since air and water pollution arising from SO_2 have public good characteristics the free-rider problem would make it difficult for the people adversely affected to offer an efficient bribe (Tietenberg, 1992). Negotiations to compensate for lost non-use values associated with damage to biodiversity in sensitive upland ecosystems are hampered by the absence of market values for these assets on which to base compensation[11].

In many instances, it is difficult to attribute damage to resources to the effects of a single pollutant. In the case of forest damage for example, a range of factors including climatic stress, disease, and the action of other pollutants such as ozone, have been implicated (Department of the Environment, 1993). Also, the complexity of pollutant transport in the atmosphere makes it unlikely that damages can be attributable to a particular source. For example, although acid rainfall was identified as the cause of damage to the paint work of several vintage cars, the failure to identify the 'culprit' meant that the motorists were unable to make a claim for the cost of restoring their vehicles (Scotsman, 1996).

[11] Compensation for lost non-use values was at the centre of the controversial debate over the use of the Contingent Valuation Method in the USA. The debate arose following moves by the Alaskan State government to sue the company responsible for the Exxon Valdez oil spill for damage to the natural environment.

3.3 Government Intervention to Control Emissions of Sulphur Dioxide

The presence of negative environmental externalities provides government with a strong incentive to intervene to control pollution. The government can choose to intervene by a command and control approach, or through the introduction of market mechanisms which encourage polluters to reduce their level of emissions by increasing their costs of production. These alternative approaches are reviewed briefly below.

3.3.1 The Command and Control Philosophy

Regulatory instruments, supported by a complex system of monitoring and penalties for non-compliance have formed the traditional approach to environmental protection in most industrialised nations. Product or process standards, or the imposition of emission/discharge standards are the main instruments of the command and control approach.

In some countries, notably the USA and Germany the command and control approach is strictly implemented with ambient pollution standards enforced through standardised end-of-pipe control technology (Teitenberg, 1995). The British approach is considerably more flexible and is based on the principle of Best Practicable Means (BPM), and more recently the closely-related EU concept of BATNEEC (Best Available Technology Not Entailing Excessive Cost). The approach combines an awareness of the need to minimise the cost burden on industry with the need to bring about a gradual lowering of emissions as technology and other circumstances fit (Pearce and Turner, 1990).

A traditional feature of the British system is the opportunity for negotiations with the polluter regarding control options.. This flexibility is regarded as one of the system's strengths and one of its greatest weaknesses. On the one hand, it typically results in a high degree of cooperation and compliance from the polluter, but it is often regarded by environmental groups to be weak and susceptible to manipulation.

Although technology-specific controls are inexpensive to administer and are likely to remove excesses of unregulated activity quickly and with certainty, they are not well suited to meeting national emission targets such as apply in the LCPD. In this situation regulation typically imposes higher costs to industry than intervention through market mechanisms. A least-cost strategy to meet a specific emission target through the command and control approach would require administrators to process vast amounts of data and is typically beyond the capacity of a centralised administration (London Economics, 1992).

3.3.2 Controlling Acid Rain through Market Mechanisms

There are two principal market mechanisms for controlling pollution, both of which have the potential to achieve pollution targets at lower cost than a command and control approach. These involve either charging industry for each tonne of pollution produced (taxation), or creating a market for the right to emit pollution (marketable pollution permits).

3.3.2.1 Taxation

When the marginal cost of abatement varies across individual polluters, government charges (taxation) can achieve a specified level of pollution abatement at lower cost than a traditional regulatory approach (Baumol, 1972). Plants facing a marginal abatement cost per tonne of SO higher than the level of tax, would prefer to pay the tax. Polluters with abatement costs lower than the tax level would choose to invest in abatement technology or to reduce output. Taxation also encourages polluters to search continually for lower cost technologies for reducing pollution in order to reduce overall costs (Shortle and Abler, 1991).

Although penalties exist for excessive pollution under current regulations, they are only used if the polluter fails to take appropriate abatement measures and pollution recurs. Pollution taxation which is directly related to the level of emissions produced by a firm is rarely used. Pearce and Turner, (1990) suggest two main reasons for this. Firstly, taxation of any form is resisted by industry because of concerns that the level of taxation will be determined by government expenditure requirements rather than achieving the stated environmental objective at least cost. Secondly, government and industry may be reluctant to replace the existing legal and institutional framework with an entirely different approach. The introduction of charges would require the government to invest both time and money. Industry may also be unwilling to give up a system which offered opportunities for manipulation to their own advantage (e.g. rent-seeking, technological barriers to new entrants).

3.3.2.2 **Tradeable Permits**

The introduction of a permit market for SO_2 emissions would also meet a specified emissions target at least-cost by exploiting the variable cost of abatement facing polluters (Pearce and Turner, 1990). Plants with high marginal costs will tend to buy permits from operators with lower costs. Since permits can be sold polluters have an incentive to continue to reduce emissions if the cost of abatement is lower than the market value of the permit (dynamic efficiency).

A permit system has several advantages over taxation. Under taxation the government needs to have reliable information on the marginal abatement cost function if the emissions target is to be met. An incorrectly set tax rate, as a consequence of misjudging the MNPB function, will either fail to meet the emission target if set too low, or will impose unnecessary costs on industry if set too high. With a permit system the government can meet the desired standard by issuing the appropriate number of permits (London Economics, 1992). The standard can be varied simply issuing more permits or buying them up. Also, since the price of permits is determined by supply and demand, inflation which would require taxation rates to be adjusted upwards, is automatically taken account of (Pearce and Turner, 1990).

To date the USA is the only country which has introduced tradeable permits on a large scale to control emissions of SO_2. In 1974 the Environment Protection Agency (EPA) introduced a form of internal trading called 'netting' whereby plants could create new sources as long as total emissions did not increase. Other modified forms of trading, such as banking and bubbles, subsequently emerged and brought about substantial cost-savings without any major negative impacts on air quality (Pearce *et al.*, 1989). Most recently an amendment to the Clean Air Act in 1990 allows power generators who succeed in reducing emissions below its allowance, to trade the difference within a national bubble. Rico (1995) estimates that trading will result in savings equivalent to 50% of the cost of meeting the national emission's target under the traditional regulatory approach.

3.3.3 The Spatial Dimension to SO_2 Abatement Policy

Unlike emissions of carbon dioxide the location of SO_2 deposition matters. The rate and extent of acidification at any particular damage site is influenced by local meteorological conditions, soil type, and prevailing wind direction. For each tonne of SO_2 produced, the Kincardine power station in Central Scotland, for example, is estimated to cause three times as much damage to important UK conservation sites as the Drax station in Yorkshire (JNCC,

1996).

When location influences the degree of damage sustained in the environment, emissions trading may not necessarily meet the required pollution objective (Tietenberg, 1992). A more appropriate approach would require deposition permits or charges (Montgomery, 1972). The total number of permits available would be determined by the assimilative capacity of the receiving site. Each plant would then have to buy a sufficient number of permits to cover their own deposition. To bring about the most cost-effective reduction, the system must be able to link emissions to deposition at all receptor sites using transfer coefficients for each emitter/site.

Ambient (deposition) permit systems would involve a complicated set of transactions since each plant would have to acquire separate permits for each site (Pearce and Turner, 1990). A simpler permit system, which incorporates the spatial dimension, is the zonal emission permit system (Tietenberg, 1995). The control area is divided into zones within which emissions of SO_2 correlate reasonably with deposition damage. The most strict design would allow trading only between zones but not between zones.

Although a deposition permit system to control pollution has yet to be implemented, the spatial dimension of SO_2 damage has been recognised in the UNECE's Second Sulphur Protocol. Using sophisticated models of pollutant transport and Critical Load mapping the UNECE have attempted to address the trade off between cost and acid deposition. Variable emission targets for each country were established on the basis of emission reductions which will achieve a 60% reduction in the gap between current emission levels and the Critical Load throughout Europe. Trading between countries using predefined trading ratios which reflect the proportionate damage arising from SO_2 emissions from each country is also permitted.

3.3.4 The UK Approach to SO_2 Abatement

Although national emission ceilings for SO_2 have been determined at international level, the UK retains some flexibility in achieving the necessary reductions. The UK National Plan for SO_2 emissions combines a regulatory approach, with certain features of market mechanisms.

The command and control approach is most strongly applied to new plants where emission limits are specified and must be met under the LCPD and UNECE Protocol. However, these regulations are sensitive to cost considerations as smaller plants have been granted more generous allowances because the unit cost of abatement is higher (see Figure 2.2). Control of emissions from existing plants is exercised through the BATNEEC principle with individual plant standards negotiated with the polluter. The objective is to meet the emissions target without placing excessive costs on industry and the consumer.

Industry can reduce SO_2 emissions through output reductions and through investment in alternative technologies. In the latter case the actual cost of curtailing SO_2 emissions depends on plant size and the technology adopted. There are three main technology options available to emitters: 1) end of pipe measures (e.g. flue gas desulphurisation); 2) technology switching (e.g. fluidised bed combustion, combined cycle gas turbines); and 3) fuel switching (e.g, natural gas or low sulphur coal). For options 1 and 2 most of the cost is associated with new capital investment and running costs, and is not generally influenced by the concentration of SO_2 in the fuel. This cost will vary depending on plant size with abatement costs considerably higher for small plant size. The cost of option 3 depends on the premium on alternative, low sulphur fuels achieve in the market place. Low sulphur coal is slightly more expensive with a delivered price differential of about 20% for a 0.5% difference in sulphur content (London Economics, 1992).

The government, through the EA (SEPA in Scotland), agrees reductions for major sources with industry. Although constrained by the total quota for the industry bubble, significant reductions in emission levels are generally only sought when new plant is installed or existing plant is altered substantially. In an attempt to take account of the spatial dimension, emission quotas are less flexible for plants which cause most environmental damage (Acid News, 1996). Negotiations over individual plant emission standards are also influenced by abatement costs, with polluters allowed to alter the distribution of emissions within an industry bubble, to favour plants where the marginal cost of abatement are lower (London Economics, 1992). For example, PowerGen and National Power can switch power stations, or fuel (coal to gas), or invest in abatement technology in selected plants in order to meet their annual quota. If a station reaches 95% of its allocated quota the operator has to notify the EA and provide an offset if it is to operate beyond its yearly allocation.

The UK approach, which derives largely from the BPM control philosophy, contrasts with a more strict implementation of a command and control approach by the FRG in the 1980's. Here coal-fired power stations, producing over 12GW of power, were forced to close while other stations, producing 38 GW, have been forced to invest over 14 billion Dmarks in FGD technology (Newbery, 1994).

The total cost of implementing the UK national plan under the LCPD has been predicted by ERL (1990) to be £10.1 billion over a 30 year period (at an 8% discount rate). Separate figures are not available for Scotland, but as the marginal cost of reducing SO_2 are higher in Scotland (London Economics, 1992), expenditure is expected to exceed the cost expected on a pro-rata basis (i.e. £0.75 billion). The additional costs of implementing the emission reductions planned for the UK under the Second UNECE Sulphur Protocol have been estimated to be £627 million per year (Sliggers and Klaassen, 1994)[12].

A study by London Economics (1992) suggests that

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[12]Again these estimates are likely to exaggerate implementation costs as they are based on end of pipe technology

substantial savings could be achieved by adopting a more radical emission trading system. Currently trade between bubbles is not possible, yet the marginal costs of abatement vary considerably across the different bubbles[13] (Newbery, 1994). A less restrictive policy which allowed trading within a single bubble for England and Wales was found to result in resource savings of between £30 and 40 million per year (London Economics, 1992).

3.3.5 Optimal Emission Targets

The focus of abatement policy has been the requirement to meet specific reduction targets with a 5 to 15 year period. To date, economic analysis has typically tended to concentrate on identifying least-cost strategies for achieving these targets. Comparatively little research has been devoted to assessing the efficiency of these targets by comparing costs with predicted benefits.

Emission standards are traditionally set with reference to some threshold value, usually related to some aspect of human health, below which there are no observable effects (ApSimon, 1994). In the case of acidification this concept has been extended to the semi-natural environment through the Critical Load approach. Critical Loads represent the level of deposition above which the environment is unable to buffer acidic inputs and suffers damage. The long term aim of European policy to reduce deposition to below the Critical

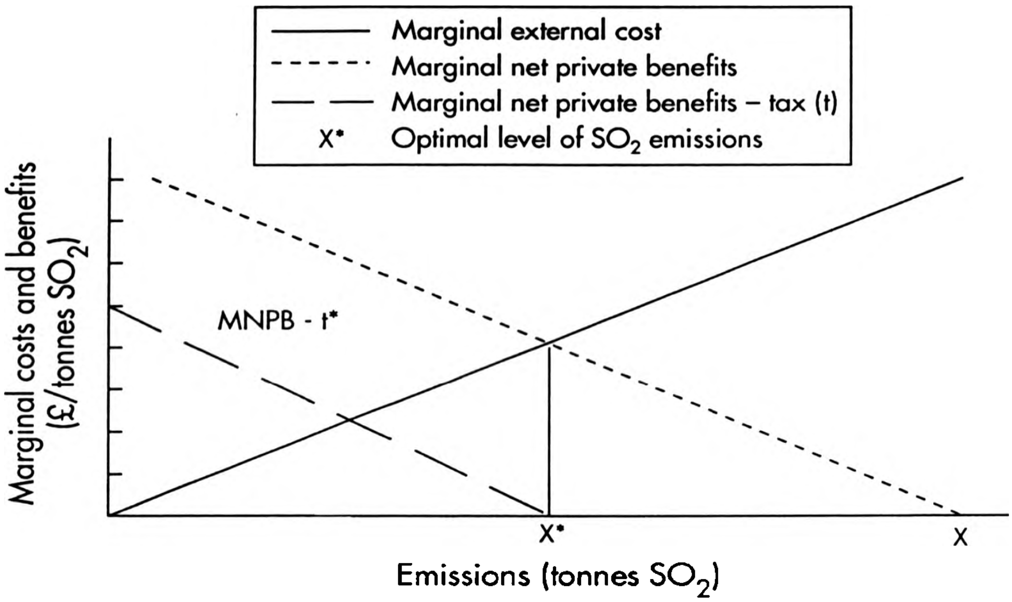
[13]Costs of emission control are minimised when the marginal costs of abatement across polluters is constant.

Load is broadly consistent with a zero pollution policy, and reflects the strong environmental agenda of the current debate surrounding acidification.

In Figure 3.1 it was shown that the optimal level of emissions does not necessarily coincide with the point where pollution is absent (i.e. where the marginal external cost equals zero). From an economist's viewpoint the optimal abatement policy would aim to bring about a reduction in emissions to point x*. If the government is aware of both the marginal cost and benefit functions for SO_2 emissions it could introduce charges, or permits, to achieve an optimal level of emissions.

The principle of charging to achieve optimal pollution was suggested by Pigou (1920). The level of taxation required to bring about optimal emission levels being set equal to marginal external damage costs at the optimum (Figure 3.2). Alternatively permits could be issued or auctioned that would allow the optimal emission target to be achieved. Environmental taxes have recently been introduced in Europe, where they aimed at achieving an efficient reduction in emission levels and at raising revenue to cover costs of pollution control. In Sweden, for example a tax on the sulphur content of fuels (over $4 per kg of sulphur) and has been extremely effective and resulted in Sweden meeting its national target several years ahead of schedule (OECD,

Figure 3.2: Achievement of optimal emissions of SO_2 through taxation



1996).

In setting the level of tax, or regulatory control required, policy-makers need information on the benefits of pollution abatement with the costs to industry. As the marginal cost of control can be expected to rise with increasing levels of abatement, the costs to industry and society in general of a zero pollution goal will be extremely high. Unfortunately estimating the environmental benefits arising from reduced emissions of SO_2 presents economists with a number of difficulties, and early benefit estimates have been considered too unreliable to have any significant influence on abatement decisions, (for example see NAPAP, 1991). In the next section some of the problems of benefit estimation are described with special reference to the semi-natural environment.

3.4 Estimating Benefits of SO_2 Abatement

Since the late 1970's economists have been attempting to place values on the damage caused by air pollution to a variety of important economic resources. These include human health (e.g. Gerking and Stanley, 1986), visibility (e.g. Schulze *et al.*, 1983), agricultural production (e.g Mjelde *et al.* 1984; Adams et al. 1982; Forster 1984), forest growth (e.g. Ewers 1986; Callaway *et al.* 1986), fish stocks (e.g. Mullen and Menz, 1985; Navrud, 1989), and building materials (e.g. Heinz, 1994). For a comprehensive review of these and other studies see Pearce *et al.* (1992).

An overall assessment of the damage costs linked to acidification in the UK by ECOTEC, (1992) is presented in Table 3.1. More recent work in the United States and in Europe has calculated the external cost of SO_2 emissions from oil and coal fuel cycles (ESEERCO, 1995; ETSU and IER, 1995). Table 3.2 lists selected damage costs for SO_2 emissions for the oil sector from the European study.

Comparison of damage estimates for different receptors indicate health effects to be the most significant cost of acid emissions. These costs arise from higher levels of morbidity and mortality, particularly among the elderly and those who suffer from respiratory complaints living in urban areas with low air quality. Damages to materials and the agriculture and forestry sectors are secondary, but still significant. Due to difficulties with valuation no estimates have been provided for biodiversity or semi-natural ecosystems (ETSU and IER, 1995).

Although the calculation of acidification costs has become increasingly sophisticated there remains considerable uncertainty regarding damage estimates. For example, damage costs in the European study for most receptors are associated with only moderate or low certainty (Table 3.2). Forest damage in particular has proven difficult to estimate reliably due to disagreements concerning the effect of SO_2 on forests. For example, European estimates of forest damage are relatively large (ETSU and IER, 1995; ECOTEC, 1992; Nilsson, 1991) whereas the National Acid Precipitation

Table 3.1 Economic damage costs of acidification to different receptors in the UK (ECOTEC, 1992)

Receptor	**Damage Costs(£m pa)**
Buildings	408
Health	2000
Crops	110
Trees	790
Fish	Not available
Vegetation	Not available

Table 3.2 Selected damage estimates for emissions of SO2 from the oil fuel cycle[14]

Damage Category	Valuation estimate (mECU/kWh)	Confid. level
Health:Acute Mortality	9.1	low
Health:Respiratory Infections	0.0042	medium
Health:Restricted Activity Days	0.86	medium
Agriculture:Wheat	0.008-0.032	medium
Agriculture:Barley	0.004-0.017	medium
Agriculture:Soil liming	0.007	medium
Forests:spruce decline[15]	0.007-0.019	low
Materials:Stone Refacing	0.002-.0051	medium
Materials:Galvanised Steel	0.15	medium
Materials:Cultural Impacts	NA	NA

[14]Source: ETSU and IER, 1995. EXTERNE: Externalities of Energy. Vol.4 Oil and Gas. DG XII, European Commission, Luxembourg.

[15]From acid deposition and ozone

Assessment Programme (NAPAP) in the USA concluded that, with the exception of red spruce at high elevations, most forests were not significantly affected by acid rain (NAPAP, 1991).

The need for complex dose response functions, together with the difficultly of measuring non-market effects, particularly non-use benefits, explains why there are few reliable benefit estimates for in the semi-natural environment.

3.4.1 Unreliable Dose Response Models

Although some of the variability in damage estimates can be attributed to the range of economic models and assumptions used[16], one of the major difficulties faced is establishing reliable dose-response functions which link emission levels to changes in environmental quality. Such links involve complex modelling of the physical relationship between the emission, transport, and chemical reactions in the atmosphere of SO_2, and its impact on the environment.

Although good dose-response functions exist for some damage receptors, such as health and buildings (ETSU and IER, 1995), functions for agricultural crops and forests are less reliable. In the case of ecosystems, major problems exist with regard to linking emissions to damage due to the complexity of environmental changes associated with acid deposition.

The construction of a dose-response function for acidification damages involves a number of stages linking

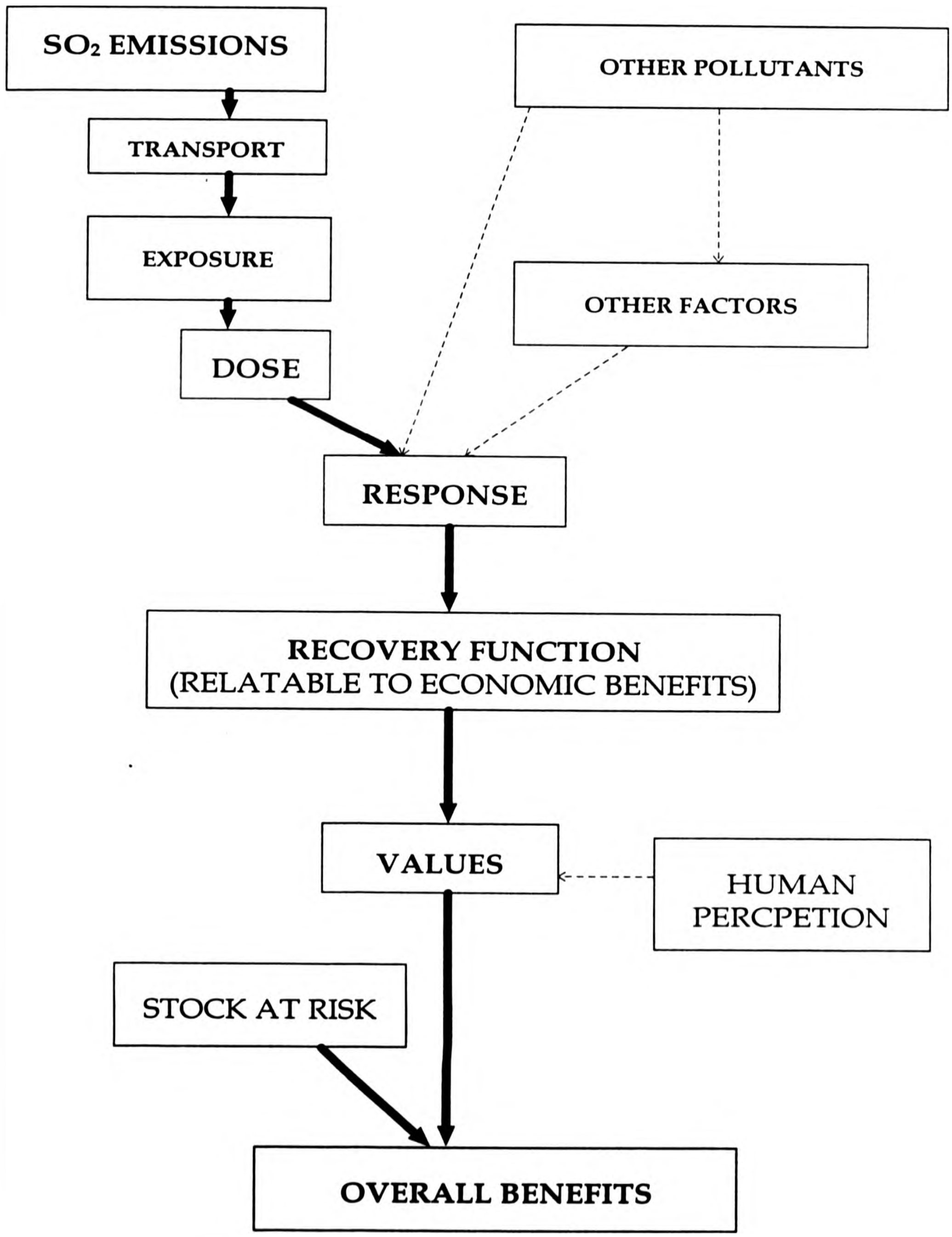
[16] These are discussed in more detail in Chapter 5

emission levels ultimately with an economic measure of damage (Figure 3.3). ApSimon, (1994) reviews the major problems with establishing reliable dose-response functions for acidification. Firstly, the link between emission levels and exposure is often poorly understood and modelled. Many studies proceed beyond this stage by assuming a response to emission levels, based on expert opinion (e.g. Ewers 1986; ECOTEC, 1992; Strand, 1981; Callaway *et al.*, 1986.). Where dose-response functions exist they often relate physical damage to predicted exposure, rather than to the level of dose. Dose differs from exposure because it also takes account of the pre-disposition of the receptor to pollution. For example, plants are more sensitive to exposure when suffering from climatic stress such as drought.

Failure to recognise the role of other stress factors in the physical response to pollution also partly explains why many economic studies which rely on laboratory studies fail to reflect impacts in the real world. Disease, climate and various pollutants are known to act synergistically to influence the response of trees to acidification (Department of the Environment, 1993). Models to predict damage from pollution which do not take account of these other variables will be poorly specified and provide unreliable estimates.

Several characteristics of the acidification process in the semi-natural environment also cause problems for establishing a reliable dose-response function. Firstly, acidification can act primarily as a stock pollutant whereby the assimilative capacity of the environment is gradually

Figure 3.3 A dose response function for SO_2 Abatement



Adapted from Apsimon (1994)

eroded over very long time periods. When the level of acidification reaches some threshold point serious damage (e.g. loss of fish populations) can occur. Dose-response functions must therefore have to model the long term impact of acidification and identify threshold levels. Secondly, damage to freshwater organisms is often associated with peak episodes of acidity (acid flushes) which can wipe out entire populations. Many dose-response models are based on average values of exposure over relatively long time periods and are not sufficiently sensitive to account for catastrophic events.

Even when scientific knowledge and models combine to produce reliable dose-response models, the economist has several other problems to overcome. Often environmental models link air pollution to a physical response which is difficult to relate to environmental changes which are meaningful to economics (OECD, 1989). For example, reduction in the pH value of a lake may not obviously linked to economic welfare unless a link is established with, for example, fish catch. Also, due to heterogeneity of the physical environment dose-response models tend to operate at a micro-scale (e.g. at a sub-catchment level). Economic analysis is typically concerned with the aggregate effect at national or international level, so that problems with scaling up and benefit transfer can emerge (Antle and Capalbo, 1993).

3.4.2 Valuation of Non-Use Benefits

Despite the focus on protecting the semi-natural environment in the current international effort to reduce emissions of SO_2, up until recently there have been no attempts to value the non-use (existence) benefits of recovery. The likely magnitude of the non-use benefits that might arise from environmental recovery from acidification in the semi-natural environment are not known. However, evidence from a study by Navrud, (1989) of the willingness to pay of the general public for recovery in fish stocks, suggest that non-use values are likely to be large. In the case of the Scottish uplands where the main areas affected by acidification are relatively remote, and infrequently used, it is likely that they may also be the dominant component of the general public's TEV.

Non-use value can be broadly defined as a person's willingness to pay to preserve a resource for which he/she has no current or future plans to use (McConnell,1997). Since the services which give rise to non-use value are available to all without the possibility of exclusion, and the enjoyment of one does not impair that of others, they have the characteristics of a public good and are not traded in the market place.

Lacking market prices, or any other behavioural information, benefit valuation is difficult and only currently possible through the application of techniques based on hypothetical markets such as the Contingent Valuation Method (CVM). Although CVM is controversial, for example concerning the treatment of altruism (see section 5.2.3), it is now widely applied, and has been approved for economic appraisal in government in both the US and UK (NOAA, 1993; HM Treasury, 1991).

3.5 Concluding Remarks

Valuing the market and non-market benefits of recovery from acidification in the semi-natural environment is central to any economic analysis of the efficiency of abatement. The difficulties of doing so are enormous and focus on the establishment of reliable dose response functions and the valuation of non-use values. However, advances in economic valuation methods and environmental modelling suggest that the potential now exists for benefit estimation, particularly if inter-disciplinary approaches can be taken.

CHAPTER 4 ECOSYSTEM EFFECTS OF ACID DEPOSITION IN THE SEMI-NATURAL ENVIRONMENT

4.1 Introduction

Rainfall is naturally acidic and the soil and vegetation of some regions of Europe have been slowly acidifying since the last ice age[1]. In the 20th Century, anthropogenic emissions of SO_2 have greatly enhanced this process and led to dramatic changes in the ecology of semi-natural areas.

In this chapter the main effects of acid deposition in semi-natural areas are reviewed. Although considerable uncertainty surrounds the impact of abatement on upland ecosystems, prospects for recovery are assessed. This will form the basis for the development of realistic recovery functions for evaluating the economic benefits of abatment to the semi-natural ecosystem.

4.2 Transport and Deposition of SO_2

Acidifying pollutants are introduced to the semi-natural environment via dry or wet deposition processes.

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[1] In the atmosphere, water (H_2O) in equilibrium with carbon dioxide (CO_2), forms carbonic acid (H_2CO_3), a weak acid solution with a pH of 5.6. The pH of rainfall can be reduced to below 5.0 by contributions of sulphur by natural sources, such as volcanoes and sea spray (UKRGAR, 1983).

Dry deposition refers to the direct uptake of gases and particulates by vegetation and other surfaces. Wet deposition follows when SO_2 and NO_x are oxidised in the atmosphere, and react with water, to form dilute acids such as sulphuric acid (H_2SO_4). These secondary pollutants are then absorbed by clouds before being deposited in rain or snow (acid rain), or captured by trees and other foliage from mist or cloud (occult deposition). Rainfall can be highly acidic under certain meteorological conditions. In Pitlochry, Perthshire for example, a pH of 4.4 has been recorded[2].

Dry deposition rates are linked to the concentration of pollutants in the atmosphere and are therefore highest in areas close to emissions sources. In the semi-natural environment wet deposition of the sulphate anion is the main cause of acidification (UKRGAR, 1983). It is greatest at high altitudes in western regions of the UK where, although the concentration of pollutants is lower than in eastern parts of the country (Figure 4.1a), the total acid load is considerably greater due to higher precipitation rates (Figure 4.1b).

[2] Values of pH range from 0 to 14 and it is the commonest measure of acidity. Defined as the negative logarithm of the H^+ activity (mol l^{-1}) a pH of 4.4 is more than 1000 times more acidic than a pH of 5.6 (Likens *et al.*, 1979).

Figure 4.1a: Acidity of rainfall (PH units)

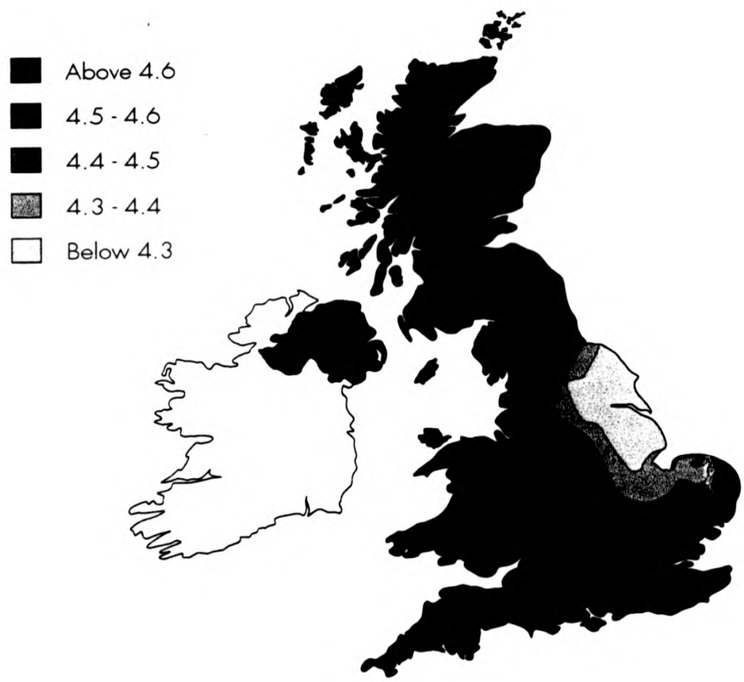
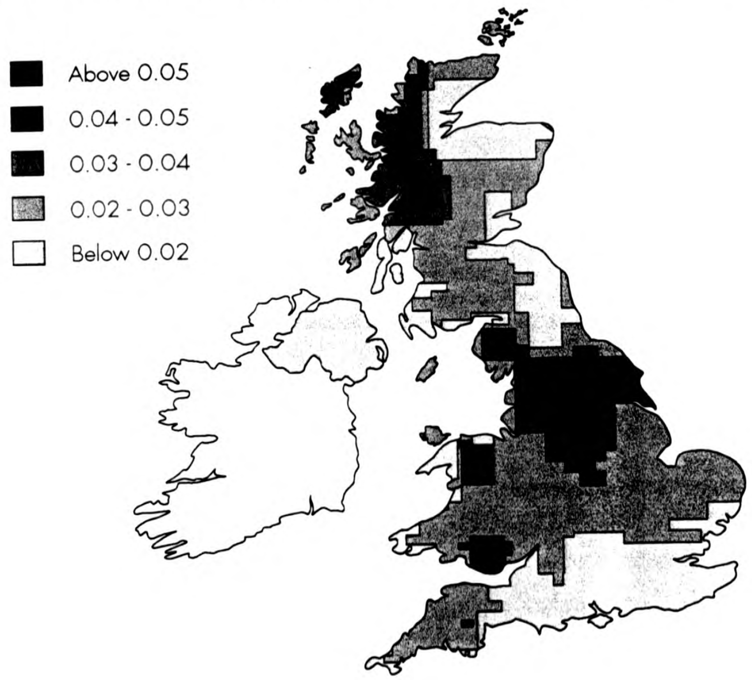


Figure 4.1b: Acidic load (grams of hydrogen ion per square metre)



4.3 Extent of Acidification Damage in the Semi-Natural Environment

The area of acidification damage in the semi-natural environment is of considerable significance to the evaluation of total recovery benefits. Although there has been no attempt to systematically survey the semi-natural environment for acidification it is well known that certain areas have suffered damage as a result of the dispersal and deposition of SO_2 from industrial sources.

Areas that are most susceptible to acidification have only limited capacity for neutralising acid inputs. They are typically located in areas of high rainfall and are characterised by high rainfall, slow-weathering granitic bedrock, and shallow, organic-rich, soils. Recent commercial afforestation has also exacerbated acidification in certain sensitive upland catchments. Fast growing coniferous species such as Sitka spruce (*Picea sitchensis* Bong. Carr) increase the interception of pollutants from the atmosphere and also remove base cations from the soil during growth (Miller *et al.* 1991).

Critical Load Maps perhaps provide the most soundly-based scientific assessment of the extent of acidification damage in Europe. The approach identifies a damage threshold for the response of ecosystems to acidic deposition: the critical load is the 'highest deposition load that will not cause chemical changes leading to long term harmful effects on ecosystem structure and function'

(Sverdup *et al.* 1990).

A number of Critical Load maps for different receptors (*e.g.* buildings, freshwaters and soils) have been commissioned by the UK's Department of the Environment as a contribution to international negotiations about the second UNECE sulphur protocol. The basis for calculating Critical Load values for soils and water were agreed at an international UNECE meeting at Skokloster, Sweden in 1988, and have been calculated using a number of models (UKCLAG, 1991).

A Critical Loads map for soils in Scotland has been produced at the Macaulay Land Use Research Institute (MLURI) for acidity (kmolH^+ $km^{-2}yr^{-1}$) and sulphur ($kgha^{-1}yr^{-1}$). Soils with large stores of leachable, available base cations and weatherable soil minerals have a high Critical Load. Conversely soils with a low Critical Load have relatively small available stores that can be depleted quickly. Two different, but complimentary, approaches have been used to establish critical loads for freshwaters: the diatom model and the steady-state chemistry model (CLAG, 1994)

The most reliable guide to the likely distribution and extent of acidification damage in Scotland can be obtained from soil exceedance maps. These are produced by taking the Critical Load of the dominant soil type within each 20 by 20 km^2 grid square and overlaying the predicted deposition rates. Figure 4.2 illustrates the areas of Scotland where the Critical Load is exceeded under 1988 deposition levels. Figure 4.3 shows the effect of the reductions planned under

Figure 4.2: Area of Scotland where the Soils Critical Load exceeded by greater than 1 keg H^+ ha^{-1} yr^{-1} (under 1988 deposition levels of SO_2)

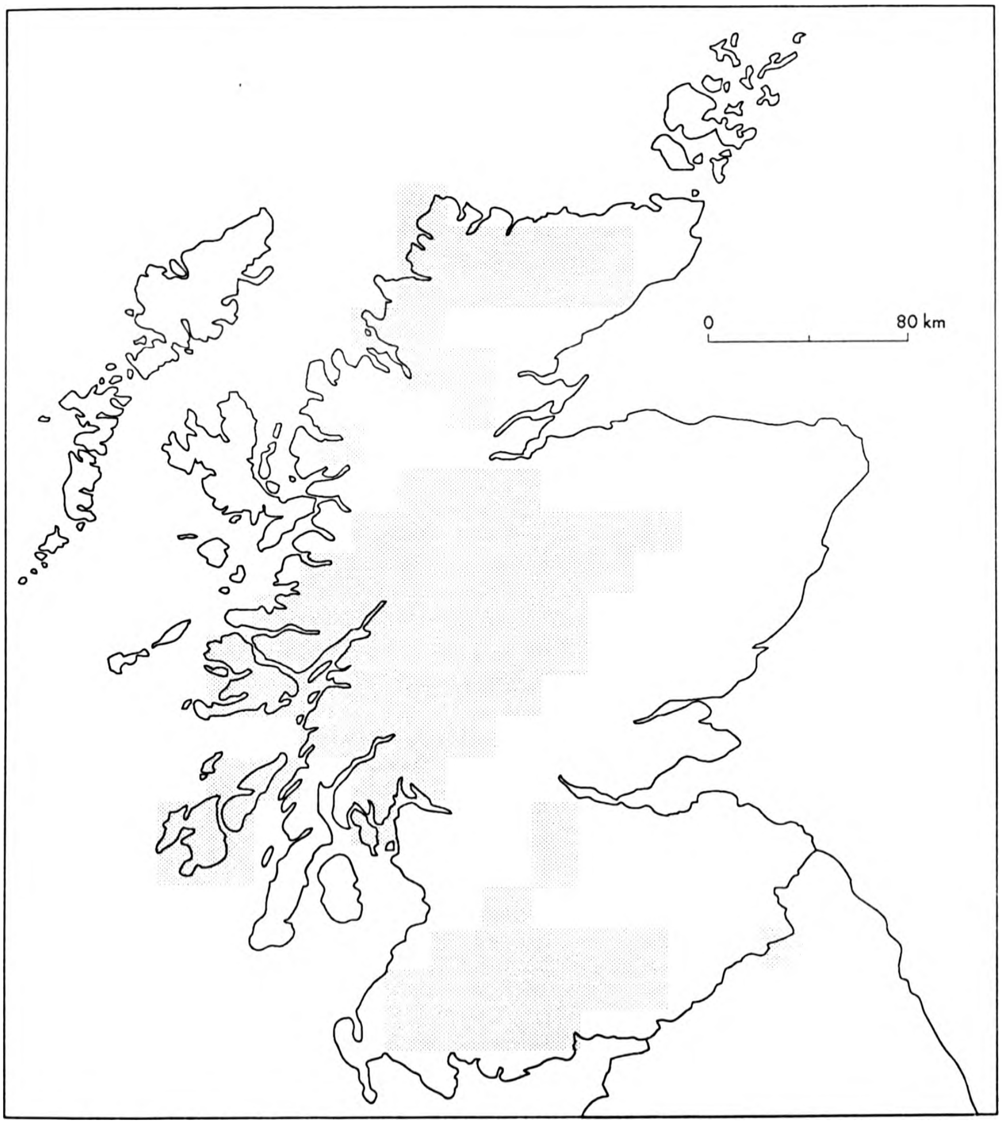
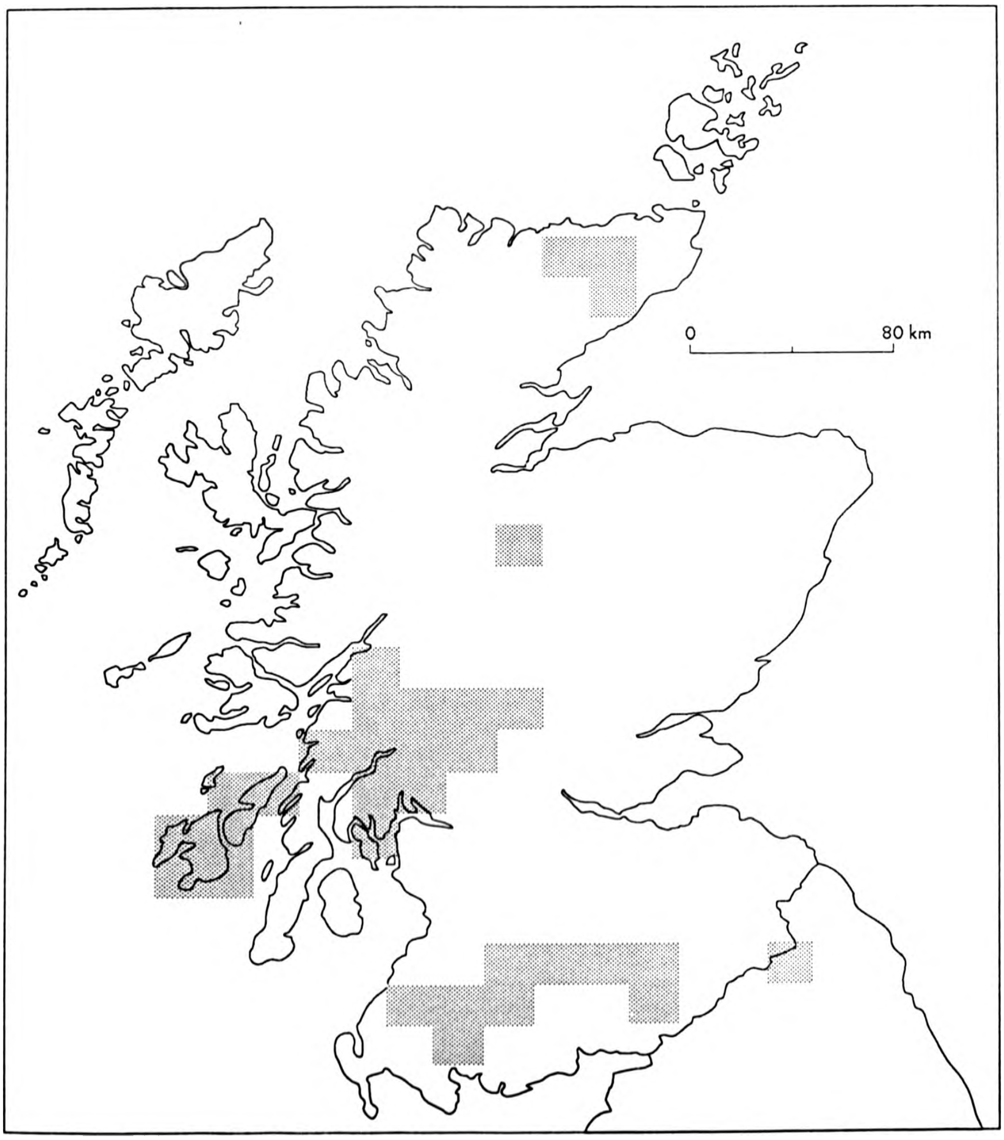


Figure 4.3: Area of Scotland where the Soils Critical Load is exceeded by greater than 1 keq H^+ ha^{-1} yr^{-1} under a 60% reduction of SO_2



the EU's Large Combustion Plant Directive on the area of acidified soils by the year 2003.

The areas that continue to be acidified under reduced emission levels are sensitive to relatively low acidic inputs, and are primarily located in Galloway, the south-west Highlands and Caithness. This distribution corresponds fairly well with the location of SSSI's damaged by acidification (Figure 1.1).

4.3 Ecosystem Impacts of Acidification

Changes in the chemical status of soil and freshwaters during acidification have resulted in a wide range of impacts on the diversity and abundance of flora and fauna in acidified areas. 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 of acidification are also important. For example, as acidification proceeds, humic matter is precipitated with the result that lakes become clearer. The change in the light and temperature regime which subsequently occurs can cause widespread changes in the composition of flora and fauna (Grahn *et al.*, 1974; Dickson,

1978).

In this section the processes of acidification in soils and water are briefly described, and is followed by a description of the main impacts on the flora and fauna of acidified areas.

4.3.1 Soil and Freshwater Acidification Processes

Acid inputs from the atmosphere influence soil chemistry by increasing the concentration of H^+ ions in the soil solution, (thus lowering pH), and by increasing the adsorption of Hydrogen (H) and Aluminium (Al) on to cation exchange sites on clay surfaces and organic matter (Wilson *et al.*, 1988). This latter process reduces the percentage of base cations on the exchange sites, leading to reduced buffering capacity, and the leaching of Calcium (Ca), Magnesium (Mg) and Potassium (K) which are essential for plant growth. Under very acidic conditions Al is released directly into the soil solution where it can damage root processes and can precipitate Phosphorus, thus making it unavailable for plant growth. Soils that are most vulnerable to acidification are typically derived from slow-weathering, metamorphic and igneous rocks such as schists, gniesses, and granites which have low buffering capacities.

During the course of acidification a lake generally passes through three distinct phases (Henrickson *et al.*, 1989). During the first phase the buffering capacity of the lake declines, but the pH is unaffected. There are few impacts on freshwater biota. In the second phase the buffering capacity becomes exhausted and pH becomes unstable. Concentrations of calcium, and other base cations decline, alkalinity levels change, and toxic metals such as aluminium, copper, lead and zinc, are released in to the water. Complex relationships between acidification and freshwater biota can emerge. Many organisms, including fish, are affected, but some invertebrates can survive relatively low ph levels if calcium levels are high.

During the final phase pH level stabilises at a low level, all fish are eliminated, and new top-level predators, usually invertebrate species, emerge.

4.3.2 Impacts on Flora and Fauna

Although changes in the flora and fauna of the semi-natural environment have undoubtedly occurred, there are few documented examples of changes to individual populations as result of acidification (Rimes, 1992). This partly reflects the slow rate of acidification which limits the potential for monitoring studies (Fry and Cooke, 1987), but also the difficulties in identifying the specific role of acid inputs in the process of change. The effects of acidification have been inferred from laboratory experiments, from comparisons of freshwater biota along a pH gradient (Flower *et al.*, 1986), and from comparison of present day flora with historical surveys (Jones *et al.*, 1986).

The most documented and best understood effects of acid deposition on the terrestrial and aquatic habitats of the semi-natural environment are now summarised. This account is not intended to be exhaustive but rather to provide a useful basis for establishing dose-response relationships for the economic valuation of recovery from acidification in the semi-natural environment. (More detailed information about acidification damage may be found in the literature cited.)

4.3.4.1 Aquatic Plants

Aquatic micro-flora and higher plants are the primary producers in upland lakes, and the impact of acidification on their ecology influences the entire food chain in affected areas. The sensitivity of certain species of diatoms to acidic conditions is well known and several studies have analyzed the change in composition of fossil diatom assemblages preserved in lake sediments to infer historical changes in lake acidity (Batterbee *et al.*, 1988; Raven, 1985). The method depends on establishing a quantitative relationship between diatoms and pH in present day samples, and provides important evidence for acidification in the UK. In general, acidophilous species become more dominant at the expense of acid-sensitive species, but there tends to be no overall decrease in the phytoplankton biomass (Flower and Batterbee 1983). For example, in one Scottish loch, Loch Laidon, the pre-1860 diatom assemblage was dominated by circumneutral taxa such as *Anomoeneis* and *Cyclotella* which were gradually replaced by acidophilous species, such as *Tabellaria flocculosa* and *Frustulia rhomboides* (Flower *et al.*, 1988).

Changes in species composition induced by acidification also triggers disturbance to the delicate process of plant competition. For example, certain species of sphagnum, which can survive acid conditions, are very efficient scavengers of the available nutrient pool. As these lakes become progressively more oligotrophic, many other micro and macro plant species are lost (Friberg *et al.*, 1980).

There is very limited evidence of acidification impacts on higher plants in the aquatic environment. Studies in North America and Scandinavia have demonstrated that acidification has been accompanied by replacement of calciole species with Juncus bilbosus (Hendrey and Vertucci, 1980; Farmer, 1990). However, no such trend among macrophytes in the UK has been observed, but this may simply reflect a lack of the necessary research.

4.3.4.2 Trees

In the late 1970's premature defoliation and reduced growth of Norway spruce (*Picea abies* (L.) Karst.), European beech (*Fagus sylvatica* L.), and silver fir (*Abies alba* Mill.) in the forests of central Europe were claimed to be dramatic evidence for widespread pollution by SO_2 and other emissions (Rose, 1990).

Acidic inputs are thought to harm trees through several mechanisms. Firstly, high atmospheric concentrations of pollutants can directly harm leaf processes and tissue. Under controlled experimental conditions exposure to high concentrations of gaseous SO_2 have been shown to effect stomatal functioning and the control of water loss (Freer-Smith, 1985); susceptibility to spring frosts (Cape *et al.*, 1990) and aphid infestation (Warrington and Whittaker, 1990). However, the concentrations of SO_2 which induced these effects are generally well in excess of those commonly experienced in semi-natural areas.

Second, the process of soil acidification arising from wet deposition is also detrimental to tree health. Nutrients essential for growth are leached out, and by increasing the concentration of heavy metals which interfere with water uptake by the roots can be increased (Department of the Environment, 1993). These effects are now believed to be the most likely cause of damage to the forests of central Europe. In this region, soils are generally deficient in Magnesium (Mg). Acidic inputs tend to exacerbate this deficiency by enhancing the rate of leaching of Mg in the soil, and by stimulating growth rates as a result of increased N availability in the soil (McLeod *et al.*, 1990).

There is little evidence from field observations that acid deposition actually seriously affects tree health in the UK. A review of tree health surveys by the Forestry Commission, found that in only 16 out of 10,500 cases of tree damage was air pollution the probable or definite cause of tree injury (Innes and Boswell, 1990). Trends in crown dieback and thinning in several major forest species were found to be primarily related to climate variation, and not air pollution (Forestry Commission, Press Release 274/93).

In the UK, Mg deficiency is not considered to be a likely mechanism for damage to forests because atmospheric inputs of Mg are much higher than in central Europe (Roberts *et al.,* 1989).

A clear understanding of acid deposition on tree health has yet to emerge. Trees are relatively long lived, and it is difficult to separate out the effects of SO_2 from other pollutants, such as ozone, and environmental stresses such as climate fluctuations, pathogens attack, and human disturbance, which can act additively, synergistically or antagonistically with high levels of dispersed secondary pollutants (McNeill and Whittaker, 1990; Cowling, 1989). Despite extensive laboratory and field investigations, the link between acid pollution and tree health has yet not been proven conclusively. A major review of the effects of SO_2 on tree health in the UK on behalf of the Department of the Environment concluded only that '*the air pollution climate in some areas of the UK may be detrimental to tree health*' (Dept. of Environment, 1993).

4.3.4.3 Moorland Communities

Although there has been considerably less research into the effects of air pollution on other higher plants, it would appear that heathland communities are affected by acid deposition. Presst *et al.,* (1974) reports that in areas of the Peak District bog myrtle (*Myrica gale*), sundew (*Drosera intermedia*) and heather (*Calluna vulgaris*) have declined in health and vigour over the past 200 years, although the specific effect of SO_2 is not clear. Thompson and Baddeley, (1991) suggest that higher levels of nitrogen, heavy grazing pressure, and high visitor numbers are also causative factors in the decline of species characteristic of moorland communities, and their replacement with cotton grass (*Eriophorum vaginatum*) and bilberry (*Vaccinium myrtillus*).

4.3.4.4 Mosses and Lichens

Non-rooting plants rely on the atmosphere to provide nutrients for growth and, as a result, tend to be more vulnerable to high levels of air pollution than higher plants (Le Blanc and Rao, 1973). For example, sphagnum communities on peatlands in the UK are believed to have been replaced by acid tolerant species, as a result of high concentrations of SO_2 and NO_x (Woodin *et al.*, 1987).

4.3.2.5 Freshwater Invertebrates

Acidification generally leads to a loss of diversity and a simplification of community structure among micro-invertebrates. If the acidification process is advanced only a relatively few species of cladocerans, rotifers and copepods dominate. Macro-invertebrate populations also become impoverished, with a reduction in the numbers of crustacea, leaches, gastropods and molluscs. Insect species such as mayflies and gammarids are often absent, with stoneflies and chirominids plentiful (Harriman and Morrison, 1982; Sutcliffe and Carrick, 1983).

These effects arise as a result of a range of direct and indirect processes. Low pH, and high levels of toxic metal released during the acidification process, are major causative factors, since they directly harm the metabolic processes of invertebrates. Invertebrate communities are also affected by changes in nutrient and food cycles, induced by diminishing rates of detritus breakdown by acid-sensitive bacteria, and by the loss of major predators (Fry and Cooke, 1987). Reduced levels of Calcium also inhibits shell development among molluscs.

4.3.2.6 Fish

Species of fish vary in respect to their sensitivity to acidification. Salmon, brown trout, roach, and char are the most sensitive, and perch and pike are the most tolerant (Bergquist, 1990). The decline of trout (*Salmo trutta*) and salmon stocks (*Salmo salar*), as a result of acidification has been an important focus for concern in the UK, Scandinavia and North America. This reflects both their ecological significance as top predators in freshwater ecosystems, but also their economic importance to commercial and recreational fisheries in these areas.

Although historical information on the abundance and condition of fish populations is sparse in the UK, acidification is thought to have initiated a significant decline in fish stocks over 50 years ago (Harriman *et al.*, 1987). Since fish have a longer life cycle than many other aquatic species, they are more vulnerable because they are more likely to encounter severe acidic episodes, following heavy rain for example.

Evidence for the direct effects on fish populations from field experiments is rare. Some scientific surveys have implicated acidification in the loss of brown trout from a number of forested catchments in Wales (Stoner and Gee, 1985) and Scotland (Harriman and Morrison, 1982). However, the role of commercial afforestation, either through enhancing the acidification process, or through other impacts, was not determined in these studies. On one or two occasions, large numbers of fish have been reported to have been directly killed by pulses of extreme acidity at snow melt, or after long dry periods (Milner and Varallo, 1990; Stephen, 1990). Other reported evidence for acidification damage include tail deformities in trout (Fry and Cooke, 1987) and the loss of the arctic charr (*Salvelinus alpinus*), the rarest salmonid species in UK, from some lochs in south-west Scotland (Rimes, 1992).

Experimental studies have shown that salmonids are most sensitive to low pH. Lacroix (1987) found, for example, that the production of salmon smolts increased exponentially between pH 4.5 and 6.0[3]. Low pH levels disturb the salt balance within the body tissues with sodium, in particular, being lost from the body faster than it can be taken up (McWilliams, 1982). The action of hatching enzymes such as chorionase may also be inhibited (Haya and Waiwood, 1981). Elevated concentrations of labile aluminium, leached

[3] Of the main salmonid species, the Atlantic salmon appear to be more sensitive than either sea or brown trout to the effects of acidification (Rosseland and Skoghiem, 1984)

out of soils by acid water moving through the soil profile, are also believed to impair respiratory processes in salmonids, with damage to gill filaments reported especially at low Ca concentrations (Rosseland, 1980). Fish appear to be most sensitive to these effects acidification during the hatching, fry and smolting stages of development, with recruitment failure is thought to be the primary cause of population decline.

4.3.2.7 Amphibians

Amphibians are extremely sensitive to acid conditions, and are restricted in their range by the acidity of their breeding sites (Cummins, 1986). Low pH, together with high concentrations of heavy metals are believed to be harmful to amphibians. Under experimental conditions, these conditions have been shown to cause embryonic mortality, delayed hatching, and spinal deformities (UKAWRG, 1988).

Decline in the numbers and range of the natterjack toad (*Bufo calamita*) in south-east England has been related to pond acidification in their heathland range (Beebee *et al*., 1990; Beebee, 1987). In south-west Sweden the breeding failure of a common frog, *Rana temporaria*, is reported to have been caused by lake acidification (Hagstrom, 1981). In Scotland, mortality rates among common frogs were also correlated with pond pH (UKAWRG, 1988).

4.3.2.8 Mammals

As a group, mammals have been relatively unaffected by acidification. The possible decline in the population of the otter (*Lutra lutra*), as a result of acidification has attracted considerable public attention. It is known to be absent from many acidic headwaters in some parts of the UK (Rimes, 1992), and its loss in the Glentrool of Galloway has been attributed to the combined effects of acidification and afforestation (Green and Green, 1987).

The evidence for the role of acidification in the decline in otter distribution is, as in many other cases, anecdotal. However, there is no doubt that depleted fish stocks as a result of acidification is likely to have had repercussions for the distribution of the otter.

Wild deer meat from an area of Sweden with high acid inputs was found to be contaminated by heavy metals (Lindvall, 1984). Similar concentrations have not been reported in Scotland, or elsewhere in the UK. Also, there is no published evidence to indicate that deer or sheep have been adversely affected by acid deposition.

4.3.2.9 Birds

The population of the Dipper (*Cinclus cinclus*), a passerine species, is in decline in acidified areas of north Wales. It is believed that a reduced supply of acid-sensitive prey species, such as benthic invertebrates and small fish is partly responsible (Ormerod *et al.,* 1988). Population levels of other bird species have also been affected by acidification. Hasselrot and Hultberg, (1981) report that thin-egg shells as a result of disrupted calcium balance, is thought to be adversely affecting the reproduction of the pied flycatcher (*Ficedula hypoleuca*).

Other effects are less direct, but equally significant. In the Netherlands, for example, calcium deficiency in the soil has resulted in a decline in the population of snails, which require Calcium to build their shells. Populations of several woodland bird species that prey on them have declined as a result (The Economist, 1994). High concentrations of mercury, which becomes more mobile in the environment as acidification increases, have also been found in the eggs of red-throated divers, and are thought to threaten successful reproduction (Acid News 5, 1993).

4.4 Prospects for Recovery Following SO_2 Abatement

The review of scientific research in the preceding section describes the damage that has been caused by acidification. Unfortunately, current understanding of colonisation dynamics and biological processes of affected biota are generally insufficient to provide expected timescales and patterns of recovery (Weatherly, 1995). The obvious assumption to make is that affected ecosystems will gradually recover to pre-acidification levels in areas where acidic deposition levels fall below the Critical Load.

There is some limited evidence from research to support this argument. For example, during the 1980's, when the deposition of sulphate in Galloway fell by 50%, diatom communities, pH and, sulphate levels in affected catchments have shown a rapid recovery toward pre-acidification levels (Batterbee *et al.,* 1988). Even so, no clear recovery in base cation concentrations was established and it is likely that if a full recovery is possible, it may take a much longer time period (Harriman, 1990). Experiments with liming in Scandinavia indicate that some species, like trout, respond quickly to improve conditions, but the general recovery pattern is unclear. For example, floral and invertebrate assemblages tend to show a marked delay in response to improved water quality (Weatherly and Ormerod, 1991).

Since ecosystems are dynamic they will not necessarily adapt from acidified to less acidified conditions along the same ecological pathway as occurred during the acidification period. Some doubt must therefore exist as to the extent to which acidified ecosystems will recover to pristine conditions. In some areas full recovery simply might not be possible. For example, fish species which have become locally extinct, may be unable to recolonise because of physical barriers created by hydro-electric development. Other species, such as molluscs which are relatively immobile may be unable to recolonise suitable habitat for a very long time. In this circumstance it would be reasonable to assume that acidification has resulted in irreversible damage.

Since the nature of recovery in the semi-natural environment will also be determined by other factors, such as the concentration of other pollutants, environmental stress and climate change, it is possible that recovery may not mirror the original damage process. Scientific understanding of these interactions is partial. Predictions about future recovery must therefore, to some extent to, considered to be speculative.

In order to assess the net impact of reduced SO_2 inputs in semi-natural areas it is also necessary to consider the nature of future damage from acidification under the status quo (i.e. if no further action is taken). Fortunately there is a considerable amount of cross-sectional and temporal data available which gives some indication of future damage trends. For example, evidence from sites which are the most highly acidic, suggest that species diversity will continue to decline, with local extinction of a range of species, including fish, passerine birds and otters, possible (Black *et al.*, 1990).

4.5 Conclusions

Decades of anthropogenic acid inputs have had a marked effect on the biodiversity of the semi-natural environment of the Scottish uplands. There has been widespread disruption of the macrophyte, invertebrate and phytoplanktonic communities, and a reduction in the populations of higher level species such as fish, birds and mammals. The loss of salmon and trout is of particular concern because of their importance for recreational fishing in affected areas.

Although the Critical Load approach provides the most reliable indication of the likely extent and location of recovery following abatement (and damage under the status quo), the exact nature of recovery is not known with any certainty. This reflects the complexity and range of the environmental changes initiated by acidification, and the interaction between acid deposition and other environmental stresses. It will be important to incorporate this uncertainty about future environmental recovery and damage in the valuation of the economic benefits of abatement.

CHAPTER 5 MEASURING THE ECONOMIC BENEFITS OF SO_2 ABATEMENT

5.1 Introduction

An economic benefit is defined as any good or service for which individuals have a positive preference, and which can be measured in terms of their willingness to pay (Pearce, 1983). The net benefits to society of reducing SO_2 deposition are determined by estimating changes to consumer and producer surplus (Hanley and Spash, 1993). Since acidification of freshwater and terrestrial ecosystems has had little impact on the productivity of agricultural and forestry systems in the Scottish uplands, benefit estimation must necessarily focus on changes in consumer surplus. In the context of SO_2 abatement this is the value an individual or household places on the prevention of future damage, and ultimately, on ecosystem recovery.

The non-market benefits of SO_2 abatement will accrue to a diverse range of individuals and groups in society. These will include those directly affected by acidification such as recreational anglers but also non-users, such as concerned members of the general public. This chapter identifies and describes appropriate methodologies for measuring these environmental benefits. Firstly, the concept of total economic value will be introduced in order to identify the range of benefits that might arise from recovery in the Scottish uplands and the likely beneficiaries. The remainder of the chapter will then be devoted to the theory and practice of measuring non-market benefits.

5.2 Total Economic Value

Conventionally a household's spending behaviour in the market place is considered to be an accurate signal of its' preference for exchangeable goods and services. However, households can also derive satisfaction from environmental goods and services which are not marketed. These include recreational opportunities, improved health and safety, and higher levels of environmental quality.

The concept of total economic value, introduced by Randall and Stoll, (1983) embraces this wider notion of economic utility and has become the accepted basis for resource valuation. The main components of total economic value, and their relevance to environmental recovery from acidification in the Scottish uplands, are described below.

5.2.1 Use Values for Recovery in the Semi-Natural Environment

Use benefits arise from all consumptive and passive means by which a household makes use of the environmental resource (Hanley and Spash, 1993). Passive use values include the benefit people derive from access to a resource but which do not entail physical use. Examples of passive use of the environment include photography, the pleasure derived from watching television documentaries about nature, or simply viewing the landscape from a passing car. Consumptive use of the environment includes output of marketable products such as timber or livestock, water abstraction for industrial or domestic consumption, and recreational activities such as fishing and hiking.

The decline in salmon and trout catch has been one of the more noticeable impacts of acidification in many areas (Waters and Kay, 1988). In a recent survey by the Scottish Anglers National Association (SANA, 1991), approximately 80% of members considered acidification of lochs and rivers to be the most serious threat to fish stocks.

Salmon angling on rivers in Scotland is a particularly valuable activity in rural areas, generating an estimated £50 million a year in income, with a capital value of approximately £300 million (Radford *et* al., 1991). Since catch is an important element in the satisfaction that can be derived from a day's angling, it is likely that anglers would be prepared to pay something to secure recovery in salmon stocks as a result of SO_2 abatement. Trout have also been affected but fishing for this species is of much lower market value. Affected trout fisheries are generally restricted to fairly remote and inaccessible lochs for which most anglers are unwilling to pay significant sums of money (Stephens, A. pers com).

Although acid deposition is known to affect the vigour of heather moorland in some upland areas, other factors such as over-grazing, and recreational pressures are thought to be more important (ECOTEC, 1992). There is no evidence to suggest that acidification has reduced the quality of pasturage available for livestock and game (e.g. grouse and deer).

Many upland water catchments which supply potable water to urban areas in Scotland 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. However, because leaching also occurs naturally in the soft waters of the Scottish uplands water treatment processes are in place to mitigate these problems and the cost of additional treatment (*e.g.* through liming and filtration) are considered to be negligible (Hooper, pers com).

Hill-walkers and ramblers are perhaps the only other significant user-group that may have been affected by environmental change in acidified areas of the semi-natural environment. However, the extent to which walkers actually visit acidified sites, many of which are very remote, and have also had their enjoyment affected as a result of acidification, is not at all clear. For example, the environmental changes that will be initiated by abatement, in comparison to other forms of pollution, are likely to be subtle and not easily observable to anyone except the keen naturalist. This suggests that the benefits associated with walking and the appreciation of landscape are likely to be restricted.

5.2.2 Option Value

Originally defined by Wiesbrod (1964), option value is the amount that people will pay for the right to enjoy a resource for a specified price at a specified point in the future (Bishop, 1982). The option price for environmental resources is equal to E(CS) + OV, where E(CS) is expected consumer surplus from use and OV is the option value (Desvousges *et al.,* 1987).

The sign and relative magnitude of option value can vary. If individuals are risk averse and there is uncertainty concerning future supply OV will be positive (Freeman, 1984), but when different attitudes to risk are considered and uncertainty is extended to the demand side (taste, income, *etc.*) OV may be positive or negative (Pearce and Turner, 1990). In general where the resource in question is unique and subject to irreversible change, OV will be large (Mitchell and Carson, 1989). For ex-ante evaluations in the presence of uncertainty, such as in the case of acidification, the appropriate welfare measure is the option price, which includes option value.

Also related to uncertainty is quasi-option value (QOV). This is the value placed on preserving options for future use given some expectation of the growth of knowledge (Arrow and Fisher, 1974). Since it represents the value of information Pearce and Turner (1991) suggest that it will always be positive and, in most cases, support the postponement of development so that a better decision can be made later. However, in the case of investment in SO_2 abatement the effect of QOV on TEV is moot. This is because information can be gathered following abatement or if abatement is delayed (Mitchell and Carson, 1989). For example, an SO_2 abatement programme will generate a QOV by providing information about the effectiveness of abatement technology, and the opportunity to learn more about acidification by delaying potentially irreversible damage. On the other hand, by not investing in abatement technology, more information will be gathered about the extent to which abatement is actually needed.

5.2.3 Non-Use Values

This category of benefit derives from the notion that individuals can derive utility from environmental resources outwith expected personal use. 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). 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' (i.e. non-use) value in the context of uniqueness and irreversibility, it is now accepted that non-use values can be held for a wide range of environmental goods and services (Randall and Stoll, 1983).

Including non-use value within the framework of total economic value has attracted criticism from some economists. The debate has tended to focus on measurement issues and the treatment of altruism towards people and other sentient beings in CBA. In utilitarian terms non-use benefits reflect the value people derive from the existence of environmental resources outwith expected personal use. However, Milgrom (1993) and Diamond and Hausman (1994) have argued that non-use values are not permissable in cost benefit analysis because they are not motivated by selfish concerns. This view is not widely supported since, as Hanemann (1994) points out, economic theory does not preclude altruism, since individuals are free to maximise welfare as they conceive it.

The nature of altruistic concern is relevant to economic appraisal however. Some individuals have the view that resources (e.g. wildlife, the planet earth) have a value, an intrinsic value, quite unrelated to human welfare. For example, Stevens *et al.*, (1991) found that 70% of respondents in a CV survey believed that animal species should be preserved irrespective of usefulness to humans, and were consequently unwilling to pay for preservation programmes. Intrinsic values for non-human species based on a moral or ethical prerogative arises from the way in which some individuals view the world (Pearce and Turner, 1990).

Intrinsic value clearly does not fit within a strict neo-classical utilitarian framework but it is not clear to what extent 'genuine altruism' (Kennet, 1980) prevails in the human population. However, it remains an important issue for valuation and CBA.

Considerable debate also surrounds the treatment of altruism directed at other human beings. While some economists have argued that altruism will lead to double-counting (Bergstrom 1982, Milgrom, 1993), others consider altruistic values to be valid if, and only if, it is focused on the utility of others with respect to the environmental change concerned (Jones-Lee, 1992; Johansson, 1992).

McConnell (1997) argues that while non-paternalistic altruism is not relevant to the outcome of CBA, paternalistic altruism is, and should be allowed. In the former the general public care only about the well-being of the user and not about the value of the services to the user. Paternalistic altruism on the other hand is motivated by the use of the resource itself, and the altruist is not concerned about the value of the services to the user. In this case non-use benefits are relevant and raise the prospect of project approval. McConnell (1997) concludes that while a mixture of altruistic motives are likely to be present in the general public paternalistic altruism is most relevant to natural resource allocation (and hence most CV studies).

Since 'there appear to be neither obvious or subtle behavioral trails that can provide information' about non-use values they cannot be measured by revealed preference techniques (NOAA, 1993). Hypothetical market approaches, such as the Contingent Valuation method (CVM), have been developed, but 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, (1989) have argued that CVM 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 chapter.

Until recently CVM studies have tended to focus on user-related environmental changes, and it has been difficult to distinguish between non-use and use values. Research which has attempted to do so, based on a variety of assumptions about motivation, have concluded that non-use values represent an important, and usually dominant, component of TEV. For example, Willis (1990) found from a study of wildlife sites that non-use values, even among visitors, was approximately 80% of total WTP. Other wildlife studies, including the Brookshire *at al.*, (1983) study of grizzly bears, and Stoll and Johnson's, (1985) study of whooping cranes, also discovered a significant non-use component.

Even when non-use values are relatively small, they can greatly increase the total benefits of an environmental project when aggregated across the entire population (Loomis and Walsh, 1986). For example, Grandstaff and Dixon's

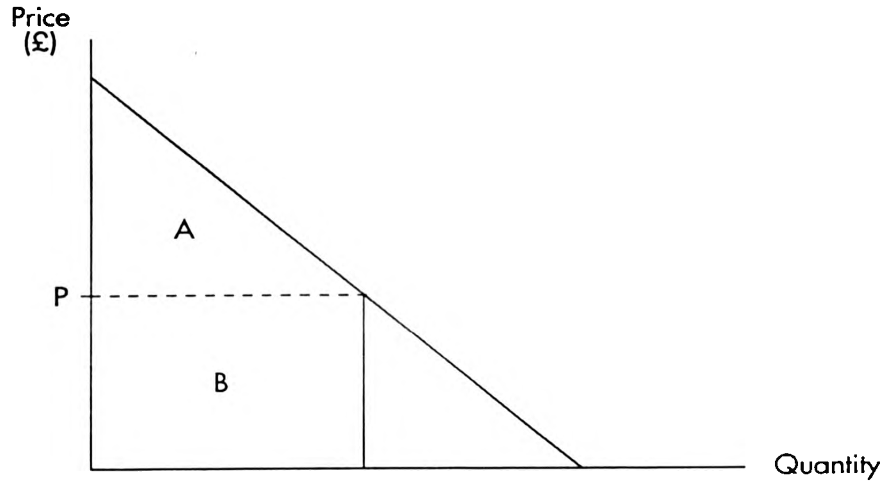
(1986), valuation study of Lumpinee public park in Thailand found that when the values of non-users were aggregated across the 3 million residents of Bangkok, the total economic value of the park increased from US $6 to $58 million.

The likely magnitude of the non-use benefits that might arise from environmental recovery from acidification in the semi-natural environment are not known. However, evidence from a study by Navrud, (1989) of the willingness to pay of the general public for recovery in fish stocks, suggest that non-use values are likely to be large. In the case of the Scottish uplands where the main areas affected by acidification are relatively remote, and infrequently used, it is likely that they may also be the dominant component of the general public's TEV.

5.3 Choice of Welfare Measure.

The economic benefits of environmental improvement are measured in terms of what people are willing to pay (WTP) for it, or are willing to accept (WTA) compensation to forgo it. In the case of marketed goods, total WTP is equal to expenditure plus the consumer surplus (CS): that is the area under the demand curve, but above the price line (Figure 5.1). Since markets are absent for many environmental goods and services economic valuation must necessarily focus on the measurement of CS.

Figure 5.1: Total benefit under a Marshallian demand curve



A = Consumer surplus
B = Total expenditure
A + B = Total Benefit

The approach of measuring consumer benefits on the basis of changes in consumer surplus was developed originally by Dupuit (1844) in the 19th century, and later refined by Marshall. If income is held constant, Marshallian demand curves describes how a change in the quality or quantity of environmental goods will affect consumer surplus. However, in utility theory, it is strictly required that utility, not income, is held constant, with the extent to which a consumer is actually better off depending on both income effects, and relative prices.

Hicks (1941) suggested four alternative theoretical measures for welfare change which hold utility constant. These are the compensating variation, compensating surplus, equivalent variation, and equivalent surplus. Compensating measures relate to the amount of compensation paid or received which would keep the consumer at the initial welfare level, after the change had taken place. Equivalent measures are the compensation, paid or received, necessary to bring the consumer to the subsequent welfare level, if the change did not take place (Mitchel and Carson, 1989).

Variation measures are used when the consumer is free to vary the quantity of the good consumed whereas surplus measures are used when the consumer is constrained to purchase fixed quantities of the good. Surplus measures are therefore most appropriate when considering changes in the levels of goods which are determined by government action, rather than the market (e.g. public goods).

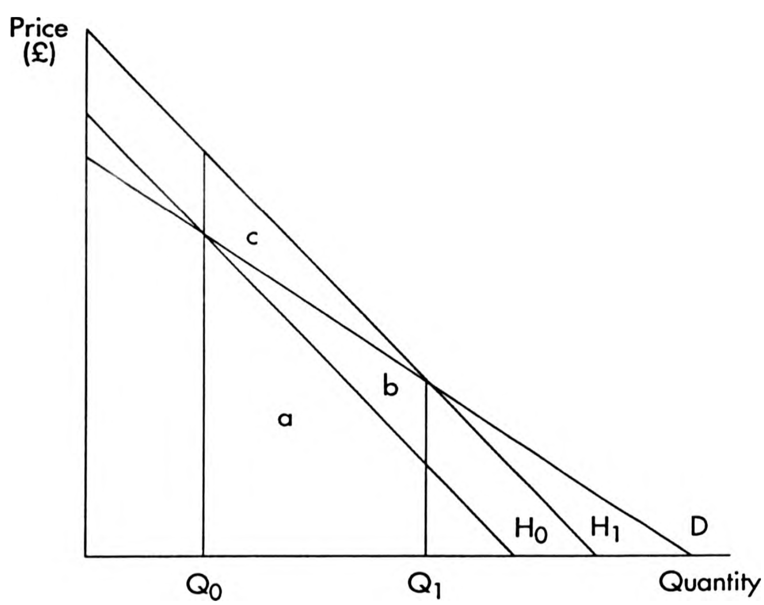
For the public good case, Marshallian CS is equal to or larger that compensating surplus, but less than, or equal to, equivalent surplus when an increase in supply is considered (Figure 5.2). When the quantity is decreased, the relationship between the two measures is reversed. The difference between the Hicksian measures depend on the income elasticity of demand (Willig, 1976), and the substitution elasticity for the good (Hanemann, 1991). Although higher income elasticities will yield larger differences between the two measures, Randall and Stoll (1980) showed that in most practical situations, (i.e. involving small quantity changes or small budget items), the difference between the two is relatively insignificant. Hanemann (1991), on the other hand argues that large differences between compensating and equivalent measures can occur if private goods are a poor substitute for the public good[1].

5.4 Valuation Methods

Over the last 30 years a range of approaches have been taken to estimate the monetary value of environmental change. Each differ with respect to the type of benefits they measure, their data requirements, and statistical techniques used. In this section the main valuation methods are described and assessed for their applicability to

[1]The divergence between WTP and WTA measures are discussed in more detail in section 5.4.3.

Figure 5.2: Marshallian and Hicksian surplus measures



$Q_0 \longrightarrow Q_1$ quality change
D = Marshallian demand curve
H_1 = Hicksian demand curve for utility level U_1
H_2 = Hicksian demand curve for utility level U_2
a + b = Consumer surplus
a = Compensating surplus
a + b + c = Equivalence surplus

estimating the benefits of acid rain abatement in the semi-natural environment of the Scottish uplands.

5.4.1 The Hedonic Price Method (HPM)

The HPM is derived from the characteristics theory of value developed by Lancaster (1966) which proposes that each commodity can be described as a combination its characteristics. For a range of goods environmental quality is an important determinant of price. The HPM exploits a statistical relationship between environmental quality variables, and the observed variation in the price of market goods, in order to derive a demand function for environmental quality.

The method has been widely used in environmental economics, and has the primary advantage that it generates 'hard' estimates of value based on observed market behaviour (Winpenny, 1992). It has been applied to valuing countryside attributes using house prices (Willis and Garrod, 1991), soil erosion using agricultural land values (King and Sinden, 1988; Palmquist and Danielson, 1989), and noise and urban air pollution using residential property values (O'Bryne *et al.*, 1985; Brookshire *et al.*, 1982).

Estimating the benefits of environmental change with the HPM requires a two stage approach. Using the residential property market as an example, Stage 1 involves regressing the prices of houses with a range of variables thought to influence the purchase price of the property. These may include features of the property (*e.g.* the size and number of rooms), accessibility to local amenities, and environmental quality (*e.g.* noise levels). Calculating the derivative of the dependant variable, with respect to the environmental attribute of interest, yields the marginal cost associated with an increase or decrease in the attribute of interest (*ceteris paribus*). This implicit price is assumed to be equivalent to marginal WTP (OECD, 1989).

Since marginal willingness to pay will vary depending on household income and other characteristics, a second stage is necessary in order to estimate the inverse demand curve for the environmental attribute. This is achieved by regressing the implicit price for the environmental attribute of interest against environmental quantities, household incomes and other relevant variables.

The Hedonic demand curve estimates the economic benefits of changes in environmental quality only if a series of assumptions are made about the market. Firstly, the conditions necessary for a fully functioning market must exist. For example, each household is assumed to have access to full information, can recognise differences in environmental quality, and is able to buy a house with all of the desired characteristics. Secondly, the price paid should be equivalent to the highest bidders maximum WTP. Thirdly, the supply of housing is typically assumed to be fixed.

Application of the method has a number of drawbacks. The HPM requires a significant amount of cross-sectional, or preferably, since policies typically bring about environmental change over some time period, time series data on market prices. Also, since regression analysis is used, the implicit price of the environmental attribute is influenced by the choice of explanatory variables, the presence of multicollinearity among the variables, and the choice of functional form for the hedonic price equation (Hanley and Spash, 1993). In many applications the environmental variables have low significance in the regression equation, and the regression equations themselves have low explanatory power (Brookshire *et al.*, 1982). Non-use values cannot be estimated using a hedonic model.

Most applications of the HPM to pollution issues have focused on the relationship between urban environmental quality (e.g. noise and health effects), and residential property values. In areas affected by acid deposition in the semi-natural environment, few people actually live there and it is not obvious how acidification would affect house prices. The potential for applying the HPM to quantify the benefits of improved environmental quality through changes in property markets would therefore appear to be limited.

However, since the right to fish for salmon is exchanged in the market place, and is known to be influenced by variables which have been affected by acidification (Radford *et al.*, 1991) it should be possible to estimate the economic benefits of recovery in salmon stocks through the HPM. To do so would require linkages to be established between changes in water quality, catch and salmon fishing rents.

5.4.2 Travel Cost Method (TCM)

Developed over 30 years ago, the TCM has been widely applied in the United States and the United Kingdom to estimate the value of non-market outdoor recreational sites (Willis and Garrod, 1991; Hanley 1989; Willis and Benson, 1989), and changes in environmental quality which affect recreational activity (Walsh *et al.*, 1989; Huppert, 1989). The model is based on Hotelling's simple proposition that observed behaviour can be used to derive a demand curve for unpriced environmental goods and services.

The TCM utilises data on the costs of consuming the recreational services of a site as a proxy for variable admission prices. Such costs include travel costs to the site and to substitute sites, as well as entry fees (OECD, 1989). As with the HPM, weak complementarity between the environmental asset and consumption expenditure is a central assumption (Hanley and Spash, 1993).

The basic method developed by Clawson and Knetsch (1966), involves grouping visitors into zones and estimating a trip generating function for each zone of the form:

$$V_{zj} = V\ (C_{zj},\ Pop_z,\ S_z)$$

where z = zone $z = 1....Z$

V_{zj} = visits from zone z to site j

Pop_z = population of zone z

S_z = socio-economic variables

To estimate Consumer Surplus per visit a second stage demand curve can be traced out by predicting how visitation rate responds to changes in travel costs (e.g. increased entry fees).

A variant of the method is the Hedonic travel cost model developed by Brown and Mendelsohn, (1984) which can be used to estimate the demand curve for individual characteristics of recreational facilities. With this approach a further stage is required where the individual site characteristics (*e.g.* picnic sites, waterfalls, broadleaved woodland) are regressed against the predicted marginal value of the characteristic in different zones (based on their regression coefficients estimated from a travel cost function) and other socio-economic variables.

Although the Travel cost method has attracted considerable attention from economists, and has the advantage of being based on observed behaviour, it has a number of important limitations. Data collection and analysis is time consuming and expensive, and difficulties remain with respect to the treatment of travel time, multi-purpose trips and congestion (Hanley *et al.*, 1996). Also, the TCM assumes people necessarily know the enjoyment they will get from a trip before they set out, which might be reasonable for regular visitors, but not for first time visits.

TCM results are highly sensitive to the statistical models used to specify the demand relationship (OECD, 1989). For example, Hanley (1989) found that CS estimates for Queen Elizabeth Forest Park varied between £73,948 to £1,497,858 depending on the functional form employed. Willis and Garrod, (1991) found that TC estimates based on the simple zonal estimates were significantly higher than those derived from an individual TC model. Multi-collinearity between environmental variables can result in a mis-specified relationship between visitation rates and changes in environmental characteristics, and low statistical power. Also, the price at which demand falls to zero, required to estimate consumer surplus, may lie outside the range of the observed data, and therefore be estimated inaccurately (Hanemann, 1994)

The TCM is most suitable for estimating the use value of parks, or other attractions in the countryside which attract a large number of visitors from a range of distances. It has not been used to any great extent with respect to acidification damage in the semi-natural environment, partly because of the diffuse nature of the damage, but also because the effects on visitation rates are not clear. For example, Mullen and Menz (1985), who used the TCM to estimate acidification damage to the Adirondack fishery in New York, had to assume a very simple binary visitation model based on whether a site had fish or had not.

The potential to apply TCM to recovery in the semi-natural environment is limited for a number of additional reasons. Firstly, since these areas are generally remote and inaccessible, visitors to affected areas are rare. Secondly, salmon fishing, which is the main angling interest of these areas, is controlled privately and market prices are therefore available to measure preferences for enhanced fishing opportunities. Finally, the method cannot measure non-use values.

5.4.3 The Contingent Valuation Method (CVM)

CVM is a survey based method which allows people to buy (WTP) or sell (WTA) non-market environmental goods in a hypothetical market situation. In order to establish a satisfactory market exchange a CVM survey should: 1) describe the environmental change; 2) identify the method of payment; and 3) define the 'market'. Socio-economic and preference questions should also be included in order to validate WTP/WTA responses (Hoevenagel, 1993a).

Several important assumptions from consumer theory underpin the theoretical validity of the Contingent Valuation Method. Firstly, individual economic agents have well-defined preference orderings, and can recognise the shape of their indifference curves. Secondly, an economic agent will make choices which will maximise his/her overall level of utility and lastly, that these choices are made on the basis of perfect information. The revealed trade-off between WTP and the quantity of the environmental good consumed will reflect true values or preferences only if these assumptions are valid (Cummings *et al.*, 1986).

Although the potential for using surveys to establish WTP for the public goods was recognised over 40 years ago (Ciricacy-Wantrup, 1952), it was not until much later that applications began to emerge. A study of the aesthetic benefits of reduced air pollution by Randall *et al.* (1974) 'provides the essence of contemporary applications' of CVM (Cummings *et al.*, 1986). CVM has now been applied to a range of environmental issues such as improvements to recreational fishing (Huppert, 1989; Bonnieux, 1993; Olsen *et al.*, 1991), wildlife viewing (Brookshire *et al.*, 1983) and disposal of toxic waste (Burness *et al.*, 1983). The method has also found applications in the field of human health and transport planning.

An important landmark in the application of the technique came with the publication of the report by an expert panel of economists, including Nobel laureates, Solow and Arrow, on behalf of the US National Oceanic and Atmospheric Administration (NOAA). The panel concluded that the method was 'reliable enough to be the starting point of a judicial process of damage assessment, including lost non-use values' (NOAA, 1993). A number of recommendations were also made with respect to implementation of CVM surveys. These include:

* A discrete choice format should be used
* A minimum response rate from the target sample of 70% should be achieved
* In-person interviews should be employed, not mail shots
* WTP not WTA measures should be sought

* A test should be made of whether WTP is sensitive to the level of environmental damage

The primary advantages of the CVM over other valuation methodologies, is that it can estimate non-use values, and can be applied to a wider range of situations than other methods. For example, it can be used where market data required for the application of the TCM or HPM does not exist, or where the policy options under consideration lie outside the range of available data (Mitchell and Carson, 1989). Also, since WTP measured by CVM is a Hicksian ex-ante welfare measure of WTP, it is directly equivalent to the integral of the compensated inverse demand function (Freeman, 1986).

CVM studies are associated with a wide range of biases (Table 5.1). Although many of these potential biases can be eliminated or controlled by careful design and implementation of the CV survey (Mitchell and Carson, 1989), the method is considered unreliable by some economists for a number of reasons.

Firstly, it is difficult, if not impossible, to directly assess the accuracy of CVM estimates for non-use values since 'there appear to be no obvious or subtle behavioral trails' that can provide information about non-use values (NOAA, 1993)[2]. Secondly, in consumer theory, upon which CVM rests, it is assumed that individuals have true, but hidden, preferences for all combinations of purchased

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[2]If we could assess accuracy directly we would not need CVM

Table 5.1. Typology of Potential Response Effect Biases in CV Studies

1. ***Incentives to Misrepresent Responses***

Biases in this class occur when a respondent misrepresents his or her true willingness to pay (WTP).

A. *Strategic Bias:* where a respondent gives a WTP amount that differs from his or her true WTP amount (conditional on the perceived information) in an attempt to influence the provision of the good and/or the respondent's level of payment for the good.

B. *Compliance Bias*
   1. *Sponsor Bias*: where a respondent gives a WTP amount that differs from his or her true WTP amount in an attempt to comply with the presumed expectations of the sponsor (or assumed sponsor).
   2. *Interviewer Bias*: where a respondent gives a WTP amount that differs from his or her true WTP amount in an attempt to either please or gain status in the eyes of a particular interviewer.

2. ***Implied Value Cues***

These biases occur when elements of the contingent market are treated by respondents as providing information about the "correct" value for the good.

A. *Starting Point Bias*: where the elicitation method or payment vehicle directly or indirectly introduces a potential WTP amount that influences the WTP amount given by a respondent. This bias may be accentuated by a tendency to yea-saying.

B. *Range Bias*: where the elicitation method presents a range of potential WTP amounts that influences a respondent's WTP amount.

C. *Relational Bias:* where the description of the good presents information about its relationship to other public or private commodities that influences a respondent's WTP amount.

D. *Importance Bias*: where the act of being interviewed or some feature of the instrument suggests to the respondent that one or more levels of the amenity has value.

E. *Position Bias*: where the position or order in which valuation questions for different levels of a good (or different goods) suggest to respondents how those levels should be valued.

3. ***Scenario Misspecification***

Biases in this category occur when a respondent does not respond to the correct contingent scenario. Except in A, in the outline that follows it is presumed that the intended scenario is correct and that the errors occur because the respondent does not understand the scenario as the researcher intends it to be understood.

A. *Theoretical Misspecification Bias*: where the scenario specified by the researcher is correct in terms of economic theory or the major policy elements.

B. *Amenity Misspecification Bias*: where the perceived good being valued differs from the intended good.

1. *Symbolic*: where a respondent values a symbolic entity instead of the researcher's intended good.

2. *Part-Whole*: where a respondent values a larger or a smaller entity than the researcher's intended good.
   a. *Geographical Part-Whole*: where a respondent values a good whose spatial attributes are larger or smaller than the spatial attributes of the researcher's intended good.
   b. *Benefit Part-Whole*: where a respondent includes a broader or a narrower range of benefits in valuing a good than intended by the researcher.
   c. *Policy-package Part-Whole*: where a respondent values a broader or a narrower policy package than the one intended by the researcher.

3. *Metric*: where a respondent values the amenity on a different (and usually less precise) metric or scale than the one intended by the researcher.

4. *Probability of Provision*: where a respondent values a good whose probability of provision differs from that intended by the researcher.

C. *Context Misspecification Bias*: where the perceived context of the market differs from the intended context.
1. *Payment Vehicle*: where the payment vehicle is either misperceived or is itself valued in a way not intended by the researcher.
2. *Property Right*: where the property right perceived for the good differs from that intended by the researcher.

3. *Method of Provision:* where the intended method of provision is either misperceived or is itself valued in a way not intended by the researcher.
4. *Budget Constraint*: where the perceived budget constraint differs from the budget constraint the researcher intended to invoke.
5. *Elicitation Question*: where the perceived elicitation question fails to convey a request for a firm commitment to pay the highest amount the respondent will realistically pay before preferring to do without the amenity. (In the discrete-choice framework, the commitment is to pay the specified amount.)
6. *Instrument Context*: where the intended context or reference frame conveyed by the preliminary nonscenario material differs from that perceived by the respondent.
7. *Question Order*: where a sequence of questions, which should not have an effect, does have an effect on a respondent's WTP amount.

Potential Sampling and Inference Biases in CV Surveys

1. *Sample Design and Execution Biases*

A. *Population Choice Bias*: where the population chosen does not adequately correspond to the population to whom the benefits and/or costs of the provision of the public good will accrue.

B. *Sampling Frame Bias*: where the sampling frame used does not give every member of the population chosen a known and positive probability of being included in the sample.

C. *Sample Nonresponse Bias*: where the sample statistics calculated by using those elements from which a valid WTP response was obtained differ significantly from the population parameters on any observed characteristic related to willingness to pay; this may be due to unit or item nonresponse.

D. *Sample Selection Bias*: where the probability of obtaining a valid WTP response among sample elements having a particular set of observed characteristics is related to their value for the good.

2. *Inference Biases*

E. *Temporal Selection Bias*: where preferences elicited in a survey taken at an earlier time do not accurately represent preferences for the current time.

F. *Sequence Aggregation Bias*

1. *Geographical Sequence Aggregation Bias*: where the WTP amounts for geographically separate amenities that are substitutes or complements are added together to value a policy package containing those amenities, despite the fact that the amenities were valued in an order (for example, independently) different from the appropriate sequence.
2. *Multiple Public Goods Sequence Aggregation Bias*: where the WTP amounts for public goods that are substitutes or complements are added together to value a policy package containing those amenities, despite the fact that the amenities were valued in an order (for example, independently) different from the appropriate sequence.

goods and services based on perfect information. Individuals select purchases such that equimarginal conditions obtain (i.e. the ratios of marginal utilities to prices for all purchased commodities are equated). With some unfamiliar, environmental goods such as biodiversity, it can be argued that individuals are not fully informed, and unlikely to provide a rational purchase choice (Cummings *et al.*, 1986).

Dependence on self-reporting and the hypothetical nature of the payment question has generated considerable scepticism regarding the reliability of CVM responses (Smith, 1986). One concern has centred on the likely prevalence of strategic bidding, particularly when public environmental goods are being valued. Samuelson, (1954) postulates that respondents would misrepresent their hypothetical WTP/WTA about public goods in order to 'snatch some selfish benefit in a way not possible under the self-policing competitive pricing of private goods'. Misrepresentation could take the form either of 'free-riding' (understating WTP) or exaggerating their WTP to improve the likelihood of provision. However, there is little empirical evidence available that suggests the presence of strategic behaviour (Bohm, 1972; Mitchell and Carson, 1981; Brookshire *et al.*, 1976; Rowe *et al.*, 1980) and the concern that strategic bias might be a serious problem for CVM has diminished.

Where comparisons between real payments and hypothetical payments have been made there is a strong indication that hypothetical amounts exceed real ones (Bishop and Heberlein, 1990; Dickie *et al.*, 1987; Cummings *et al.*, 1986). However the differences are not always significant and the studies involved have not always used best CVM practice (Hanemann, 1994). No study has yet satisfactorily compared real with hypothetical payments for public goods where the major component is non-use value.

Another serious criticism of CVM is the observed difference in many empirical studies between WTA and WTP measures for equivalent welfare changes (Rowe *et al.*, 1980; Knetsch and Sinden, 1984; Huppert, 1989). In theory, if the ratio of consumer's surplus to income and the income elasticity of demand for the good is sufficiently low WTA and WTP should be broadly equivalent (Willig, 1976).

From psychology there is a growing body of experimental evidence showing that individuals do actually treat losses differently from gains. Loss aversion, together with 'overtones of involuntary exchange' associated with WTA questions, particularly in one-off CVM situations where the individual respondent is unfamiliar with having something and giving it up, are believed to generate very high WTA values (Kahneman and Tversky, 1979). Consequently, Kahneman (1986) suggests that WTA should be avoided in CVM due to the 'compensation structure' of the question.

Economists are divided on this issue and it is possible that the discrepancies between WTA and WTP measures can be explained by existing economic theory. For example, while the amounts involved may be small in relation to total income they are not small in relation to disposable income, particularly for people on very low total income (Smith, 1986). Hanemann (1991), building on the work of Randall and Stow (1980), showed that under plausible values for the income elasticity of demand for the good being valued, and the elasticity of substitution between that good and all other goods, the differences between WTP and WTA can be quite large. Also, Hoehn and Randall (1987) have argued that if risk aversion is prevalent among respondents, it is likely that in a one-off situation such as in a CVM, WTP will be under-estimated, and WTA overestimated. This is supported by experimental work by Coursey and Schulze, (1986) who found that final WTP and WTA were not significantly different amongst respondents participating in a Vickery style auction.

The prevalence of an embedding effect in some CV studies has aroused considerable controversy amongst economists (Kahneman and Knetsch, 1992). Embedding manifests itself in three ways. Firstly CVM results have been shown to be insensitive to the scope of the environmental change. One explanation, based on the Andreoni's theory of impure altruism and warm glow giving (Andreoni, 1990), suggests that individuals are expressing an egoistic satisfaction from giving, rather than expressing concern for the public good in question. If true, this would almost certainly undermine the usefulness of the method for project appraisal since CVM will provide benefit estimates that are likely to be insensitive to the scope of the project.

Sequencing and sub-additivity are further aspects of the embedding issue. Sequencing refers to the situation where the value of an environmental good depends on the order it is placed in a list of items to be valued, with the highest value achieved when it is placed first (Kahneman and Knetsch, 1992). Sub-additivity occurs where the WTP for a composite change in a group of public goods is less than the sum of WTP for the individual changes. Unlike the scope effect, both of these embedding effects can be consistent with economic theory if the goods are substitutes (Hanemann, 1994).

Although concern among the general public suggests that recovery from acidification in the biodiversity in the semi-natural environment implies that substantial non-use benefits will be generated there have been, as yet, no published CVM studies. The effects of acidification on the ecology of semi-natural areas are complex, pervasive, and occur over very long time periods, and hence pose a considerable challenge to existing CVM practice. Previous studies have therefore tended to focus on particular resources, such as fish stocks, where there is a large, use-related demand from anglers (Navrud, 1989).

5.4.4 Stated Preference (SP)

The SP method originated in mathematical psychology and was developed primarily as an applied market research tool in transport planning (Louviere, 1988) and latterly in economic valuation (Adamowiz *et al.*, 1994). SP involves presenting respondents with different sets of multi-attribute alternatives of some environmental change, including cost, and recording their preferences.

There are three main variations of the SP method where individuals are asked either to:

1. select the preferred option between paired alternatives
2. rank all alternatives in the choice set
3. rate the choices on a numerical or semantic scale.

In order to control for respondent fatigue and boredom the number of attributes used in the choice experiment rarely exceeds 5, with the total number of choices, restricted to 10 or less (Pearman, 1994).

The approach differs from CVM in that respondents are not asked directly for a monetary amount. WTP is inferred from the SP choice information about the respondents relative evaluation of the attributes described. To estimate a benefit measure requires the application of a relevant model and estimating procedure. (A common approach, based on random utility theory, uses a multinomial logit model).

SP shares some of the advantages of CVM. It can be used to evaluate environmental goods which are not marketed and can be used to estimate ex-ante welfare measures options which do not currently exist. Since respondents are more aware of individual attributes, and collinearity in data sets is avoided, there is potential scope for transferring benefits with stated preference techniques.

Disadvantages of SP relative to CVM include respondent fatigue and boredom with the process, limitations on sample size, and problems of sensitivity of the welfare measure to the experimental design used, and the attributes selected. The method is most suited to environmental changes which can easily be attributised, and careful pre-testing of the instrument is required to select appropriate attributes and to express them in a meaningful way. The basic assumption that the environment is a sum of its parts, ignoring the possibility of super or sub-additivity, is questionable. Also, respondents are forced to make choices only on the attributes of interest with little other contextual information which might be highly relevant to policy.

Applications of the SP method in the environmental field are only now emerging. For example, Adamowicz *et al.*, (1994), has applied SP to hunting in Alberta, Canada. Recovery from acidification, is not particularly suited to an SP approach because the pervasive ecosystem effects of abatement are not easily described by a small number of attributes.

5.4.5 Production Function Approach.

Production Function approaches relate changes in environmental quality to changes in production relationships for affected resources (Hanley and Spash, 1993). For firms this might mean either a reduction in market output, or an increase in expenditure on inputs to mitigate the effects of environmental degradation. An important advantage of the production function approach is that, from the point of view of the decision-maker, hard estimates of the costs or benefits of environmental change are generated. This contrasts with the somewhat 'softer' values generated by approaches such as CVM (Hoevenagel, 1994b). Two of the most widely applied production function approaches are described below.

5.4.5.1 The Avoided Cost (AC) Approach

The AC approach is based on the assumption that the value individuals place on environmental damage can be inferred from what they are prepared to spend on expenditure to prevent being harmed. Examples include double glazing to combat noise pollution, the use of inhalers by asthmatics to relieve the morbidity effects of air pollution, or bottled water to replace contaminated mains supplies.

In the case of an increase in environmental quality agents are assumed to maximise the benefits they can obtain from that improvement by reducing their consumption of defensive expenditures until the marginal costs of environmental damage equals the marginal cost of defensive expenditure and mitigated damage. If environmental quality and the avoided cost are perfect substitutes, and output is unaffected, the welfare change associated with higher levels of environmental quality can be approximated by the change in expenditure (Smith, 1991).

The approach is most useful when physical effects are well perceived and where there is a possibility of prevention (Winpenny, 1991). In general AC estimates underestimate the benefits of environmental change because, for all but the marginal user, individuals will engage in averting expenditure as long as the marginal cost is less than the marginal benefit (Hanley and Spash, 1993). The AC approach will also understate WTP if the defensive expenditures are not perfect substitutes for the reduction in environmental quality incurred (Winpenny, 1991). For example, the cost of replacing an historic building with a modern replica will not reflect the lost cultural values associated with the original! In some cases avoided cost can overstate the costs of environmental deterioration if the expenditure accrues additional benefits to the agent. For example, if double glazing is installed to reduce noise pollution, it would also lead to improved heat conservation in the home.

While the method has the advantage that it is based on actual observed behaviour it is limited to a range of marketed inputs and outputs. It cannot provide estimates for non-use values, nor can it be used in situations where there are no obvious substitutes for the environmental service: for example, the role of carbon dioxide in plant growth.

The approach has been applied to acid deposition and air pollution issues generally. Studies include effects on human morbidity and mortality (Dickie and Gerking, 1989), damage to buildings and materials (Hienz, 1994), and to forestry crops (Braat and Ruyssenaars, 1994). The application of lime to restore fish populations and protect vulnerable surface waters from further acidification is also a type of defensive expenditure. In Sweden, liming of freshwaters currently costs the government around 200 million SEK per year (SEPA, 1994).

In the UK, liming was rejected as a long term solution by the Department of the Environment because there was evidence that liming upland catchments led to detrimental effects on the natural ecosystem, including the release of toxic forms of aluminium. It also appears to offer ineffective protection under flood conditions (Woodin and Skiba, 1990). Liming is therefore a poor substitute for the SO_2 abatement, and will, on this basis, poorly represent the benefits of environmental improvement. Liming cost as a basis for benefit assessment would also ignore the value individuals may place on a more natural recovery based on reduced pollution levels.

5.4.5.2 Dose-Response Production Functions

This approach does not attempt to measure preferences for environmental change directly, but relies on a technical or biological relationship between pollution and the output level of some marketed commodity (OECD, 1989). The most applicable dose-response relationships for economic analysis are those which relate the environmental effect directly to economic outputs (e.g. timber), rather than to more fundamental changes (e.g. root physiology).

The method has been applied to a range of environmental problems including the impact of soil erosion on losses to hydro-power, irrigated agriculture and fish production in Thailand (Hufschmidt, 1986), the impact of airborne residuals on agricultural crops (Adams *et al.*, 1982; Page, *et al.*, 1982) and on forestry crops (Skeffington, 1994).

At its most sophisticated, the approach takes account of changes in production costs, supply conditions, and the demand curve for the output involved (Freeman, 1979; Freeman, 1993). However, this is usually very data demanding and a more simplistic method is often applied. The traditional model assumes that market price is unaffected by changes in output (if the environmental impact is relatively localised then this is a reasonable assumption), and the benefits (or quasi-rent) associated with environmental enhancement can be calculated by multiplying the change in output by the product price[1] (Hanley and Spash, 1993).

This simple dose-response approach involves four basic stages (OECD, 1989). These are:

i) Estimate a physical damage function of the form

$R = R(P, \text{other variables})$

where R = physical damage

P = pollution

ii) Calculate the coefficient of R on P through multiple regression analysis ie $\Delta R/\Delta P$ (Δ = change in)

---

[1] In some cases market prices do not reflect the value of output and have to be adjusted upwards or downwards. A good example are agricultural commodities in the EU which are heavily subsidised.

iii) Predict the change in pollution level due to the policy

iv) Calculate $V.\Delta P.\Delta R/\Delta P = V.\Delta R = \Delta D$

where V = Price; D = Benefit (damage avoided).

By ignoring price effects the traditional approach will normally generate upper-limit estimates of the economic benefits of environmental change (OECD, 1989). Even if there is no price effect it is possible that estimates based on loss of output will over-estimate economic damage because producers may adjust their behaviour to mitigate the environmental change (e.g. by using more resistant varieties or crops). For example, Adams *et al.*, (1982) found that estimates derived from the simple approach were 20% higher than a more sophisticated procedure which modelled the effect on the demand and supply of all crops produced, taking account of prices, and the cost of variable and fixed inputs. However, the simple, traditional method is still widely used since it has the merit of being straightforward to apply and has low information requirements.

Apart from its simplicity, the dose-response approach is also capable of providing benefit estimates when complex, often unobservable, environmental changes affect production. For example, losses in yield may be attributable to a wide range of factors (e.g. crop yields can be affected by stress factors including climate, and insect infestation, as well as pollution), and it may only be possible to link economic losses or gains through the application of scientifically complex dose-response functions (Hoevenagel, 1993b).

However, one of the problems with applying the approach to recovery from acidification is the problem of constructing reliable dose-response functions for SO_2 abatement. In the United States, for example, the National Acid Precipitation Assessment Programme (NAPAP), was unable to generate cost estimates for damage to a range of receptors due to inadequate scientific knowledge. In relation to the semi-natural environment, a dose-response approach would appear to have little practical application in any case since opportunities for marketed activities (e.g. agricultural and timber production) are extremely limited, and pollution impacts on productivity have, so far, not been detected.

5.5 Conclusions

The environmental damage caused by acidification in the semi-natural environment is characterised by complex changes in the composition of flora and fauna. The remote nature of affected areas in the Scottish uplands suggests that non-use values are likely to be important to overall benefit estimation. The Contingent Valuation Method would therefore appear to be an appropriate methodology to apply in this study.

Application of the Travel Cost Method, on the other hand, appears to be impractical to implement because acidified areas are infrequently visited and because the effect of acidification damage on recreational use is far from clear. Due to the restricted potential of acidified areas for agriculture and other productive activities application of production function approaches are likewise limited.

Salmon angling is the only major use-related activity which has been affected by acidification in the semi-natural environment. If links between water acidity, fish catch and market value can be established it should be possible to estimate the economic benefits of recovery in salmon stocks through the a variety of valuation techniques. Navrud, (1989) used the CVM to estimate the value placed by the general public on recovery in the fish stocks of southern Norway. The method is most appropriate for non-market benefits, rather than for salmon fishing rights in Scotland which are privately owned. It would be very difficult to repeat such an exercise with salmon anglers since beat owners may be unwilling to divulge information about clients, due to commercial confidentiality.

The Travel Cost Method has also been used to measure the economic value of public access fishing. For example, Mullen and Menz, (1985) used the TCM to estimate the costs of acidification to fisheries in New York state. It is most suitable for popular public fisheries where there is a clear link between travel cost and visitation rate. In the case of the salmon angling in Scotland the approach seems inappropriate for a number of reasons. Firstly, there would be difficulties with finding a sufficiently large population of anglers to interview. Secondly, since over 50% of all salmon anglers travel to fish from outwith the UK (Mackay Consultants, 1989), it is not clear how the relationship between travel cost and visitation rate could be modelled. For example, some visitors would face very high travel costs to get to Scotland, but very low daily costs to fish. How these costs are partitioned, particularly when fishing is combined with other activities (e.g. family holiday) is not obvious.

As the right to fish for salmon on rivers using rod and line (R&L), are privately owned and can be bought, sold or leased in the market place, the Hedonic Price Method would appear to have some potential. Major salmon rivers are divided into a large number of beats which are either owned by individuals, local angling clubs, or hotels and time shares. If a hedonic price function could be established to link changes in fish catch (as determined by changes in water quality), to market value it should be possible to estimate the economic benefits to the Galloway fishery as a result of SO_2 abatement.

Damage estimation involving the Hedonic Price and Contingent Valuation methods, is complicated by the complex and uncertain nature of the changes that are likely to follow abatement, and the long time periods involved. The following chapters describe the development and application of these methods using approaches which attempt to overcome these difficulties. Chapters 6, 7 and 8 deal with the Contingent Valuation Method, while Chapters 9 and 10 are concerned with the application of the Hedonic Price method to salmon fisheries.

CHAPTER 6 THEORETICAL FRAMEWORK FOR CVM STUDY

6.1 Introduction

Ecosystem recovery in the Scottish uplands will involve complex environmental changes over a very long time period. As yet, neither the extent of the environmental gain expected from SO_2 abatement, nor the rate at which the environmental recovery will take place are known with certainty. Acidification of the semi-natural environment therefore presents an opportunity to test the extent to which CVM can produce reliable WTP amounts consistent with theoretical expectations regarding non-use benefits, scope effects, and valuation in a risky world (uncertainty). Many environmental issues share these characteristics (e.g. global warming), and the ultimate usefulness of CVM to economic analysis will depend on the extent to which it can provide reliable benefit estimates under these circumstances.

The aims of this CVM study are twofold. Firstly, to produce benefit estimates for SO_2 abatement for the most plausible environmental recovery scenarios and, secondly, to assess the extent to which CVM is sensitive to the scope of environmental recovery and the presence of scientific uncertainty.

6.2 Theoretical Framework

Individuals can be expected to value environmental improvements in the Scottish uplands for a number of motives. These include (i) an improvement in the quality of recreational experience for those currently using upland areas directly (walkers, fishermen, ornithologists, etc), (ii) the potential for such use for those not currently using the area, (iii) an increase in satisfaction on the part of those who currently, and in the future, will only use the area indirectly (e.g. by reading about the area, or watching TV programmes); and (iv) an altruistic motivation to bequeath a particular level of environmental quality for future generations. Motives (i) and (ii) lead to a direct use value; motive (iii) to a passive use value, and motive (iv) to a pure existence value. With CVM, unlike other valuation methods, it is possible to measure all of these potential benefits of abatement.

6.3 Scope Effects

A basic axiom of economic theory is that, unless the individual is satiated, he/she will prefer more of an environmental good, and less of an environmental bad (Diamond, 1993). The NOAA report (1993) states that for a CVM study to be considered reliable, it should be able to demonstrate this scope effect. Although WTP estimates from a range of studies have varied considerably with the level of environmental change (Walsh *et al.,* 1992; Carson and Mitchell, 1993) and there remains a general concern that CVM results do not vary sufficiently with scope (Hanemann, 1994; Diamond and Hausman, 1994).

In the context of SO_2 abatement individuals can be expected to place a greater value on an abatement programme which avoids higher levels of environmental damage in the future, or which leads to higher levels of environmental recovery. Also, assuming positive time preference, individuals would be expected to prefer a programme of abatement which led to environmental benefits sooner rather than later.

In order to assess the sensitivity of the CVM estimates of WTP to environmental quality and recovery time three propositions are to be tested. These are described below.

6.3.1 Sensitivity to Recovery Level.

For a policy that ensures a higher level of environmental quality than possible under the *status quo*, with certainty, the maximum payment that the individual would offer to secure the policy is defined as:

$$V(p,\ z^1,\ y - WTP_j\ ) = V(p,\ z^0,\ y)$$

where p is a vector of prices of private goods,

y is fixed annual income.

z^1 is the steady state level of environmental quality under the status quo

z^0 is the alternative, higher, steady state level of environmental quality available following abatement

WTP_j is the maximum WTP which the individual would give up to pay for abatement in order to restore him/herself his/her original utility level before abatement given the increase to a higher level of environmental quality.

Assuming individuals prefer more environmental recovery to less one would expect:

$$WTP_a > WTP_b > WTP_c \qquad \textit{Proposition 1}$$

where *a, b,* and *c* are, respectively, pristine, intermediate, and no recovery from 1990 levels of damage.

6.3.2 Sensitivity to Damage Level Under the *Status Quo*

Assuming less environmental damage is preferred to more, one would also expect:

$$WTP_x > WTP_y \qquad \textit{Proposition 2}$$

where x, y are, respectively, depictions of realistic low and high levels of damage under the *status quo*. WTP is the maximum WTP which the individual would give up to pay for abatement in order to restore him/herself to his/her original utility level prior to abatement, given the decrease to a lower level of environmental quality under the status quo.

6.3.3 Rate of Recovery

The timing of recovery to the steady state which might result from reduced deposition is not known at present. However, given positive time preference, individuals will prefer a more rapid recovery to a slower one. The following proposition can therefore be assessed:

$$WTP_r > WTP_t \qquad \textit{Proposition 3}$$

where r is recovery time = 20 years; t is recovery time = 120 years. These time intervals represent the range in recovery periods suggested by current scientific research (see section 4.4).

6.4 Uncertainty

As is implicit above, considerable scientific uncertainty exists as to both the level of recovery to steady state conditions following abatement, and the future level of damages under the *status quo*. Whether society wishes to undertake a risky activity, (in this case abatement), is a normative question which requires the expected benefits and costs of a risky activity to be compared.

While the CVM can evaluate this uncertainty directly, most CVM applications ignore risk by placing the outcome of an environmental programme in a riskless world. If respondents are aware of this uncertainty, then the CVM researcher will not be estimating the appropriate welfare measure (Segerson, 1986; Johannson, 1990).

6.6.1 Risky Recovery

Following Johansson (1988), the CVM instrument will be used to estimate abatement benefits when a risky recovery (environmental gain) is proposed. In one subsample, respondents are informed that there is a 50% chance that recovery will achieve pristine conditions (level a) and a 50% chance that recovery will not exceed the current level of environmental quality (level c).

If individual agents are risk-averse[1] (that is they would prefer the certainty of receiving the expected price of a lottery rather than the lottery itself), we would expect them to prefer a project with a certain outcome lying midway between a and c. In this case outcome b rather than a stochastic programme with b as the expected outcome. Hence:

$$WTP_b > AP \qquad \textit{Proposition 4}$$

[1] Under the Expected Utility Approach (EUA), the standard preference model economists use to describe decision making under risk (Querner, 1994), individuals are conventionally assumed to be risk averse

where WTP_b is the willingness to pay for certain recovery to level b (intermediate) and AP is an *ex ante* equivalent surplus WTP measure for uncertain recovery. If respondents are on average risk seeking, (i.e. has an indirect utility function that is strictly convex in z and y) the AP measure would be expected to exceed WTP_b.

6.6.2 Risky Damage under the *Status Quo*

In the above, the *status quo* level of damages was considered to be certain. In the final proposition WTP was estimated where there was uncertainty with respect to the steady state damage level under the status quo. In one sub-sample, individuals are informed that there was a 50% chance that minimal damage (level x) and a 50% chance that high damage (level y) will occur by some specified date in the future. The respondents in the other sub-sample are informed that an intermediate level of damage (w) is certain. In both cases, respondents are told that the recovery to pristine conditions (level a) is assured. Assuming risk aversion we have our final proposition:

$WTP_w > AP$ *Proposition 5*

where WTP_w is willingness to pay recovery to level c when intermediate damage is expected with certainty under the status quo. The AP is the ex-ante equivalent surplus WTP measure when the status quo level of damage is uncertain. As above, if respondents are risk loving, on average, the AP measure will exceed WTP_w.[2]

6.5 Overall Design

The design of the scope tests is presented in Figure 6.1. All together there are 15 damage/recovery scenarios, 13 of which relate to certainty concerning future gains and losses. In the next chapter the development and implementation of this design through a mail questionnaire is described.

[2]While probabilities over damage/recovery can be numerically quantified for respondents, the challenge with Propositions 4 and 5, will be to quantify the 'amount' of biodiversity gain that is achieved in each case.

Figure 6.1 Scenarios used in CVM survey

With Certainty

Recovery Level (Recovery time=20 years)

Damage Level

	Pristine(a)	Intermediate(b)	None(c)
low (x)	•	•	•
high (y)	•	•	•

Recovery Level (Recovery time=120 years)

Damage Level

	Pristine(a)	Intermediate(b)	None(c)
low (x)	•	•	•
high (y)	•	•	•
intermediate	•		

With Uncertainty

Risky Recovery (Recovery time=120 years)

	Recovery = 50% pristine/50% none
Certain Damage	
high (y)	•

Certain Recovery (Recovery time=120 years)

	Pristine (level a)
Risky Damage	
50% low/50% high	•

CHAPTER 7 DEVELOPMENT AND IMPLEMENTATION OF THE CVM QUESTIONNAIRE

7.1 Introduction

Since the areas affected by acid deposition in the semi-natural environment are very remote, and only infrequently visited by the general public, the benefits of recovery are likely to be dominated by non-use values. Up until the mid-1980's the CVM was considered to be too unreliable for evaluating non-use benefits. Indeed, several of Cummings *et al.*,'s (1986) Reference Operating Conditions (ROCs) for the CVM virtually ruled out its application to non-use values on the grounds of unfamiliarity (ROC 1), and previous valuation experience (ROC 2). Up until the mid 1980's applications were typically restricted to estimating user benefits from environmental resources (e.g. recreation and hunting).

Many of the ROC's of Cummings *et al.*, now seem unnecessarily restrictive. Freeman, (1986) for example, suggests that in instances where people do not know their preferences for non-use type goods, the errors associated with a lack of familiarity would generate WTP estimates that were imprecise, but not biased. Secondly, drawing on psychological research, Hanemann, (1994) argues that cognition can be a constructive process, and rational choice is not necessarily dependent on familiarity or experience.

Respondents to a CVM questionnaire construct their memories, attitudes and judgements, and tend to make bottom-up decisions. Over the past ten years considerable effort has, therefore, been invested in improving the design, implementation, and analysis of CVM studies of non-use environmental goods. In 1993, the *'blue ribbon'* NOAA panel gave the method qualified support for litigation involving non-use values. However, application of CVM to test issues of scope and uncertainty about environmental change were acknowledged by the panel to be an important aspect in the search for reliability.

Testing for scope and uncertainty with respect to recovery from acidification presents particular challenges for CVM. Firstly, there are difficulties with presenting changes in environmental quality to the general public which are likely to be pervasive, complex and subtle. Secondly, the survey instrument must overcome a degree of unfamiliarity not normally experienced with CVM applications to environmental valuation. For example, a recent survey found that less than 10% of Scottish residents 'knew a great deal' about acid rain and its effects (Government Statistical Service, 1991). Because of this low level of awareness about acidification damage, respondents can be expected to depend, to a very great extent, on the information set presented to them in the CV instrument. In this chapter the development and implementation of a CVM questionnaire to meet these challenges is described.

7.2 Method of Implementation

Prior to the development of the questionnaire a decision had to be made regarding whether the survey would be implemented by mail, by telephone, or with in-person interviews. This is because the method of implementation dictates the style and content of the questionnaire to a considerable extent.

The NOAA panel recommended that mail surveys should not be used because they were considered to be less reliable due to problems with self-reporting, and because low response rates, below 50%, which can introduce non-response bias, are frequently encountered. However, mail surveys remain popular because they can be executed at much lower cost than other methods. They also permit respondents a considerably greater amount of time to respond to the questionnaire, which is important when dealing with a complex subject such as acid rain. Also Whittington *et al.*, (1992) considered that respondents given more time to respond to the payment question were more likely to consider their budget constraints and competing needs. Mail questionnaires are not prone to interviewer bias and simple random sampling is possible. (In-person interviews usually require a careful stratification, such as clustering, to reduce costs and to ensure a representative sample.)

After consideration of all these factors, particularly the expenditure involved, it was decided to send the questionnaire to respondents by post. In order to ensure that the CVM questionnaire was easy to understand and complete considerable care was taken in question design and presentation of the environmental change.

7.3 Development of the CVM Questionnaire

The basic objective of a CVM questionnaire is to create a hypothetical, but realistic market where individuals are given the opportunity to express a monetary preference for environmental goods. The questionnaire is composed of four different elements:

1) a description of the environmental change,
2) a description of the method of payment,
3) a definition of the market, and
4) questions for validating WTP responses.

In this section the development of these four components of the CVM questionnaire used in this study is described.

7.3.1 Definition of Environmental Change.

Describing changes in the semi-natural environment following abatement to the general public is extremely difficult because the changes are likely to be complex and because respondents are unlikely to be well informed about acid deposition. Considerable effort was therefore spent on developing an information set which would convey the nature of the environmental damage and recovery path from acidification in a simple, but accurate fashion. Following Fischoff and Furby (1988), the CVM questionnaire aimed to provide precise information on the 'substantive' and 'formal' definition of the environmental change.

7.3.1.1 Substantive Definition

The substantive definition involves a description of the attributes of the environment which have been affected. For recovery from acidification in the semi-natural environment these effects are complex and multi-scaled, ranging from the unobserved (e.g. reduced pH), the barely perceptible (e.g. changes in micro flora assemblages), and the more readily apparent or dramatic (e.g. loss of fish populations). The degree of certainty surrounding the effects of acidification on different receptors also varies (see chapter 4 for more details). Three approaches to defining the substantive change were initially considered. These were:

OPTION A

All of the relevant attributes are described using full graphical and textual information (e.g. pH levels, aluminium concentrations, effects on individual species etc.).

OPTION B

A simplified attribute list including only the most obvious effects, for example the effects on *charismatic* species such as fish, trees and birdlife. This approach has the advantage of focusing on attributes which people are most likely to care about or associate with. It can also be justified on the grounds that, ultimately, all changes in the semi-natural environment (e.g. on soil and water chemistry) will eventually be manifested in changes to the higher trophic levels of biota.

OPTION C

Dividing the change in environmental quality into groups of attributes and valuing each group separately. For example, by separately valuing recovery in the terrestrial and aquatic ecosystems, or by focusing on individual species. This approach simplifies the substantive definition, but would involve a number of different CVM surveys and, due to substitution effects, is likely to cause a problem with double counting (Diamond, 1993).

Due to the potential difficulties associated with Option C it was decided to develop formal definitions of environmental change based on Options A and B only.

7.3.1.2. **Formal Definition**

The formal definition of the change involves describing the reference (without project) and target (with project) level of each affected attribute, the geographical extent of the environmental change, the time period involved, and also the probability that the changes will actually take place (in either reference or target states).

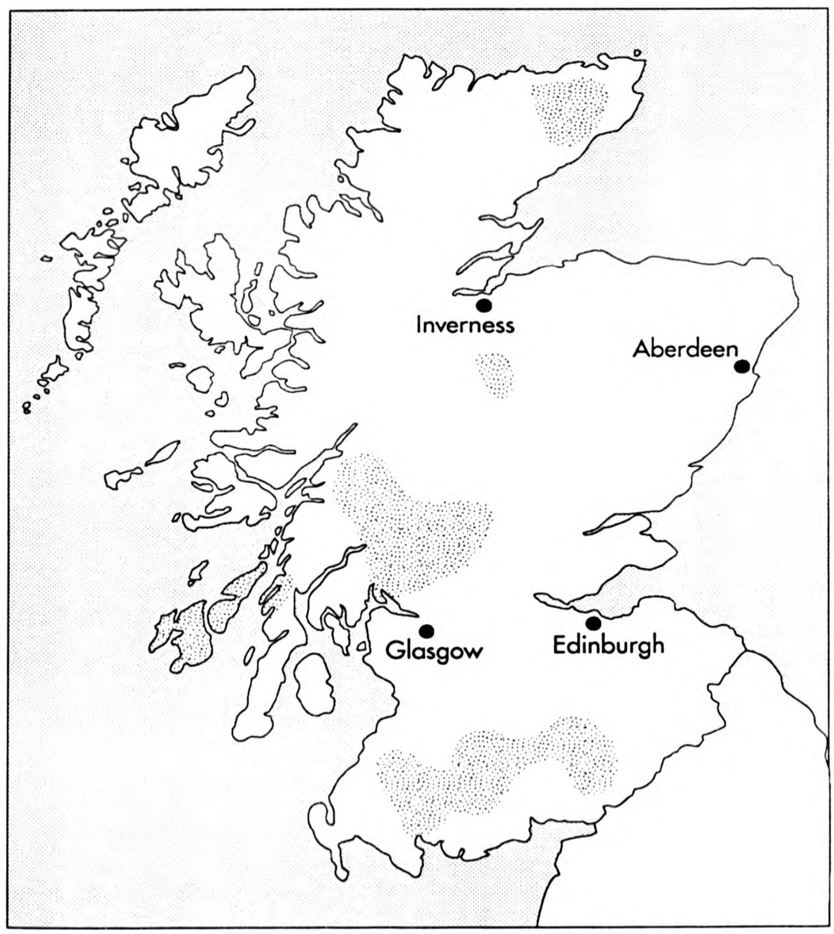
With the exception of geographical extent Propositions

1 to 5, set out in Chapter 4, provide the basis for testing the sensitivity of the CVM instrument to variation in these facets of the formal definition. Three target levels following recovery (levels a, b, and c), and two reference levels of damage (x and y) were defined in Propositions 1 and 2. Proposition 3 examines sensitivity to two different time periods for recovery (r and t) and Propositions 4 and 5 address uncertainty.

Geographic extent was not included because it was considered to be the one aspect of the formal definition where it could be assumed that there was a reasonable degree of certainty. Critical Load maps provide a reliable scientific base upon which to predict the effect of SO abatement on the extent of acidification damage in Scotland. A simplified version of the Critical Load map for Soils in Scotland (Figure 4.4), which illustrates the area affected by acidification under the current reduction plans was used. This is presented in (Figure 7.1) and, in accordance with a recommendation from the NOAA report (1993), shows the extent and location of undamaged substitute sites (unmarked areas).

Considerable thought was given to the presentation of the environmental change scenarios. Carson and Mitchell (1989) emphasise the need to describe differences in the formal definition of environmental change (levels) which are meaningful to respondents. Under Option A, the description of the reference and target levels required a considerable amount of textual information, supplemented by graphical

Figure 7.1: Illustration showing area damaged by acidification in questionnaire



depiction of the change in environmental quality. This information is presented in Figure 7.2 on pages 138-139.

Under Option B, it was decided to describe changes in environmental quality using pictorial information. 'Species boxes' were used to convey changes in species composition and population levels in a quantitative, yet interesting and stimulating style (Figure 7.3). Historic levels of environmental quality and predicted future environmental level under the *status quo,* were indicated by an individual species box. For those scenarios where risk regarding future damage was investigated, uncertainty was indicated by a question-mark between alternative levels of future damage (Figure 7.4). The full range of scenarios used in the study are presented in Appendix 1.

7.3.2 Description of the Payment Method.

A substantive and formal definition is also required to describe the method of payment. The substantive definition describes the payment vehicle, (i.e. how the payment will be levied, for example through higher taxes and prices, or admission fees) and the people who must pay (e.g. taxpayers, users, etc.). The formal definition identifies the reference level (i.e. the current level of disposable income) and the target level (i.e. the maximum willingness to pay/minimum compensation received).

The choice of payment vehicle is important for two reasons. Firstly, an appropriately chosen payment vehicle can enhance the realism of the hypothetical market.

Figure 7.2: Description of environmental change: Option A

In these areas the quality of the environment will continue to deteriorate and it is expected that over time the numbers and diversity of species will decline to levels much lower than existed before pollution. Many animals and plants will have difficulty reproducing and maintaining a healthy population. Salmon and trout stocks will become very rare and fisherman will no longer be able to catch any fish. Woodland trees will continue to lose their leaves and many will eventually die, leaving only scattered individuals.

Please look at Chart 1 below.

CHART 1

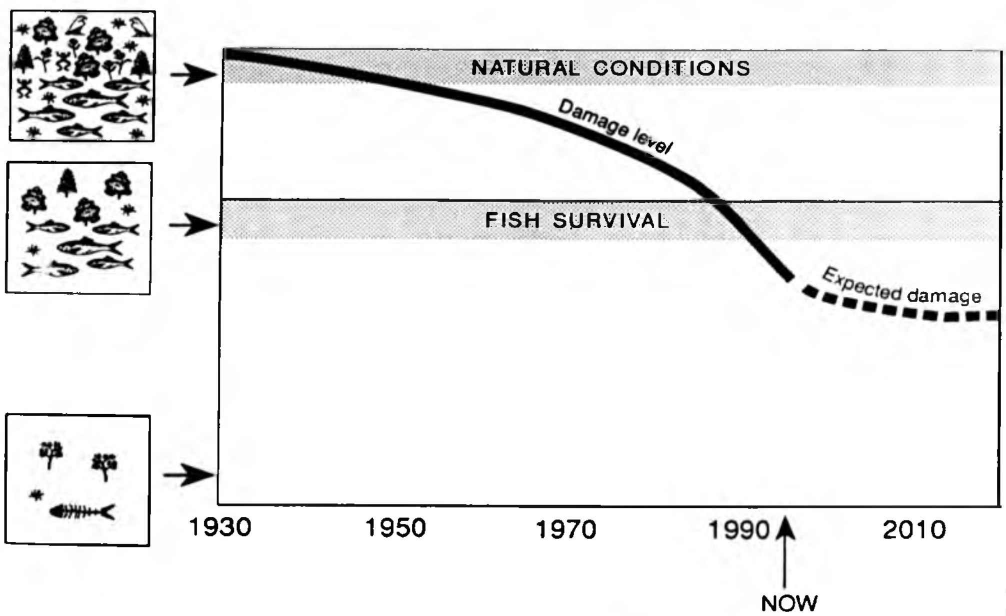


Figure 7.2 (cont.): Description of environmental change: Option A

In order to protect these areas from further damage and allow the environment to recover further cuts will have to be made. While these new cuts in **Acid Rain** will cost money, scientists expect the environment will recover fully so that all populations of plants, insects and animals will return to levels found under natural conditions prior to industrial pollution.

Full recovery is expected to take 20 years. **Please look at Chart 2 below.**

CHART 2

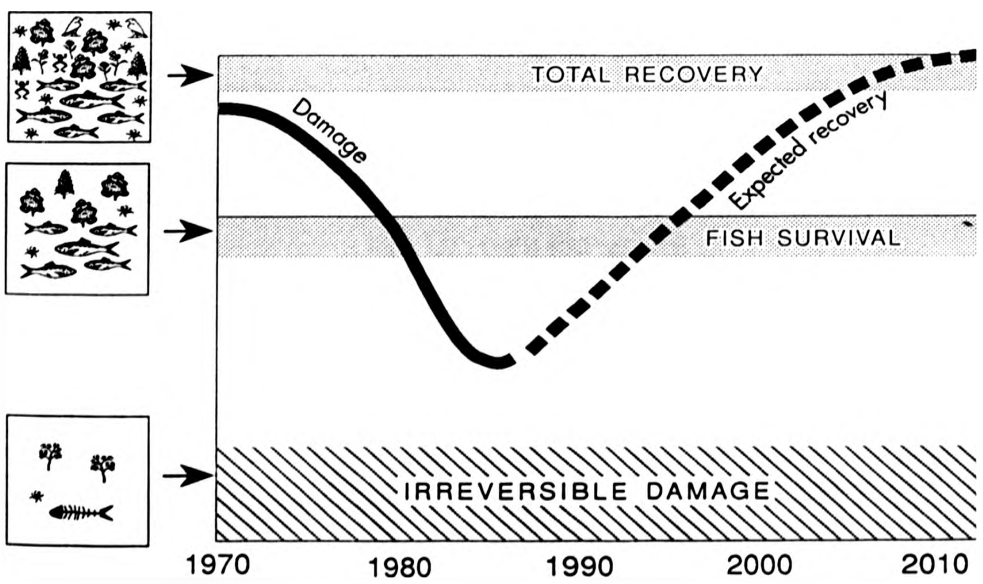


Figure 7.3 Description of environmental change (with certainty): Option B

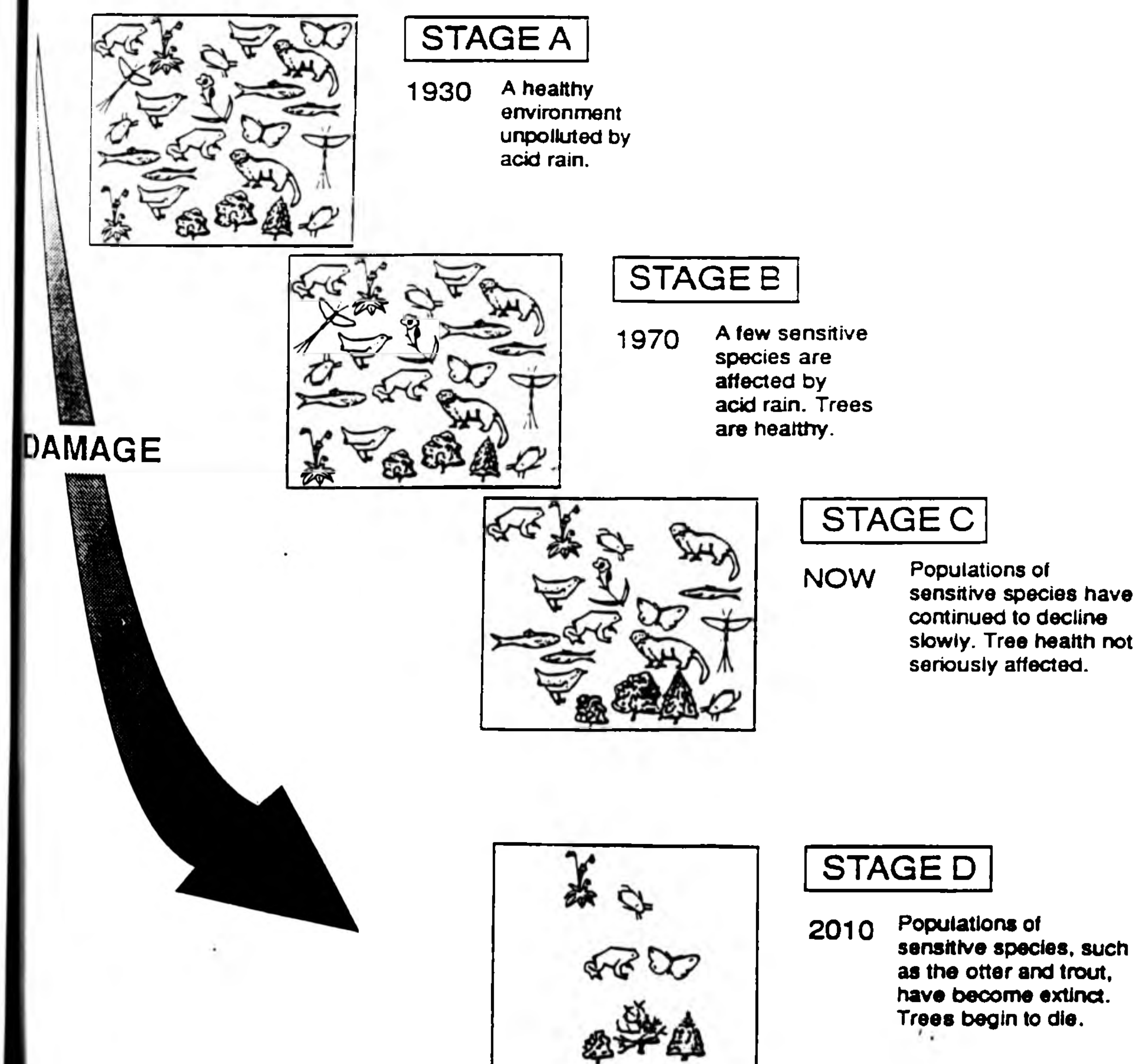
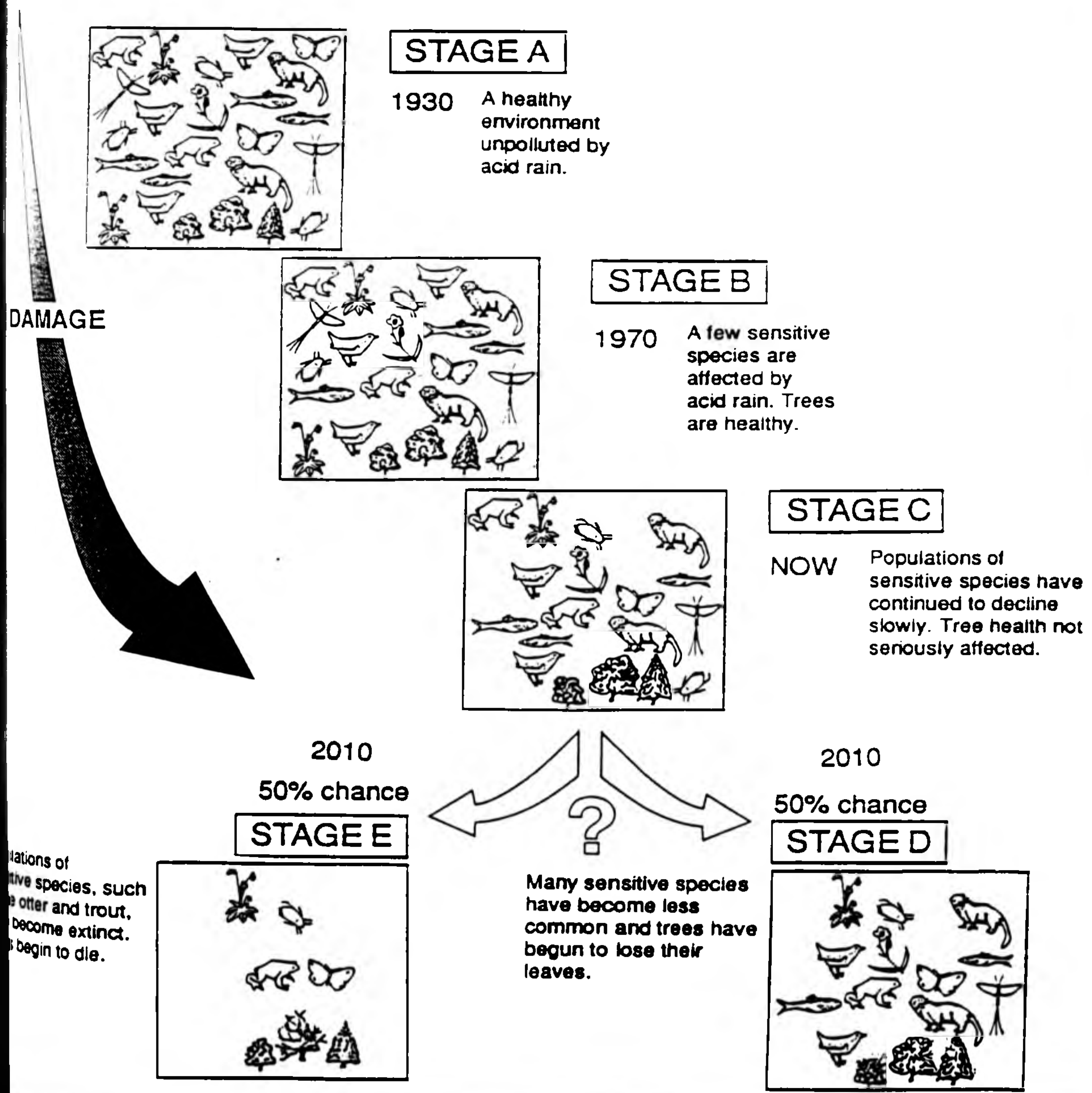


Figure 7.4 Description of environmental change (with uncertainty): Option B



Secondly, WTP is known to differ across payment vehicles (Cummings *et al.*, 1986)[1]. Certain payment vehicles are less desirable than others. For example, it is possible that the payment vehicle can give rise to a relational bias. Relational bias occurs where respondents base their valuation on a 'typical' price for some comparable good rather than expressing their maximum WTP. Charitable donations and entrance fees are examples of payment vehicles where this may occur (Hoevenagel, 1994a). A donation to charity is also unsuitable since it may imply a contribution to a good cause and encourage warm-glow effects (Seip and Strand, 1992; Hanemann, 1994).

After screening for biases, the choice of payment vehicle is a pragmatic one. The payment vehicle that is most relevant to the environmental change in question, which would likely be used if the change is actually implemented, and which discourages protests bidding should normally be adopted (Hoevenagel, 1994a). In this study it was decided to use higher prices on a wide range of commonly purchased consumer items (e.g. electricity, cars, and central heating) as the payment vehicle. This was felt to be a realistic option since prices would be expected to rise following stricter pollution control.

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[1]As Randall, (1986) points out there is nothing irrational about WTP varying across payment vehicles, rather it is an expected and valid determinant of WTP.

responsible for the pollution. This shared-responsibility also enhances the link between intended and actual behaviour, and therefore improves reliability. Bishop and Heberlein, (1986) suggest that respondents who are encouraged to accept personal responsibility for environmental degradation, have a stronger tendency to turn intentions into acts. It was also made clear that all households would be required to pay something. This was regarded as fair, in the sense that it reflected the 'polluter-pays' principle, and would reduce the possibility that respondents would blame other groups.

Respondents were asked if they would be willing to pay a specified rise in prices on an annual basis. WTP creates a 'purchase structure' for the environmental good and is generally preferred over the 'compensation structure' of a WTA scenario (section 5.4.3). In order to enhance realism, respondents were reminded of their budget constraint (Burness *et al.*, 1983), and were asked to specify which budget category they would reduce spending on to pay for the environmental good.

The choice of payment elicitation method used is an extremely important one since each method is associated with types of bias and other problems. Table 7.1 briefly reviews the suitability of the most commonly used elicitation methods. For this study it was decided to use the discrete choice (DC) method. In a DC contingent valuation study, each individual in the sample is offered a bid, and the respondents must state whether they would be willing to pay

Table 7.1 Review of elicitation methods

Bidding Game

Some of the earliest CVM applications used a bidding game format (Davies, 1963; Randall et al., 1974). It involves face to face interviews with the interviewer starting with either a low bid and working upwards or a high bid working downwards. The game terminates when the respondent indicates he/she would not pay more/less for the good than the bid offered. It is considered to be an effective method because it allows people, placed in the unfamiliar context of a hypothetical market, the opportunity to 'learn' their WTP. However, it has to be implemented using in person interviews, and great care is required to prevent boredom setting in. Also, the bidding process runs the risk of upwardly biasing WTP through unintentional 'arm twisting' by the interviewer (Cummings *et al.,* 1986) and the initial starting bid may suggest the range of the 'appropriate' bid for the good in question thus creating a starting point bias (Boyle *et al.,* 1984).

Payment Card

First used by Mitchell and Carson, (1981) and seen as one way to get around starting point bias it can also be used in postal surveys. It involves asking the respondent to circle, from a set of alternative values printed on a 'payment card', the amount which comes closest to his/her maximum WTP. Prone to anchoring bias with upward rounding a prevalent strategy (Kahneman, 1986).

Open-Ended (OE)

Individuals are asked to state their maximum WTP for the environmental change in question. The primary advantage of the OE method is that the estimation of mean WTP is straightforward. However, the format is considered to be mentally demanding and the payment question is often left unanswered (Desvousges *et al.,* 1983). Opportunities also exist for strategic bidding (Randall, 1986).

Discrete Choice (DC)

First used by Bishop and Heberlein, (1979) this method involves offering respondents a single bid and asking them if they are prepared to pay it to obtain the described environmental improvement. Although each respondent is offered a single bid level, different respondents are offered different bid levels, according to some design. McConnell, (1990) showed how the different approaches to the

7.1 (contd.) Review of Elicitation Methods

analysis of binary response data to estimate is compatible with economic theory. One strength of the method is that it asks people to make a decision relative to a price and in this respect it reflects behaviour in every day life. Seller *et al.*, (1985) found that respondents considered their own DC bid responses more accurate that estimates from the OE method. The method is also incentive compatible since it is in the individual's own best interest to respond truthfully (Randall, 1986). Although strategic bidding in this sense is not considered to be a major problem, DC values typically exceed OE values. Yea-saying, that is when respondents accept the offered bid level even when true WTP falls below this amount, is believed to bring this about. This bias can occur if there is preference uncertainty, a desire to please the interviewer, or to register support to the environment.

Referendum

A version of the DC approach, the Referendum format was favoured by the NOAA panel (1993). It sets the payment question in the context of a political market, by asking respondents if they would be willing to vote for some policy with a specified cost. The referendum approach is intuitively appealing for public goods since they are normally delivered by government. In some states, such as California, referendums are commonly used for public projects and the approach has obvious potential. However, referenda are rare in many other political systems. In Scotland, for example, referenda have only been carried out on two occasions and cannot be considered familiar to most members of the public.

Double-bounded DC

This approach has also developed from the DC method and involves elements of a bidding game. Respondents are asked two DC payment questions with a second bid, lower or higher than the original bid (depending on the initial response) is offered. Although favoured by many CVM researchers over the single-bounded version since it is statistically more efficient and therefore cheaper to implement (Hanemann *et al.,* 1991) it is suspected of encouraging upward bias through yea-saying, and is suited best to in-person interviews.

that sum to achieve a specified level of environmental improvement.

Hanemann, (1984) showed how DC data could be analysed using logit models to provide welfare measures consistent with the hypothesis of utility maximisation. The crucial assumption is that although an individual knows his utility function with certainty, it contains components which cannot be observed by the investigator and which can be treated as being stochastic. This random error provides the structure of the binary response statistical model. Hence, an individual will accept the bid if

$$v(h_1, y - A; s) + e_0 \geq v(h_0, y; s) + e_1$$

where h_1 = higher level of environmental quality

h_0 = current level of environmental quality

y = income

A = Payment level

s = vector of socio-economic varaibles

e_0 and e_1 = random error variables with zero means.

Although the design of the bid distribution across respondents is relatively complex and estimation of mean WTP is complicated, the DC approach offers some important advantages. The NOAA report recommends the Discrete Choice (DC) approach on the basis that it is incentive compatible, since it is in the individual's own best interest to respond truthfully (Randall 1986), and because it presents the respondent with an easy and familiar purchase decision. Unlike the double-bounded approach it can also be readily implemented in a postal survey. As recommended by the NOAA panel, it was decided to introduce a 'Don't Know' option so that respondents who are unsure, either of their preference, or about the questionnaire (confused) are not 'forced' to declare a preference.

Immediately prior to the DC payment question a screening question was also introduced. Respondents where asked if they would be willing to pay anything, even a very small amount, toward reducing acid deposition. This question has several main functions. Firstly, it can be used to identify protest responses; that is individuals who may have a positive preference for the environmental change, but who are not prepared to pay anything because they object to one or more aspects of the questionnaire. Secondly, for statistical reasons it is necessary to screen out genuine zero bids from other non-protest responses. Also, since it gives the respondent an opportunity to register support initially for the environmental project, it may help to reduce yea-saying (see Table 7.1).

7.3.4 Validation Questions

Since there are no market data for non-use acidification damages to compare with the hypothetical payments solicited through the CVM, some form of internal validation is required. This can be achieved by introducing behavioral, attitudinal and socio-economic questions to the CVM questionnaire. Answers to these questions can then be used to explore the extent to which the response to the payment question is related to variables one would expect to influence WTP from economic theory, and to assess the internal consistency of responses so that some confidence can be acquired that the *'answers corresponded to some reality'* (NOAA, 1993). Validation questions, if they are relevant to the environmental change of interest and are imaginatively designed, can also stimulate the respondent to think more carefully about the payment question, and help to maintain interest.

In this questionnaire twenty validation questions were incorporated. These ranged from standard questions on income, age and education, to questions which attempted to identify levels of concern for environmental issues, including acid rain, and behavioural evidence for environmental altruism (e.g. recycling, membership of environmental organisations etc.). The NOAA panel also recommended that questions which ascertain the motives behind responses to the payment question are incorporated. For Yes responses, this question provides some indication of the motive for paying, while also helping with the identification of protestors among those refusing the offered bid. These questions were inserted immediately following the payment question in the questionnaire.

7.4 Testing the Questionnaire

Considerable effort was spent on testing the questionnaire in order to ensure that the information set and market context would allow respondents, unfamiliar with the commodity and inexperienced at valuation in a hypothetical situation, to formulate their WTP. The testing process involved five stages.

7.4.1 Discussions with MLURI Scientists

These discussions were concerned with the veracity of the environmental change scenarios depicted in the questionnaires. The scientists are all currently involved in MLURI's research programme on acidification processes in the semi-natural environment. Options A and B were presented to each scientist. Although generally happy with the broad interpretation of acidification damage and recovery in both options, the scientists thought option A, which relied on more substantial graphical and textual information about the environmental change, was more scientific, and hence preferable.

7.4.2. Focus Groups

Two focus groups involving people less familiar with acid rain were convened in August 1993. Each group was comprised of 5 persons drawn from the non-scientific staff at MLURI, and from their friends and relatives. Each meeting lasted for approximately one hour and involved a discussion of the questionnaire led by myself. All of the main comments were recorded and evaluated with regard to making improvements to the questionnaire.

Focus groups permit in-depth exploration of the cognitive responses of individuals, and can highlight aspects of the information survey which are unclear, or are misleading. Of particular concern were 'framing effects', that is where unexpected interpretations are made with regard to wording and visual aids (Schuman and Presser, 1981) or problems with amenity mis-specification bias (Mitchell and Carson, 1989).

In contrast to the MLURI scientists, Option B was favoured by participants in both focus groups. Some members had difficulty comprehending the graphical presentation of change in recovery level in option A, and preferred the use of simple statements supported by the depiction of changes in species diversity using the 'Species Boxes'. Minor changes in phraseology and presentation of several questions was also made on the basis of comments received. Option A was discarded at this stage.

7.4.3. Individual Interviews

The revised questionnaire, based on option B was then presented to individuals to complete in my presence. Whereas the focus groups were primarily aimed at assessing respondent cognition with respect to the environmental change, the individual interviews were intended to assess the ease with which the questions could be answered, and the overall time taken. Fifteen interviews were carried out covering each one of the fifteen scenarios to be used in the study. The interviews were organised in three sets of five, so that revisions could be incorporated at the end of each set.

As respondents completed each question they were encouraged to 'think aloud' and report their thoughts and difficulties about each question (Willis *et al.*, 1991). Non-verbal reactions were also observed, such as hesitations and facial expressions, to establish where difficulties lay. Probing questions were asked after each question to gain an appreciation of whether the written response actually represented the perceived response (i.e. they reported what they actually intended to report). Since the survey was to be implemented by mail, advice or comment from myself was kept to a minimum during completion of the questionnaire.

This was a particularly important stage in the testing of the questionnaire since it gave a good indication of the questions or passages of text which tended to involve greatest mental effort. Apart from minor problems with several questions the main difficulty lay with the length of time taken to complete the questionnaire with some individuals taking more than 25 minutes. The text was therefore substantially reduced, without leaving out pertinent information, in order that the questionnaire could, on average, be completed in 15 minutes. Other changes were quite subtle (placing commands in bold for example), but greatly improved the flow of the questionnaire.

7.4.4 Small-Scale Piloting

Prior to a full scale piloting of the questionnaire with a random sample of the general public, a small piloting exercise was organised at the Universities of Stirling and Aberdeen. Graduate students in environmental management were asked to complete the questionnaire at home and to add written comments or criticisms where necessary. Few problems emerged during this stage, apart from minor grammatical errors, and the questionnaire was therefore prepared for a full scale pilot survey.

7.4.5 Mailed Pilot Survey

A mail pilot survey was carried out on the general public in September 1993 in order to test for survey design flaws and establish the response rate from a mail shot. An open ended elicitation method was adopted for the pilot. This was to done to aid bid design in the main survey by providing data on the mean and distribution of WTP values among the general population.

In total, 254 respondents were selected from a systematic sample of Scottish households listed in the most recent telephone directory. A response rate in excess of 50% was achieved and there did not seem to be any significant problems with item non-response or protesting. The results of the pilot survey are presented in the next chapter, together with the optimal bid distribution which was subsequently generated using Cooper's optimal design method (see section 7.5.3 below).

7.5 Sampling

The objective of the study was to estimate the aggregate WTP of Scottish residents for recovery from acidification in the semi-natural environment. The aim of the sampling design was therefore to produce an accurate, precise estimate of mean household WTP which could be used to generate an unbiased aggregate figure for Scotland.

7.5.1 Sampling Frame

The sample was drawn from residential households listed in the telephone directory. Although telephone directories are considered to be incomplete and therefore biased, (for example some very poor families may not have telephones), it is probably the most comprehensive national listing publicly available. British Telecom confirmed in a telephone conversation, that they estimated that upwards of 90% of all Scottish households were listed in the telephone directory (British Telecom, pers. com.). A systematic sampling procedure was employed with one household selected from the same position from every fourth page and entered into a computer database. The sample was then randomised using a random number generator.

7.5.2 Sample Size

To obtain an acceptable degree of precision for mean WTP from DC studies very large sample sizes are required because of the large variance in the WTP responses (Mitchell and Carson, 1989). The decision was taken therefore to sample as many households as the budget for the project would allow. This was calculated to be 3000 questionnaires, distributed evenly between each of the 15 scenarios.

7.5.3 Bid Distribution

In a DC contingent valuation study, each individual in the sample is offered a single bid level. However, this bid level is varied so that each respondent receives a different bid amount. Cooper, (1994) has developed a model which predicts the sample design for any specified total sample size (N) which minimises the mean square error of the welfare measure. Cooper's optimal design model uses open-ended pilot data, and specifies the number of bid amounts (m), the actual bid level values ($b_1 \ldots b_m$), and the sample size corresponding to each bid value ($n_1 \ldots n_m$). As such it is an improvement on other, more *ad hoc* approaches, based on informal criteria. For example, Duffield and Patterson's, (1991) approach which determined the optimal allocation of the total sample size among the different bid amounts but could not generate the bid vector, and that of Bergland *et al.,* (1987) which selected bids using equal log-linear increments.

7.6 Implementation

The biggest concern regarding implementation through a postal survey was the overall response rate. In order to ensure a high response rate Dillman's, (1979) total design method for mail surveys was used. This approach involves meticulous attention to presentation of the questionnaire and adoption of an intensive follow up procedure.

In terms of presentation the overall impression of the questionnaire was considered to be very important, particularly to encourage the less enthusiastic individuals to respond. The following design features were therefore incorporated:

* The questionnaire was limited to six double-sided sheets of A4 and presented in a booklet form.
* The last page left space for comments.
* A large font size was used to improve readability.
* Generous space was allowed for each question to avoid a 'crammed' appearance.
* Use of tick boxes to record answers, with the boxes aligned down the page.
* Instructions to the respondent were highlighted in bold.
* An attractive front cover with illustrations was incorporated, to encourage the respondent to open the questionnaire
* The questionnaire was reproduced on coloured paper to help the questionnaire stand out amongst other mail

A full version of the questionnaire is presented in Appendix 2, together with the covering letter sent out with the form. This letter was intended to encourage household's to complete the questionnaire and return it by emphasising the importance of their views, and the role they could play in influencing policy.

Following the initial mailing, a reminder post-card was sent out after seven days to households which had not replied. When three weeks had elapsed from the date of the initial mailing a second questionnaire was despatched to those that had yet to reply.

Chapter 8 CVM RESULTS AND DISCUSSION

In this chapter the results of the CVM survey are presented, including estimates of mean WTP of Scottish households for alternative acidification scenarios, calculated using a new approach to the analysis of discrete choice data. The validity and reliability of the results, and related methodological issues are discussed.

8.1 Results of the Pilot Study

A response rate in excess of 50% from the pilot mail survey provided over one hundred useable responses for Cooper's optimal design programme. There were no obvious problems with the questionnaire, and annual payments ranged from £0 to £1000, per household, per annum, with a mean WTP of £75.

Using the offered payments (excluding the zero bids[1]), Cooper's bid design programme CVMMLOGIT selected the number (m_i) and size of bid amounts (b_i), and the number of respondents given each bid amount (n_i), which minimised the mean square error term. The resultant optimal design suggested that forty bid levels were required for the main survey. Since this would have resulted in six hundred different versions of the questionnaire, it was decided to

[1] Zero bids were excluded because in the discrete choice survey non-payers would be screened out prior to the DC question.

restrict the maximum number of bid levels to twenty. The resulting optimal bid design for this number of bid levels is shown in Table 8.1.

8.2 Main Discrete Choice Survey

8.2.1 Response Rate

Out of 3000 questionnaires mailed out to households in Scotland, a total of 1820 were returned. This is equivalent to an overall response rate, disregarding questionnaires which were undelivered to the intended household, of 67%. Of these, 1669 respondents had answered the valuation question to give an effective response rate of 61% (Table 8.2). The remaining responses were considered invalid, either because they were improperly completed (1%), or were classified as protest responses (4%). Protesters are people who refused to pay the preferred bid because they objected to some aspect of the questionnaire. Since their valuation of the environmental change is not known they are excluded from the analysis. (Protest responses are examined in more detail in section 8.6.5.2).

The proportion of respondents prepared to pay something, who also agreed to pay the offered bid amount, is described in Table 8.3 for each bid level. Almost half (47%) stated they were willing to pay the highest bid level offered (£396). This is clearly unsatisfactory with respect to modelling the upper tail of the bid curve and a further

Table 8.1 Optimal bid distribution (X_i) and number (N_i) for twenty bid levels (M =20)

X_i	N_i
£11	14
£17	33
£21	34
£26	43
£31	51
£37	55
£42	57
£48	69
£55	81
£63	88
£71	99
£81	114
£92	133
£106	155
£122	184
£143	224
£170	285
£209	384
£270	603
£396	294

Table 8.2 Summary of responses to main survey

Category		
Responded to specified bid	Yes	638
	No	399
	Don't Know	352
Refused to pay anything		288
Total number of Valid Responses		**1669**
Protests		118
Incompletes		33
Total completed questionnaires		**1820**
Refusals		110
Non-responses		790
Total delivered to intended household		**2720**

Table 8.3 Bid distribution (X_i) and observed probability of a YES response

X_i	P(Yes)
£11	.67
£17	.75
£21	.82
£26	.88
£31	.79
£37	.85
£42	.90
£48	.87
£55	.77
£63	.82
£71	.69
£81	.59
£92	.74
£106	.72
£122	.76
£143	.63
£170	.68
£209	.62
£270	.48
£396	.47

30 questionnaires had to be dispatched incorporating an increased bid price of £798. The observed proportion responding Yes to this bid level was 0.22 which was considered to be satisfactory for modelling purposes. (The problem with the design of the bid distribution and possible solutions are discussed in section 8.6.5.4).

8.2 Logistic Regression of Discrete Choice Data

Since the dependent variable is derived from an observation y, which can only be zero (bid rejected) or one (bid accepted), and cannot fall outside this range, linear regression methods suitable for variables with a normal distribution, are inappropriate. The appropriate method for analysing discrete choice data are through logistic regression. Non-linear models, such as the logit or probit model are most frequently used because they constrain the dependent variable within the range 0 to 1. The logit is usually preferred because it is more easily manipulated (Langford and Bateman, 1993).

If there are no covariate data, the logistic regression equation is of the form:

$$E(y_j) = 1/\{1 + \exp[-\boldsymbol{\beta}(\mathbf{x}_j - U]\} \qquad \textit{Equation 8.1}$$

where $E(y_j)$ is the probability that a respondent accepts bid x_j, and $\boldsymbol{\beta}_i$ and U are parameters to be estimated. Mean WTP can be estimated by fitting the logistic curve to bid level, and integrating under this curve (Hanemann, 1984)[2].

Equation 8.1 describes the situation where bids can be positive and negative. However, in this study it was assumed at the outset that negative bid amounts (i.e. willingness to accept compensation) were unrealistic for acidification abatement. Also, it may be inappropriate to model WTP and WTA using the same mathematical function since they may be based on entirely different lines of reasoning (Hanneman, 1991). No information was available to model WTA, hence it was considered desirable to restrict the bid (x) to be non-negative number.

8.2.1 Logistic Regression Where Bid is Non-Negative

Where negative bids are considered to be inappropriate, as in this study, it is necessary to restrict the bid vector by truncation, or by transforming bid to a non-negative number. Three different approaches were considered:

8.2.2 Left Truncation

Left truncation is a relatively simple but crude approach to eliminating negative bids and has the effect of collapsing the negative left-hand tail of the distribution onto the point where bid = 0 (Ready and Hu, 1995).

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[2] Cameron, (1988) describes an alternative approach using censored logistic regression. Although mean WTP estimates should not differ between the two (Kristrom, 1990), they differ in their assumptions about the decision-making process. The Hanemann approach assumes only that the individual knows whether the change will make he/she better or worse off while the Cameron approach assumes the individual knows their valuation of the environmental change and whether it exceeds the offered payment (McConnell, 1990).

Consequently it tends to produce a rather poor fit to the data, particularly at small bid levels (Buckland *et al.*, 1996).

8.2.2 Log Transformation

The log-linear model[3] has been more widely used since it often outperforms other specifications based on goodness of fit statistics (Langford and Bateman, 1993). Under a log-transform Equation 8.1 becomes

$$E(y_i) = 1/\{1 + \exp[-\beta\{\log(x_i) - U]\}\} \quad \text{Equation 8.2}$$

Since the upper tail of the distribution is lengthened by a log transformation, estimates of mean WTP are very sensitive to observations at the upper tail. Upwards bias can result where there are few bids that correspond to a mean probability of a Yes response close to zero, or if the fit of the model is poor in the upper tail (Buckland *et al.*, 1996). Truncation of the bid distribution corresponding to the largest bid level offered (Boyle *et al.*, 1988), or at some arbitrary point in the distribution (McCollum, 1990) has been used to reduce extreme values for WTP, but these estimates are very sensitive to the point of truncation (Langford and Bateman, 1993).

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[3] Although not entirely consistent with utility theory (Hanneman, 1984) it can be considered to be a first order approximation for a utility difference (Bowker and Stoll, 1988).

8.2.3 Reciprocal Bid Transformation

An alternative approach is to take the reciprocal of the bid. This transformation adequately restricts x to be non-negative in the lower tail, but does not affect the behaviour of the upper tail. The transformation takes the form $w = x - a/x$ for some a, where a is an estimate of some unknown parameter α whose value can be chosen from a search procedure which minimises the residual deviance (Buckland *et al.,* 1996). Under this transformation Equation 8.1 becomes:

$$E(y_j) = 1/\{1 + \exp[-\beta(x_j - (\alpha/x_j) - \mu)]\} \qquad \text{Equation 8.3}$$

Since this particular transform is less likely to bias the estimate of mean WTP it was considered to be the most appropriate approach to restricting x. (In Chapter 8.4.1 estimation of mean WTP using the other two methods is compared to the reciprocal transformation).

8.2.2 Logistic Regression with Covariates

Logistic regression with covariates was performed on the dependant response variable (y). It is useful to include covariate data, in addition to bid level for a number of reasons. Firstly, covariate data provides some idea of the extent to which the responses to the payment question can be explained by respondents characteristics which one might expect to influence WTP from theory. Secondly, by modelling environmental scope, as described in

Propositions 1 to 3 (e.g. recovery level, damage level, rate of recovery), as ordinal data in the regression analysis it is possible to determine their statistical significance in relation to WTP. Finally, covariate information needs to be taken into account when estimating mean WTP otherwise a biased estimate will be obtained (Buckland *et al.,* 1996).

The regression equation takes the form:

$$E(y_j) = 1/1 + \exp[-\boldsymbol{\beta}_0 - \sum \boldsymbol{\beta}_i z_{ij}] \qquad \text{Equation 8.4}$$

where $E(y_j)$ is the probability that a respondent accepts bid x_j

$\boldsymbol{\beta}_i$ are coefficients to be estimated, $i \geq 0$,

z_{ij} is the value of covariate i for respondent j, $i \geq 0$.

A step-wise logistic regression procedure was used to identify the covariates which significantly influenced the probability of a Yes response E(y). Twenty-nine covariates, including the transformed value for bid (transbid), and ordinal variables for future recovery level, future damage level, the rate of recovery, and socio-economic variables were initially included. These covariates are described in Table 8.4.

The step-wise logistic regression was carried out on the 1037 respondents who either accepted or rejected the specified bid amount. Excluded from the step-wise regression analysis were respondents who had received

Table 8.4 Description of covariates used in the logistic regression

COVARIATE	DESCRIPTION	TYPE	Q
transbid	transformed bid value	continuous	13
income	annual pretax household income	ordinal (1-9)	20
age	age of respondent	ordinal (1-6)	19
child	respondent has/has not children	binary	24
understand	level of understanding about information presented on acid rain in survey	ordinal (1-4)	12
member	memberships of environmental organisations	continuous	9
pollu	pollution as a priority amongst 6 social issues	ordinal (1-6)	6
tv/radio	opinion about TV/radio programmes about nature	ordinal (1-4)	4
govt	role of government in protecting the environment	ordinal (1-5)	7
return	number of mailings required before response	ordinal (1-3)	
acid	concern about acid rain	ordinal (1-4)	8
gender	male/female	binary	17
papers	number of different newspapers read	continuous	25
regional	concern index for regional environmental issues which affect UK	ordinal (1-8)	8
abroad	concern index for environmental issues not related to UK	ordinal (1-8)	8
global	concern index for global environmental issues	ordinal (1-8)	8
concern	overall concern index for environmental issues	ordinal (1-32)	8
envaction	participation in actions to improve the environment	ordinal (1-9)	5
passive	participation index for passive outdoor pursuits	ordinal (1-12)	1
user	participation index for active outdoor pursuits	ordinal (1-27)	1
college	attended college/university	binary	23
awareness	awareness about acid rain before reading survey	binary	10
recovery	level of recovery presented (a,b, or c)	ordinal (1-3)	
dam	level of damage under the status quo	ordinal (1,2)	
rate	rate of recovery from damage	ordinal (1-2)	

questionnaires referring to the uncertain scenarios. These could not be modelled as ordinal data within the regression analysis. Also, respondents who replied 'Don't Know' to the bid question were excluded because their preferences for recovery from acidification have not been revealed (see section 8.5.2). Respondents who reported, prior to answering the bid question, that they were unwilling to pay anything, even a very small amount were excluded. Accounting for 17% of all valid responses, this group presents several difficulties for modelling the bid function.

Firstly, since the logistic curve is continuous there should not be a discrete 'lump' of probability anywhere along the bid function. Secondly, logistic regression requires that there should be a bid that everyone would accept. Failure to identify and screen out non-payers would have meant that the upper asymptote of the curve would not approach unity. These respondents are not lost from the analysis all together however, since their zero valuation is used to weight the estimate of mean WTP (section 8.4).

8.3 Regression Results

Transbid was entered first with the remaining covariates selected using a step-wise procedure. Only covariates with a student's t value significant at the 5% level were included in the final model. Future damage level was the only variable relating to the scope of the environmental change which significantly influenced y, with a high damage level associated with a higher probability of saying Yes to the payment question. Future recovery level and rate of recovery did not significantly influence the bid response even when forced into the model first. The covariates of the fitted model are listed in Table 8.5, together with their parameter estimates, standard errors, and t-values.

As expected the probability of a Yes response decreased with bid level and increased with *income*. Environmental improvement following abatement is therefore a normal good, with people on higher incomes paying more. A greater understanding about acid rain (*understand*)[4], and whether the respondent was a member of an environmental charity (*member*) also increased the probability of a Yes response. People who believed had an important role in protecting the environment (govt), who ranked pollution (*pollu*) higher in importance among 6 social issues (e.g. employment, education etc.) and returned their questionnaire earlier (*return*) were also more likely to say Yes. Respondents who were particularly concerned about environmental issues outwith the UK were also more likely to accept the offered bid level. This is slightly surprising in that the survey deals only with damage in Scotland, but perhaps suggests that alturistic concern for big environmental problems underpins

[4]The coefficient on understand is negative because respondents who had greatest understanding about the acid rain issue were ranked 1, while respondents who did not understand were ranked as 4.

Table 8.5 Estimates of regression coefficients included in final logistic regression model.

Covariate	Estimate	s.e.	Student's t
Constant	1.079	0.699	1.54
transbid	-0.004	0.001	-8.72
income	0.419	0.048	8.70
govt	-0.427	0.147	-2.91
return	-0.406	0.122	-3.34
understand	-0.372	0.129	-2.89
member	0.371	0.146	2.55
pollu	-0.173	0.075	-2.31
dam	0.306	0.153	2.01
abroad	0.132	0.066	2.01

WTP for recovery from acid rain. Overall the attitudinal and behavioural signals picked up by the questionnairre, and which serve as proxies for preferences in CVM, are correlated with WTP in a way one would expect from theory.

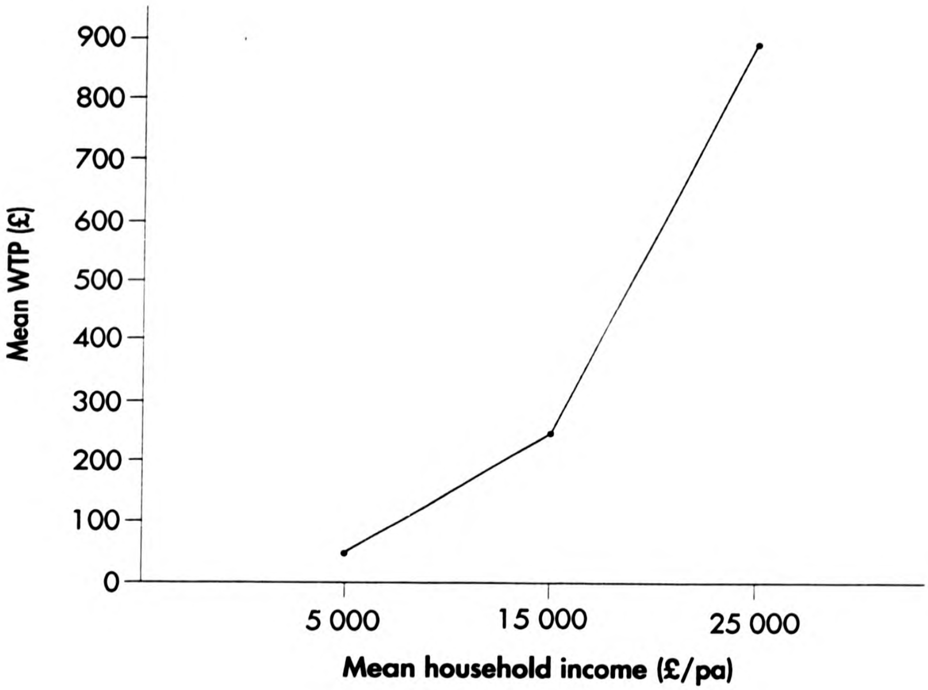
Table 8.6 describes the effect of each covariate on the residual deviance. Income level exhibited greatest explanatory power, reducing the deviance by 104.8. *Transbid*, which was forced to enter the model first, reduced deviance less than income. Goodness of fit of the model can be approximated by referring to the residual deviance. The closer the residual deviance is to the residual degrees of freedom the better the fit of the model (Bateman et al., 1994). In this case the fit appears reasonable.

The income elasticity of environmental improvements has recently come under scrutiny. Kristrom and Reira (1996) provide empirical evidence from a range of CVM studies that income elasticity for environmental improvements is less than one, and propose that the environment is not a luxury good as widely supposed. In this study income elasticity was calculated from predicted mean WTP for three income groups, (less than £10000, £10-20000, and £20-30000). Figure 8.1 reveals that WTP increases as a proportion of total income. Arc income elasticities of over 2 were obtained and provide evidence for supposing that environmental improvements stemming form SO_2 abatement is a luxury good.

Table 8.6 Accumulated analysis of deviance

Change	d.f.	deviance
Regression	9	225.4
+transbid	1	45.6
+income	1	105.7
+govt	1	24.3
+return	1	14.8
+understand	1	11.7
+member	1	8.6
+pollu	1	8.2
+dam	1	4.5
+abroad	1	4.1
Residual	733	718.7
Total	742	942.1

Figure 8.1: Relationship between household income and mean WTP



8.4 Estimating Mean WTP

In order to assess the economic efficiency of SO_2 abatement for policy purposes it is necessary to derive an aggregate measure of WTP. This requires estimating the mean WTP from the sample of households, adjusted for any sampling bias, and aggregation to the population level by multiplying the estimated mean WTP by the total number of households in Scotland.

In the presence of covariates, the mean WTP is conventionally estimated by fitting a single logistic curve to bid level, having averaged over all the other covariates (Cameron, 1988) and integrating under this curve (Hanemann, 1984). The integral is equivalent to the mean WTP of a household or individual, whose covariate values correspond to the mean for the population.

While useful for statistical comparison between two or more sub-samples (Bowker and Stoll, 1988), this practice can generate a biased estimate of WTP because of the non-linear relationship between the covariates and WTP in a logistic model (Buckland *et al.*, 1996). For example, if many people with an average income were asked to pay £1000, it is likely that all would refuse, whereas if people with a representative range of incomes averaging £1000 were asked to pay the same amount, some of those with very high incomes might accept.

This bias was avoided by using the fitted logistic model (Equation 8.4) to predict the probability (y_j) that each individual respondent (j) would accept the bid (x) presented to them for each damage level. At each bid level, the individual predictions were averaged to obtain a valid sample estimate of the overall probability of a positive response by bid level in the sample population (p_k). A logistic curve was then fitted to these mean probabilities, weighted by the term $n_k/(p_k(1-p_k))$, to account for sample size variance across bid levels, using a curve fitting routine. This logistic curve is univariate, with bid as the independent variable, and it can be used to estimate mean WTP by integration.

Table 8.7 shows the predicted probabilities of a 'Yes' response and the predicted number of 'Yes' responses, and compares them with the observed values. There appears to be little evidence of systematic departure of the predicted proportion of respondents accepting the bid and the observed proportion. This is confirmed when observed and predicted responses are plotted in Figure 8.1. Hence this curve gives a reasonable fit to the data. By way of comparison, the fitted curve following left truncation and log transformation of bid are shown in Figures 8.2 and 8.3 respectively. The left truncation gives a markedly poorer fit at the lower tail, while the log transformation gives a good fit except at the highest bid level, which, if the curve is not truncated, would lead to a massive estimate for WTP.

Numerically integrating the fitted logistic equation (with covariates) generates a mean annual WTP for those

Table 8.7 Predicted and observed proportion accepting the bid

Bid Level (x)	Number offered bid	Proportion of accepting bid	
		Observed	Predicted
11	7	.86	.93
17	6	.67	.89
21	8	.88	.87
26	11	.91	.85
31	12	.83	.84
37	9	.78	.82
42	18	.89	.81
48	19	.84	.80
55	14	.86	.79
63	25	.84	.78
71	28	.71	.77
81	29	.59	.76
92	38	.82	.75
106	42	.79	.74
122	49	.78	.72
143	56	.59	.70
170	73	.78	.68
209	80	.64	.64
270	141	.50	.58
396	69	.54	.46
798	9	.22	.16

Figure 8.2 Scatterplot of proportion of respondents accepting bid (o) and of mean predicted probability of acceptance (x), against bid level under the reciprocal bid transformation. Also shown is the fitted logistic curve from which mean WTP is estimated.

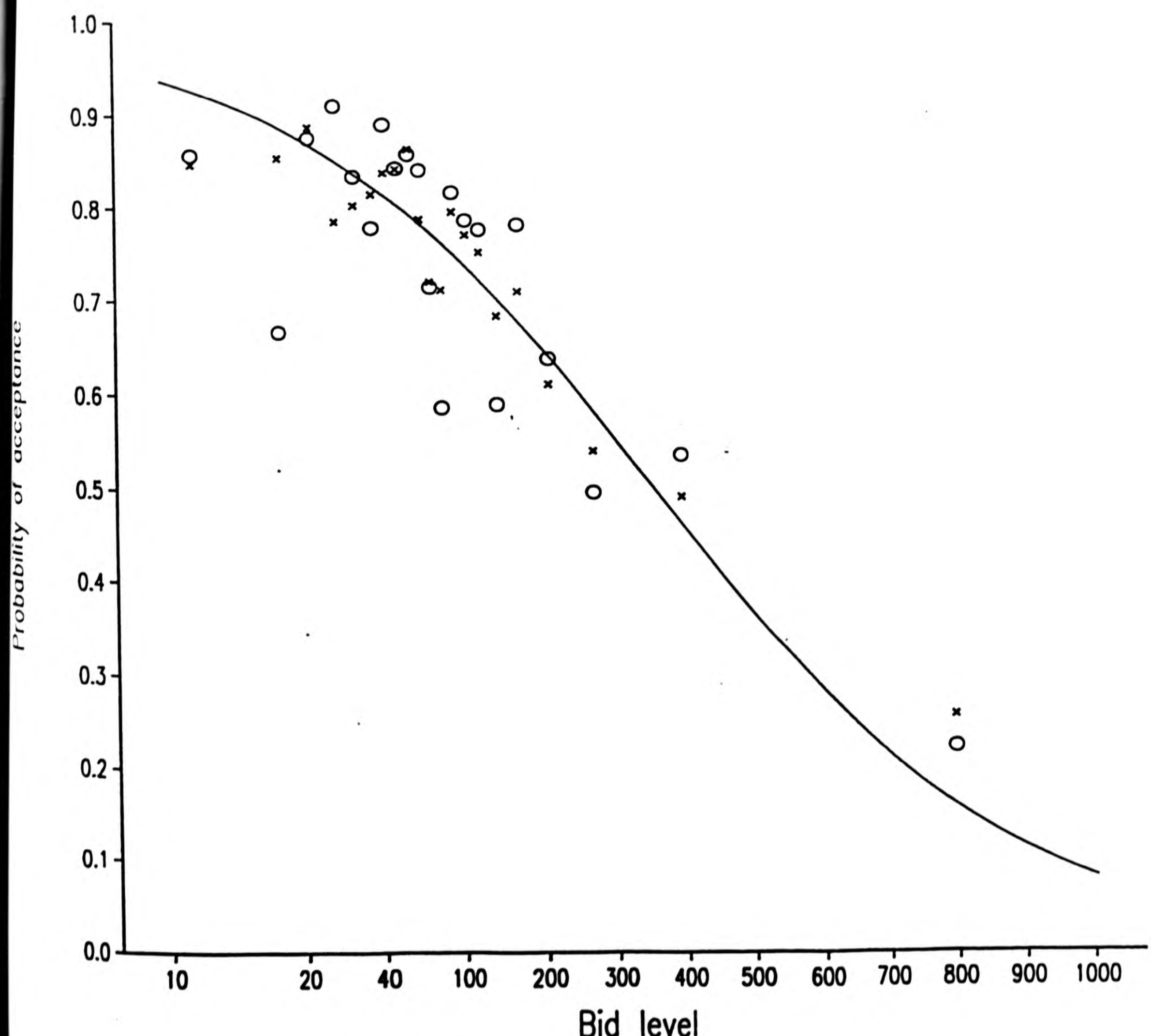


Figure 8.2 Scatterplot of proportion of respondents accepting bid (o) and of mean predicted probability of acceptance (x), against bid level under the reciprocal bid transformation. Also shown is the fitted logistic curve from which mean WTP is estimated.

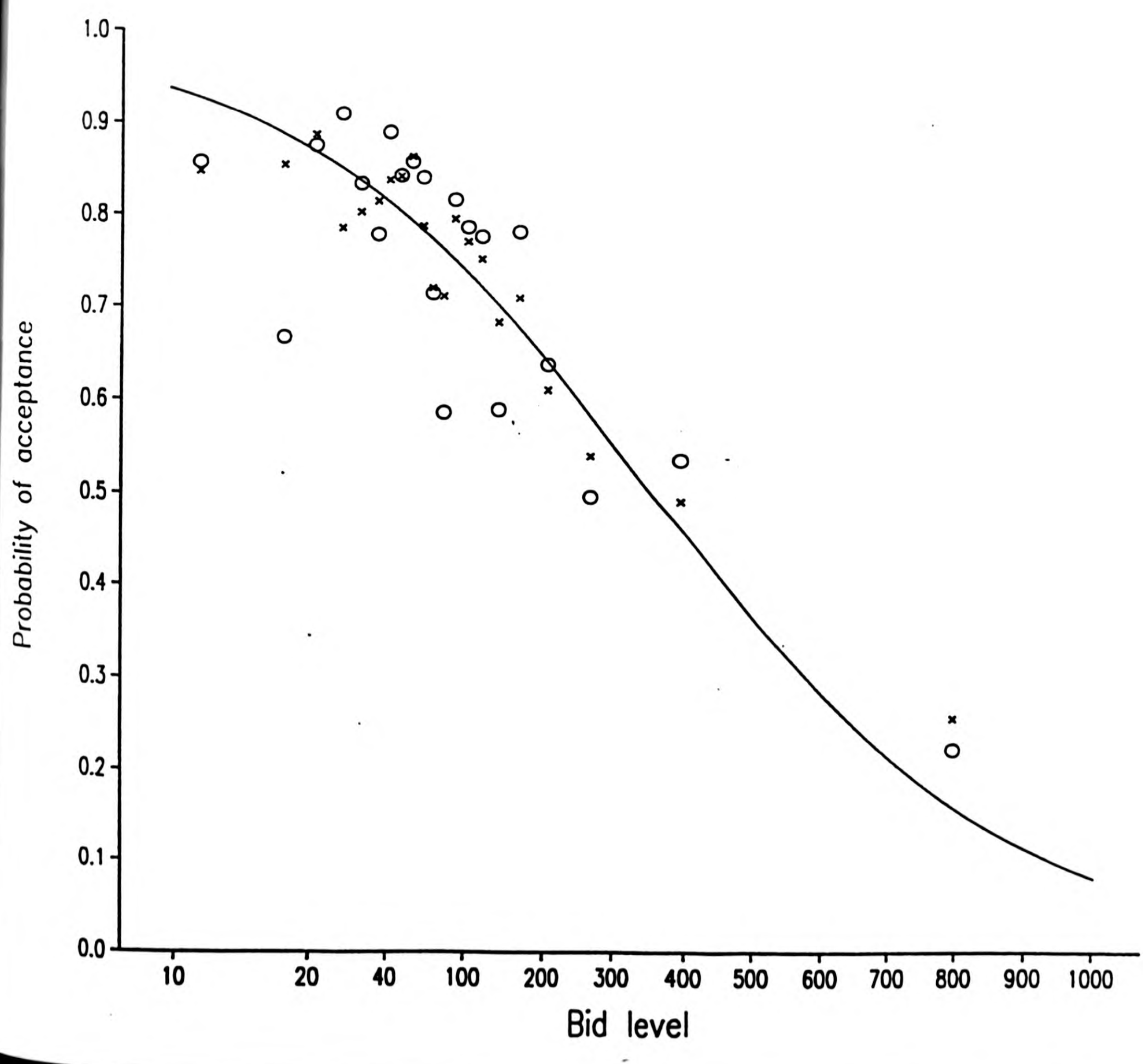


Figure 8.3 Scatterplot of proportion of respondents accepting bid (o) and of mean predicted probability of acceptance (x), against bid level under left truncation of the WTP curve, fitted using untransformed bid level. Also shown is the fitted logistic curve from which mean WTP is estimated.

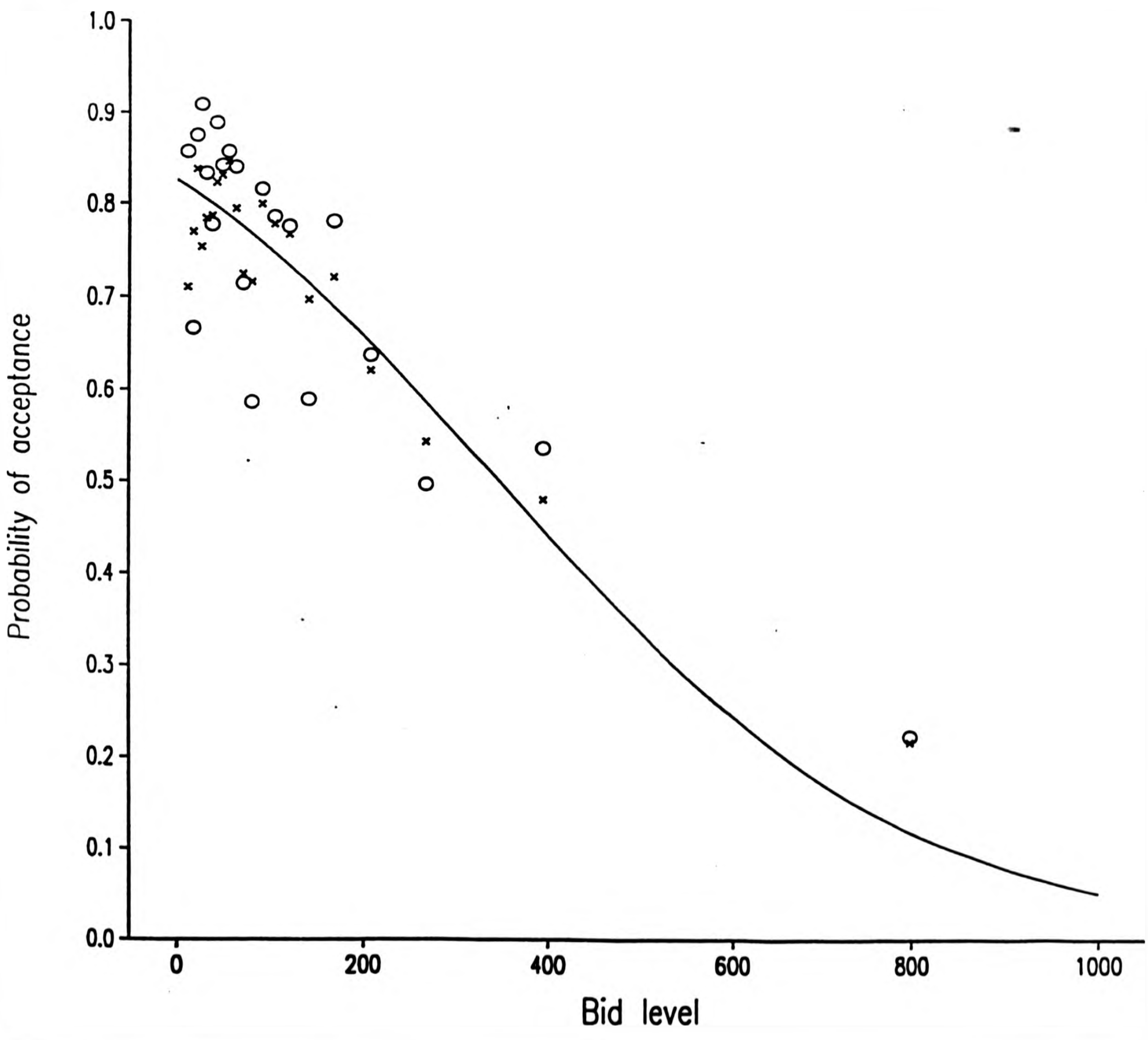
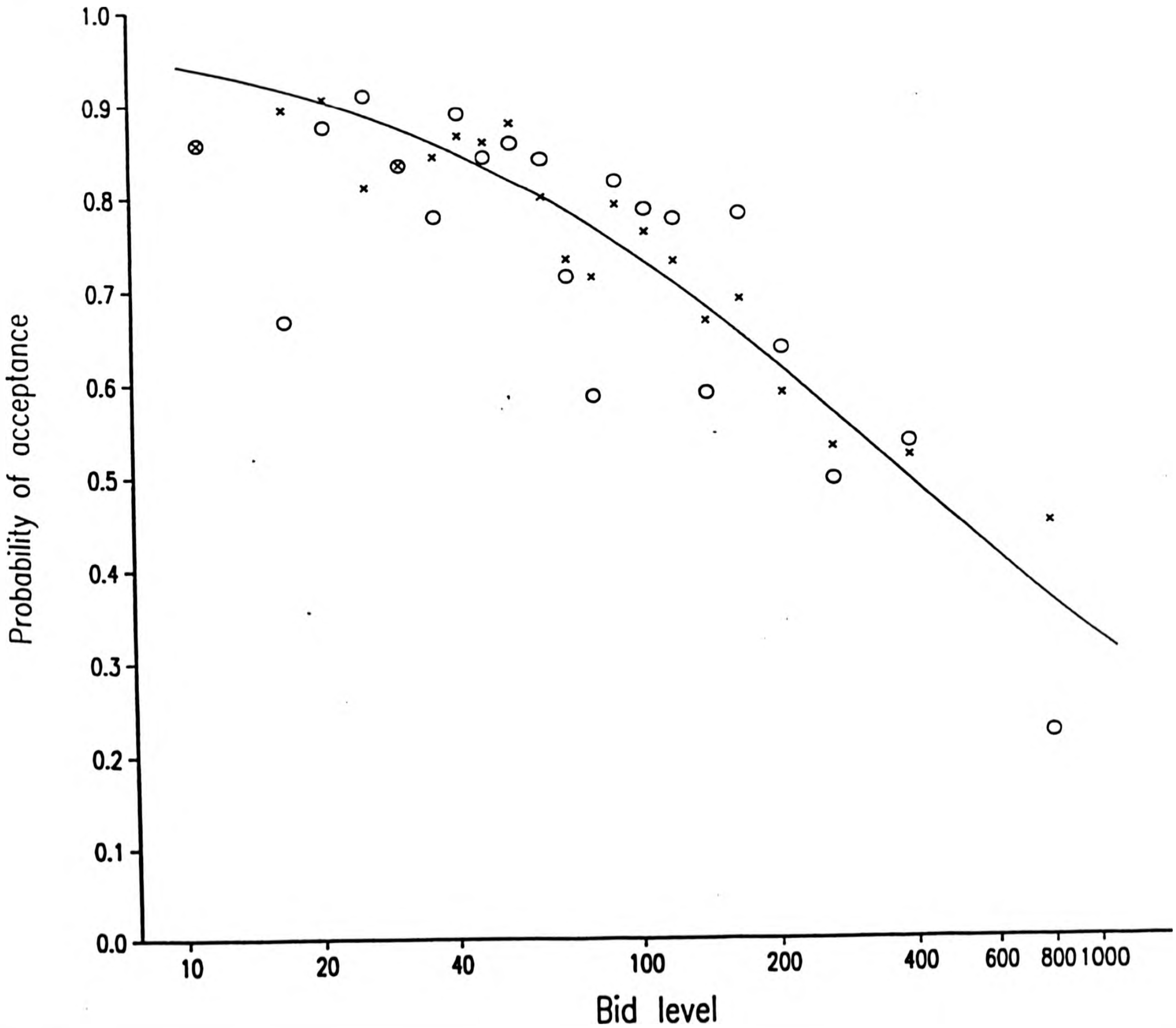


Figure 8.4 **Scatterplot of proportion of respondents accepting bid (o) and of mean predicted probability of acceptance (x), against bid level under logarithmic bid transformation. Also shown is the fitted logistic curve from which mean WTP is estimated.**



willing to pay something of £425, with a 95% confidence interval, quantified using the bootstrap procedure[5] of between £333 and £596. This estimate has to be adjusted for those respondents unwilling to pay anything by multiplying the estimate by the proportion of non-payers (21% in this case). The resultant mean (and 95% CI) for the sample population is £335 (£261,£475).

Table 8.8 compares the mean estimates derived by the preferred method with other model specifications. The logarithmic transformation generates an extremely high WTP as a result of a poor fit in the upper tail (Figure 8.3). Left truncation provides a reasonably close estimate of WTP because the curve fits the data poorly only in the lower tail where the fit does not greatly influence the mean. For comparison, the mean WTP estimated using average covariate values is also shown. This estimate of mean WTP is only slightly greater than the preferred approach (£341), because the model is a poor fit (Figure 8.4) only at low bid levels, (i.e. individuals on average income are more likely to accept low bid levels, whereas most on low incomes would reject these bid levels).

8.4.2 Mean WTP Under Alternative Damage Scenarios

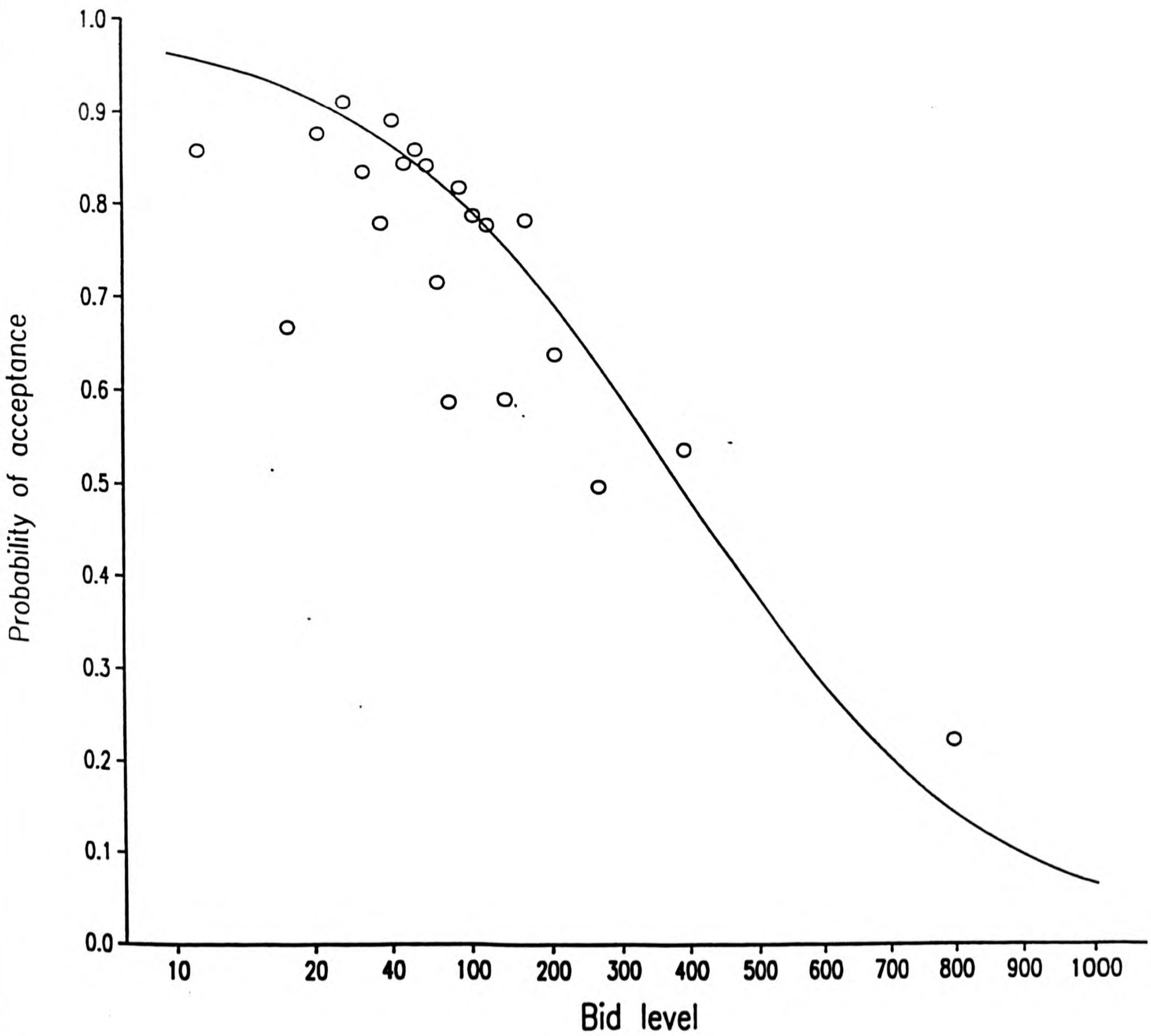
Since future damage level under the status quo was the only scope attribute which significantly influenced the

[5]Bootstrapping is a method of generating variance estimates based on resampling the sample (size = n) with replacement. The resampling continues until the sample size is equably to n, with some data points sampled more than once, and others not at all.

Table 8.8 Estimated mean WTP under alternative approaches

Transformation	Truncation	Covariates Averaged	Estimated mean WTP (£)
Reciprocal	None	No	335
None	Left-Truncation	No	308
Logarithmic	None	No	4442
Reciprocal	None	Yes	341

Figure 8.5 Scatterplot of proportion of respondents accepting bid against bid level. Also shown is the fitted logistic curve obtained by averaging over covariates other than bid level.



dependant variable (y), mean WTP was calculated for the high and low damage scenarios, with the data pooled across both recovery time and level of recovery. The covariates selected from the stepwise logistic regression for high and low damage are presented in Table 8.9 and Table 8.10.

Covariate selection appears to be reasonably consistent with the results from the overall data set. *Transbid, income,* and *govt* are selected in all three regressions and the coefficient estimates have the same sign. Under high damage three covariates which did not appear in the other regression models have been selected. These are *college, child* and *pollu.* Respondents who gave a higher ranking to pollution amongst other social issues, did not attend university or college, and do not have any children, were more likely to accept the payment question.

The calculated mean household WTP per annum for low and high future damage, together with 95% confidence intervals, calculated using the bootstrap procedure, are presented in Table 8.11. The mean WTP for high future damage is £394 and, as expected from the regression analysis, is significantly greater than mean WTP for the low damage level (£281).

Table 8.9 Estimates of regression coefficients for low damage

	Estimate	s.e.	Student's t
transbid	-0.006	0.001	-5.10
income	0.361	0.073	4.93
govt	-0.457	0.228	-2.01
return	-0.474	0.183	-2.58
member	0.514	0.210	2.44

Table 8.10 Estimates of regression coefficients for high damage

	Estimate	s.e.	Student's t
transbid	-0.005	0.001	-4.04
income	0.526	0.093	5.67
govt	-0.529	0.246	-2.15
college	-0.758	0.308	-2.46
child	-1.036	0.424	-2.45
pollu	0.318	0.133	-2.39

Table 8.11 Mean WTP (£) estimates for alternative damage scenarios*

Damage level	Mean (£)	95%C.I.(£)
low (x)	281	217-343
high (y)	394	286-573

* Averaged over rate of recovery and recovery level

8.5 Aggregation

In order to raise the mean household estimates to a valid estimate of the total WTP of the Scottish population it is necessary that the sample should represent the population as a whole. Ideally, this should be done by comparing the mean values for the covariates which significantly influenced mean WTP with those of the population as a whole. Unfortunately comparable data for the entire population are strictly limited, hence the assessment of representativeness must depend on comparisons based on a rather smaller sub-set of socio-economic variables which are available from national statistics. Table 8.12 compares the age, income, and gender of the Scottish population, from the 1991 national census, with that of the sample. For the purposes of aggregation it would appear that the sample is broadly similar to the national statistics[6].

Aggregation to the national level is relatively straightforward and is based on the total number of households in Scotland, which in 1991, was estimated to be 1.96 million (Census, 1991). The aggregate WTP for further abatement of SO_2 in Scotland is estimated to range from £772 million per annum if high damage occurs under the status quo, and £551 million for a low damage scenario.

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[6] Testing for statistically significant differences was not possible because the groupings for these data in the survey were not compatible with the national census.

Table 8.12 Comparison of socio-economic characteristics of the sample with the Scottish population[7] (modal values for head of household).

Covariate	Sample	Scottish Population
Average Age	46-55	45-64
Sex	67%	71%

[7] Source : Census 1991

8.5 Uncertainty About Future Damage and Recovery Levels

So far the data analysis has dealt with future environmental changes that are known with certainty (Propositions 1, 2, and 3). The effect of introducing uncertainty on the mean WTP for future recovery and damage levels in Propositions 4 and 5 is now examined.

In both cases the number of useable responses was less than one hundred and the curve fitting routine for the predicted probabilities, described in Chapter 8.4, did not converge. It was, therefore, necessary to estimate mean WTP using a standard logit programme using *bid* (truncated at zero), rather that transbid. In Table 8.13 mean WTP estimates are presented for uncertain recovery (AP_{rec}) and damage (AP_{dam}), and for their certain equivalents (WTP_b and WTP_w).

For both uncertain scenarios (the AP measures), WTP was lower than their certain equivalents, suggesting that respondents were risk averse with respect to environmental gains, and to environmental losses. Due to the small sample size involved parameter estimates for the covariates were imprecise and few were found to have a significant influence on mean WTP (Table 8.14).

Table 8.13 Mean WTP (£) estimates for uncertain and their certain equivalents

Scenario		mean (£)	95% C.I. (£)
Uncertain Damage	AP_{dam}	184	113-278
Certain Damage	WTP_w	328	201-1174
Uncertain Recovery	AP_{rec}	187	63-1023
Certain Recovery	WTP_b	241	171-624

Table 8.14 Estimates of significant[8] regression coefficients for uncertain and their certain equivalents

Covariate	Scenario			
	AP_{dam}	WTP_w	AP_{rec}	WTP_b
bid	-0.021	*	*	-0.005
return	*	-2.780	*	*
income	*	*	1.386	-0.502
pollu	*	*	-2.094	*

[8] T-statistic significant at the 95% level.

8.6 Discussion

In this section issues concerning the measurement of environmental scope and uncertainty using the CVM are discussed in relation to the results of this study. The accuracy, reliability and validity of the estimates for policy purposes are also assessed. The section concludes with a discussion of several important methodological issues raised by this study.

8.6.1 Sensitivity to Environmental Change

The results of the regression analysis suggest that respondents were sensitive only to the level of future acidification damages. The level of environmental recovery and the rate of recovery did not significantly influence the bid response, even when forced into the model first. Lack of sensitivity to these aspects of recovery is contrary to theoretical expectations (Johansson, 1988).

Carson and Mitchell, (1993) arguing in defence of the CVM, regard absence of a scope effect for environmental change as symptomatic of poor survey design and administration. This seems an unlikely explanation, since future damage level was found to influence mean WTP, and pre-testing clearly showed that respondents were able to distinguish between the three environmental levels depicted in the acidification scenario. The clear scope effect associated with the level of future damage also tends to undermine arguments that the study has been affected by symbolic bias. This bias has been proposed as a cause of insufficient variation in levels of environmental goods (Kahneman and Knetsch, 1989), and is said to result from respondents reacting to the symbolic significance of environmental change, rather than to the specific levels of provision.

An alternative, and more plausible, explanation may lie in the way individuals perceive and value losses and gains relative to some common reference point. Kahneman and Tversky (1979), using results from experimental economics, proposed a value function which is steeper for losses than for commensurate gains. Greater sensitivity to losses than gains has been shown in a range of experimental situations, and has been put forward as a possible reason for the divergence between WTA and WTP measures in CVM studies (Knetsch, 1993).

The scope effects described in this study are not unique and may actually reflect a real perception amongst respondents that damage level matters more than the level of future recovery. This is supported by evidence from a recent review of scope effects in CVM studies (Carson and Mitchell, 1993). In four of five studies described, where significant differences between WTP for different levels of some environmental good have been established, WTP to avoid a loss, rather than WTP to obtain a gain was involved (e.g. WTP to avoid negative mining effects in Kakadu National Park or WTP to avoid water shortages in southern California).

Insensitivity to the scope of environmental improvement is widely reported in the literature. For example, Hanley and Owen (1994) found that WTP for one Site of Special Scientific Interest (SSSI) in southern England was not significantly different from the WTP for all regional SSSI's. Also, Green and Tunstall (1991) found mean WTP for alternative higher levels of water quality in English rivers were not significantly different.

The greater apparent concern about future damage among respondents may also have been heightened by the long time period before the final damage state was attained (30 years from present). Knetsch, (1992) reports that individuals have greater aversion for long delayed 'dreaded events' rather than to more immediate ones. Some form of negative discounting amongst respondents would therefore have enhanced the effect of damage level on WTP.

Unfortunately, time preference for environmental change was only examined for future recovery (Proposition 3). In this case, whether recovery was achieved in 20 or 120 years appeared to have had no affect on WTP. Absence of any evidence for positive discount rate on future gains may imply insensitivity of the CVM instrument, but it could also be consistent with the notion of hyperbolic discount rates (Lowenstein and Prelec, 1992), with rates falling to zero when gains occur far into the future. Low or zero discount rates for environmental gains is also consistent with the bequest motive, expressed strongly by the majority of respondents (see Table 8.17).

8.6.2 Uncertainty

The results for proposition 4 suggest that respondents are risk averse when gains are in prospect. This is equivalent to concavity in the utility function (Kahneman and Tversky, 1979) and supports the earlier empirical work of Johansson, (1988) where individuals were asked to value alternative species preservation programmes. In general, risk averseness appears to be a prevalent, but not universal, strategy when gains are considered. For example, risk seeking has been observed, amongst hunters in Sweden who gained satisfaction from the uncertainty of the hunt (Johansson, 1990).

Turning to uncertainty about future damage, Kahneman and Tversky (1979) present a body of evidence from experiments using money gambles that suggests individuals are risk-seekers when losses are in prospect, but risk-averse when gains are in prospect (the "reflection effect"). However, no evidence for risk-seeking amongst respondents was found for future acid rain damages, with the mean WTP to avoid a certain loss exceeding the equivalent uncertain loss. Hence, respondents also appear to be risk averse when faced with future environmental damage.

Cautious interpretation of the results from the final two propositions is necessary for two reasons. Although every effort was made to do so, it was not possible to precisely identify the median damage level using the described presentation format (i.e. species boxes). Hence, AP and WTPb/WTPw measures may not be expressing equivalent biodiversity gains/losses to respondents. Secondly, the sample size for each scenario was very small and the estimates of mean WTP were imprecise, and differences between the certain and uncertain measures were not statistically significant.

8.6.4 Accuracy and Validity

If CVM estimates of mean WTP for non-market benefits are to be of use in guiding policy and project appraisal, they have to be accurate and reliable. Unfortunately, the 'true' value of the public's willingness to pay for recovery from acidification in the semi-natural environment is unknown. Thus for non-use type environmental goods it is not possible to directly measure accuracy and it becomes necessary to fall back on some assessment of validity and reliability.

8.6.4.1 Accuracy

Before discussing validity of the CVM estimates produced in this study it is important to highlight one important feature of CVM estimates with respect to accuracy: the propensity for hypothetical values to overstate actual WTP. A number of comparisons between hypothetical and actual payments, for either private goods or quasi-public goods, have shown an upward bias in hypothetical payments. Examples include WTP for new TV programmes (Bohm, 1972), and Bishop and Heberlein's, (1979) study of sales of goose hunting permits.

However, the comparison of real payments with hypothetical ones in many of these studies is inconclusive because of inconsistencies between the real and hypothetical markets, and the type of statistical analysis performed (Hanemann, 1994). For example, Mitchell and Carson, (1981) found no significant differences between the real and hypothetical WTP in both Bohm's and Bishop and Heberlien's studies when outlying data points were excluded. There have also been several studies which have not shown any significant difference between real and hypothetical payments (Dickie *et al.*, 1987; Sinden, 1988).

Few studies have compared hypothetical with real payments for non-use environmental resources, where there are considerable difficulties obtaining equivalent estimates for actual WTP outside an experimental setting. One might expect, however, that since individuals are often unfamiliar with the type of environmental goods they are being asked to value, and have no previous choice experience with respect to consumption levels, that hypothetical payments may be substantially different from real payments. Seip and Strand, (1992) found that the number of individuals who subsequently joined a wildlife charity following a hypothetical survey was significantly lower than the number who indicated an intention to join in the survey. Navrud and Veisten (1996) found that real WTP to preserve forests in Norway were between 2-4 times lower than hypothetical payments. Kealy *et al.,* (1990) performed a small scale experiment involving payments to prevent acidification to a regional aquatic system. Actual payments were approximately half ($5.40) of stated hypothetical WTP ($10.10).

However, these studies do not provide particularly robust evidence for an upward bias for hypothetical payments. Hanneman, (1994) criticised the Seip and Strand study on the basis of a flawed methodology, since the hypothetical commitments to join were obtained on the telephone by members of the environmental charity concerned, while respondents who actually joined did so as a result of a mail shot. Also, soliciting an intention to make a charitable donation is a poor test of CVM because it invites less commitment than, for example, an intention to vote for higher taxes (Hanemann, 1994). The Kealy *et al.*, (1990) study is also not conclusive because it involved a small group of students in an experimental setting, and no significant differences were observed between real and hypothetical payments were found. In general, therefore, it would seem that CVM estimates of WTP are higher than actual payments, but the extent of this bias is, as yet, undefined.

8.6.4.2 Validity and Reliability

In the absence of real payments to compare with the hypothetical estimates calculated in this survey it is necessary to establish their validity and reliability based on a series of tests. These are considered below.

8.6.4.2.1 Convergent Validity

This test of validity compares the extent to which WTP estimates from CVM converge on estimates generated through other measurement techniques, such as the Travel Cost Method, and other CVM studies involving comparable types of environmental change. Table 8.15 lists the results of all known CVM studies for reducing acid rain damage in Europe, including the recent project carried out by ECOTEC on behalf of the Department of the Environment in 1993. Since each study differs in respect to implementation and elicitation method, the resource affected, and the expected recovery from acidification, the comparison cannot be considered to be very conclusive.

However, the overall impression is that mitigation of acidification damage in Europe, with the exception of Navrud's and ECOTEC's study, is highly valued. The Navrud study generated comparatively low estimates but only recovery in fish stocks were considered. The ECOTEC study is the only other UK study described in Table 8.15, but reveals a considerable lower annual WTP for environmental recovery than was found in this study. However the ECOTEC study was concerned with the aquatic environment only, and not terrestrial effects (e.g. tree damage). It also used an open-ended format for the WTP question, which consistently gives lower estimates of WTP than the DC format (see section 8.6.5.1).

In this study the open-ended payment from the pilot study was somewhat closer at £75. The divergence is obviously still substantial but the two estimates are unlikely to be very close, in any case, because of the

Table 8.15 Comparison of CVM estimates for SO_2 abatement

AUTHORS	COUNTRY/ YEAR	PAYMENT METHOD	SO_2 CUT	PAYMENT	$[9]
Navrud	Norway 1988	Payment Card	30% 70%	478NOK 603NOK	54 69
Johansson & Kristrom	Sweden 1988	Discrete Choice	100%	4500SEK	453
ARA Consultant	Canada 1982	Open Ended	Not Given	$320	256
ECOTEC	UK 1993	Open Ended	80%	£28	44
Macmillan	Scotland 1993	Discrete Choice	100%	£281 £394	441 618

[9] Converted to 1994 US Dollars, using Purchasing Parity Conversion Rates in OECD : Main Economic Indicators (1997).

differences in environmental scope and survey design. The Johansson and Kristrom, (1989) study is probably the most comparable in terms of methodology and level of abatement and gives very similar estimates of WTP.

CVM estimates have often been compared with user values derived form the Travel Cost Method (Knetsch and Davis, 1966; Hanley 1989; Bishop and Heberlein 1979; Seller *et al.*, 1985). TC estimates of Marshallian consumer surplus and the Hicksian measures from CVM should not differ greatly if the income effect is small (Willig, 1976) and a literature search of studies comparing CVM and TC estimates by Carson et al., (1996) suggests that the two are not significantly different.

Mullen and Menz, (1985) have tried to estimate the economic damage of acidification to recreational fisheries in the Adirondack mountains in the US, but comparison of their results with the non-user benefits arising from recovery in the semi-natural environment in Scotland does not seem appropriate.

8.6.4.2.2 Content Validity

Probably the weakest test of validity, content validity should be established prior to implementation of the survey. It is concerned with whether the scenarios presented in the questionnaire are realistic and consistent with current understanding of the environmental effects of acidification. No serious problems of this nature were encountered when the information set was reviewed by scientists involved in the acidification programme at MLURI.

8.6.4.2.3 Instrument Validity

The extent to which the questionnaire was accurately and comprehensively completed by respondents is the basis for assessing instrument validity. Failure to respond to questions, or the prevalence of ambiguous or difficult to interpret answers, suggests serious cognitive difficulties among the respondents, This, together with the failure of some other respondents to take the questionnaire seriously, reduces the overall number of useable questionnaires.

The question of instrument validity can be addressed in a number of ways. Firstly, an extremely high response rate (67%) for a mail survey was obtained and suggests that the survey instrument was well designed and simple to complete. Secondly, only 33 out of 1820 returned questionnaires were considered to be unusable for subsequent analysis, because of illegible or incoherent responses to questions. In general these questionnaires were returned by very old people, who had difficulty writing, or found the questions difficult to answer.

A significant number of other questionnaires could not be used in the regression analysis due to 'missing responses'. Table 8.16 lists, for each question used in the statistical analysis, the percentage of questionnaires where no response was given. Q.20, regarding household income, was the most frequently unanswered question, with seventy missing responses (6% of total). While this may lead to a

Table 8.16 Missing values among the covariates

COVARIATE	Missing Responses	% of Total
transbid	0	0
income	70	8
age	9	1
child	0	0
understand	11	1
member	10	1
pollu	40	5
tv/radio	11	1
govt	8	1
return	0	0
acid	9	1
gender	12	1
papers	94	10
regional	8	1
abroad	8	1
global	10	1
concern	8	1
envaction	19	2
passive	16	2
user	15	2
college	33	3
awareness	22	2

slight problem with non-response bias (if the non-response was correlated with WTP) it is a problem common to all CVM studies. Q.6, was a challenging question which involved ranking six alternatives, was unanswered on forty occasions (5%). Overall item non-response was therefore low and does not indicate that there were any serious problems with the instrument. Evidence to support this conclusion also comes from Q.12 where respondents were asked to indicate their level of understanding about the information presented in the questionnaire. Less than 10% of respondents considered that they 'didn't understand most of it'.

8.6.4.2.4 Construct Validity

Perhaps the strongest test of validity available for this study is the extent to which responses to the payment question can be explained in relation to variables which one might expect to influence WTP. Economic theory directs that WTP should be dependent on preferences for acidification recovery constrained by available net income. While neither of these attributes can be easily measured using survey techniques, approximations can be determined using questions about socio-economic status, attitude toward the environment, and membership of environmental groups. CVM studies which are invalid in this respect are either poorly designed, or suggest that the underlying model or theory is inappropriate. In either case little confidence can be placed in the resultant estimates of WTP.

In this study construct validity is high. The covariates which entered the step-wise regression model influenced the dependent variable in line with *a priori* expectations. The probability of responding Yes to the payment question decreased with bid but increased with *income*, level of understanding about acid rain (*understand*), and whether the respondent was a member of an environmental charity (*member*). People who ranked pollution (*pollu*) higher in importance among 6 social issues (employment, education etc) and returned their questionnaire earlier (*return*) were also more likely to say Yes. The omission of covariates related to active participation in outdoor activities was unexpected, but perhaps reflects the non-use nature of the benefits of environmental recovery. Alternatively it might suggest that involvement or commitment to the outdoors was poorly modelled by the survey question. Significant covariates selected in the step-wise procedure were fairly consistent across the three models (low damage, high damage and the pooled data set).

NOAA, (1993) also suggest that, following the payment question, respondents should be given the opportunity to state why they are willing, or unwilling, to pay for the environmental change (Q.15 and Q.16). These reasons must reasonably be expected to be compatible with economic circumstances, and with the posited environmental change. Tables 8.17 and 8.18 list the range of reasons given by respondents.

The most common reason among respondents who were prepared to pay for environmental recovery was *'concern for*

Table 8.17 Reasons for paying the offered bid amount

	% of Protestors
I care about animals and plants	29
I would catch more fish	4
Future generations should have an unpolluted environment	37
All pollution is bad	14
Own Reason	6
Don't Know	8
Total	100

Table 8.18 Reasons for not paying the offered bid amount

	% of Protestors
I can't afford to pay this amount	54
I don't think these cuts are worth this amount to me	14
There are large areas of Scotland unaffected by acid rain	11
Own Reason	9
Don't Know	12
Total	100

future generations'. This is clearly associated with a bequest motive and is consistent with the long time-scale of environmental change expected following abatement. It is also consistent with the absence of any significant influence of rate of recovery on WTP (Proposition 3). The non-use nature of the environmental change is emphasised in these responses, with only 4% of respondents paying because they would benefit directly by catching more fish. Excluding protestors, most respondents who were unwilling to pay the offered bid price reported that they were unable to afford the yearly payment. Again, this is consistent with theory.

8.6.4.3. Reliability

Reliability relates to the extent that variation in WTP derives from random sources introduced by sampling, and from error introduced by the survey instrument (Hoovenagel, 1994a). Confidence intervals for mean WTP generated from the DC data are typically wide, reflecting considerable unexplained variation across respondents in the regression model. In this study the 95% C.I. for the overall mean was wide, ranging from £261 to £475. For sub-groups the 95% percentile was wider still and directly reflects the reduced sample sizes available in these groups. Estimates for Propositions 4 and 5 involving uncertainty were sufficiently imprecise that significant differences between samples could not be detected. Clearly if more reliable estimates are to be produced far larger sample sizes are required. This would be costly, but is perhaps necessary on studies where mean WTP for different sub groups are to be compared as in tests for scope.

Instrument reliability can be tested using a test-retest procedure on the same sample population. The instrument is considered to be reliable if WTP is not significantly different and the explanatory variables are consistent across the two time periods. It was not possible to do such an exercise in this study but a range of test-retest studies have been carried which show a fairly high correlation between WTP over time (Jones-Lee *et al.*, 1985, Loomis, 1990, Reiling *et al.*, 1990; Hanley *et al.*, 1996).

8.6.5 Methodological Issues

The results of this study raised important methodological issues concerning elicitation method, the treatment of respondents who are unsure about accepting the bid, non-response bias, the use of mail surveys and aggregation. Each of these issues are discussed in turn below.

8.6.5.1. Elicitation Method

The open-ended (OE) approach employed in the pilot revealed a mean WTP of £75 per household, while mean WTP in the Discrete Choice (DC) survey was close to four times this figure. The divergence between the two question formats is frequently reported in the literature and is a form of procedural variance, (i.e. a quantitative difference in value for the same theoretical construct, measured using normatively similar procedures). Langford and Bateman (1993), for example, reported that 97% of all respondents gave an OE WTP of less than 500, yet only 79% refused this bid level in the equivalent discrete choice survey, with mean WTP from the OE study approximately half of the DC estimate. While DC estimates are sensitive to the form of the model used (Boyle and Bishop, 1988) this divergence appears to be consistently reported (Seller *et al.,* 1985; Walsh *et al.*, 1989; Kealy and Turner, 1993).

A number of reasons have been proposed to help explain this anomaly. Anchoring, whereby respondents link their valuation to the bid level offered is considered to be a prevalent strategy among some respondents, particularly those that are uncertain about their valuation and may consider the bid to be an implied value cue (Kahneman et al., 1982). Since mean WTP, in the presence of anchoring, is affected only by the location of the bid vector there is no *a priori* direction for anchoring bias (Kristrom, 1993). Thus, anchoring alone cannot account for higher mean WTP estimates from DC.

The tendency for 'yea-saying' amongst respondents can upwardly bias WTP estimates (Kanninen, 1995). 'Yea-saying' occurs where a respondent accepts the offer price even though their actual WTP is less than this amount. It is thought to be prevalent amongst respondents who wish to register their environmental vote. In other words they are reluctant to say 'No' since this would result in their positive preference for the good being ignored (Loomis, 1987). Respondents who are uncertain of their WTP, or are reluctant to research their preferences may also say 'Yes' because they prefer to give a socially acceptable response to please the surveyor, or if the bid level falls within their WTP 'range'.

Question format clearly matters and it remains a contentious issues for CVM methodology. While DC is incentive compatible and more closely resembles a market situation, open ended estimates provide a conservative estimate which policy makers may find more 'believable'. A comparison between OE, DC and real payments would provide some indication of the more reliable method, but this would have to be strictly controlled for other survey influences.

8.6.5.2 Unsure and Protest Respondents

Respondents who were unsure (i.e. who answered 'Don't Know' to the payment question) and protestors accounted for a quarter of all responses received, yet their preferences for environmental recovery are unknown and have, so far, been ignored in the analysis of the survey data. These two groups are now examined in more detail below.

8.6.5.2.1 Protesters

Protestors accounted for only 5% of all respondents. Protest respondents are those respondents who may, or may not, place a positive value on the environmental change described, but have indicated that they are unwilling to pay anything because they objected to one or more aspects of the scenario presented in the questionnaire. The range of reasons given by protesters for not paying are presented in Table 8.19.

The most common reasons for protesting were that '*industry should be made to pay*' or that it was '*a government responsibility*'. Although the questionnaire emphasised mutual responsibility for acidification damage, some respondents clearly felt that they were not personally to blame. Only 26% of all protests respondents were unhappy with particular aspects of the contingent market established in the questionnaire (payment method, and information given). This suggests that the instrument was reasonably effective.

Mitchell and Carson, (1989) observe that many protestors are of low income and education, and that mean WTP may be biased upwards if they are dropped from the sample. The possibility for such a bias was investigated by comparing the average values for all covariates, which significantly influenced WTP in the step-wise regression model, among protestors, respondents unwilling to pay anything (non-payers), and respondents who rejected or accepted, the bid offered in the payment question. This comparison is shown in Table 8.20.

Protestors appear to be quite dissimilar to non-payers with respect to some important determinants of WTP. For

Table 8.19 Reasons for protesting

Reason	% of Protestors
Industry should pay	38
Government responsibility	28
Payment method	7
Don't trust everyone will pay	5
Disagree with information given	19
Others	3
Total	100

Table 8.20 Comparison of covariate values between Protestors, Non-payers, No's (bid refused) and Yes's (bid accepted), and 'Don't Know's.

Covariate	Respondent Type				
	Protest	Non-Payers	No's	Yes's	Don't Know's
income	3.48*	2.66	2.32	4.26	3.33*
understand	1.53*	2.71	1.98	1.48	3.03†
member	0.22**	0.09	0.15	0.24	0.17
pollu	4.13	4.26	4.07	3.68	4.03
govt	1.70*	2.19	1.98	1.65	1.84
abroad	4.93	4.10	4.96	8.01	5.17

* Significant difference from non-payers at 95% level

** Significant difference from non-payers at 99% level

† Significant difference from Yes's at 95% level example, income, and membership of environmental charities, which are both positively correlated with WTP, are significantly higher among protesters. There is therefore no evidence from this study to support the claim that protestors should be treated as zero-bidders.

8.6.5.2.2 Unsure Respondents

Over twenty percent of all respondents answered 'Don't Know' to the payment question. This is a far greater proportion than one might expect from theory to be truly indifferent to the environmental change at the offered bid price (Svento, 1994). Although unsure respondents were not asked to explain their reasons for being uncertain, it is likely that many simply could not make up their minds under the constraints imposed on them by the survey process. Since their preference remain undetermined these respondents, as with protesters, were dropped from the sample.

Given the number of respondents in the 'Don't Know' group in relation to the total sample size, the potential for biasing the mean WTP either up or down by excluding this group is considerably greater than with the protest respondents. In Table 8.19 the average covariate values of unsure respondents are also presented. Comparison with the other respondent groups suggest that unsure respondents are broadly similar to those respondents who either accepted or rejected the offered bid, rather than non-payers. They significantly differ from Yes respondents only with respect to the level of understanding they have about acid rain, a characteristic which probably explains their response to the payment question.

Since unsure respondents do not share similar characteristics to non-payers and cannot be identified as either potential Yes or No bidders, it seems appropriate to exclude them from the analysis. However, given the proportion of respondents in this group, future research effort should be directed at investigating the Don't Know' group in more detail. This could be accomplished through a follow-up survey by telephone, or where this is not feasible, by introducing a follow-up question which seeks to explore the cause of their uncertainty.

8.6.5.3 Negative Bids

In this survey it was decided at the outset to exclude negative bids because they were considered to be unrealistic in relation to the environmental scenarios involved. Also, considerable difficulties are associated with asking WTA questions (Kahneman, 1986). Comparison of the fitted curves and estimates of mean WTP indicate that the reciprocal bid transformation should be used wherever possible. The other alternatives are less satisfactory. Left truncation provides a poorer fit to the lower tail of the logistic, while a log transformation generates extremely high estimates of WTP. One solution to this problem would be to estimate the mean log(WTP) and then back transform. however, this is a strongly biased estimate (Buckland *et al.*, 1996).

The most satisfactory approach would be to allow negative bids, and elicit WTA from a sample of respondents so that the logistic can be fitted without the need for transformation or truncation. Although this may present problems in establishing a feasible market context, WTA compensation also provides the opportunity to establish the opportunity cost of the environmental change. In many situations involving landscape, and land use change, for example, where the *status quo* may be preferred by some individuals, it may be necessary to include WTA questions in order to estimate the net benefits of the change.

8.6.5.4 Bid Design

The discrepancy between open-ended and discrete-choice elicitation methods also created a problem for the original bid design based on Cooper's approach. The very high percentage of respondents (47%) who were prepared to pay the top bid of £396 meant that the upper tail of the bid curve was not identified and a further mailing was required. If this had not been possible, an alternative strategy would have been to truncate the data at the highest bid level (Park *et al.*, 1991) or to 'pinch' the upper tail so that the cumulative density function reaches 1 at a some higher bid level (Ready and Hu, 1995). However, these options are not very satisfactory. Truncation generates a lower bound estimate of mean WTP which is very sensitive to the point of truncation, while 'pinching' involves extrapolation beyond the range of the data.

A better approach would be to implement the DC survey in two stages. In the first stage only half of the bids would be used, but selected to cover a wide spread of money values. Data from this stage can then be used, if necessary, to adjust the remaining bid levels to ensure that the upper tail is identified.

8.6.5.5 Mail Surveys

Mail surveys are generally the cheapest way to implement CVM studies. However, the NOAA Panel (1993) favoured in person or telephone interviewing since they considered them more likely to give reliable benefit estimates. This recommendation was not supported with evidence from comparative studies of reliability, but stemmed primarily from perceived weaknesses of mail surveys such as low response rate, biased sample listings, reliance on self reporting, and problems with question order effects.

In the context of this study some of these criticisms appear to be unfounded. Firstly, although mail surveys often generate response rates of around 40% (Loomis, 1987) the overall response rate in this study was 67%. This rate of return is attributable to the strict implementation of the Total Design Method (Dillman, 1979) and careful pre-testing of the survey instrument. Face to face surveys are often assumed to achieve a response rate of 100% but actual response rates, based on the number of people approached and refused to participate is rarely recorded. Evidence from follow-up surveys of non-respondents by telephone also suggest that the WTP of recalcitrant households are not significantly different from respondents, with general, rather than survey specific reasons, leading to non-response (Fredman, 1994)[10]. Overall, therefore, non-response bias in mail surveys is probably overstated as a problem with all mail surveys.

The criticism that mail samples drawn from listings are biased because some social classes may be under-represented, may be less important in the UK than the US. For example, in this study the sample was drawn form the national telephone directory which lists approximately 90% of all Scottish households (British Telecom, pers com). The equivalent figure for the US is less, due to the greater proportion of non-listed or incorrect numbers, and non-phone households. Electoral roles also provide a reasonable basis on which to sample, but at this point in time, is probably biased because many low income households dropped out of the register to avoid the 'poll-tax'.

Self-reporting seems to be a viewed as a disadvantage of mail surveys on the assumption that only individuals favourably pre-disposed to the environment will respond. This criticism seems unfounded, and an equally plausible and less worrying explanation, from the point of view of mail surveys, is that only people who are willing to fill in forms are likely to respond.

The NOAA panel were also concerned with question order

[10]The conventional strategy of assigning zero WTP (Bishop and Boyle, 1985) to non-respondents is likely too produce a mean WTP that is strongly biased downwards.

effects. The inability of mail surveys to control for question order effects is certainly a drawback but one which can be overcome by designing the instrument to be free of such effects. However, where double-bounded formats, and other more elaborate survey designs are required, in person approaches are probably most suitable.

Mail surveys also offer other advantages over in-person approaches, apart from cost. Perhaps most important, is that mail shots give people time to process the information, research their preferences and decide about their response. Face to face approaches on the other hand, can involve awkward social exchanges between interviewer and interviewee which does not simulate real life market situations (Cummings *et al.,* 1986). Under pressure respondents in a face to face interview, may also not attempt to research their preferences, but will instead be influenced by spurious, irrelevant factors which can exaggerate symbolic bias and introduce upward bias through a desire to please the surveyor ('yea-saying').

8.6.5.6 Unbiased Aggregation

The issue of aggregation to the population level from discrete choice data has received little attention in the literature. Loomis, (1987) looked at difficulties arising from low response rates, and weighting to account for a biased sample. No study has yet addressed the bias associated with the conventional approach of averaging across the covariates in a non-linear model. In this study an unbiased mean was estimated by predicting the average probability of a Yes response at each bid level using the covariate data from individual respondents. Unfortunately, this approach requires a relatively large sample size and is unlikely to be successfully applied to samples of less than 100.

An alternative approach would be to substitute the covariate values for each respondent in turn into the logistic regression in order to obtain a logistic curve for each individual which is a function of bid alone. Assuming the respondents are a random sample from the population, the average across all individuals is an estimate of the mean WTP. However, this approach assumes that only the intercept of the curve changes across individuals, whereas a change in shape might also occur, in which case some bias can be expected (Buckland *et al.,* 1996).

8.7 Conclusions

The results of the CVM exercise reported in this chapter predict that recovery from acidification in the semi-natural environment will give rise to substantial non-use benefits amongst the general public. WTP for abatement was highest when more severe damage was predicted under the status quo, but was not significantly affected by the level of recovery, or the rate of recovery. Concern with future losses is consistent with research in other fields and suggests that further research be directed at investigating the influence of possible irreversible damage (e.g. extinction of species) on public perception of environmental change.

In addition to non-use benefits, recovery from acidification can also be expected to give increased fish catch from the Scottish salmon fishery. From the point of view of the policy maker these user benefits are important to estimate because of their implications for the rural economy of Scotland, and because they represent hard monetary estimates of economic benefits. In Chapters 9 and 10 the development and results of a bioeconomic model to predict the benefits of recovery in salmon fisheries is described.

CHAPTER 9 ESTIMATING THE BENEFITS TO THE ROD AND LINE SALMON FISHERIES: METHODOLOGY

9.1 Introduction

One of the earliest, and most dramatic, effects of acidification in the semi-natural environment was the loss of fish populations. The decline in the Atlantic salmon *(Salmo salar)* has been of particular concern because of its importance to the rod and line (R&L) salmon fishery, which contributes over £30 million annually to the rural economy in Scotland (Mackay, 1989).

Salmon recruitment in the upland headwaters of acid-sensitive areas has been particularly affected by acidification, with substantial declines observed in major salmon rivers in south-west Scotland (Waters and Kay, 198[illegible]). Since average annual salmon catch is one the most important variables influencing the value of a fishery (Radford, 1991), this decline is likely to have resulted in a substantial decrease, in real terms, in the market value of the Scottish salmon fishery. Abatement of SO_2 should therefore, lead to economic benefits through it's effect on salmon recovery, and hence fish catch.

Estimating the marginal benefits of recovery in r&l salmon fisheries presents the economist with several challenges. Firstly, complex modelling of dynamic, environmental processes, over very long time periods is required to link reductions in sulphur emissions to improvements in water chemistry. Secondly, although considerable scientific evidence has emerged to suggest that anthropogenic acidification has caused serious denudation of salmon stocks, the complexity of salmon population dynamics and the absence of reliable stock, catch, and effort data, both for the high seas and rivers, has frustrated attempts to quantify the effect of changes in water chemistry on fish catch (Waters and Kay, 1988). Previous studies have, therefore, generally used fairly simplistic damage functions, such as binary fish/no fish responses to estimate the economic cost of acidification (e.g. Mullin and Menz, 1985).

This chapter describes the approach taken to predicting the economic benefits to the R&L salmon fishery. The research was carried out in collaboration with Dr. R. Ferrier at MLURI, who provided the water chemistry data from the MAGIC model.

9.2 General Approach

The modelling approach used to estimate the marginal economic benefits arising from higher market values for the R&L salmon fishery involved three main stages:

(1) Prediction of changes in water chemistry and fish population status in response to reduced SO_2 emission

levels using the MAGIC model.

(2) establishing a relationship between changes in fish population status and fish catch per unit effort, and

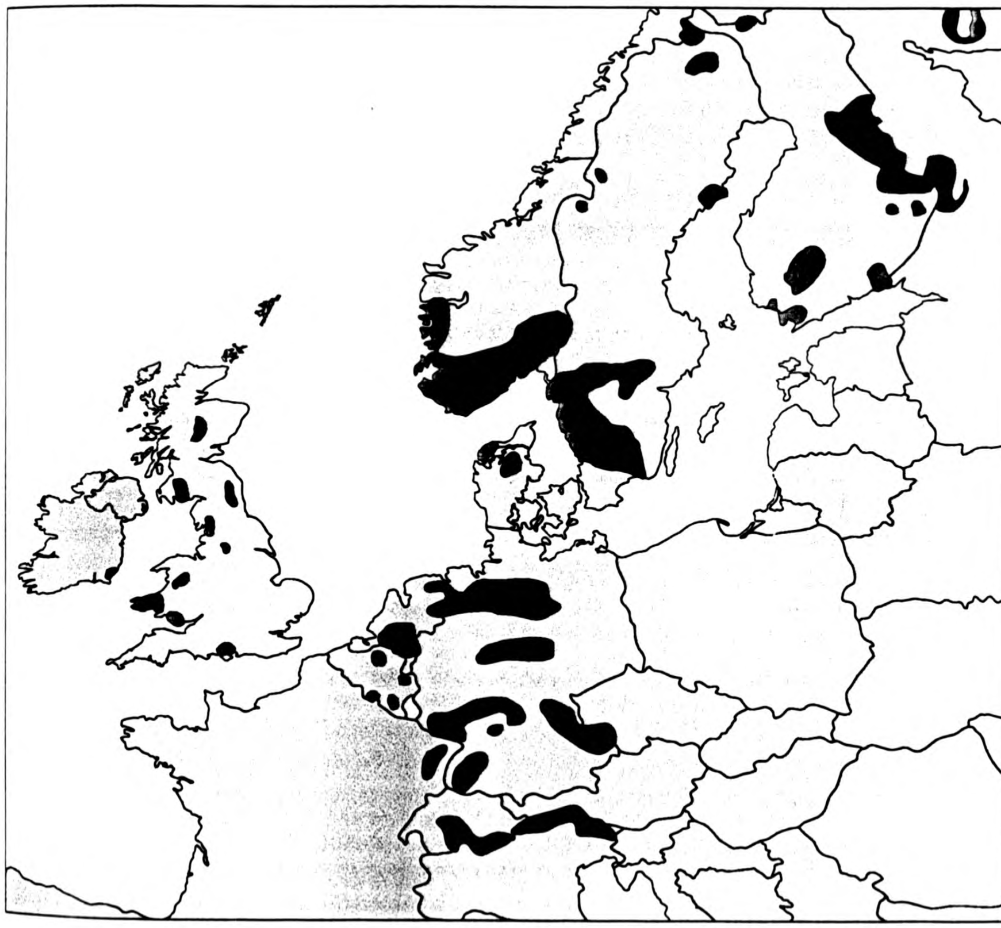
(3) predicting the effect of changes in average, annual salmon catch on the capital value of the fishery through a hedonic price model.

9.2.1 Study Area

The study area chosen to develop the methodology was the Galloway R&L salmon fishery in south-west Scotland. Critical Load maps recognise Galloway to be one of the most acidified regions in Europe (Figure 9.1). Evidence for the acidification of the headwaters has also emerged from water chemistry samples which indicate that many the region's rivers regularly fail to meet the standards of pH established under the EC's Directive (78/659/EEC) on the quality of freshwaters for fish life (Solway River Purification Board, 1992).

The uplands of Galloway is an acid-sensitive environment where slow-weathering, granitic bedrock is overlain by shallow, organic-rich, acidic soils with only a limited capacity for neutralising acidity (Wright *et al.*, 1994). Acidification in the area has also been exacerbated by the expansion of commercial forestry plantations, which enhance the process of acidification by increasing the interception of pollutants, and by removing base cations from the soil as the crop canopy expands (Miller *et al.*, 1991).

Figure 9.1. Acidified areas of Europe



Although not one of Scotland's most prestigious locations for salmon angling, a number of major salmon rivers arise in the Galloway hills which, taken together, account for approximately 10% of the total Scottish Catch (SOAFD, 1987-1991). Considerable evidence now exists of the damage done by decades of anthropogenic acid inputs to salmonoid fish populations and other aquatic organisms (Harriman *et al.*, 1987, Flower *et al.*, 1988). Waters and Kay, (1987) report that rivers in the Galloway region have suffered a 50% reduction in the average salmon and sea trout catch during the period 1972 to 1981 and suspect that acidification may be to blame. This is supported by anecdotal evidence which indicates that the upper reaches of rivers which are most affected by acidification, have experienced the greatest decline in fish recruitment and catch (Stephens, pers com).

9.3 Changes in Water Quality and Fish Populations

The effects of reducing SO_2 emissions on soil and water processes and fish health in Galloway was predicted using the MAGIC model.

9.3.1 The MAGIC Model

MAGIC (Model of Acidification of Groundwaters In Catchments) is a process-oriented, intermediate-complexity model which has been widely used in the UK and North America for predicting future water chemistry at the catchment level (Cosby *et al.*, 1990). The following features of MAGIC make it particularly suited to modelling acidification effects on salmon fishing in Galloway.

1) A dynamic fisheries population model is incorporated into the framework of MAGIC. The algorithms of the population model are based on a statistical evaluation of the salmonoid fish population status from the extensive Norwegian thousand lakes survey (Henriksen *et al.*, 1990, Bulger *et al.*, 1993).
2) MAGIC takes account of both the past rate (accumulated pollution) and the current rate of deposition when predicting future environmental recovery following abatement.
3) Output can be validated by comparing re-constructed acidification history of individual catchments from diatom records with hindcast predictions from the model.
4) Predictions of water chemistry under alternative SO_2 abatement strategies can be made over decades to centuries.
5) The interaction between land use, particularly coniferous plantations, and acidification processes is incorporated.

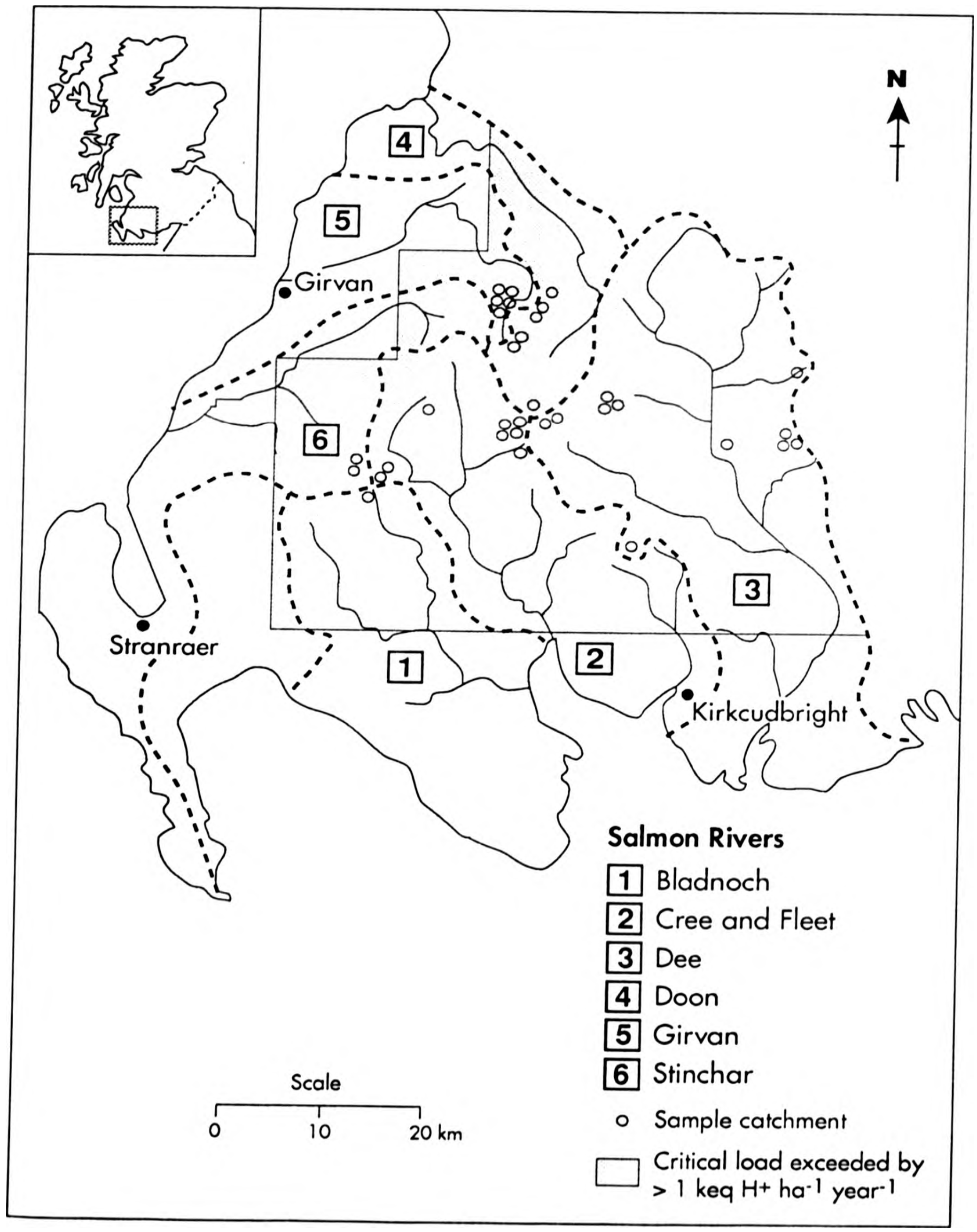
9.3.2 Application of MAGIC to Galloway

The MAGIC model was run for the Galloway region using data gathered from a sample of over 30 individual upland catchments, located in areas considered to be vulnerable to acidification. In the absence of detailed information on the extent of acidification damage in Galloway, the affected area was estimated from the Critical Load map for soils which identifies the location of soils where acidic inputs exceed the buffering capacity of the soil (Department of Environment, 1992a). A substantial proportion of the upper reaches of 6 major salmon rivers (Cree & Fleet, Doon, Bladnoch, Dee, Girvan and Stinchar) are included within this area (Figure 9.2).

Input data for each sample catchment included deposition chemistry (Na, M, Ca, K SO_4, NO_3, NH_4^+, Cl, and H) from projections made by the Harwell Trajectory model, soil chemistry variables including Cation Exchange Capacity (CEC), depth, bulk density, and porosity, and land use history (Wright *et al.*, 1994). These data, summarised at the catchment level, were incorporated into a few readily described equations which summarise the many chemical and biological processes active in the catchment. These are:

(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, and the dissolution and precipitation of aluminium, inorganic and organic carbon.

Figure 9.2: Location of sample catchments in the Galloway Fishery



(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 (Ca, M, Na, K, NH_4^+, Cl, SO_4, NO_3), mineral weathering, net uptake in biomass, and loss in runoff. The effects of afforestation on net uptake of cations from soil and on dry and occult deposition are also incorporated.

(3) Predictions of fish population status depend upon water chemistry variables such as pH, alkalinity, and acid neutralising capacity (ANC). The output describes the probability that the fish population will fall into three distinct categories (Wright *et al.*, 1994): (i) Healthy (a vigorous population unaffected by acidic deposition), (ii)Marginal (a sparse population, either historically thin or damaged by acidic deposition), and (iii) Extinct (population lost).

Uncertainty in the estimation of output water chemistry parameters is dealt with using a Monte-Carlo technique which selects values within specified ranges for each parameter. The optimisation procedure is carried out 10 times to produce 10 calibrated models for each catchment.

For this study the model was used to predict changes in water chemistry, and predicted fish population status, in the acidified waters of the Galloway fishery over a 50 year period up to 2038. Three SO_2 abatement scenarios were investigated. These were:

1 *Status Quo*: Emission levels to remain constant at 1988 levels.

2 60% Abatement: Emissions reduced by 60% from 1980 levels by 1993. This is equivalent to the UK's current commitment to reduce acidification under the Large Combustion Plant Directive of the EC.

3 90% Abatement: Emissions reduced by 30% in 1993; 60% by 2003 and 90% by 2008. This abatement scenario is believed to be consistent with the an emission level that will not cause damage to the semi-natural environment (UKAWRG, 1988).

9.4 Estimating the Effect on Catch Levels

One approach to predicting future catch levels would be to establish a predictive function based on observed historical changes in water chemistry and fish catch per unit effort. While MAGIC could be used to reproduce acidification history for individual catchments, modelling the effect of acidification on fish catch would require information on both population abundance and fishing effort.

Identifying the specific effect of acid rain on salmon abundance alone, would be extremely demanding due to the complex life-cycle of the Atlantic salmon. After hatching in freshwater they migrate seawards to distant ocean feeding grounds before returning, after several years, to their river of origin to spawn. During this cycle the salmon are exposed to a wide variety of biological and environmental stresses and large natural fluctuations in the size, age structure, and timing of returning populations are common (Shearer, 1987). During the years 1981 to 1984 for example, the percentage of salmon entering the North Esk, a river in eastern Scotland, during the netting season varied between 23% and 44% (Dunkley, 1986).

Any attempt at modelling historical changes in salmon abundance as a consequence of changes in acidification status would, in any case, be futile due to the complete absence of reliable data on fishing effort, both on the high seas,[1] and in estuaries and rivers (Williamson, 1987). Improved efficiency of commercial netting stations, and the closure of unprofitable stations are believed, for example, to have caused large fluctuations in R&L catches in some salmon rivers (Waters and Kay, 1988).

In order to establish a relationship between improvements in water chemistry and, catch per unit effort, it was necessary to turn to a more controlled fishing system where many of the external influences on fish status are excluded, and for which reliable records of catch per unit of effort were available. Correspondence with local angling clubs revealed that detailed catch data, per unit of effort, were available from the Balloch Fishing Club for Loch Reicawr, one of the catchments sampled in the Galloway application of MAGIC. The records were for the period 1945-1970, and included detailed information on catch and catch

[1]SOAFD does collect data on annual salmon catch but not on effort.

effort (boat days).

Average catch per unit of effort (C) was estimated for each year and regressed against a range of water chemistry and fish health parameters, predicted for Loch Reicawr from a hindcast projection by MAGIC. The closest correlation was found to be between C and H (the percentage probability of there being a healthy fish population achieving a healthy status). This relationship is described in Figure 9.3 and Equation 9.1 with summary statistics presented in Table 9.1.

$$C = 24.2 - 0.974\ H \qquad \textit{Equation 9.1}$$

where C = average fish catch per boat day;

H = % probability of the fish population being healthy, as predicted by MAGIC using water chemistry variables.

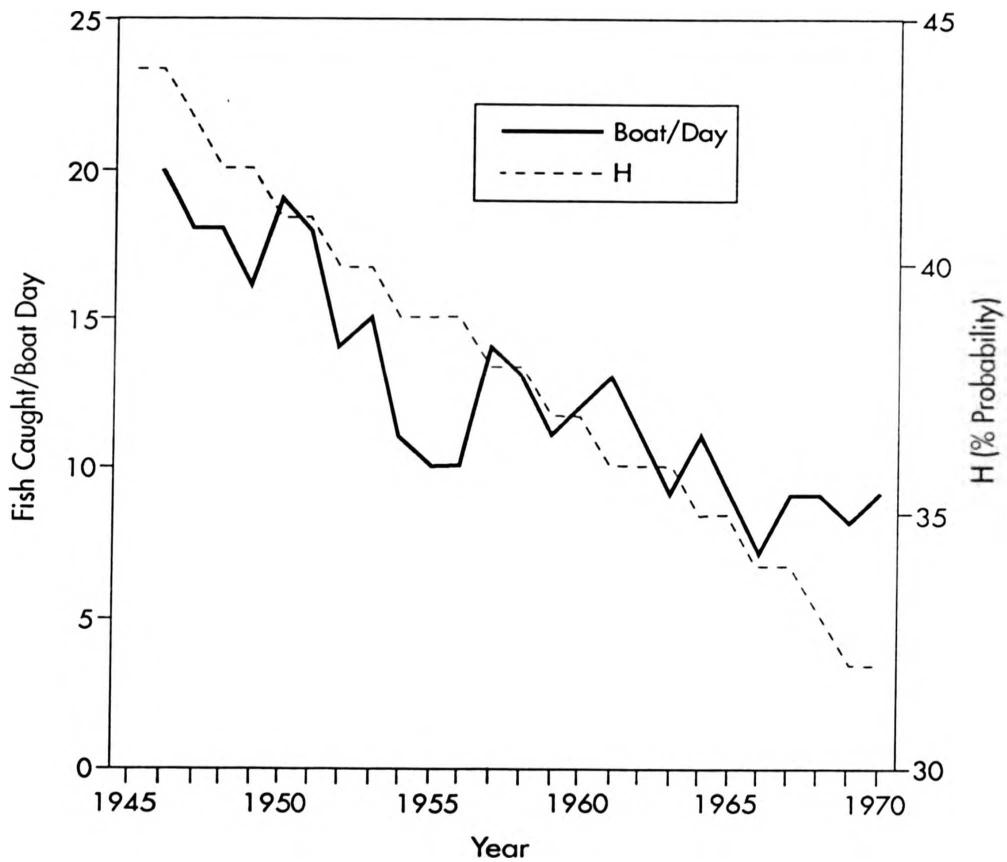
Lacking data on fish catch per unit effort from other catchments it was necessary to assume that this relationship reflected the response of catch, to changes in H, for all waters in the fishery. On this basis the percentage change in C at 5 year intervals for the period 1988-2038 predicted from the estimated average value of H for the Galloway fishery as a whole (H_T), could be translated into increases in the actual number of fish caught over this period in the fishery (C_T). This was considered to be a more realistic approach than estimating changes in catch only from the

Figure 9.3

Table 9.1 Summary statistics for Equation 9.1

Variable	Coefficient b_H	Standard Error
H	.974	.114
Constant	24.2	4.386
F = 61.59 sig. .000		$AdjR_2$= .71

Figure 9.3: Relationship between H (% probability of a healthy population) and fish caught per boat day



acidified areas of the region (see Figure 9.2). This was necessary for two reasons. Firstly, catch data are only published for Salmon Fishery Districts, which generally include one or more recognised salmon rivers. Data are not available for acidified river stretches alone. Secondly, even if it were possible to estimate the proportion of total catch from these stretches, changes in water chemistry following abatement also lead to changes in water chemistry in downstream areas of the catchment.

Since there were no data on water quality outside the affected area, it was necessary to assume that the unacidified area had a value for H of 1. A regional value for H (H_T) was then calculated by weighting the predicted values for H by their respective catchment areas[2].

9.5 Economic Value of Enhanced Salmon Catch

In Galloway, as in other salmon fisheries, the five or ten-year average salmon catch is an important determinant of beat quality[3], and hence, value. Higher average catches due to improved water chemistry as a consequence of SO_2 abatement would be expected to lead to a higher WTP amongst potential

[2] The values of H for each catchment are shown in Appendix 3

[3] A salmon beat represent separate stretches of a river which can be let out and fished individually. Each beat is different with respect to physical characteristics and catch, thus beats can vary in value even within a single ownership.

purchasers for the right to catch salmon in the Galloway fishery. The resultant change in the market value, predicted by the HPM would therefore represent a measure of the user benefits to the salmon fishery resource of SO_2 abatement.

9.5.2 Application of the HPM to the Galloway Fishery

The HPM has been widely used by economists to estimate the benefits of environmental improvements in other capital markets, particularly for residential properties (Willis and Garrod, 1991; Brookshire *et al.*, 1982). It derives from Lancaster's, (1966) theory of value which proposes that each commodity can be described by a combination of its characteristics, one or more of which may be linked to environmental quality. Non-linear regression methods can be used to establish the marginal change in value associated with changes in environmental quality from the observed variation in the prices of market goods (Hanley and Spash, 1993).

The Hedonic approach requires several assumptions. Firstly, a fully functioning market is assumed to exist for salmon beats (i.e. anglers are able to select from a range of beats which exhibit a comprehensive combination of different characteristics likely to influence the satisfaction derived from ownership). Also, the extra cost of securing the right to fish the chosen beat, rather than some other fishery with a lower catch, is also assumed to equal his/her maximum WTP for the additional catch.

In this study cross-sectional, or ideally time series data, would have to be collected on the price (P_b) of salmon beats offered for sale in the Galloway region, together with data on attributes of the individual beats, such as average salmon catch (S_b), number of holding pools (H_b), number of rods usually allowed (R_b), access (A_b), location (L_b), and prestige 'value' of the beat (T_b). A hedonic price function of the form below could then be estimated using multiple regression methods:

$$P_b = f\ (S_b,\ H_b,\ R_b,\ A_b,\ L_b,\ T_b) \qquad \textit{Equation 9.2}$$

The above hedonic price function estimates a point on each angler's demand curve, with the slope of the relationship representing a locus of points on the demand curve of many anglers.

Unfortunately, as salmon beats are only infrequently traded in the open market, there is relatively little data on which to develop a Hedonic Price Model for the Galloway fishery. However, a previous study by Radford *et al.*, (1991), has estimated the marginal value of a change in the annual salmon catch in England and Wales with the Hedonic approach. Using a mail survey of all fishery rod owners in England and Wales, the market value of individual beats, based on an estimated sale value suggested by the owner, was established.

The predicted market values were regressed against a range of factors thought to influence the market value.

Significant variables were selected through a step-wise procedure. The best fit model is given below, with standard errors and f-statistic presented in Table 9.2:

$$\ln P_b = 1.550 + 0.547 \ln S_b + 0.423 \ln N_b + 0.337 D_b \quad \textit{Equation 9.3}$$

where $\ln P_b$ = sale value of the salmon beat b.

$\ln S_b$ = log salmon average catch for each beat

$\ln N_b$ = log number of named pools for each beat

D_b = double bank dummy variable for each beat

This is a multiplicative log-linear relationship where the increase in value is dependent on the level of catch taken, that is, as the level of catch increases, so the market value of the beat increases, but at a declining rate (Figure 9.4). Hence, as one might expect from theory, the marginal value of an additional salmon caught depends on the overall level of the catch.

The implicit price of a change in salmon catch can be derived by partially differentiating Equation 9.3 with respect to lnS, the log of average salmon catch *(ceteris paribus)*. Thus a one unit increase in the salmon catch variable will result in a 0.547 unit increase in the price variable $\ln P_b$.

A second stage of the hedonic technique is required to combine the quantity and price information to identify the inverse demand curve (Freeman, 1993). This can be done by regressing implicit prices for catch against lnS, relevant

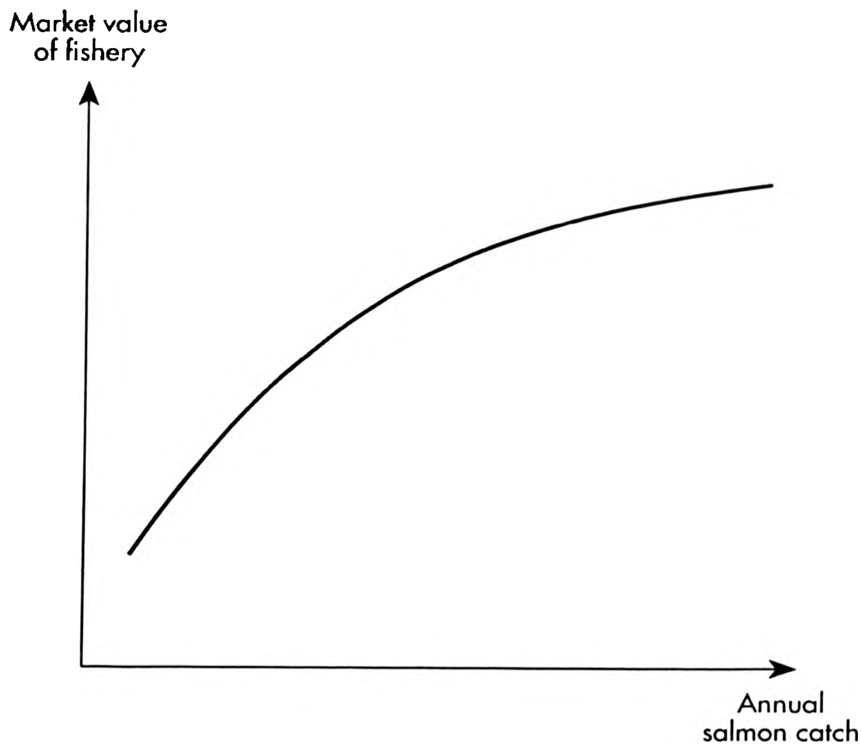
Table 9.2 Summary statistics of Equation 9.3

Variable	b	se b	T	Sig T
constant	1.550	0.173	8.963	.000
lnS	0.547	0.062	8.820	.000
lnN	0.423	0.096	4.424	.000
D	0.337	0.138	2.445	.016

F = 70.392   AdjR_2= .564

Sig F = .000

Figure 9.4: Relationship between average annual salmon catch and market value of the salmon fishery



socio-economic variables, such as income, and age, and other taste or preference variables (Hanley and Spash, 1993)[4]. However, in common with many other empirical applications, since marginal changes in the market value of the fishery were of primary interest in the Radford study, this second stage was not implemented.

Changes in the market value of the Galloway fishery, for any specified time i, was estimated in Equation 9.4 from the change in fish catch (predicted from Equation 9.1), using Radford's model.

$$\ln P_i = \ln P_0 + 0.547(\ln C_{Ti} - \ln C_{T0}) \qquad \textit{Equation 9.4}$$

where $\ln P_0$ = Natural log of the estimated market value of the salmon fishery in Year 0

$\ln P_i$ = Natural log of the estimated future market value in next time period.

$\ln C_{T0}$ = Natural log of the estimated 5-Year average salmon catch of salmon fishery in Year 0

$\ln C_{Ti}$ = Natural log of the predicted future 5-year average salmon catch in next time period.

---

[4] If one assumes that all agents are identical with respect to income and their utility functions then the implicit price function is the inverse demand function. It is also necessary to control for the supply side in the hedonic model. However, in the case of salmon beats, supply can be considered to be fixed since most salmon rivers are fished to capacity (Mackay, 1989) and it is unlikely that new beats could created in response to improved salmon catch (Stephen, pers. com.).

The predicted market value could then be obtained by taking the exponential $\ln P_i$.

9.5.3 Initial Market Value of the Galloway Fishery

Since no figures are collected on the market value of salmon fisheries in Scotland it was necessary to estimate the initial market value of the salmon fishery (P_t). This was done, as in the Radford study, by multiplying the estimated capital value of a salmon by the average, annual total salmon catch for the Galloway fishery. Rather than use Radford's estimate of capital value for England and Wales, it was decided to use the comparable Scottish figure of £3420 per salmon, produced by Mackay, (1989).

CHAPTER 10 SALMON FISHERY BENEFITS: RESULTS AND DISCUSSION

In this chapter the results of the modelling exercise to predict the economic benefits of recovery in the Galloway fishery are presented for alternative SO_2 emission abatement levels. Issues of accuracy and reliability, together with policy implications are discussed.

10.1 Predicted Change in Fishery Status.

The effect of alternative abatement scenarios on fish population status for the Galloway fishery (H_T), is presented in Table 10.1 and Figure 10.1. Under the *status quo,* the MAGIC model predicts a gradual decline in H_T from a value of 0.620 in 1988 to 0.592 in 2038. A rapid recovery in fish population status, under both 60% and 90% abatement levels, is predicted with H_T rising to 0.692 and 0.710 respectively by the year 2028. However, this recovery is not sustained, and population status declines gradually between 2028 and 2038. Many sensitive upland catchments would, therefore, appear to be unable to adequately buffer acidic inputs, even under very low levels of atmospheric inputs of SO_2.

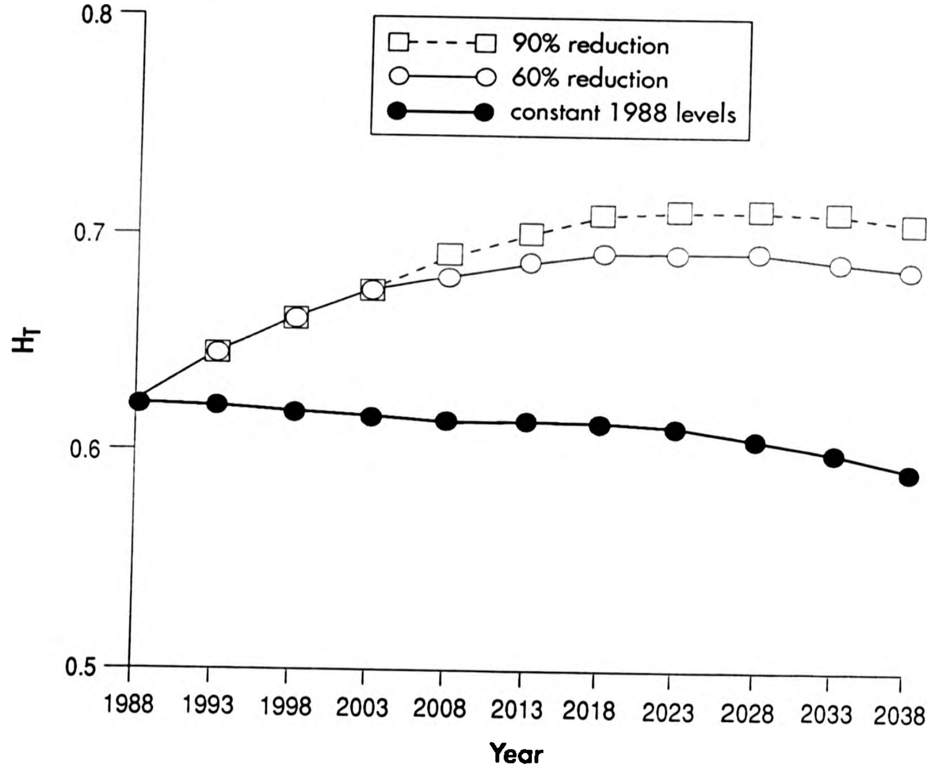
10.2 Predicted Change in Salmon Catch

The changes in the average 5-year salmon catch from the

Table 10.1: Predicted change in fish population status (H_T) for the Galloway Fishery under alternative abatement scenarios

YEAR	STATUS QUO 0%	60%	90%
1988	0.620	0.620	0.620
1993	0.620	0.644	0.644
1998	0.617	0.660	0.660
2003	0.616	0.674	0.674
2008	0.613	0.681	0.689
2013	0.614	0.687	0.701
2018	0.613	0.692	0.710
2023	0.611	0.692	0.711
2028	0.605	0.692	0.711
2033	0.599	0.688	0.710
2038	0.592	0.685	0.706

Figure 10.1: Predicted change in fish population status (H_T) for the Galloway fishery under alternative abatement scenarios



Galloway fishery (C_T) predicted from Equation 9.1, using the values of H_T, are presented in Table 10.2 and Figure 10.2. Under the *status quo*, catch levels drop from 3676 salmon in 1988 to 3000 salmon in 2038, which is equivalent to a 10% decline. Following abatement, salmon catch rises rapidly to a peak of 4706 fish under a 60% reduction, and 4977 fish under a 90% reduction in SO_2. By 2038 fish catch has increased by a total of 25% and 33% respectively, on 1988 levels and is 39% and 48% higher than predicted catch under the *status quo*.

10.3 Predicted Change in Market Value of the Galloway Fishery

The effect of the predicted change in salmon catch on the market value of the Galloway salmon fishery is presented in Table 10.3 and Figure 10.3. Under the *status quo*, the market value of the fishery declines gradually from £12.6 million in 1988 to £11.8 million in 2033.

By contrast 60% and 90% reductions in SO_2 levels initiate a rapid recovery in market value. Under a 60% reduction, the market value of the fishery rises to a peak of £14.4 million in 2028, falling back to £14.2 million by the end of the forecast period. This represents a 14% increase over the estimated 1988 value during the 50 year period, and a net gain over the status quo of £2.44 million (21%).

Table 10.2 Predicted change in 5-year average salmon catch (C_T) in the Galloway Fishery under alternative abatement scenarios

YEAR	STATUS QUO 0 %	60%	90%
1988	3676	3676	3676
1993	3676	4009	3901
1998	3629	4242	4102
2003	3606	4451	4350
2008	3575	4544	4660
2013	3583	4629	4830
2018	2575	4698	4954
2023	3544	4706	4977
2028	3451	4698	4970
2033	3366	4644	4946
2038	3300	4598	4900

Figure 10.2: Predicted change in 5-year average salmon catch (C_T) in Galloway fishery under alternative abatement scenarios

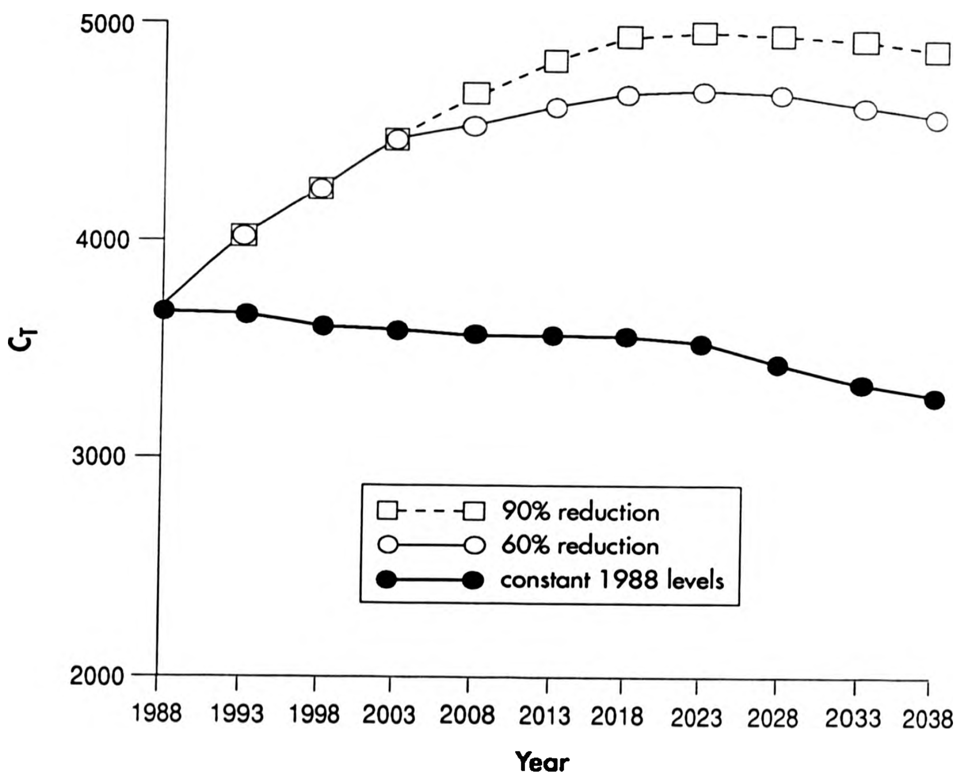
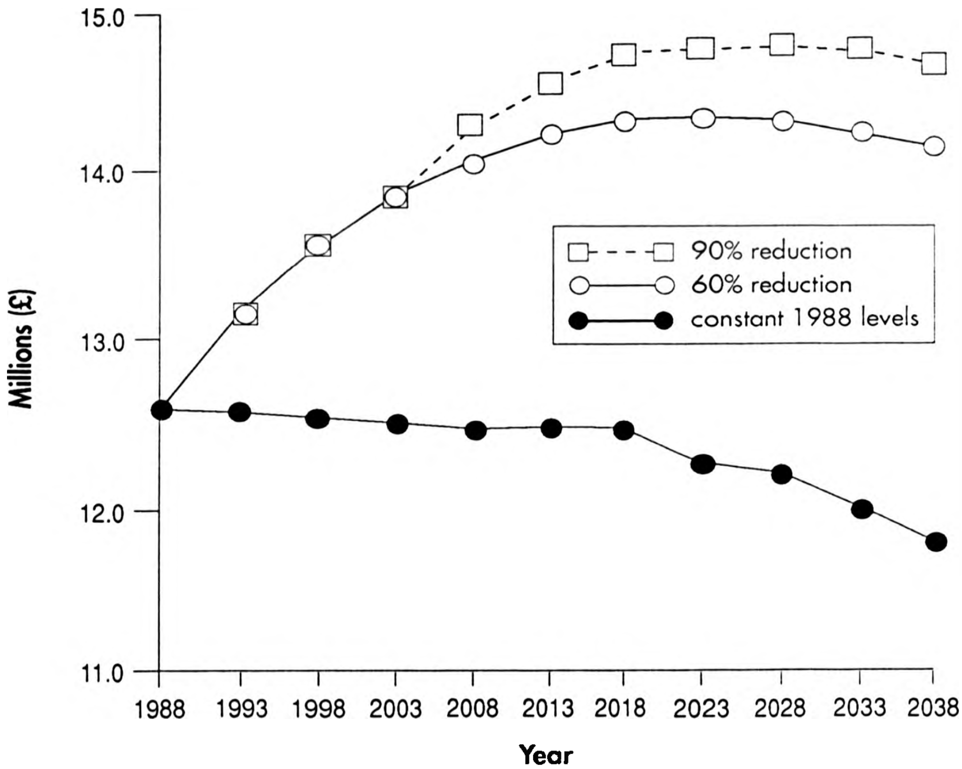


Table 10.3 Predicted market value (£ Millions) of the Galloway Fishery under alternative abatement scenarios

YEAR	0 %	60%	90%
1988	12.60	12.60	12.60
1993	12.60	13.21	13.21
1998	12.57	13.62	13.62
2003	12.51	13.93	13.93
2008	12.48	14.16	14.43
2013	12.49	14.33	14.60
2018	12.48	14.38	14.79
2023	12.30	14.38	14.81
2028	12.19	14.38	14.81
2033	11.95	14.29	14.79
2038	11.78	14.22	14.70

Figure 10.3: Predicted market value (£ millions) of the Galloway salmon fishery under alternative abatement scenarios



A 90% reduction in SO_2 levels generates a peak in market value of £14.8 million in 2023. The final value of £14.7 million represents an increase of 17% over the initial market value in 1988, and a 25% increase over the *status quo* value of the fishery in 2038.

The predicted rise in market value represents the increase in the capitalised future benefits of owning the rights to fish for salmon in Galloway. As such they are an appropriate measure of the economic benefits of SO_2 abatement to the Galloway salmon fishery. In economic appraisal it is necessary to take account of the long time period associated with these future benefits by discounting to provide present value estimates for the marginal benefits of the alternative abatement strategies.

The discount rate used can vary, but the UK Treasury currently employs a discount rate of 6% for environmental schemes (H.M. Treasury, 1991). Applying this discount rate to the Galloway fishery, yields a present value for the increase in market value over the status quo, of £1.13 million for a 60% reduction, and £1.21 million under a 90% reduction. The marginal benefits of increasing abatement from 60% to 90% is therefore only £0.08 million. More recent studies (e.g. ETSU and IER, 1995), have applied a discount rate of 3%. The respective benefit figures under 3% are £1.61 million for 60% and £1.73 million for 90%.

10.4 Accuracy and Precision

Forecasting the future market value of the Galloway R&L salmon fishery as a result of alternative SO_2 emission levels is clearly a complex task. A considerable degree of uncertainty surrounds the parameter estimates used in the model, and it is therefore desirable to indicate, as far as possible, both the accuracy and the precision of the predicted market values.

10.4.1 Accuracy

Accuracy refers to the extent to which the predicted estimates differ from the true value. In some modelling exercises it is possible to test for bias by comparing the predicted estimates with actual values measured or recorded independently from the modelling exercise (Macmillan, 1992). However, since time-series data on the response of market, or rental, values to changing salmon catch for individual fisheries is not publicly available, this type of assessment poses obvious difficulties.

A validation exercise has recently been carried out to assess the accuracy of MAGIC predictions for key water chemistry and fish population parameters (Wright *et al.*, 1994). Predicted values in 1988 for pH, acid neutralising capacity (ANC), diatom fossil counts, and non-marine sulphate concentrations based on a 1979 calibration in Galloway freshwaters were found to be in close agreement with observed 1988 values.

Also, using a hindcast technique (i.e. predicting from the present into the past) it has been shown that the fish population parameters predicted by MAGIC (i.e. Healthy, Marginal, and Extinct) agreed well with historical information on the decline in fisheries status of several Galloway lochs and estimated dates of extinction. Figure 10.4 presents a comparison of the historical change in fish population status and the predicted status from MAGIC for Loch Narroch in Galloway.

Actual extinction is observed to coincide with a predicted percentage probability of extinction of 100 around the late 1960's. Supplementary evidence for MAGIC's validity is also available from Jenkins *et al.*, (1990) where a high level of consistency between MAGIC output and independent historical data on water chemistry and diatom records from lake sediments in Galloway is noted.

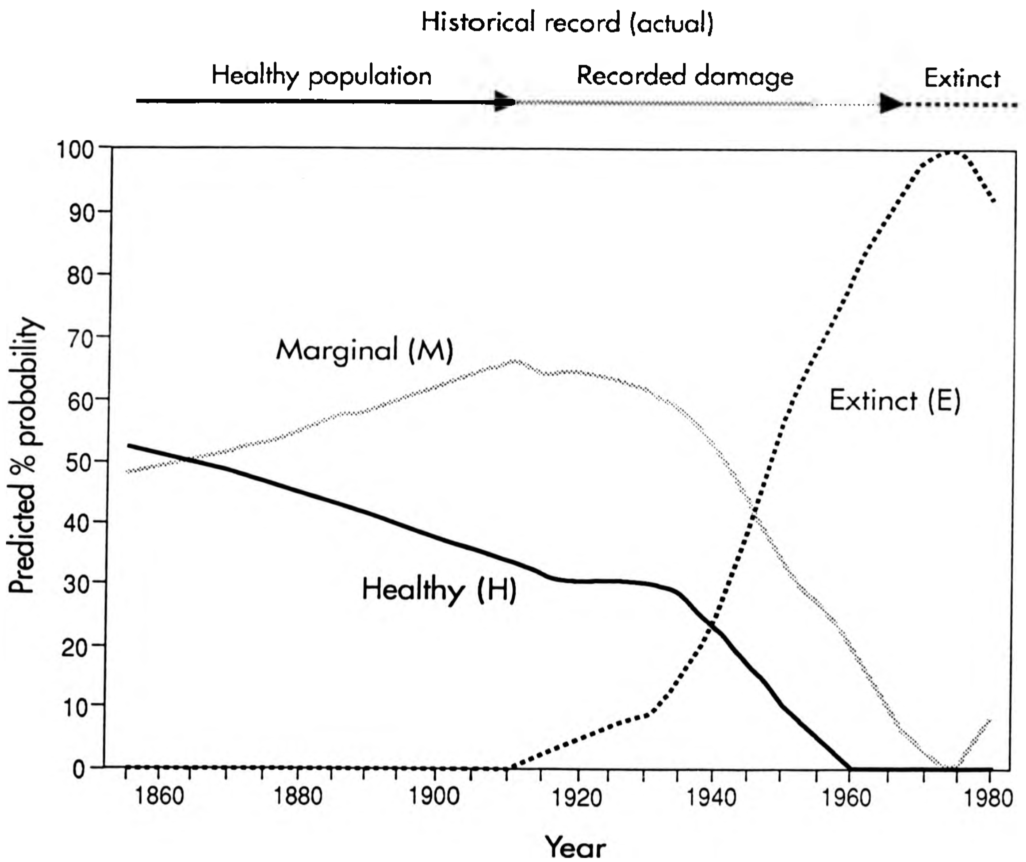
10.4.2 Precision

The extent to which the modelled predictions for market value can be considered statistically reliable depends on the unexplained variance (i.e. error) associated with the estimation of the model parameters. In this section confidence intervals for the predictions of market value are constructed for the forecast period.

A bootstrap technique was used which simulated values for the parameters used in Equations 9.1 and 9.3, based on their means and standard errors.

Parameters to be simulated were:

Figure 10.4: Comparison of predicted fish population status in Loch Narroch as reconstructed by MAGIC with historical information on fish decline



Equation 9.1

i) Value of H_T where $H_i \sim N(m_i,\ se_i)$

$H_0 \sim N(m_0,\ se_0)$

ii) Value of b_H where $b_H \sim N(0.974,\ se\ 0.114)$

Equation 9.3

iii) Value of b_{lns} where $b_{lns} \sim N(0.547,\ se\ 0.062)$

The 95% confidence intervals for the predicted market values of the Galloway fishery, based on this procedure, are presented in Table 10.4 and Figure 10.5. The intervals widen considerably during the forecast period. This reflects the increasing divergence between the mean value for the parameters in the bootstrap procedure, and their predicted values (i.e. the confidence interval is closest where the parameter values for H, b_H, and b_{lns} are close to their estimated mean value).

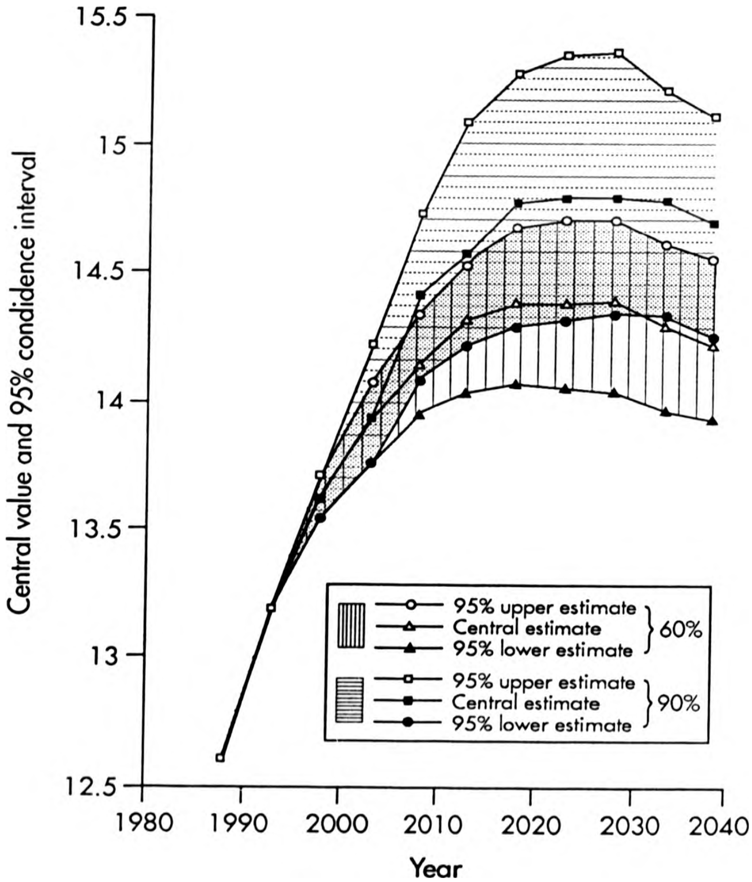
The confidence intervals presented in Table 10.4 are likely to overstate the level of precision in the predictions for two reasons. Firstly, the total market value of the fishery was based on an assumed value of 1 for the catchments not affected by acidification throughout the forecast period. In reality there is likely to be considerable variation in the value of H between unaffected catchments in the fishery, and if this uncertainty was incorporated the parameter estimates would be less precise.

Secondly, the precision of the estimates is influenced by the level of correlation that is assumed to occur between

Table 10.4 **Central values and 95% Confidence Interval for the Market Value (£) of the Galloway Fishery under 60 and 90% reductions of SO_2.**

YEAR	Central Value (95% Confidence Interval)	
	60% REDUCTION	90% REDUCTION
1988	12.60	12.60
1993	13.21(13.19,13.23)	13.21(13.19,13.23)
1998	13.62(13.54,13.70)	13.62(13.54,13.70)
2003	13.93(13.77,14.09)	13.93(13.77,14.09)
2008	14.16(13.97,14.35)	14.43(14.10,14.76)
2013	14.33(14.04,14.62)	14.60(14.24,14.96)
2018	14.38(14.07,14.69)	14.79(14.29,15.29)
2023	14.38(14.05,14.71)	14.81(14.32,15.30)
2028	14.38(14.04,14.72)	14.81(14.26,15.36)
2033	14.29(14.01,14.57)	14.79(14.35,15.23)
2038	14.22(14.00,14.42)	14.70(14.28,15.12)

Figure 10.5: Central estimates and 95% confidence intervals for the market value of the Galloway Fishery under 60% and 90% abatement scenarios



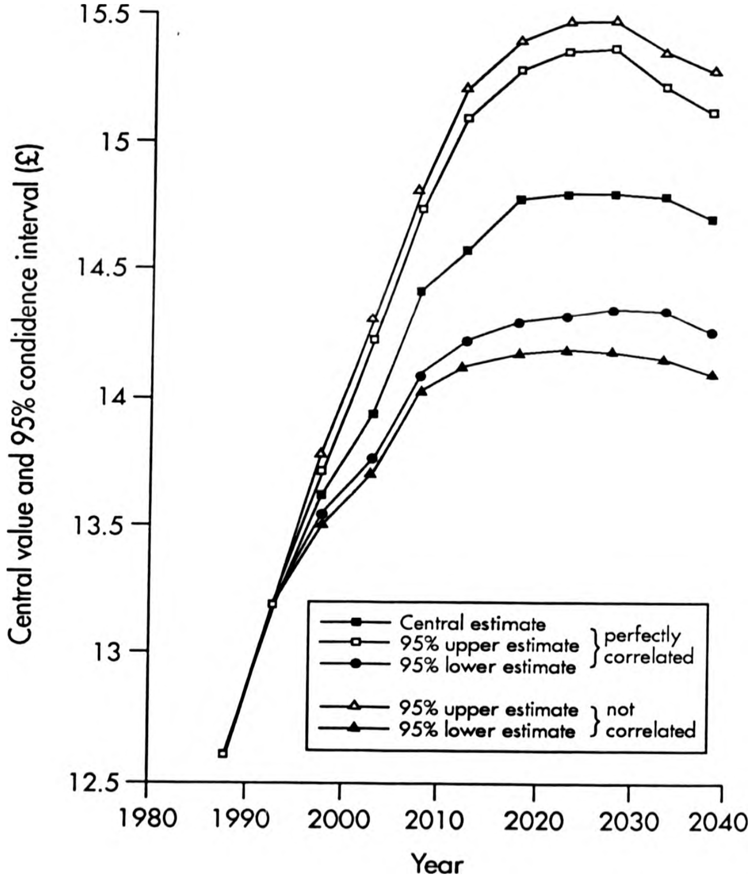
the initial value of H and it's value in future periods, (i.e. the extent to which the value of H in period T+1 is influenced by it's value in the previous period, T). The confidence estimates presented in Table 10.4 assume perfect correlation in H. If H is assumed to be perfectly uncorrelated the confidence intervals widen considerably because there is greater uncertainty about the future values of H. This is shown in Figure 10.6, where Confidence Intervals for a 90% reduction in SO_2 are depicted when H is assumed to be both perfectly correlated and perfectly uncorrelated.

10.5 Aggregation

Although the Galloway fishery is probably most affected by acidification in Scotland, it is only one of a number of fisheries that are thought to have suffered fish population loss as a result of acid deposition. In order to compare the costs and benefits of SO_2 abatement it is necessary to obtain an aggregate measure of the total benefits to the entire Scottish salmon fishery.

Estimating the total value of recovery in the Scottish R&L salmon fishery using the approach described in this study would require de-novo application of the model to other affected fisheries. Since MAGIC is a catchment based model, it would require to be recalibrated for each region. Given the paucity of historical fisheries records, and the

Figure 10.6: Central estimates and 95% confidence intervals for the market value of the Galloway Fishery under the 90% abatement scenario assuming perfect correlation and no correlation between values for H



all projects involving benefit transfer, it was decided that aggregation to the Scottish level based on the Galloway model was worthwhile.

The first step was to identify the salmon fisheries where catch has been affected by acidification. Although several studies have associated catch decline in some salmon rivers with acidification (UKAWRG, 1988; Waters and Kay, 1988), conclusive links have not been established, and it is not known with certainty which fisheries have been affected. For the purposes of this exercise, fisheries damaged by acidification were identified by overlaying the Critical Loads Exceedance map for soils onto the map produced by Waters and Kay, (1988) which indicates salmon rivers where there has been declines in average salmon catch in recent decades. Damaged fisheries were identified as those fisheries which had suffered a long term decline in catch and fell within areas indicated by the Exceedance map to be affected by acidification (Figure 10.7).

Changes in the market value for each fishery under alternative abatement scenarios were estimated by transferring the percentage change experienced in the Galloway fishery to each of the target fisheries, adjusted for the percentage of the salmon fishery catchment affected by acid rain. The initial market value of the target fisheries in 1988 was estimated, as in the case of the Galloway fishery, by multiplying the total average annual catch by the estimated capital value of a Scottish salmon (£3420). The results are presented in Table 10.5.

Figure 10.7: Predicted area of acidified salmon fisheries in Scotland

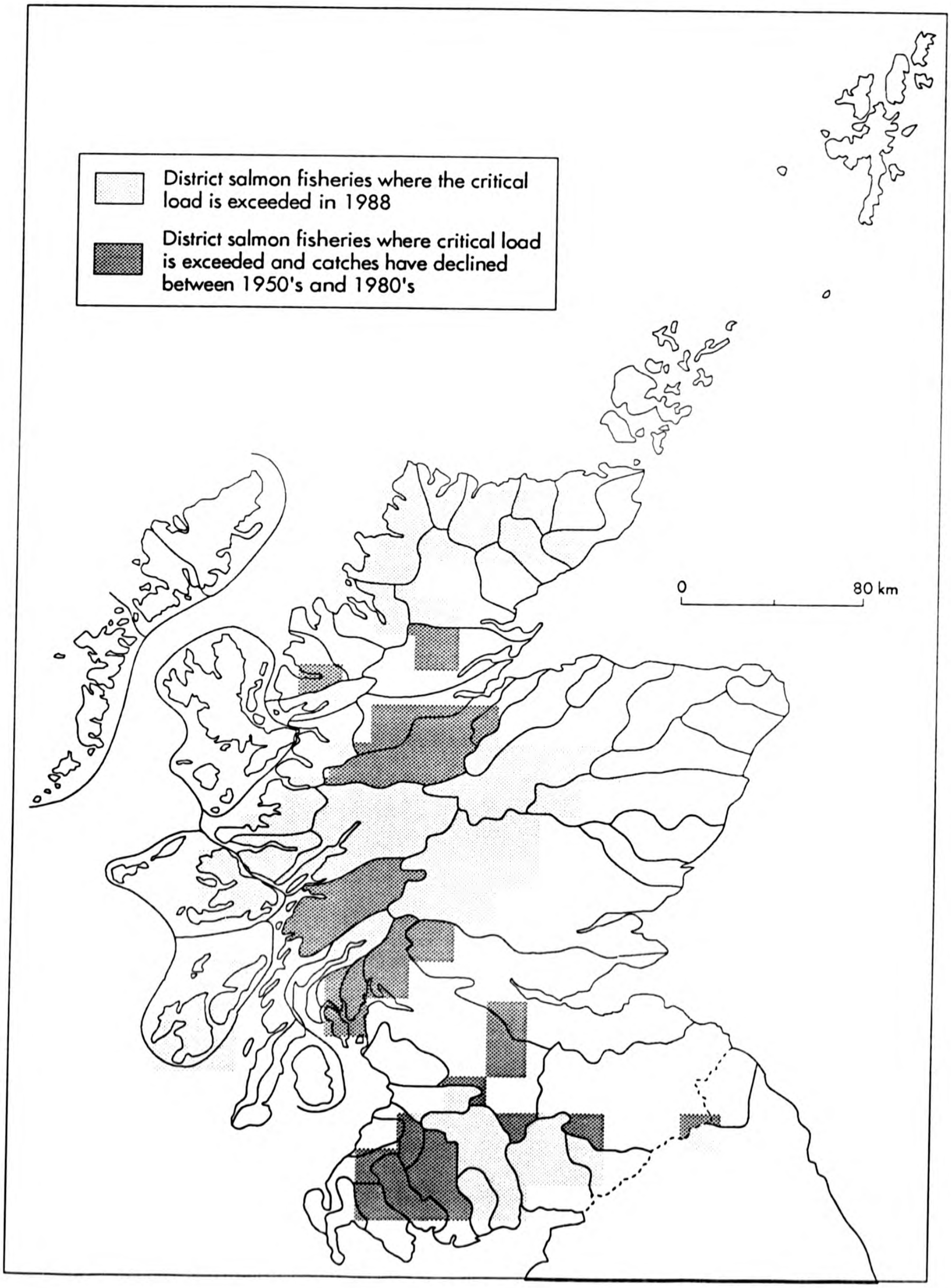


Table 10.5 Predicted change in the market value of the Scottish Salmon Fishery under alternative abatement scenarios

Fishery	% Area Acid	Initial Value £ Million	% Change in Value 1988-2038			Final market value £ Million		
			0%	60%	90%	0%	60%	90%
Galloway	56	12.6	-6	13	17	11.8	14.2	14.7
Conon/ Alness	26	4.54	-3	6	8	4.4	4.8	4.9
Badachro/ Applecross	34	1.19	-4	8	10	1.2	1.3	1.3
Ness	91	4.79	-10	21	28	4.3	5.8	6.1
Beauly	64	2.45	-7	15	19	2.3	2.8	2.9
Nairn	38	0.37	-4	9	12	0.4	0.4	0.4
Awe/Nell	100	2.14	-11	23	30	1.9	2.6	2.8
Forth	14	5.15	-2	3	4	5.1	5.3	5.4
Clyde	24	2.48	-3	6	7	2.4	2.6	2.7
Ruel/ Drummachloy	83	0.07	-9	19	25	0.1	0.1	0.1
Eachaig	100	0.34	-11	23	30	0.3	0.4	0.4
Tweed	8	39.72	-1	2	2	39.4	40.5	40.7
Totals						73.5	80.8	82.4

The estimated aggregate benefits of SO_2 over the *status quo* at 2038 for the entire Scottish salmon fishery were £7.37 million and £10.94 million for the 60% and 90% abatement levels respectively. This is equivalent to a present value (at a 6% discount rate) of £3.2 million and £3.7 million. The marginal increase in the market value of the entire fishery as a result of reducing SO_2 emissions from 60% to 90% is therefore £1.57 million, which is equivalent to a present value of £0.5 million.

10.6 Discussion

10.6.1 Policy Issues

The increase in the market value of the Galloway salmon fishery predicted under both abatement scenarios, represent a fairly substantial increase in the total value of the salmon fishery. For example, under a 90% reduction in SO_2, the market value of the fishery is predicted to increase by 25%. However, the increase in the market value of the entire Scottish fishery is less dramatic. Using a capital value of £3420 per salmon Mackay (1989), the total value of the Scottish R&L salmon fishery can be considered to be in the region of approximately £350 million. The increase in value brought about by reductions in SO_2 of 60% and 90% therefore represent increases of only 2.1% and 2.6% respectively.

In comparison to the value placed on the non-use benefits estimated using the CVM, where annual benefits ranged from £551 million to £772 million, fishery benefits appear to be relatively small. However, recovery in salmon fisheries is extremely important with respect to its implications for rural areas. The economies of areas affected by acidification are relatively fragile, and salmon fishing often makes an important contribution to local employment and tourist expenditure. Mackay, (1989) for example, estimates that R&L salmon fishing in Scotland generates over 3000 jobs and gross expenditure of approximately £50.4 million annually. This aspect of the recovery, together with the estimates of benefit based on 'real money' as opposed to hypothetical payments, will undoubtedly raise the significance of recovery in the salmon fisheries in government appraisal.

The results from the hedonic model are also significant for policy appraisal in several other respects. Firstly, the marginal increase in the market value of the Galloway fishery as a result of decreasing SO_2 emissions from 60% to 90% was only £0.08 million. This compares to a marginal increase over the status quo under the First UNECE Sulphur Protocol (60%) of £1.13 million. This would suggest that the marginal benefits of improving water chemistry for the benefit of angling diminish rapidly as abatement increases. The benefits of the UNECE's Second Sulphur Protocol, which will bring about a lower marginal reduction in SO_2 than the 90% level, are likely to be lower than predicted here.

Second, the long term aim of reducing sulphur emissions to non-damaging levels may not be sensible for the simple reason that very sensitive upland catchments may become acidified as a result of the deposition of sulphur from natural sources. The MAGIC model clearly predicts that acidification will resume in sensitive catchments of the Galloway fishery by 2028 under both abatement scenarios. This suggests that abatement may therefore only be delaying what is an irreversible process of acidification in sensitive catchments.

If this is the case, then the CVM estimates may also need to more cautiously interpreted since the scenarios investigated did not consider this possibility. Further scientific research is certainly required to establish the long term status of sensitive upland catchments under low or negligible SO_2 inputs.

10.6.2 Methodological Issues

The hedonic model described here represents a novel approach to valuing changes in a private fishery resource as a result of complex environmental changes. A particular strength is the incorporation of a sophisticated catchment model to predict the affect of reduced SO_2 deposition on catchment water chemistry in affected fisheries, and its linkage to fish health and catch. Although relatively data demanding, the model is able to address the dynamic and long term nature of acidification processes.

In contrast, other attempts to estimate the economic benefits of SO_2 reductions have tended to incorporate considerably cruder dose-response relationships. Mullins and Menz, (1985), for example, rather than attempting to model a response function for the effects of acidification on fish catch used a simple binary no fishing/fishing relationship in their Travel Cost model. Other studies, also faced with problems of obtaining reliable data on catch per unit effort, have only been able to go as far as establishing a relationship between water chemistry and biological fish yield (Forster, 1984).

The application of Radfords' hedonic model permitted a link to be established between enhanced fish catch and market value without the need for expensive primary data collection from the Galloway fishery. However, Radford's model, in some respects, was less than ideal for this study. Firstly, the regression relationship between beat value and explanatory variables (Equation 9.3) was based entirely on data from England and Wales. Although regional fisheries were found by Radford to be relatively uniform in this respect, it is conceivable that a different relationship may hold in Scotland.

Secondly, Radford used expected sale value rather than the observed market value of individual beats obtained in a fully functioning market. Since owners may over or under-estimate the market value of their beats, it follows that the implicit price relationship may not reflect the true value of a marginal change in environmental quality. However, Freeman, (1979) reports that studies based on expected value are broadly consistent with those based on actual transactions and suggests that the error introduced by self-reporting is generally small and random.

Finally, the Hedonic model relied on cross-sectional data from a range of fisheries and, as Abelson and Markyanda, (1985) point out, this can lead to bias in the hedonic price estimates when the future level of environmental quality changes over time. The strict interpretation of the implicit price in this study is that it represents the value of a marginal change in environmental quality, where that level is assumed to hold across all future periods. The true hedonic capital price on the other hand, would consider the effect on the price caused by dynamic change in environmental quality expected in all future periods. In this case, where catch is expected to increase steadily for 40 years, the true price is likely to be higher than that estimated. However, the extent to which consumers of fishing rights are aware of, or incorporate possible future catch levels into their market valuation of a fishery is unknown.

An alternative method would have been to use time series market data from fisheries where there has been an historical decline in catch. This would have provided useful information on expectations of future catch levels, as well as data which reflected the relationship between catch level and value for individual rivers over time.

The approach used in this study, had to make a number of other assumptions due to lack of information. Perhaps the most important, and least avoidable assumption, was that the relationship between fish catch and fish population status for Loch Reiwcar was representative of the response of the entire affected fishery to increases in H. Clearly, other environmental factors can influence fish catch and they are likely to differ across catchments. Secondly, the assumption that other variables which are likely to influence catch (e.g. agricultural pollution, high sea fishing effort) would remain unchanged is difficult to envisage, but the current state of knowledge prevents any sensible future scenarios for these variables being incorporated. Since salmon can be caught almost anywhere along the river catchment the only sensible approach to estimating changes in salmon catch was to model changes in H for the entire Galloway fishery, weighted by catchment area. This approach is supportable if one assumes that: a) spawning is possible throughout the entire fishery, and changes in water quality in all acidified waters will therefore lead to increased salmon recruitment[1], and b) the value of the fish population status parameter H remained at 1 throughout the forecast period in non-acidified waters. (If the regional value was less than 1, or changed from this value during the forecast period, the resultant fish catch estimates will be biased).

Although improvements to the approach are limited by the availability of data, one obvious opportunity to improve the model would be to enhance the link between fish

[1] Although there are likely to be some small catchments which are unsuitable or inaccessible to salmon, local opinion is that salmon recruitment is possible throughout most of the Galloway salmon fishery (Stephen pers. com.).

population status and fish catch. Currently this is based on a simple empirical correlation from one catchment. It would be advantageous to incorporate a more sophisticated model which can simulate salmon population dynamics in response to a wide range of environmental factors (Ormerod, 1990).

10.7 Concluding Remarks

Despite the restrictions placed on benefit estimation for salmon fisheries by inadequate data for modelling purposes, it is clear from this study that abatement will lead to a relatively substantial increase in the market value of the Galloway fishery. However, recovery is likely to be localised and the total benefits to salmon fishing in Scotland as a whole are small compared to the non-use values predicted from the CVM study.

In the final chapter the implications of these benefit estimates for the economic appraisal of abatement policy are assessed. Some general conclusions about the methodologies used and their reliability are also presented.

Chapter 11 GENERAL DISCUSSION AND CONCLUSIONS

In this chapter some general implications of the research for the economic valuation of environmental change are outlined. The main conclusions regarding the methodology used, and the implications of the results for abatement policy, are also presented.

11.1 General Discussion

Ten years ago Bishop and Heberlien (1986) suggested that 'the direction the economics scales tip could well depend, at least over the next decade or two, on whether existence (non-use) values have credibility'. In 1996 this statement seems especially relevant to the economic appraisal of SO_2 abatement policy.

The annual non-use benefits from the CVM exercise were substantially greater than the predicted market benefits to the Scottish R&L salmon fishery. Even if the CVM estimates are reduced by one-half, as suggested by the NOAA report and by empirical comparisons with real payments (Navrud and Veisten, 1996), the discounted value for low damage is £9.3 billion[1]. This figure is over 2000 times the value of recovery in the Scottish salmon catch, which was estimated to be £3.7 million under a 90% reduction scenario.

Although the cost to Scottish industry associated with

[1]Using a discount rate of 6%.

reducing SO_2 emissions to non-damaging levels has not been published a reasonable estimate, based on the costs of reducing SO_2 under a range of other abatement scenarios for the UK as a whole (ERL, 1990), would be in the order of £1-2 billion (assuming increasing marginal costs of abatement). Even if benefits to other resources, such as health and buildings, are ignored a non-damaging emissions policy could therefore be justified, on the grounds of efficiency, by considering non-use values for the semi-natural environment alone. Ignoring non-use benefit estimates in a cost-benefit analysis of abatement policy would therefore significantly under-represent the benefits of recovery in the semi-natural environment.

While CVM estimates certainly may tip the scales in favour of abatement, the usefulness and reliability of non-use values for decision-making remains open to question. One noticeable feature of the results from this CVM study was the consistently large WTP values for all scenarios including that depicting a low level of recovery and low future damage. Insensitivity of WTP to the scope of the environmental change is a feature of many CVM studies (Desvouges et al., 1993; Diamond and Hausman, 1994), yet the presence of a scope effect was considered by the NOAA panel to be an important indicator of reliability. Some variation in WTP to reflect varying levels of environmental enhancement or damage is not only expected from theory, (unless the individual is satiated), it is important if CVM benefit estimates are to help guide decisions regarding resource allocation.

This study yielded ambiguous evidence for a scope effect. While WTP under the two alternative future damage scenarios was significantly different, this was not the case for either future recovery level or rate of recovery. Carson et al., (1996) suggest that the lack of a scope effect can be attributable to survey designs which do not make respondents aware of alternative scenarios of environmental change. However, the results in this study cannot be explained in this way since the three recovery levels were described to respondents in the depiction of environmental change arising from acidification. In the case of future damage on the other hand, where a significant difference in WTP was observed, respondents were only aware of one expected damage scenario.

One possible explanation for this inconsistent scope effect may be derived from 'prospect theory' which suggests that people value losses more highly than gains (Kahneman and Tversky, 1979). Evidence supporting the notion that the indifference curves are steeper for losses than for gains, has also been proposed by Knetsch (1992) as a hypothesis to explain the disparity between WTA and WTP for the same good. Nonreversible indifference curves of this type would suggest that traditional assumptions about equivalence and reversibility may be in error, and lead to gains from trade being overstated, too many voluntary exchanges predicted, and welfare losses being underestimated (Knetsch, 1992).

The implication of indifference to recovery level in the case of acidification is clear - the general public are prepared to pay large amounts of money, regardless of the prospects for recovery, to end pollution caused by SO_2. Although there may be doubts about the magnitude of non-use values generated by CVM in this study (see below), the results therefore supports the precautionary principle of zero pollution behind the Critical Loads approach to European abatement policy.

Whether that target is actually achievable however, is thrown into question by the results from the salmon fisheries model. The model predicts that around 2030 the market value of the salmon fishery, even under the 90% reduction scenario, will fall as the acidification process is resumed in the upland catchments of Galloway. Where soils have a very low buffering capacity against acid inputs, very low sulphur inputs from natural sources can give rise to acidification. Indeed acidification is a natural phenomenon which has been affecting some upland catchments since the last ice-age. Although these sensitive catchments are only likely to be locally significant, it is clear that a dramatic reduction in SO_2 emissions will only lead to a slowing down in an inevitable process of acidification. Whether the 'zero' acidification target of current UNECE policy for SO_2 abatement is realistic in these locations is doubtful.

The non-use values estimated from the CVM study were substantial. This is not unusual: CVM studies typically generate large values which often greatly outweigh other project costs and benefits. UK examples include Hanley and Craig's study of afforestation in the flow country (1991), and the ESA programme in England (Garrod and Willis, 1995) and Scotland (Hanley *et al.*, 1996).

While large values *per se,* may be welcomed by policy-makers anxious to justify environmental programmes, their contribution to decision-making will be limited if projects which incorporate non-use values almost always pass the cost-benefit test (Hoehn and Randall, 1987). It is interesting to note that many CVM studies sponsored by government departments (e.g. Forestry Commission, MAFF) were carried out for existing programmes, where high benefit values are desirable to justify earlier decisions (e.g. afforestation and the Environmentally Sensitive Areas programme), rather than as part of an ex-ante appraisal of possible options.

The concern that benefit figures from CVM studies are simply too big when aggregated over the general population, and thus cannot possibly represent a true economic commitment is widespread amongst many of CVM's critics (e.g. Diamond and Hausman, 1994). Alternative hypotheses regarding the motivation behind CVM responses which support this line of reasoning include Kahneman and Ritov's (1993) suggestion that individuals may only be expressing an attitude towards a public good, expressed in a monetary scale, or Andreoni's theory of 'warm-glow' giving (Andreoni, 1990). Also Diamond *et al.*, (1993) have proposed that individuals may actually be expressing what they think should be done, in a sort of casual cost-benefit analysis, rather than deciding on the basis of a rational economic trade-off between the good and income.

In order to provide some empirical evidence of hypothetical bias a limited number of comparative studies, which have compared hypothetical payments with real payments, have been undertaken (*e.g.* Dickie et al., 1987; Neill *et al.*, 1994; Kealy *et al.*, 1990; Duffield and Patterson, 1992; Seip and Strand, 1992; Navrud and Viesten, 1996). Typically, these studies have shown that stated willingness to pay exceeds the corresponding real payment by a factor of between 2 and 10.

Although some of these studies can be faulted in their method of implementation (Hanneman, 1994), there is a general consensus amongst economists that hypothetical payments are biased upwards. This bias appears to be greater when non-use public goods are considered (Navrud and Viesten, 1996), and has been attributed to unfamiliarity among respondents about the good and its exchange in a market context.

While there are no real WTP data to compare with the hypothetical payments calculated for this study, there is strong evidence from the regression analysis that individuals are making genuine economic commitments when responding to the valuation question. Diamond and Hausman, (1994) state that if WTP is a true reflection of economic preferences then they would expect patterns of answers across different individuals that are consistent with theoretical expectations (construct validity). In this study there was considerable variation in willingness to pay across individuals with respondents on higher income, positive attitudes toward the environment, and who were members of environmental charities, more likely to accept the bid offered to them.

Calibration of hypothetical responses to bring them closer to the level of payment that would reflect a real economic commitment is one approach to the problem of excessive WTP values (Harrison *et al.*, 1993). Calibration is common in marketing designed to predict purchases (Diamond and Hausman, 1994), and the NOAA panel made the somewhat arbitrary recommendation that CVM damage estimates used for litigation should be divided by two.

Recent research has attempted to calibrate CVM responses using statistical bias functions for both private goods (Blackburn et al., 1993), and to correct for hypothetical and free-riding bias in the case of local public goods (Harrison et al., 1993). While some encouraging results have emerged, more comparisons between real and hypothetical payments are required to improve our understanding of the extent to which hypothetical payments differ from real payments under a range of valuation contexts (e.g. private/public goods; use/non-use values). This would also allow robust statistical bias functions, which can be applied to these varying circumstances and potential biases, to be identified.

Another aspect of the debate about hypothetical bias in

CVM studies which has been overlooked concerns the need to also estimate the opportunity cost, as well as the benefits, of environmental projects. Diamond and Hausman, (1994) suggest costs, (presumably motivated by altruistic concerns) associated with lost jobs opportunities as a result of environmental. Macmillan, (1996) has estimated the opportunity costs associated with restoring native woodlands on open moorland using WTA questions, and found that net benefits were significantly reduced when landscape and wildlife benefits of the existing land use were incorporated. While opportunity costs may be difficult to estimate since they will necessarily involve WTA questions (Kristrom, 1994), and may not be appropriate for all projects, they should be routinely investigated. This would not only be good economic practice, but is likely to offset the hypothetical bias associated with CVM estimates of WTP for environmental projects.

The large values which CVM generates tend to support, rather than conflict with ethical and environmental demands for more environmental investment. Somewhat surprisingly, when viewed in this context, CVM has attracted considerable opposition from environmental groups (Bowers, 1992). Rather than grasping the opportunities CVM offers in influencing economic analysis, environmentalists have been outspoken in their criticism. There may be several reasons for this position. Firstly, some environmentalists appear to be fundamentally opposed to any form of economic analysis of environmental issues, believing that environmental concern transcends economics (Pearce and Turner, 1990). Secondly, CVM results, derived from the views of an uninformed public, cannot substitute for expert environmental judgement (particularly in the eyes of those experts!). Thirdly, CVM may be considered by some to be prone to manipulation in order to achieve the 'desired answer'. While CVM researchers may go to great effort to ensure that the best available scientific evidence is incorporated, and that information sets are sufficiently informative, there is no doubt that opposition from environmentalists holding these beliefs will persist for some time to come.

Another cause for concern about reliability is apparent when the non-use values from this study are compared with those from a concurrent CV study by ECOTEC, (1993) on behalf of the Department of the Environment. Taking the UK as a whole, this study estimated the annual benefit value of recovery in the aquatic ecosystem to be £622 million. This figure falls within the range estimated for low and high damage (£551 million and £772 million per annum respectively) for the Scottish population alone. While the ECOTEC study focused only on the aquatic ecosystem, and there were substantial differences in survey methodology, the disparity between the results is dramatic.

If CVM estimates for non-use values are to gain greater credibility future research must be concerned with understanding why CVM estimates for broadly comparable environmental projects vary so widely. The ECOTEC study used an open-ended format which is known to produce much lower estimates of WTP than the discrete choice approach used in this study. However, elicitation method (mail/in person), the information set used, and form of presentation can all influence individual responses and further research is required to define these biases and ultimately to establish through validation with real payments, the survey instrument which most closely reflects real economic commitments.

The absence of significantly higher WTP among respondents for a faster rate of recovery from acidification following abatement is also of significance to project appraisal. In economic analysis the overall costs and benefits of SO_2 abatement are conventionally required to be discounted in time to present values. Since discounting has the effect of reducing the value of temporally distant costs and benefits to almost nothing, the practice has drawn considerable criticism on the grounds of inter-generational equity (Hanley and Spash, 1993).

Economists provide several justifications for applying a positive rate of discount to investment projects. Firstly, people are assumed to have a positive time preference for the consumption of goods and services. Positive time preference is enhanced as a consequence of economic growth: as consumption grows so the marginal utility of consumption will decline. The second rationale takes account of rates of return from alternative investments, i.e. the opportunity cost of capital (Pearce *et al.*, 1990). Selection of the actual rate of discount used in the public appraisal of policies is highly complex and discretionary (Henderson and Bateman, 1993), but generally takes account of both these considerations.

In this study comparison of mean WTP in the CVM survey for alternative recovery periods showed that respondents were indifferent to the rate of recovery in the semi-natural environment. This is consistent with the bequest motivation which most respondents gave for paying higher prices following abatement and concurs with Knetsch's (1993) view that people are less sensitive to time frame when considering the avoidance of 'dreaded events'[2].

A low or zero time preference for environmental change observed in this study provides some empirical support for the lower discount rate which the UK Treasury currently applies to some environmental projects, such as afforestation on bare land. However, manipulating the discount rate to favour the environment is considered to be undesirable by many economists. Not only do variable discount rates give government agencies the opportunity to manipulate project selection (Henderson and Bateman, 1993), but Pearce et al., (1990) argue that it would be difficult in practice to separate environmental projects from other types of investment.

Government appraisal of commercial afforestation projects in the 1980's provides an interesting illustration

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[2]Although respondents were not directly asked if they considered future acidification damage to be a dreaded event, responses to Question 7 showed that acid rain ranked second highest amongst a range of environmental concerns.

of the potential pitfalls from adopting a discretionary lower rate of discount. The Forestry Commission have in the past evaluated all investments in tree planting using a reduced discount rate of 3%, on the basis that tree planting provides rural employment opportunities, and non-market environmental benefits. As a consequence afforestation proceeded rapidly throughout upland Britain and some areas of high conservation interest were lost or damaged. In these areas the positive environmental benefits of commercial forests were considered by the RSPB, and other environmental organisations, to be outweighed by the environmental losses. Forestry in these locations (notably the uplands of England and the 'Flow Country' in northern Scotland) were subsequently considered by the government to be undesirable from an environmental perspective, and grant aid was withheld[3].

11.2 Conclusions and Recommendations for Further Research

A number of conclusions can be drawn from this research into the benefits of recovery from SO_2 deposition on the semi-natural upland environment of Scotland. Some of these conclusions relate to the economic appraisal of abatement policy, while others are relevant to environmental valuation methodology. Some suggestions regarding future research are

[3] The withdrawal of tax concessions for private investors in the 1988 budget can also be partly attributable to this conflict with conservation interests.

outlined at the end.

11.2.1 Conclusions

The non-use benefits associated with recovery in the semi-natural environment greatly outweigh user benefits, which are primarily related to angling activity. Taken alone non-use benefits are likely to exceed the costs of reducing SO_2 below the Critical Load for soils in Scotland.

Although previous studies which have compared real and hypothetical payments suggests that CVM estimates for non-use environmental goods are biased upwards, non-use values calibrated for hypothetical bias are still likely to dominate abatement benefits in the semi-natural environment. A range of validity checks on the CVM data suggest that the responses to the payment question were a reflection of a real economic commitment.

Respondents to the CVM survey appear to be more concerned about future environmental losses than gains. This would tend to suggest that policy should be targeted towards preventative policies rather than enhancement projects such as tree planting on farmland. There was also some limited evidence from the CVM survey that the general public do not exhibit a positive time preference for environmental recovery. Although further research will be required to examine this issue in more detail this result does offer some support for the Treasury's current practice of applying a lower discount rate for environmental projects.

Although the user benefits to the rod and line salmon fishery were of a much lower order of magnitude than the CVM estimates, recovery in salmon catch is of considerable importance to the rural economy in terms of income and employment. This consideration, together with the use of a valuation methodology which is incorporates market, rather than hypothetical data, suggests that these user estimates may have considerable impact on policy development.

Some important conclusions can also be made with respect to valuation methodology. Acidification, in common with many other environmental issues, is characterised by scientific complexity and uncertainty. This study attempted to incorporate sophisticated dose-response functions for environmental recovery using a mixture of expert knowledge and scientific modelling techniques. The significance of economic analysis to the development of environmental policy will increasingly depend on the extent to which economists, working with scientists, can reliably model environment-economy links (Pearce, 1996).

The very high response rate, together with satisfactory validity checks suggest that CVM surveys implemented via direct mailings have few serious drawbacks. The case for mail surveys is particularly strong when dealing with environmental issues where passive use and existence values are likely to feature prominently, which place greater cognitive demands on the respondent.

The most satisfactory approach to estimating mean WTP from discrete choice data (when negative bids are not allowed), was generated by the reciprocal bid transformation. For aggregation purposes in the presence of covariates, mean WTP should not be estimated from average covariate data, but from the probability of accepting the bid level as predicted by covariate data from individual respondents.

11.2.2 Further Research

Two main areas of further research are suggested by this study. Firstly, since the inconsistent scope effects found in this study could have far reaching implications for both CVM and the evaluation of welfare gains and losses, more tests for scope are required. These should focus on determining sensitivity of WTP and WTA measures to a range of environmental losses and gains.

Secondly, CVM is vulnerable to criticisms of hypothetical bias. the few studies comparing real payments with hypothetical CVM payments suggests that the latter can be up to ten times higher. More comparative studies are urgently required to allow for the development of reliable statistical bias functions to calibrate CVM responses. Due to their very nature, the need for calibration is greatest for non-use values, and collaboration with environmental charities, or related agencies, will undoubtedly be required to obtain comparable real payments.

BIBLIOGRAPHY

Abelson, P.W. and Markyanda, A. 1985. The interpretation of capitalised hedonic prices in a dynamic environment. Journal of Environmental Economics and Management 12, 195-206.

Acid News 2, 1993. The Swedish Society for the Conservation of Nature, Goteberg, Sweden.

Acid News 5, 1993. The Swedish Society for the Conservation of Nature, Goteberg, Sweden

Acid News 2, 1996. The Swedish Society for the Conservation of Nature, Goteberg, Sweden.

Adamowicz, W., Louviere, J., and Williams, M. 1994. Combining revealed and stated preference methods for valuing environmental amenities. Journal of Environmental Economics and Management 26, 271-292.

Adams, R.M., Crocker, T.D. and Thanavibulchai, N. 1982. An Economic Assessment of Air Pollution Damages to Selected annual Crops in Southern California. Journal of Environmental Economics and Management 9, 42-58.

Andreoni, J. 1990. Impure altruism and donations to public goods: A theory of warm-glow giving. Economics Journal 100.

Antle, J.M. and Capalbo, S.M. 1993. Integrating economic and physical models for analysing environmental effects of agricultural policy on nonpoint-source pollution. Russell, C.S. and Shogren, J.F. (eds) Theory, modelling and experience in the management of nonpoint-source pollution. Kluwer Academic Publishers, Dordrecht.

ApSimon, H.M. 1994. Dose-Response Relationships for Acidifying Species. Paper presented at Workshop on 'Economic Evaluation of Damage Caused by Acidifying Pollutants'. CSERGE/Imperial College, London.

ARA Consultants, 1982. Value awareness and attitudes associated with acid precipitation in Ontario: The amenity value survey. Report to the Ontario Ministry for the Environment, Toronto.

Arrow, K.J. and Fisher, A.C. 1974. Environmental preservation, uncertainty, and irreversibility. Quarterly Journal of Economics 88, 313-319.

Baker, C.K., Colls, J.J., Fullwood, A.E. and Seaton, G.G.R. 1986. Depression of growth and yield in winter barley exposed to sulphur dioxide in the field. New Phytologist 104, 233-241.

Barrett, C. F., Fowler, D., Irwing, J.G., Kallend, A.S., Martin, A., Scriven, R.A. and Tuck, A.F. 1982. Acidity of rainfall in the United Kingdom. A Preliminary Report. Warren Spring Laboratory, Stevenage.

Bateman, I.J., Langford, I.H. Willis, K.G. Turner, R.K. and Garrod, G.D. 1993. The impacts of changing willingness to pay format in contingent valuation studies: An analysis of open-ended, iterative bidding and dichotomous choice formats. paper presented at the 4th EAERE Conference, Fontainableu, France.

Battarbee, R.W., Flower, R.J., Stevenson, A.C. and Rippey, B. 1985. Lake acidification in Galloway: a palaeoecological test of competing hypothesis. Nature 314, 350-352.

Battarbee, R.W., Flower, R.J., Stevenson, A.C., Jones, V.J. , Harriman, R. and Appleby, P.G. 1988. Diatom and chemical evidence for reversibility of acidification of Scottish lochs. Nature 332, 530-532.

Battarbee, R.W. 1989. Geographical research on acid rain 1: The acidification of Scottish lochs. The Geographical Journal 155, 353-377.

Baumol, W. 1972. On taxation and the control of externalities. American Economic Review, 62, 307-322.

Beebee, T.J.C. 1987. Eutrophication of heathland ponds at a site in southern England: causes and effects, with particular reference to the Amphibia. Biol. Conserv. 42, 39-51.

Beebee, T.J.C., Flower, R.J., Stevenson, A.C., Patrick, S.T., Appleby, P.G., Fletcher, C., Marsh, C., Natkanski, J. Rippey, B. and Battarbee, R.W. 1990. The decline of the Nattarjack toad *Bufo calamita* in Britain: Palaeoecological, documentary and experimental evidence for breeding site acidification. Biological Conservation, 53, 1-20.

Bergland, O., Musser, W., Musser, L. and Terry, K. 1987. Optimal sampling intensities in close-ended contingent valuation methods. Dept. of Agricultural Economics, Oregon State University, Corvallis, OR.

Bergstrom, T.C. 1982. When is a man's life worth more than his human capital? In 'The value of life and safety: Proceedings of a conference held by the Geneva Association, Jones-Lee, M.V. (ed), Amsterdam.

Bishop, R.C. 1982. Option Value: An exposition and extension. Land Economics 58, 1-15.

Bishop, R.C. and Boyle, K.L. 1985. The economic value of Illinois Beach State Nature Reserve. Final Report to the Illinois Department of conservation, University of

Wisconsin, Madison.

Bishop, R. C. and Heberlein, T. A. 1979. Measuring values of extra-market goods: Are indirect measures biased? American Journal of Agricultural Economics 61, 926-930.

Bishop, R. C. and Heberlein, T. A. 1986. Does contingent valuation work? In Valuing Environmental Goods: An assessment of the contingent valuation method, Cummings, R.G., Brookshire, D.S. and Schultze, W.D. (eds), Rowman and Allanheld, Totowa, NJ.

Bishop, R.C. and Heberlein, T.A. 1990. the contingent valuation method. In 'Economic valuation of natural resources. Johnson, R.L., and Johnson, G.V. (eds.), Westview Press, Boulder, Colorado.

Blackburn, M. Harrison, G.W. and Rutstrom, E.E. 1993. Statistical bias functions and informative hypothetical surveys. Economics Working Paper Series B-93-02, College of Business Administration, The University of South Carolina.

Bohm, P. 1972. Estimating demand for public goods: an experiment. European Economic Review, 3: 111-130.

Bonnieux, F., Desaigues, B. and Vermersch, D. 1993. France. In 'Pricing the European Environment', Navrud, S. (ed), Scandinavian University Press, Stockholm.

Bowers, J. 1992. Economics of the environment, the conservationists response to the Pearce report, British Association of Nature Conservationists, Telford.

Bowker, J.M. and Stoll, J.R. 1988. Use of dichotomous nonmarket methods to value the whooping crane resource. American Journal of Agricultural Economics 70, 372-381.

Boyle, K.L. and Bishop, R.C. 1984. Economic benefits associated with boating and canoeing on the lower Wisconsin river. Economic Issues No. 84, Dept. of Agricultural Economics, University of Wisconsin, Madison.

Boyle, K.L. and Bishop, R.C. 1988. Welfare measurement using contingent valuation: A comparison of techniques. American Journal of Agricultural Economics 70, 20-29.

Boyle, K.L., Bishop, R.C. and Welsh, M.P. 1986. Starting point bias in contingent valuation bidding games. Land Economics 62, 16-24.

Boyle, K.L., Welsh, M.P. and Bishop, R.C. 1988. Validation of empirical measures of welfare change: comment. Land Economics 64, 94-98.

Braat, L.C. and Ruyssenaars, P.G. 1994. Estimating the benefits of environmental improvement to the forest sector.

Paper presented at the 'Workshop on Economic evaluation of damage caused by acidifying pollutants', CSERGE, London.

Brookshire, D.S., Ives, B.C., and Schulze, W.D. 1976. The valuation of aesthetic preferences. Journal of Environmental Economics and Management 3(4), 325-346.

Brookshire, D.S., Eubanks, L.S., and Randall, A. 1983. Estimating option price and existence values for wildlife resources. Land Economics 59 (1), 1-15.

Brookshire, D.S., Thayer, M.A., Schulze, W.D. and d'Arge, R.C. 1982 Valuing public goods: A comparison of survey and hedonic approaches. American Economic Review 72, 165-176.

Brown, G. and Mendelsohn, R. 1984. The hedonic travel cost model. Review of Economics and Statistics 66, 427-433.

Buckland, S.T., Macmillan, D.C., Duff, E.I. and Hanley, N. 1996. Estimating willingness to pay from dichotomous choice contingent valuation studies. The Statistician (in press).

Bulger, A.J., Lien, L., Cosby, B.J. and Henriksen, A. 1993. Trout status and chemistry from the Norwegian thousand lake survey: statistical analysis. Canadian Journal of Fisheries and Aquatic Science 50: 575-585.

Burness, H.S., Cummings, R.G., Mehr, A.F., and Walbert, M.S. 1983. Valuing policies which reduce environmental risk. Natural Resources Journal 23, 675-682.

Callaway, J. M., Darwin, R.F., and Nesse, R.J. (1986). Economic Effects of Hypothetical Reductions in Tree Growth in the Northeastern and Southeastern United States, Pacific Northwest Laboratory, Richland, Washington.

Cameron, T.A. 1988. A new paradigm for valuing non-market goods using referendum data: maximum likelihood estimation by censored logistic regression. Journal of Environmental Economics and Management 15, 355-379.

Carson, R.T., Flores, N.E., Martin, K.M., and Wright, J.L. 1996. Contingent valuation and revealed preference methodologies: Comparing the estimates for quasi-public goods. Land Economics 72(1), 80-99.

Cape, J.N., Fowler, D., Eamus, D., Murray, M.B., Sheppard, L.J. and Leith, I.D. 1990. Effects of acid mist and ozone on frost hardiness of Norway spruce seedlings. In: Environmental research with Plants in Closed Chambers, ed. H.D. Payer, T. Pfirrman, and P. Mathy, 331-335. Air Pollution Research Report 26, CEC, Brussels.

Carson, R.T. and Mitchell, R.C. 1993a. The issue of scope in contingent valuation studies. American Journal of Agricultural Economics 75(5),1263-1267.

Carson, R.T. and Mitchell, R.C. b. The value of clean water: the public's willingness to pay for boatable, fishable, and swimmable quality water. Water Resources Research 29 (7): 2445-2454.

Carson, R.T., Hanemann, W.M. and Mitchell, R.C 1986. The use of simulated political markets to value public goods. Economics Department, University of California, San Diego.

Carson, R.T., Hanemann, W.M., Kopp, R.J., Krosnick, J.A., Mitchell, R.C., Presser, S., Ruud, P.A. and Smith, V.K. 1996. Was the NOAA panel correct about Contingent Valuation. Discussion Paper 20-96, Resources for the Future, Washington, DC.

Census, 1991. Census of population. General Register Office (Scotland). HMSO, Edinburgh.

Commission on Energy and the Environment. 1981. Coal and the Environment. HMSO, London.

Ciriacy-Wantrup, S.V. 1952. Resource Conservation: Economics and Policies. University of California Press, Berkeley.

CLAG. 1994. Critical Loads of Acidity in the United Kingdom. Critical Loads Advisory Group Summary Report. ITE, Penicuik.

Clawson, M. and Knetsch, J. 1966. Economics of outdoor recreation. The Johns Hopkins University Press, Baltimore.

Coase, R. H. 1960. The Problem of Social Cost. Journal of Law and Economics 3, 1-44.

Cooper, J.C. 1993. Optimal bid selection for dichotomous choice contingent valuation surveys. Journal of Environmental Economics and Management 24, 25-40.

Cosby, B.J., R.F. Wright, G.M. Hornberger and J.N. Galloway 1985a. Modelling the effects of acid deposition: assessment of a lumped parameter model of soil water and streamwater chemistry. Water Resour. Res. 21, 51-63.

Cosby, B.J., Jenkins, A., Ferrier, R.C., Miller, J.D. and Walker, T.A.B. 1990. Modelling stream acidification in afforested catchments: long term reconstructions at two sites in central Scotland. Journal of Hydrology 120: 143-162.

Coursey, D.L. and Schulze, W.D. 1986. The application of laboratory experimental economics to the contingent valuation of public goods. Public Choice 49 (1), 47-68.

Cowling, E.B. 1989 Recent changes in chemical climate and related effects on forests in North America and Europe.

Ambio 18(3), 167-171.

Crocker, T.D. and Regens, J.L. 1985. Acid Deposition control: a benefit-cost analysis: its prospects and its limits. Environmental Science and Technology 19(2), 112-116.

Cummings, R.G., Brookshire, D.S. and Schulze, W.D.(eds). 1986. Valuing Environmental Goods: An assessment of the contingent valuation method. Rowman and Allenhead, Totowa, NJ.

Cummins, C.P. 1986. Effects of alluminium and low pH on growth and development in *Rana temporaria* tadpoles. Oecologica (Berl.) 69, 248-252.

Davis, R.K. 1963. Recreational planning as an economic problem. Natural Resource Journal, 238-249.

Department of the Environment 1992a. The UK Environment. Government Statistical Service, HMSO, London.

Department of the Environment 1992b. This Common Inheritance. The Second Year Report. HMSO, London.

Department of the Environment 1993. Air pollution and Tree Health in the United Kingdom. HMSO, London.

Desvousges, W.H., Smith, V.K. and McGivney, M.P. 1983. A comparison of alternative approaches for estimating recreation and related benefits of water quality improvements. Report No. EPA-230/05-83-001. U.S. Environmental Protection Agency, Washington, DC.

Desvousges, W.H., Smith, V.K. and Fisher, A. 1987. Option price estimates for water quality improvements: A contingent valuation study for the Monongahela River. Journal of Environmental Economics and Management 14, 248-267.

Desvousges, W.H., Naughton, M.C., and Parsons, G.R. 1992. Benefit transfer: Conceptual problems in estimating water quality benefits using existing studies. Water Resources Research 28, 675-683.

Desvousges, W.H. et al. 1993. Measuring natural resource damages with contingent valuation: Tests of validity and reliability. In 'contingent valuation: A critical assessment. Hausman, J. (ed), North Holland Press, Amsterdam, 91-164

Diamond, P.A., Hausman, J., Leonard, G.K., and Denning, M.A. 1993. Does contingent valuation measure preferences?. In Contingent valuation: A critical assessment. Hausman, J. (ed), North-Holland, NY

Diamond, P.A. and Hausman, J. 1994. Contingent Valuation:

Is some number better than no number? Journal of Economic Perspectives 8(4).

Dickie, M., Gerking, S., Brookshire, D.S., Coursey, D.L., Schulze, W.D., Coulson, A. and Tashkin, D. 1987. Reconciling averting behaviour and contingent valuation benefit estimates of reduced symptoms of ozone exposure draft. In 'Improving accuracy and reducing costs of environmental benefit assessments. USEPA, Washington, D.C.

Dickie, M. and Gerking, S. 1989. Benefits of reduced morbidity from air pollution control: a survey. In Folmer, H. and van Ireland, E. (eds). Valuation methods and policy making in environmental economics, Elseveir, Amsterdam.

Dickson, W. 1978. Some effects of the acidification of Swedish lakes. Verh. Internt. Verein. Limnol., 20:851-856.

Dillman, D.A. 1978. Mail and telephone surveys - The total design method. Wiley, New York.

Duffield, J.W. and Patterson, D.A. 1991. Inference and optimal design for a welfare measure in dichotomous choice contingent valuation. Land Economics 67, 225-239.

Duffield, J.W. and Patterson, D.A. 1993. Field testing existence values: An instream flow trust fund for Montana rivers. University of Montana.

Dunkley, D.A. 1986. Changes in the timing and biology of salmon runs. In Jenkins, D. and Shearer, W.M. (eds). The status of the Atlantic salmon in Scotland. ITE symposium 15.

Dupuit, J. 1844. On the measurement of the utility of public works. reprinted in 'Transport: Selected readings, Munby, D (ed.), Penguin books, 1968.

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

ECOTEC, 1993. Evaluating a cost-benefit analysis of reduced acid deposition: A contingent valuation study of aquatic ecosystems. Working Paper 5, ECOTEC, Birmingham.

ESEERCO, 1996. Environmental externalities cost study. Empire State Electric Energy Research Corporation, Oceana Publications Ltd.

ETSU and IER, 1995. Externalities of Energy Vol4: Oil and Gas. DGXII, European commission, Luxembourg.

European MEP, 1994. Transboundary acidifying pollution in Europe: calculated fields and budgets 1985-1993. MSC-W Report 1/94 Norwegian Meterological Institute, Blindern.

Environmental Resources Ltd. 1990. ACIDRAIN IV. Environmental Resources Limited, London.

Ewers, H. J. (1986). On the Monetisation of Forest Damages in the Federal Republic of Germany. Kosten der Umweitverschmutzung, Umweltbundestant Berichte 7/86, 121-143.

Farmer, A.M. 1990. The effects of lake acidification on aquatic macrophytes - a review. Environmental Pollution 65, 219-240.

Fischoff, B. and Furby, L. 1988. Measuring Values: A conceptual framework for interpreting transactions with special reference to contingent valuation of visibility. Journal of Risk and Uncertainty 1, 147-184.

Flower, R.J. and Battarbee, R.N. 1983. Diatom evidence for recent acidification of two Scottish lochs. Nature, 305: 130-132.

Flower, R.J., Patrick, S.T., Appleby, P.B., Oldfield, F., Rippey, B., Stevenson, A.C., Darley, J., Higgett, S.R. and Battarbee, R.W. 1986. Palaeological evaluation of the recent acidification of Loch Laidon, Rannoch Moor, Scotland. Report for the Dept. of the Environment, 29, Palaeoecology Research Unit, ed. by S.T. Patrick and A.C. Stevenson, Dept. of Geography, University College, London.

Flower, R.J., Battarbee, R.W., Natkanski, J., Rippey, B. and Appleby, P.G. 1988. The recent acidification of a large Scottish loch located partly within a National Nature Reserve and Site of Special Scientific Interest. J. Applied Ecology, 25, 715-724.

Forster, B.A. 1984. An Economic Assessment of the Significance of Long Range Transported Pollutants for Agriculture in Eastern Canada. Canadian Journal of Agricultural Economics, 32, 489-525.

Forster, B.A. 1984. Economic impact of acid precipitation: A Canadian perspective. In 'Economic perspectives on acid deposition control. Crocker, T.D. (ed), Butterworth Publ., Stoneham, MA, 97-122.

Fredman, P. 1994. A test of non-response bias in a mail contingent valuation survey. Arbetsrapport 201, Sveriges Lantbruksuniversitet, Institutionen for Skogsekonomi

Freeman, A.M. 1979. The benefits of environmental improvement: Theory and practice. Johns Hopkins Press, Baltimore.

Freeman, A.M. 1984. The size and sign of option value. Land Economics 60, 1-13.

Freeman, M.A. 1986. On assessing the state of the arts of the contingent valuation method of valuing environmental changes. In Valuing Environmental Goods: An assessment of the contingent valuation method, Cummings, R.G., Brookshire, D.S. and Schultze, W.D. (eds), Rowman and Allanheld, Totowa, NJ.

Freeman, A.M. 1993. The Measurement of Environmental and Resource Values. Resources for the Future, Washington D.C.

Freer-Smith, P.H. 1985. The influence of SO_2 and NO_2 on the growth, development and gas exchange of Betula pendula (Roth). New Phytol., 99, 417-430.

Friberg, F., Otto, C. and Svensson, B. 1980. Effects of acidification on the dynamics of allochthonous leaf material and benthic invertebrate communities in running water. In: D. Drablos and A. Tollan (eds) Ecological Impact of Acid Precipitation (Proc), Sandefjord, Norway, pp 304-305.

Fry, G.L.A. and Cooke, A.S. 1987 Acid Deposition and its Implications for Nature Conservation in Britain. Focus on Nature Conservation No. 7, Nature Conservancy Council, Peterborough.

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

Gerking, S. and Stanley, L. 1986. An economic analysis of air pollution and health: the case of S. Louis. Review of Economics and Statistics, Vol. LXVIII (1), 115-121.

Government Statistical Office, 1991. The Scottish Environment - statistics. The Scottish Office, Edinburgh.

Government Statistical Service, 1996. Digest of Environmental Statistics No. 18, HMSO, London.

Grahn, O., Hultberg, H. and Laudner, L. 1974. Oligotrophication- a self-accelerating process in lakes subjected to excessive supply of acid substances. Ambio, 3: 377-391.

Grandstaff, S. and Dixon, J.A. 1986. Evaluation of Lumpineee public park in Bangkok, Thailand. In 'Economic valuation techniques for the Environment' Dixon, J.A. and Hufschmidt, M.M., The Johns Hopkins University Press, Baltimore, 121-140

Green, J. and Green, R. 1987. Otter Survey of Scotland 1984-85. Vincent Wildlife Trust, London.

Green, C.H. and Tunstall, S.M. 1991. The evaluation of river quality improvements by the CVM. Applied Economics 23, 1135-1146.

H.M. Treasury 1991. Economic appraisal in central government: A technical guide for government departments. HMSO, London.

Hagstrom, T. 1981. Reproductive strategy and success of amphibians in waters affected by atmospheric pollution. In Proceedings of the European Herpetological Symposium, Oxford 1980, 55-57.

Halkos, G.E. 1994. Optimal Abatement of Sulphur Emissions in Europe. Environmental and Resource Economics 4 (2), 127-150.

Hanemann, W.M. 1984. Welfare evaluations in contingent valuation experiments with discrete responses. american Journal of Agricultural Economics 66, 332-341.

Hanemann, W.M. 1994. Valuing the environment through contingent valuation. Journal of Economic Perspectives 3, 1-23.

Hanemann, W.M. 1991. Willingness to pay and Willingness to Accept: How much can they differ? The American Economic Review 81(3), 635-647.

Hanemann, W.M., Loomis, J. and Kanninen, B. 1991 Statistical Efficiency of double-bounded dichotomous choice contingent valuation. American Journal of Agricultural Economics, 73(4): 1255-1263.

Hanley, N. 1989. Valuing rural recreation benefits: An empirical comparison of two approaches. Journal of Agricultural Economics 40(3), 361-374.

Hanley, N. and Craig, S. 1991. Wilderness development decisions and the Krutilla-Fisher model: The case of Scotland's Flow Country. Ecological Economics 4, 145-164.

Hanley, N. and Spash, C. 1993. Cost-Benefit Analysis and the Environment. Edward Elgar, Cheltenham.

Hanley, N. and Owen, R. 1994. Embedding, nesting and component sensitivity in the contingent valuation of Sites of Special Scientific Interest in the UK. Working paper, Department of Economics, University of Stirling (*in mimeo*).

Hanley, N. Shogren, J.F. and White, B. 1996. Environmental Economics: In theory and Practice. Macmillan, London.

Hanley, N, Simpson, I., Parsisson, D., Macmillan, D., Bullock, C. and Crabtree, J.R. 1996b. Valuation of the conservation benefits of Environmentally Sensitive Areas. ESEG Report 1-96, Macaulay Land Use Research Institute, Aberdeen, Scotland.

Harriman, R. and Morrison, B.R.S. 1982. Ecology of streams draining forested and non-forested catchments in an area of central Scotland subject to acid precipitation. Hydrobiologia 88, 251-263.

Harriman, R., Morrison, B.R.S. Caines, L.A., Collen, P. and Watt, A.W. 1987. Long term changes in fish populations of acid streams and lochs on Galloway, south-west Scotland. Water, soil and Air Pollution, 32; 89-112.

Harrison, G.W., Beekman, R.L., Brown, L.B., Clements, L.A., McDaniel, T.M., Odom,S.L. and williams, M.B. 1993. Environmental damage assessment with hypothetical surveys: The calibration approach. Economics Working Paper Series B-93-18, College of Business Administration, The University of South Carolina.

Hasselrot, B. and Hultberg, H. 1981. Acid rain - effects on aquatic life and ground water. In: Report of the European Conference on Acid Rain, Goteberg, 1981, pp:49-58.

Haya, K. and Waiwood, B. 1981. Acid pH and chorinase activity of Atlantic salmon eggs. Bulletin of Environmental Toxiocology 27, 7-12.

Heinz, I. 1994. Monetary Value of Damages to Buildings and Materials in Germany. Paper presented at Workshop on 'Economic Evaluation of Damage Caused by Acidifying Pollutants'. CSERGE/Imperial College, London.

Henderson, N. and Bateman, I. 1993. Intergenerational discounting: Public choice and empirical evidence for hyperbolic discount rates. CSERGE Working Paper GEC 93-02, University of East Anglia and University College London.

Hendrey, G.R. and Vertucci, F.A. 1980. Benthic plant communities in acidic Lake Colden, New York: Sphagnum and the algal mat. In: D. Drablos and A. Tollan (eds). Ecological impact of acid Precipitation (Proc), Sandefjord, Norway, pp:314-315.

Henriksen, A., Lein, L., Rosseland, B.O., Traan, T.S. and Sevaldrud, I.S. 1989. Lake acidification in Norway: Present and predicted fish status. Ambio 18, 314-321.

Hicks, J.R. 1941. The rehabilitation of consumer's surplus. Review of Economic Studies 8, 108-116.

Hoehn, J.P. and Randall, A. 1987. A satisfactory benefit cost indicator from contingent valuation. Journal of Environmental Economics and Management, 14: 226-247.

Hoevenagel, R. 1993a. An assessment of the contingent valuation method. In Valuing the environment: Methodological and measurement issues. Pethig, R. (ed). Kluwer Academic Publishers, Dordrecht.

Hoevenagel, R. 1993b. A comparison of economic valuation methods. In Valuing the environment: Methodological and measurement issues. Pethig, R. (ed). Kluwer Academic Publishers, Dordrecht.

Hufschmidt, M.M. 1986. The Nam Pong water resources project in Thailand. In 'Economic valuation techniques for the Environment' Dixon, J.A. and Hufschmidt, M.M., The Johns Hopkins University Press, Baltimore, 121-140

Huppert, D.D. 1989. Measuring the value of fish to anglers: Application to central California anadromous species. Marine Resource Economics 6, 89-107.

Innes, J.L. and Boswell, R.C. 1990. Monitoring of forest condition in the United Kingdom 1989. Forestry Commission Bulletin 88. HMSO, London.

Joint Nature Conservation Council, 1996. The Relative Contribution of Different Sulphur Point Sources - Acidification on SSSI's in Britain. Joint Nature Conservancy Council Report 260, NHBS, Devon.

Jenkins, A., cosby, B.J., Ferrier, R.C., Walker, T.A.B. and Miller, J.D. 1990. Modelling stream acidification in afforested catchments: An assessment of the relative effects of acid deposition and afforestation. Journal of Hydrology 120, 163-181.

Johansson, P.O. 1988. Valuing public goods in a risky world: An experiment. In H. Folmer and E. van Ierland (eds.). Valuation methods and policy making in environmental economics, Elsevier, Amsterdam.

Johansson, P.O. 1990. Willingness to pay measures and expectations: An experiment. Applied Economics 22, 313-329.

Johansson, P.O. 1992. Altruism in cost-benefit analysis. Environmental and Resource Economics 2: 605-613.

Johansson, P.O. and Kristrom, B. 1988. Measuring values for improved air quality from discrete response data: two experiments. Journal of Agricultural Economics 39 (3), 439-445.

Jones-Lee, M.W. 1992. Paternalisitc altruism and the value of a statistical life. The Economic Journal 102, 80-90.

Jones-Lee, M.W., Hammerton, W.M. and Philips, P.R. 1985. The value of safety: results of a national sample survey. The Economic Journal 95: 49-72.

Jones, V.J., A.C. Stevenson and R.W. Battarbee 1986. Lake acidification and the land use hypothesis: a mid-post-glacial analogue. Nature 322, 157-158.

Kahneman, D. 1986. Comments on the Contingent Valuation Method. In Valuing Environmental Goods: An assessment of the contingent valuation method, Cummings, R.G., Brookshire, D.S. and Schultze, W.D. (eds), Rowman and Allanheld, Totowa, NJ.

Kahneman, D. and Tversky, A. 1979. Prospect theory: An analysis of decisions under risk. Econometrica 47(2), 263-291.

Kahneman, D., Solvic, P. and Tversky, A. 1982. Judgement under uncertainty: Heuristics and biases. Cambridge University Press, Cambridge.

Kahneman, D. and Knetsch, J.L. 1992. Valuing Public Goods: The purchase of moral satisfaction. Journal of Environmental Economics and Management 22, 57-70.

Kanninen, B.J. 1995. Bias in discrete response contingent valuation. Journal of Environmental Economics and Management 28 (1), 114-125.

Kealy, M.J., Montgomery, M. and Dovidio, J.F. 1990. Reliability and predictive validity of contingent values: Does the nature of the good matter? Journal of Environmental Economics and Management 19, 244-263.

Kealy, M.J. and Turner, R.W. 1993. A test of the equality of closed-ended and open-ended contingent valuations. American Journal of Agricultural Economics 75, 321-331.

King, D.A. and Sinden, J.A. 1988. Influence of soil conservation on farm land values. Land Economics 64(3), 242-255.

Klaassen, G. 1995. Trade-Offs in Sulphur Emission Trading in Europe. Environmental and Resource Economics 5(2), 191-219.

Knetsch, J.L. 1992. Preferences and nonreversibility of indifference curves. Journal of Economic Behaviour and Organisation 17, 131-139.

Knetsch, J.L. 1993. Environmental valuation: some practical problems of wrong questions and misleading answers. Resource Assessment Commission, Occasional Paper 5, Australian Government Publishing Service, Canberra.

Knetsch, J.L. and Davis, R.K. 1966. Comparisons of methods for recreation evaluation. In Kneese, A.V. and Smith, S.C. Water Research, Resources for the Future Inc., John Hopkins Press, Baltimore, 125-142.

Kennet, D. 1980. Altruism and economic behaviour: developments in the theory of public and private distribution. American Journal of Economics and Sociology

39, 183-189.

Knetsch, J.L. and Sinden, J.A. 1984. Willingness to pay and compensation demanded: Experimental evidence of an unexpected disparity in measures of value. Quarterly Journal of Economics 94 (3), 507-521.

Kristrom, B. 1990. A non-parametric approach to the estimation of welfare measures in discrete response valuation studies. Land Economics 66 (2), 135-139.

Kristrom, B. 1993. Comparing continuous and discrete contingent valuation questions. Environmental and Resource Economics 3, 63-71.

Kristrom, B. 1995. spike models in contingent valuation: Theory and illustrations. Arbetsrapport 210, Sverigies Lantbruksuniversitet: Institutionen for Skogsekonomi, Umea, Sweden.

Kristrom, B., and riera, P. Is the income elasticity of environmental improvements less than one? Environmental and Resource Economics, 7(1), 45-55.

Krutilia, J.V. 1967. Conservation reconsidered. American Economic Review 57, 787-796.

Lacroix, G.L. 1987. Model for loss of Atlantic salmon stocks from acidic brown waters of Canada. In 'Acid rain: Scientific and technical advances. Ponds, H. (ed), Selper, London.

Lancaster, K. 1966. A new approach to consumer theory. Journal of Political Economy 74, 132-157.

Langford, I.H. and Bateman, I.J. 1993. Welfare measurements for contingent valuation studies: Estimation and reliability. CSERGE Working paper GEC 93-04, University College London and University of East Anglia.

Le Blanc, F. and Rao, D.N. 1973. Effects of sulphur dioxide on lichen moss transplants. Ecology, 54:612-617.

Lindvall, B. 1984. The acid rain menace. Deer, 6:65-66.

Likens, G.E., Wright, R.F., Galloway, J.N. and Butler, T.J. 1979. Acid Rain. Scientific American 241(4); pp 39-47.

Loewenstein, G. and Prelec, D. 1992. Anomalies in intertemporal choice: Evidence and an interpretation. The Quarterly Journal of Economics 10; 575-597.

London Economics 1992. The Potential Role of Market Mechanisms in the Control of Acid Rain. Department of the Environment Environmental Economics Research Series, HMSO, London.

Loomis, J.B. 1987. Expanding contingent value sample estimates to aggregate benefit estimates: Current practices and proposed solutions. Land Economics 63 (4), 396-402.

Loomis, J.B. 1990. Comparative reliability of dichotomous choice and open-ended contingent valuation methods. Journal of Environmental Economics and Management 72, 78-85.

Loomis, J.B. and Walsh, R.G. 1986. Assessing wildlife and environmental values in cost-benefit analysis: State of the art. Journal of Environmental Management 22, 125-131.

Louviere, J.J. 1988. Conjoint analysis modelling of stated preferences. Journal of Transport Economics and Policy. 22:93-119.

McCollum, D.W., Gilbert, A.H. and Peterson, G.L. 1990. The net economic value of day use cross country skiing in Vermont: A dichotomous choice contingent valuation approach. Journal of Leisure Research 22, 46-56.

McConnell, K.E. 1983. Existence and bequest value. In 'Managing air quality and scenic resources at National Parks and Wilderness Areas. In Lowe, R.P. and Chestnut, L.G. (eds), Westview Press, Boulder, Colorado.

McConnell, K.E. 1990. Models for referendum data: The structure of discrete choice models for contingent valuation. Journal of Environmental Economics and Management 72, 19-34.

McConnell, K.E.1997. Does altruism undermine existence value? Journal of Environmental Economics and Management 32(1), 22-37.

Mackay Consultants 1989. Economic importance of salmon fishing and netting in Scotland. HIDB/STB, Inverness.

McLeod, A.R., Holland, M.R., Shaw, P.J.A., Sutherland, P.M. Darrall, N.M. and Skeffington, R.A. 1990. Enhancement of nitrogen deposition to forest trees exposed to SO2. Nature, 347, 277-279.

Macmillan, D.C. 1992. Predicting the general yield class of sitka spruce on better quality land in Scotland. Forestry 64 (4), 359-372.

Macmillan, D.C. and Ferrier, R.C. 1994. A bioeconomic model for estimating the benefits of acid rain abatement to salmon fishing: a case study in south-west Scotland.

Macmillan, D.C., Hanley, N., and Buckland, S.T. 1996. A contingent valuation study of uncertain environmental gains. Scottish Journal of Political Economy 43 (5), 519-533.

Macmillan, D.C. 1996. Non-market benefits of restoring native woodlands. In proc of 'International Symposium on the non-market benefits of forestry.' Edinburgh.

McNeill, S. and Whittaker, J.B. 1990. Air pollution and tree-dwelling aphids. In: Population dynamics of Forest Insects. Ed: A.D. Watt, S.R. Leather, M.D. Hunter and N.A.C. Kidd, 195-208. Intercept Ltd., Andover.

McWilliams, P.G. 1982. A comparison of physiological characteristics in normal and exposed populations of the brown trout. Biochemical Physiology 72A (3), 515-522.

Milgrom, P. 1993. Is sympathy an economic value? Philosophy, Economics, and the contingent valuation method. In 'Contingent valuation: A critical assessment'. Hausman, J. (ed), North-Holland, NY

Miller, J.D., Anderson, H.A., Cooper, J.M., Ferrier, R.C. and Stewart, M. 1991. Evidence for enhanced atmospheric sulphate deposition by Sitka spruce from evaluation of some Scottish catchment study data. Science of the Total Environment 103: 37-46.

Milner, N.J. and Varallo, P.V. 1990. Effects of acidification on fish and fisheries in Wales. In: Acid Waters in Wales, ed: R.W. Edwards, A.S. Gee, and J.H. Stoner, 121-144. Dordrecht, Kluwer.

Mitchell, R.C. and Carson, R.T. 1981. An experiment in determining willingness to pay for national water quality improvements. Report to the US Environmental Protection Agency, Washington DC.

Mitchell, R.C. and Carson, R.T. 1989. Using surveys to value public goods: The contingent valuation method. Resources for the Future, Washington.

Mjelde, J.W. Adams, R.M., Dixon, B.L. and Garcia, P. 1984. Using farmers actions to measure crop loss due to air pollution. Journal of Air Pollution Control Association 34; 360-365.

Montgomery, W.D. 1972. Markets in Licences and Efficient Pollution Control Programs. Journal of Economic Theory 5, 395-418.

Mullen, J.K. and Menz, F.C. 1985. The Effect of Acidification Damages on the Economic Value of the Adirondack fishery to New York anglers. American Journal of Agricultural Economics 67; 112-119.

National Acid Precipitation Assessment Program 1991. NAPAP -1990 Integrated Assessment Report, The NAPAP Office of the Director, Washington DC.

National Oceanic and Atmospheric Administration 1993.

Natural Resource Damage Assessments: proposed rules. Federal Register 59(5), 1062-1191

Navrud, S. 1989. Estimating social benefits of Environmental Improvements from reduced acid depositions: a contingent valuation survey. In H. Folmer and E. van Ierland (eds.). Valuation methods and policy making in environmental economics, Elsevier, Amsterdam.

Navrud, S. and Veisten, K. 1996. Validity of nonuse values in contingent valuation: an empirical test with real payments. Paper presented to 7th EAERE Conference, Lisbon.

Neill, H.R., Cummings, R.G., Ganderton, P.T., Harrison, G.W. and McGuckin, T. 1994. Hypothetical surveys and real economic commitments. Land Economics 70(2), 145-254.

Newbery, D. M. 1994. The impact of EC environmental policy on British coal. Oxford Review of Economic Policy 9 (4); 66-95.

Nillson, S. 1991. European Forest Decline: the effects of air pollutants and suggested remedial policies. International Institute for Applied Systems Analysis, Luxenburg, Austria

Organisation for Economic Cooperation and Development, 1989. Environmental Policy Benefits: Monetary valuation. OECD, Paris.

Organisation for Economic Cooperation and Development, 1996. Environmental taxes in OECD countries. OECD, Paris.

Organisation for Economic Cooperation and Development, 1997. Main Economic Indicators, Feb 1997, Statistics Directorate, OECD, Paris.

O'Byrne, P., Nelson, J. and Seneca, J. 1985. Housing values, census estimates, disequilibrium and the environmental cost of aircraft noise. Journal of Environmental Economics and Management 12, 169-178.

Olsen, D., Richards, J. and Scott, R.D. 1991. Existence and sport values for doubling the size of Columbia river basin salmon and steelhead runs. Rivers 2 (1), 44-56.

Ormerod, S.J., Bull, K.R., Cummins, C., Tyler, S.J. and Vickery, J.A. 1988. Egg mass and shell thickness in dippers *Cinclus cinclus* in relation to stream acidity in Wales and Scotland. Environmental Pollution 55, 107-121.

Ormerod, S.J., Weatherly,N.S. and Whitehead, P.G. 1990. Temporal patterns of ecological change during the acidification and recovery of streams throughout Wales according to a hydrochemical model. In 'Regional Acidification Models: Geographic extent and time development. Kamari, J., Brakke, D.F., Jenkins, A., Norton, S.A., and Wright, R.F. (eds), Springer-Verlag, Berlin.

Overrein, L.N., and Seip, H.M., and Tollan, A. 1980. Acid precipitation - effects on forest and fish. Final Report of the SNSF project 1972-1980., Oslo - As., Norway.

Page, W.P., Fabian, A. and Cieka, B. 1982. Estimation of economic losses to the agricultural sector from airborne residuals in the Ohio river basin region. Journal of Air Pollution Control Association 32, 360-365.

Palmquist, R.B. and Danielson, L.E. 1989. A hedonic study of the effects of erosion control and drainage on farmland values. American Journal of Agricultural Economics 71, 55-62

Park, T., Loomis, J.B. and Creel, M. 1991. Confidence Intervals for evaluating benefits estimates from dichotomous choice contingent valuation studies. Land Economics 67(1), 64-73.

Pearce, D.W. 1983. Cost-Benefit Analysis. Macmillan (2nd edition).

Pearce, D.W., Markandya, A., and Barbier, E. 1989. Blueprint for a green economy. Earthscan Publications Ltd., London.

Pearce, D.W., Barbier, E., and Markandya, A. 1990. sustainable development: Economics and environment in the third world. Edward Elgar, Aldershot.

Pearce, D.W. and Turner, R.K. 1990. Economics of natural resources and the environment. Harvester Wheatsheaf, Hemel Hempstead.

Pearce, D.W., Bann, C, and Georgiou, S. 1992. The social cost of fuel cycles. HMSO, London.

Pearce, D.W. 1996. Economic valuation and ecological economics. Plenary address to the european Society for Ecological Economics Inaugural International Conference, University of Versaille, Guyancourt, France.

Pearman, A. 1994. The use of stated preference methods in the evaluation of environmental change. In Pethig, R. (ed). Valuing the environment: methodological and measurement issues.

Pigou, A.C. 1920. Economics of Welfare. Macmillan, London.

Prestt, I, Cooke, A.S. and Corbett, K.F. 1974. British Amphibians and Reptiles. In: D.L. Hawkesworth (ed) The Changing Flora and Fauna of Britain. Systematics Association Special Volume No 6, 229-254.

Querner, I. 1994. The need for alternatives to the expected utility approach in environmental risk economics or 'who is afraid of Russian roulette'. In Pethig, R. (ed). Valuing the environment: methodological and measurement issues.

Radford, A.F., Hatcher, A.C., and Whitmarsh, D.J. 1991. An economic evaluation of salmon fisheries in Great Britain: Summary Report. Centre for Marine Resource Economics, Portsmouth Polytechnic.

Randall, A. The possibility of satisfactory benefit estimation with contingent markets. In Valuing Environmental Goods: An assessment of the contingent valuation method, Cummings, R.G., Brookshire, D.S. and Schultze, W.D. (eds), Rowman and Allanheld, Totowa, NJ.

Randall, A., Ives, B.C. and Eastman, C. 1974. Bidding games for valuation of aesthetic environmental improvements. Journal of Environmental Economics and Management 1: 132-149.

Randall, A. and Stoll, J.R. 1980. Consumer's surplus in commodity space. American Economics Review 70(3), 449-455.

Randall, A. and Stoll, J.R. 1983. Existence value in a total valuation framework. In 'Managing air quality and scenic resources at National Parks and Wilderness Areas. In Lowe, R.P. and Chestnut, L.G. (eds), Westview Press, Boulder, Colorado.

Raven, P.J. 1985. The use of aquatic macrophytes to assess water quality changes in some Galloway lochs: an exploratory study. Research Paper No. 9, Palaeoecology Reserach Unit, University College London, pp 76.

Ready, R. C. and Hu, D. 1995. Statistical approaches to the fat-tail problem for dichotomous choice contingent valuation. Land Economics 71, 491-499.

Reiling, S.D., Boyle, K.J., Philips, M.L., and Anderson, M.W. 1990. Temporal reliability of contingent values. Land Economics 66, 128-134.

Rosseland, B.O. 1980. Effects of acid water on metabolism and gill ventilation in brown trout and brook trout. In 'Ecological implications of acid rain', Sandefjord, Norway.

Rosseland, B.O. and Skogheim, O.K. 1984. A comparative study on salmonid fish species in acid alluminium-rich water: physiological stress and mortality rates of one and two year old fish. report of the Institute of Freshwater Research 61, 186-194 (*in mimeo*).

Rowe, R.D.,d'Arge, R.C. and Brookshire, D.S. 1980. An experiment on the economic value of visibility. Journal of Environmental Economics and Management 7: 1-19.

Rico, R. 1995. The U.S. Allowance Trading System for Sulphur Dioxide: An Update on Market Experience. Environmental and Resource Economics 5(2), 115-129.

Rimes, C. 1992 Freshwater acidification of SSSI's in Great Britain I: Overview. English Nature 1992.

Roberts, T.M., Skeffington, R.A. and Blank, L.W. 1989. Causes of Type I spruce decline in Europe. Forestry 62, 179-222.

Rose, C. 1990. The Dirty Man of Europe: the Great British Pollution Scandal. Simon and Schuster, London.

Samuelson, P. 1954. The Pure Theory of Public Expenditure. Review of Economics and Statistics 36, 387-389.

Schulze, W.D., Brookshire, D.S. and Walther, E.G 1983. The economic benefits of preserving visibility in the national parklands of the south-west. Natural Resources Journal 23, 149-173.

Schuman, H. and Presser, S. 1981. Questions and answers in attitude surveys. Academic Press, New York.

Scotsman, 1996. Acid rain confirmed as cause of car damage. Scotsman, 6/7/96, p. 8.

Scottish Anglers National Association, 1991. A survey of members environmental concerns. SANA, Edinburgh.

Scottish Office Agriculture and Fisheries Department, 1987. Scottish salmon and sea trout catches 1986. Statistical Bulletin Fisheries Series, The Scottish Office, Edinburgh.

Scottish Office Agriculture and Fisheries Department, 1988. Scottish salmon and sea trout catches 1987. Statistical Bulletin Fisheries Series, The Scottish Office, Edinburgh.

Scottish Office Agriculture and Fisheries Department, 1989. Scottish salmon and sea trout catches 1988. Statistical Bulletin Fisheries Series, The Scottish Office, Edinburgh.

Scottish Office Agriculture and Fisheries Department, 1990. Scottish salmon and sea trout catches 1989. Statistical Bulletin Fisheries Series, The Scottish Office, Edinburgh.

Scottish Office Agriculture and Fisheries Department, 1991. Scottish salmon and sea trout catches 1990. Statistical Bulletin Fisheries Series, The Scottish Office, Edinburgh.

Segerson, K. 1986. economic evaluation of air pollution damage and Control: Discussion. American Journal of Agricultural Economics 68, 479-481.

Seip, K. and Strand, J. 1992. Willingness to pay for environmental goods in Norway: A contingent valuation study with real payment. Environmental and Resource Economics 2: 91-106.

Sellar, C., Stoll, J.R., and Chavas, J.P. 1985. Validation of empirical measures of welfare change: A comparison of nonmarket techniques. Land Economics 61, 156-175.

Shaw, R.W. and Young, J.W.S. 1986. A proposed strategy for reducing sulphate deposition in North America - I. Methodology for minimising sulphur removal. Atmospheric Environment 20(1), 189-199.

Shearer, W.M. 1987. Relating catch records to stocks. In Mills, D. and Piggins, D. (eds). Atlantic salmon: Planning for the future. Proceedings of the third international atlantic salmon symposium, Biarritz, Crook Helm, London.

Shortle, J.S. and Abler, D.G. 1991. The political economy of nonpoint pollution control policy design for agriculture. Paper presented at the 24th seminar of the European Association of Agricultural Economists, Paris.

Sinden, J.A. 1988. Empirical tests of hypothetical biases in consumers' surplus surveys. Australian Journal of Agricultural Economics 32: 98-112.

Skeffington, R.A. 1994. A preliminary dose-response function for SO2 damage to forests. Paper presented at the 'Workshop on Economic evaluation of damage caused by acidifying pollutants', CSERGE, London.

Sliggers, J. and Klaassen, G. 1994. Cost sharing for the abatement of acidification in Europe: the missing link in the new sulphur protocol. European Environment 4(1), 5-11.

Smith, V.K. 1986. To keep or toss the contingent valuation method. In Valuing Environmental Goods: An assessment of the contingent valuation method, Cummings, R.G., Brookshire, D.S. and Schultze, W.D. (eds), Rowman and Allanheld, Totowa, NJ.

Solway River Purification Board 1992. Annual Report 1991-1992. Solway River Purification Board, Dumfries.

Stephen, A.B. 1990. West Galloway Fisheries Trust, Annual Report 1989-1990, Newton Stewart.

Stevens, T.H., Glass, R., More, T. and Echeverria, J. 1991. Wildlife Recovery: is Benefit-Cost Analysis appropriate? Journal of Environmental Management 33 (4), 327-334.

Stoll, J.R and Johnson, L.A. 1985. concepts of value, nonmarket valuation, and the case of the Whooping Crane. Texas Agricultural Research Station Article No. 19360, Department of Agricultural Economics, Texas A&M University.

Stoner, J.H. and A.S. Gee 1985. Effects of forestry on water quality and fish in Welsh rivers and lakes. J. Inst. Water Eng. Sci. 39, 27-45.

Strand, J., 1981. Valuation of freshwater fish as a public good in Norway. Institute of Economics, University of Oslo, Oslo, mimeo.

Sutcliffe, D.W. and Carrick, T.R. 1983. Acid rain and the river Duddon in Cumbria. Freshwater biological Association Pamphlet.

Svento, R. 1994. Testing the asymmetry of the vagueness zone in WTP answers. Paper presented at the 5th EAERE annual conference, Dublin 1994.

Sverdrup, H., de Vries, W and Henrikson, A 1990. Mapping Critical Loads. Nordic Council of Ministers, Copenhagen.

Swedish Environmental Protection Agency, 1994. Liming costs to counter water acidification in Sweden. Paper presented at the 'Workshop on Economic evaluation of damage caused by acidifying pollutants', CSERGE, London.

Tietenberg, T. 1992. Environmental and natural resource economics. Harper Collins Publishers Inc, New York (3rd edition).

Tietenberg, T. 1995. Tradeable Permits for Pollution Control when Emission Location Matters: What have We Learned? Environmental and Resource Economics 5(2), 95-113.

Thompson, D.B.A. and Baddeley, J. 1991. Some effects of acidic deposition on montane Racomitrium lanuginosum heaths. In: Focus on Nature Conservation 26, ed S.J. Woodin and A.M. Farmer, NCC.

United Kingdom Critical Loads Advisory Group 1991. Critical Load Maps for the United Kingdom. 1: Soils. Report to the Dept. of the Environment by the UK Critical Loads Advisory Group. HMSO.

United Kingdom Acid Waters Review Group 1988 Acidity in United Kingdom Fresh Waters. Second Report to the Department of the Environment, HMSO.

United Kingdom Review Group on Acid Rain 1983 Acid deposition in the UK. Report to the Dept. of the Environment, Warren Spring Lab.

Walsh, R.G., Ward, F.A. and Olienyk, J.P. 1989. Recreational demand for trees in national forests. Journal of Environmental Management 28, 255-268.

Walsh, R.G., Johnson, D.M. and McKean, J.R. 1992. Benefits transfer of outdoor recreation demand studies:1968-1988. Water Resources Research 28(3): 707-713.

Warrington, S. and Whittaker, J.B. 1990. Interactions between Sitka spruce, the green spruce aphid, sulphur dioxide pollution and drought. Environmental Pollution 65, 363-370.

Waters, D. and Kay, D. 1988. Trends in salmon catches - a critical evaluation of the United Kingdom Acid Waters Review Group interpretation. Area 19 (3), 223-235.

Weatherly, N.S. and Ormerod, S.J. 1991. The importance of acid episodes in determining faunal distributions in Welsh streams. Freshwater Biology 25, 71-84.

Weatherley, N.S. (1995). Liming to mitigate acidification in freshwater ecosystems: a review of the biological consequences. W.A.S.P.

Weisbrod, B.A. 1964. Collective consumption services of individual consumption goods. Quarterly Journal of Economics 78(3), 471-477.

Whittington, P., Smith, V.K., Okorafov, A. Okore, A., Lui, J.L. and McPhail, A. 1992. Giving respondents time to think in CV studies: A developing countries application. Journal of Environmental Economics and Management, 22(2), 205-225.

Williamson, R.B. 1987. Status of exploitation of Atlantic salmon in Scotland. In Mills, D. and Piggins, D. (eds). Atlantic salmon: Planning for the future. Proceedings of the third international atlantic salmon symposium, Biarritz, Crook Helm, London.

Willig, R. D. 1976. Consumer's surplus without apology. American Economic Review 66(4), 587-597.

Willis, G.B., Royston, P. and Bercini, D. 1991. The use of verbal report methods in the development and testing of survey questionnaires. Applied cognitive Psychology 5, 251-267.

Willis, K.G., 1990. Valuing non-market wildlife commodities: an evaluation and comparison of benefits and costs. Applied Economics 22, 13-30.

Willis, K.G. and Benson, J. 1988. Valuation of wildlife: A case study on the upper Teesdale site of special scientific interest and comparison of methods in environmental economics. In ' sustainable Environmental Management: Principles and practice, Turner, R.K. (ed), Belhaven Press, London.

Willis, K.G. and Garrod, G. 1991. An individual travel cost method of evaluating forest recreation. Journal of Agricultural Economics 42(1), 33-42.

Willis, K.G. and Garrod, G. 1991. the hedonic price method and the valuation of countryside characteristics. Countryside Change Centre Working Paper 14, University of Newcastle-upon-Tyne.

Wilson, M.J. 1988 Vulnerable soils and their distribution. In Proceedings of a symposium 'Acidification in Scotland 1988', Scottish Development Department, Edinburgh.

Winpenny, J.T. Values for the environment: A guide to economic appraisal. HMSO, London.

Woodin, S. J., Studholme, C.J. and Lee, J.A. 1987. Effects of acid deposition on ombrotrophic peatlands. In: Acid Rain: Scientific and Technical Advances ed. R. Perry, J.N.B. Bell and J.N. Lester. Selper, London, 554-561.

Woodin, S.J. and Skiba, U. 1990. Liming fails the acid test. New Scientist 1707, 50-54.

Wright, R.F. and Henrikson, A. 1980. Regional survey of lakes and streams in southwestern Scotland. SNSF Project 72/80. SNSF, Oslo.

Wright, R.F., Cosby, B.J., Ferrier, R.C., Jenkins, A., Bulger, A.J. and Harriman, R. 1994. Changes in acidification of lochs in Galloway, southwestern Scotland, 1979-1988: The MAGIC model used to evaluate the role of afforestation, calculated critical loads and predicted fish status. Journal of Hydrology 124:24-37.

Appendix 1 Damage and Recovery Scenarios used in the CVM Survey

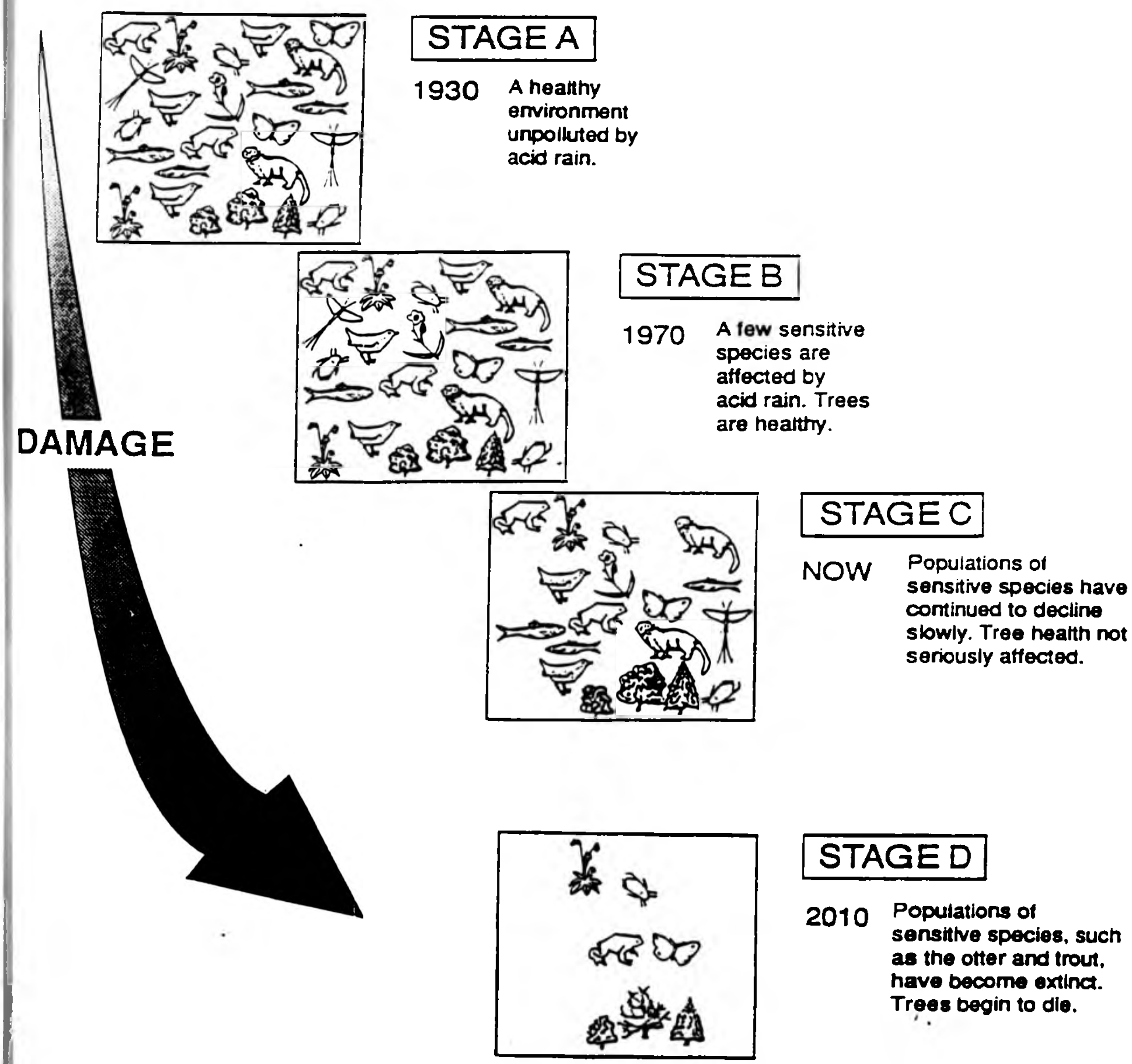
Page Numbering as Bound

filmed as seen.

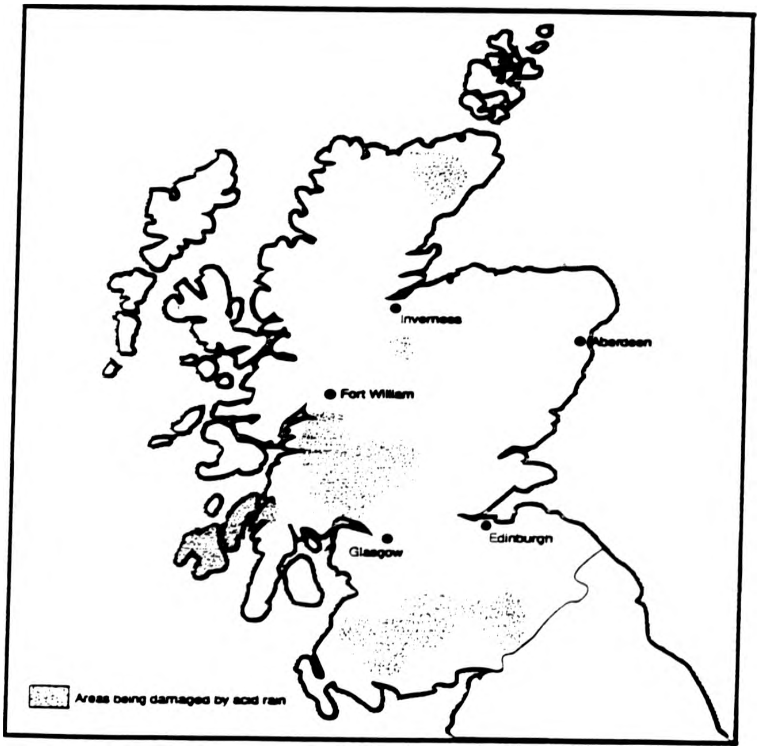
I would like to tell you a little about the effects of Acid Rain on the natural environment of Scotland.

The major cause of Acid Rain is sulphur and nitrogen gases produced by power stations, manufacturing industries and cars. These gases are picked up by water droplets in clouds and then falls as Acid Rain.

In some areas of the Scottish hills, scientists believe that the environment is being damaged. The chart below describes the damage which has occurred in these areas since 1930, and **the expected future damage, if nothing is done about Acid Rain.**



The areas where this Acid Rain damage is occurring are shown on the map below. (In other areas the environment and the species that live there are not affected).



Acid Rain damage in Scotland is now at STAGE C. By 2010 we will have reached STAGE D if nothing is done.

By taking action now to reduce Acid Rain, scientists expect a full recovery to a healthy environment (*STAGE A*). This will take *20 years*.

Reducing Acid Rain will lead to higher costs for your household through higher prices for cars, electricity and household goods.

10. If you could be guaranteed that industry was forced to act and recovery was assured, would you be willing to pay anything (even a small amount) towards reducing acid rain?

Yes ☐ 1   No ☐ 2

If No, please say why ....

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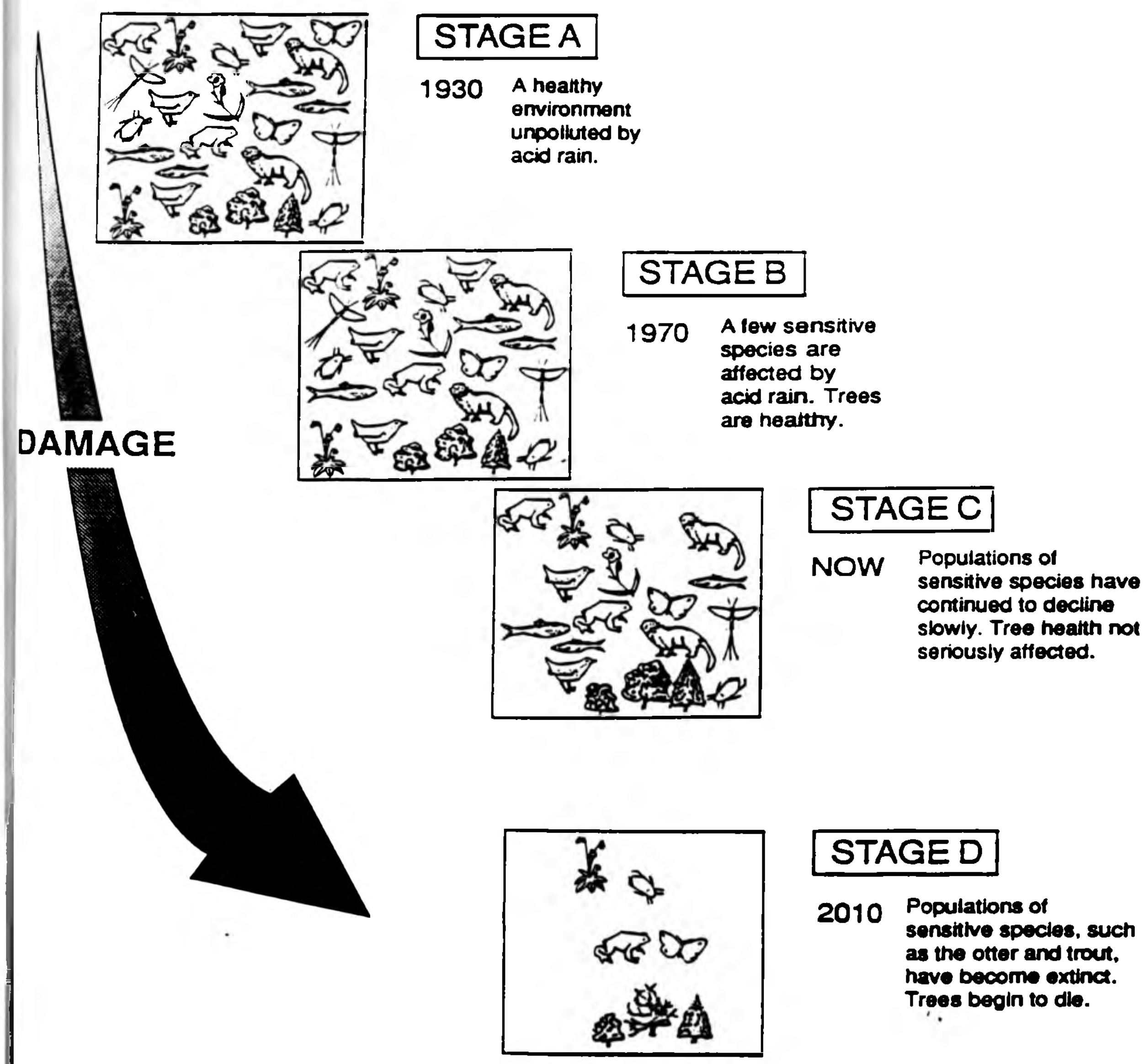

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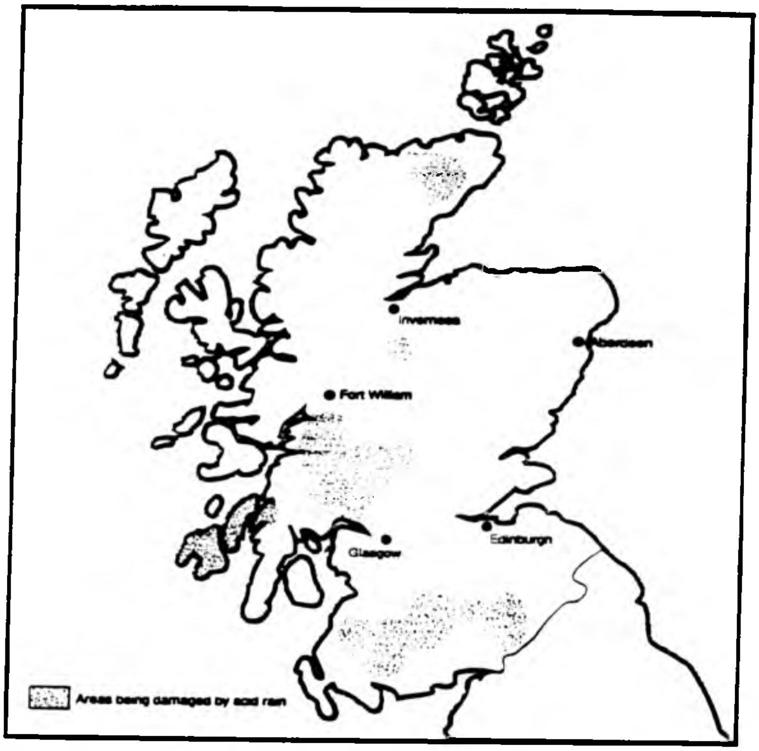

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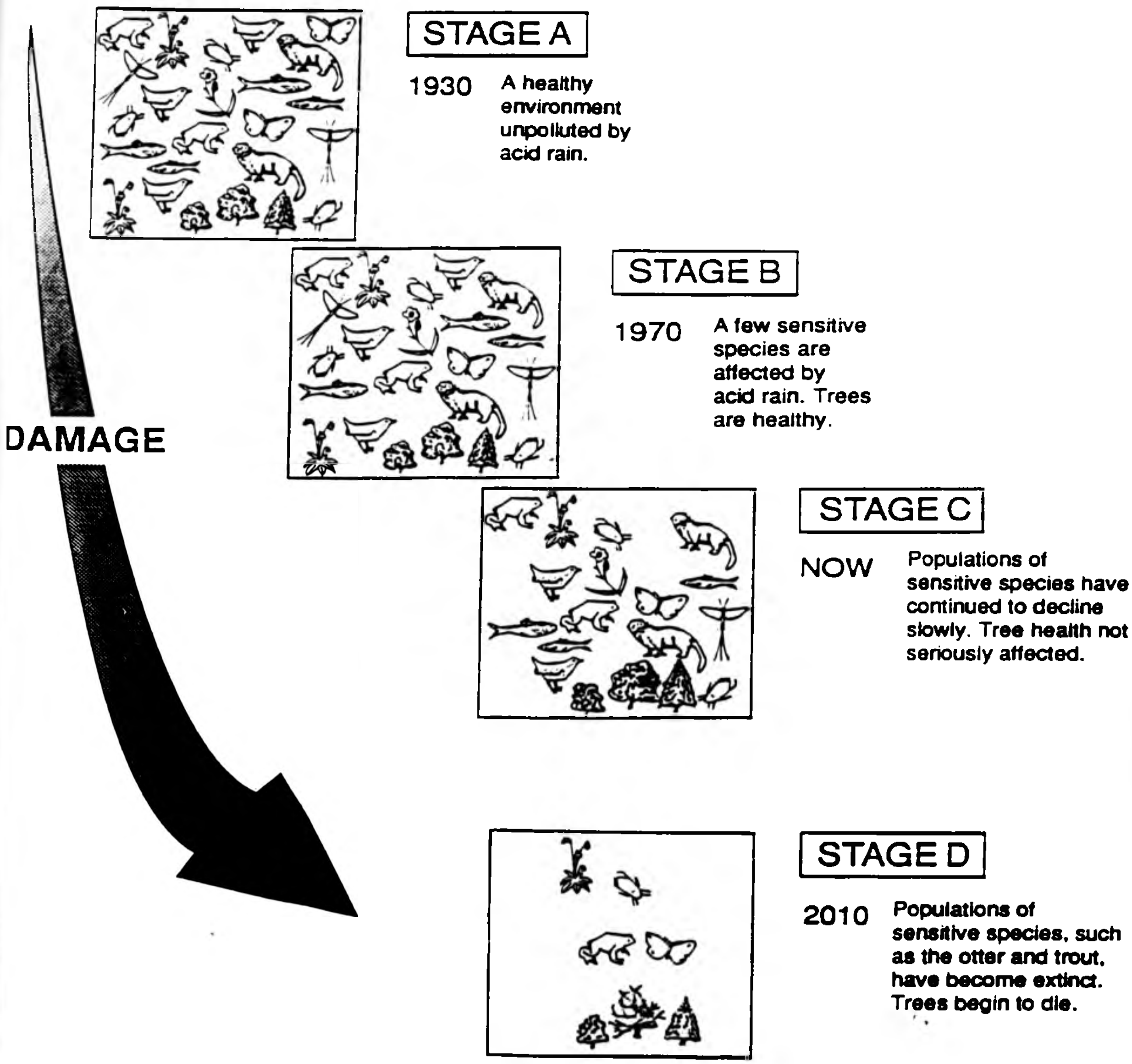

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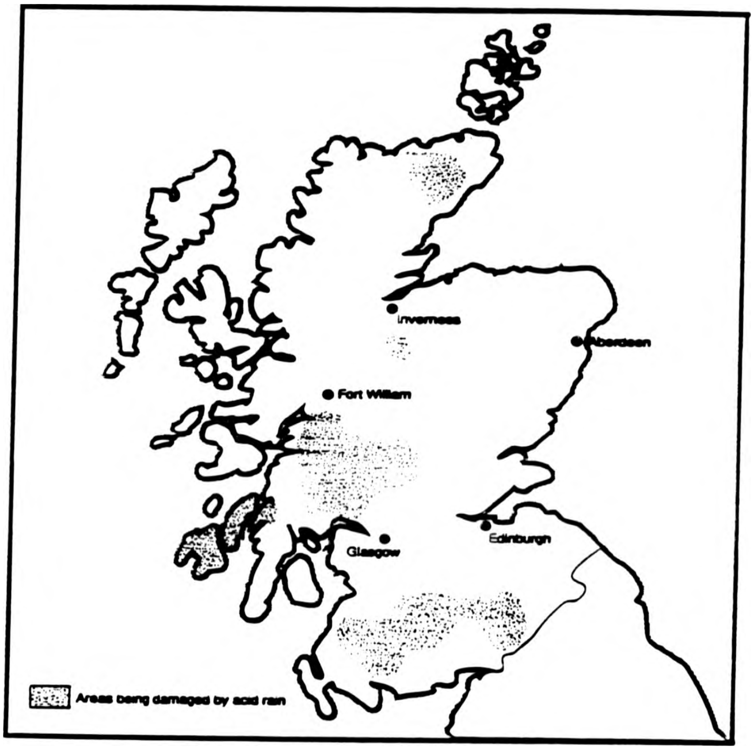

Acid Rain damage in Scotland is now at STAGE C. By 2010 we will have reached STAGE D if nothing is done.

By taking action now to reduce Acid Rain, scientists expect the environment to recover only to STAGE B. (A fuller recovery is not thought to be possible.) This will take *20 years*.

Reducing Acid Rain will lead to higher costs for your household through higher prices for cars, electricity and household goods.

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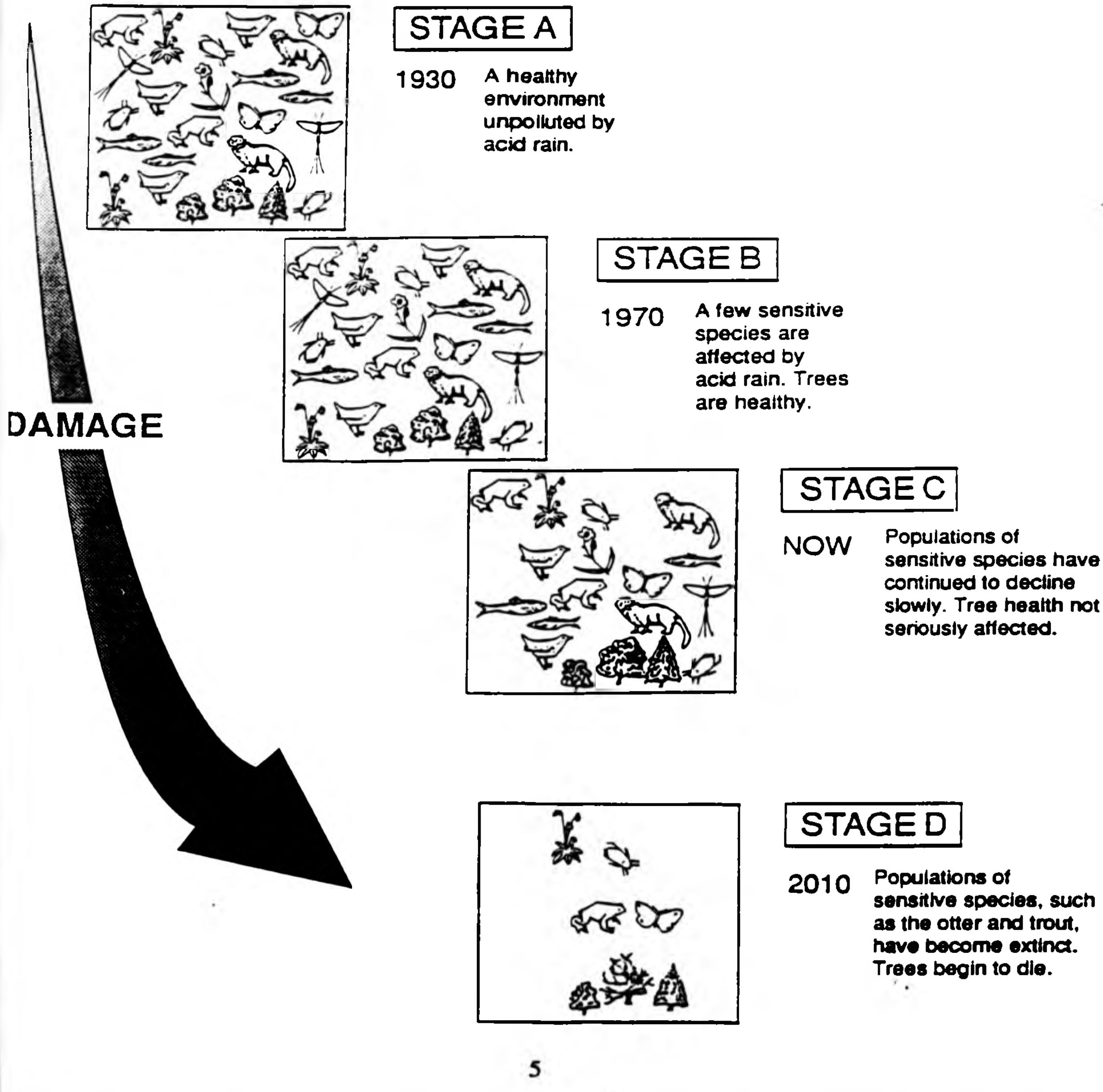

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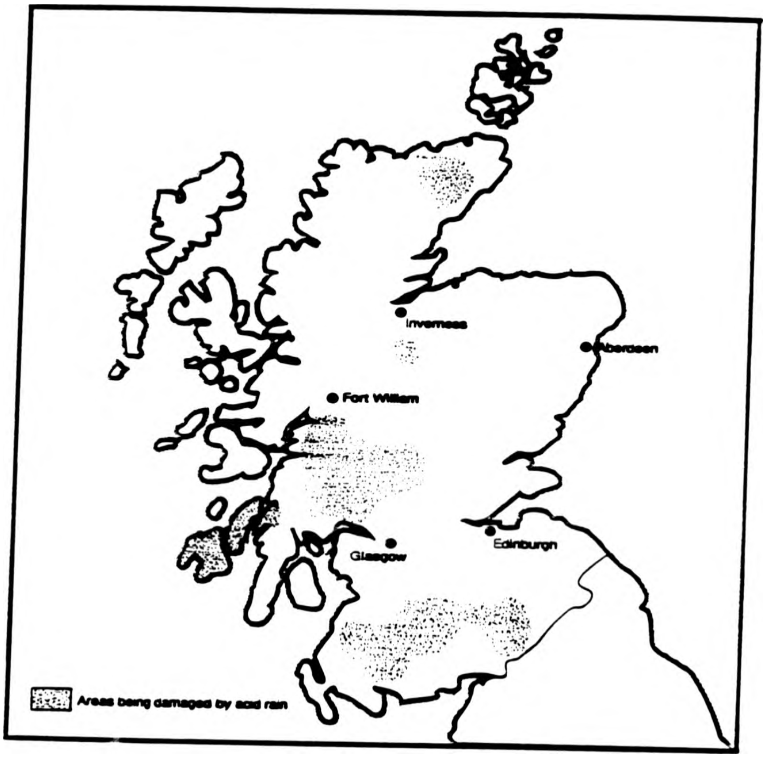

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Yes ☐ 1 No ☐ 2

If No, please say why ....

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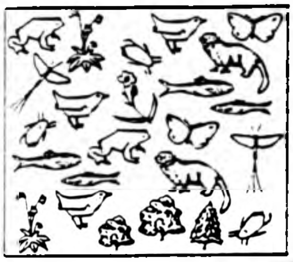
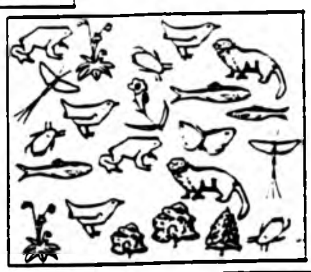


STAGE A

1930 — A healthy environment unpolluted by acid rain.



STAGE B

1970 — A few sensitive species are affected by acid rain. Trees are healthy.

MAGE

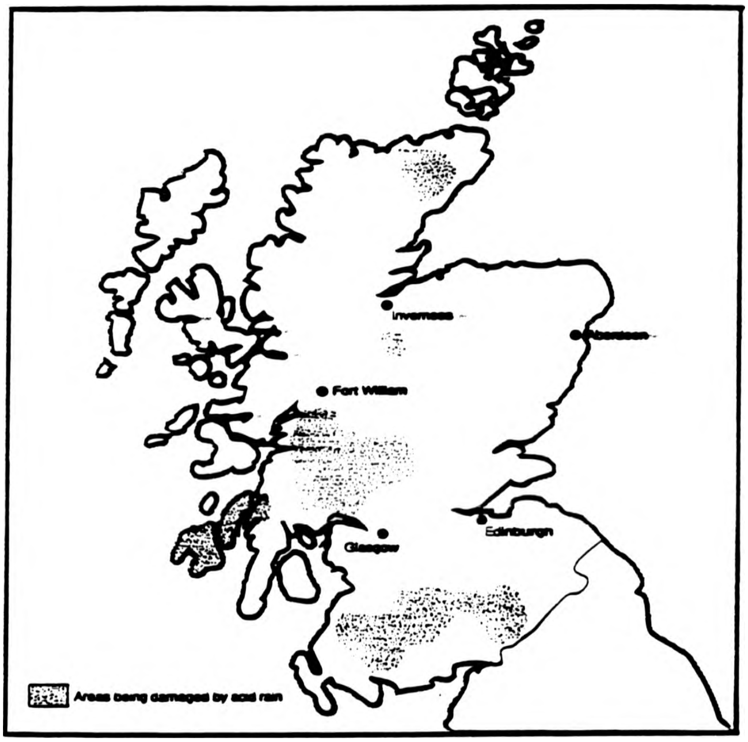
STAGE C

NOW — Populations of sensitive species have continued to decline slowly. Tree health not seriously affected.

STAGE D

2010 — Populations of sensitive species, such as the otter and trout, have become extinct. Trees begin to die.

The areas where this Acid Rain damage is occurring are shown on the map below. (In other areas the environment and the species that live there are not affected).



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Reducing Acid Rain will lead to higher costs for your household through higher prices for cars, electricity and household goods.

10. If you could be guaranteed that industry was forced to act and the environment could be kept at STAGE C, would you be willing to pay anything (even a small amount) towards reducing acid rain?

Yes ☐ 1 No ☐ 2

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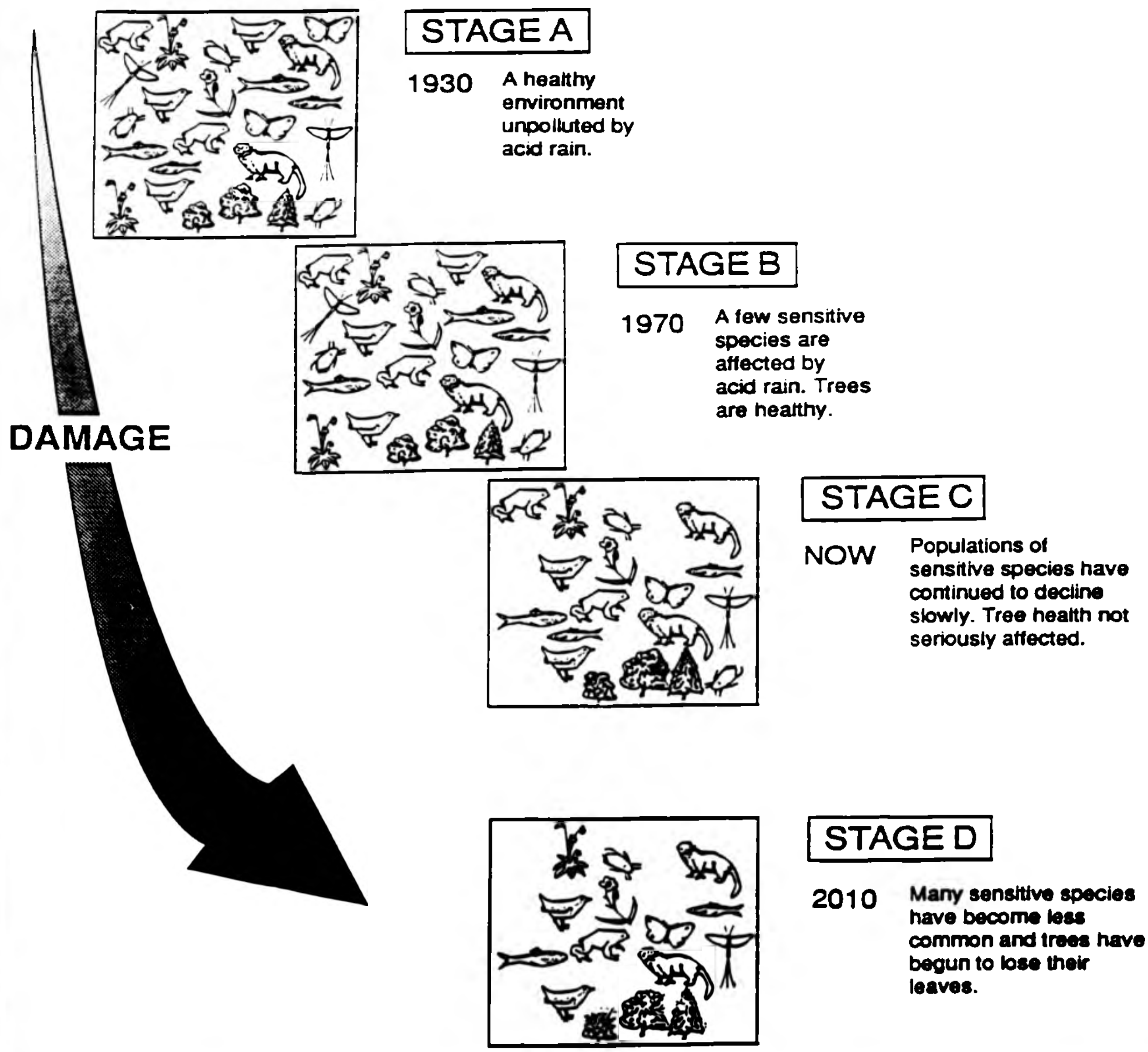

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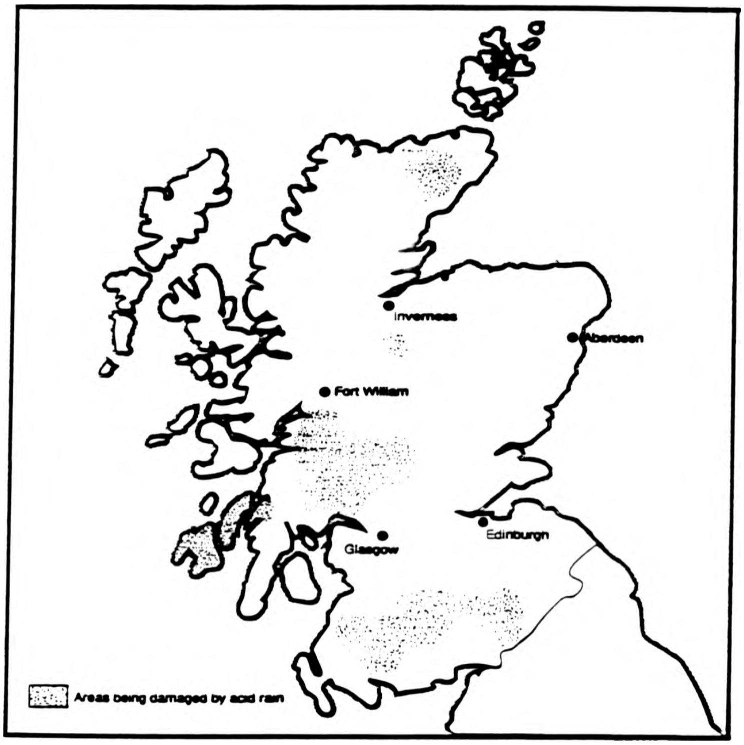

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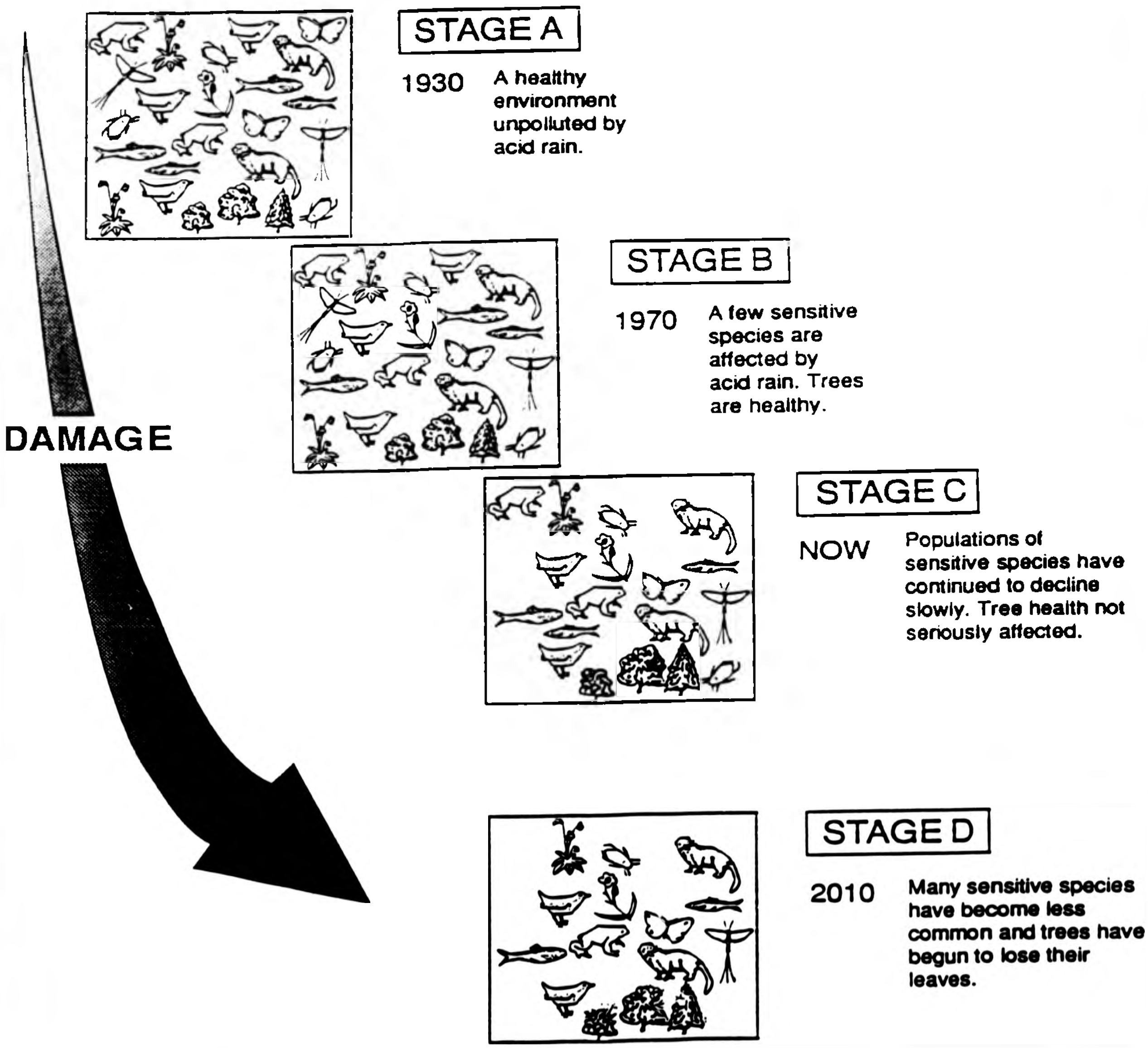

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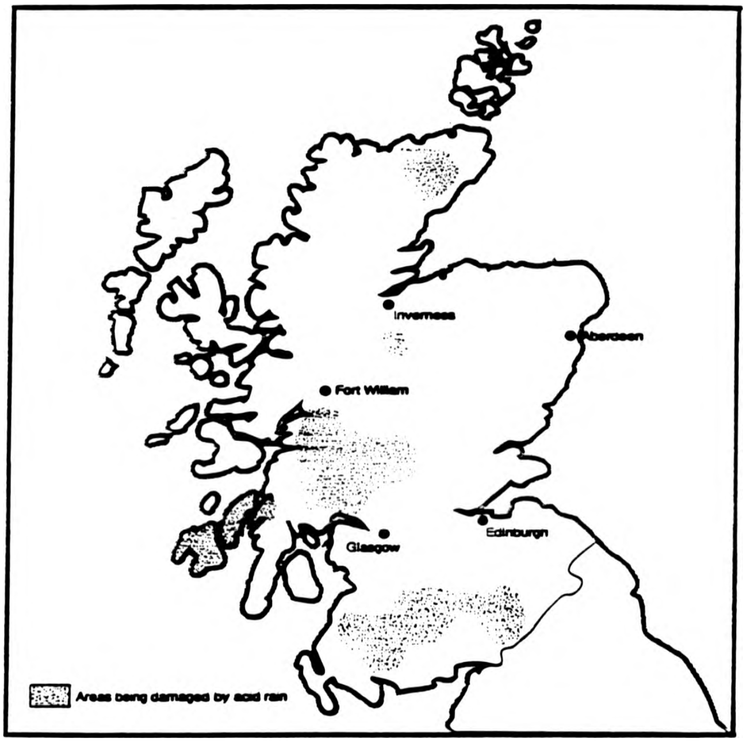

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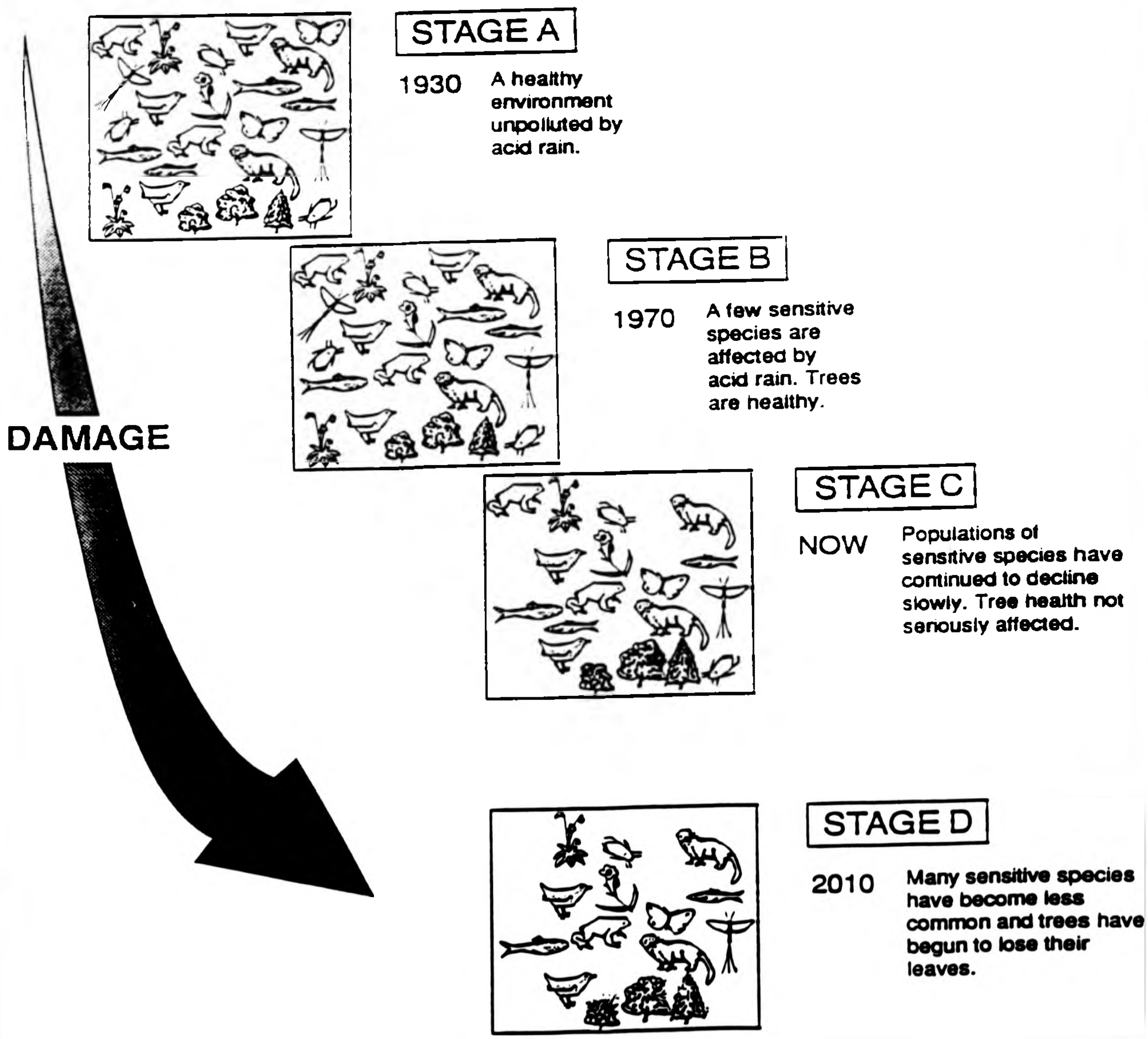

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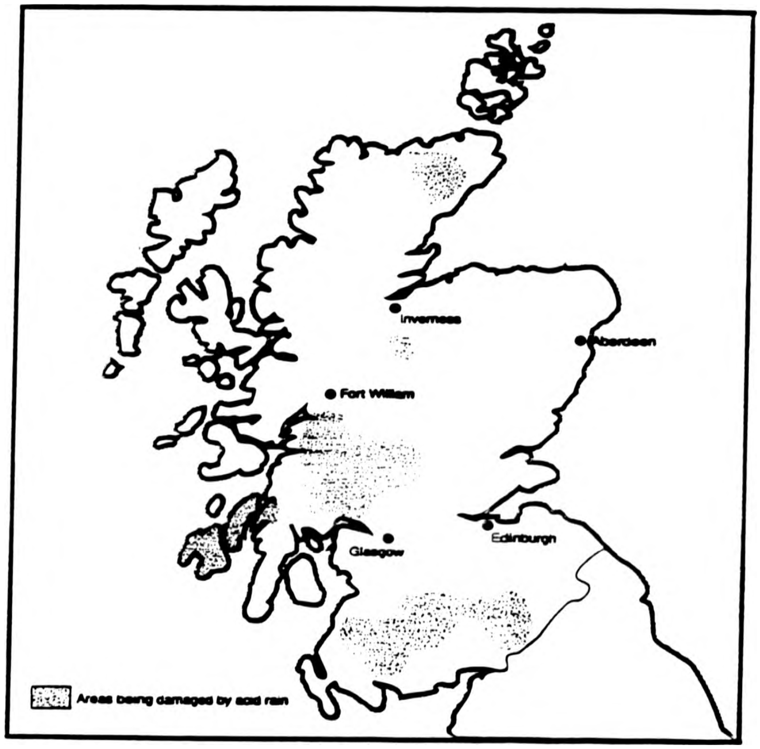

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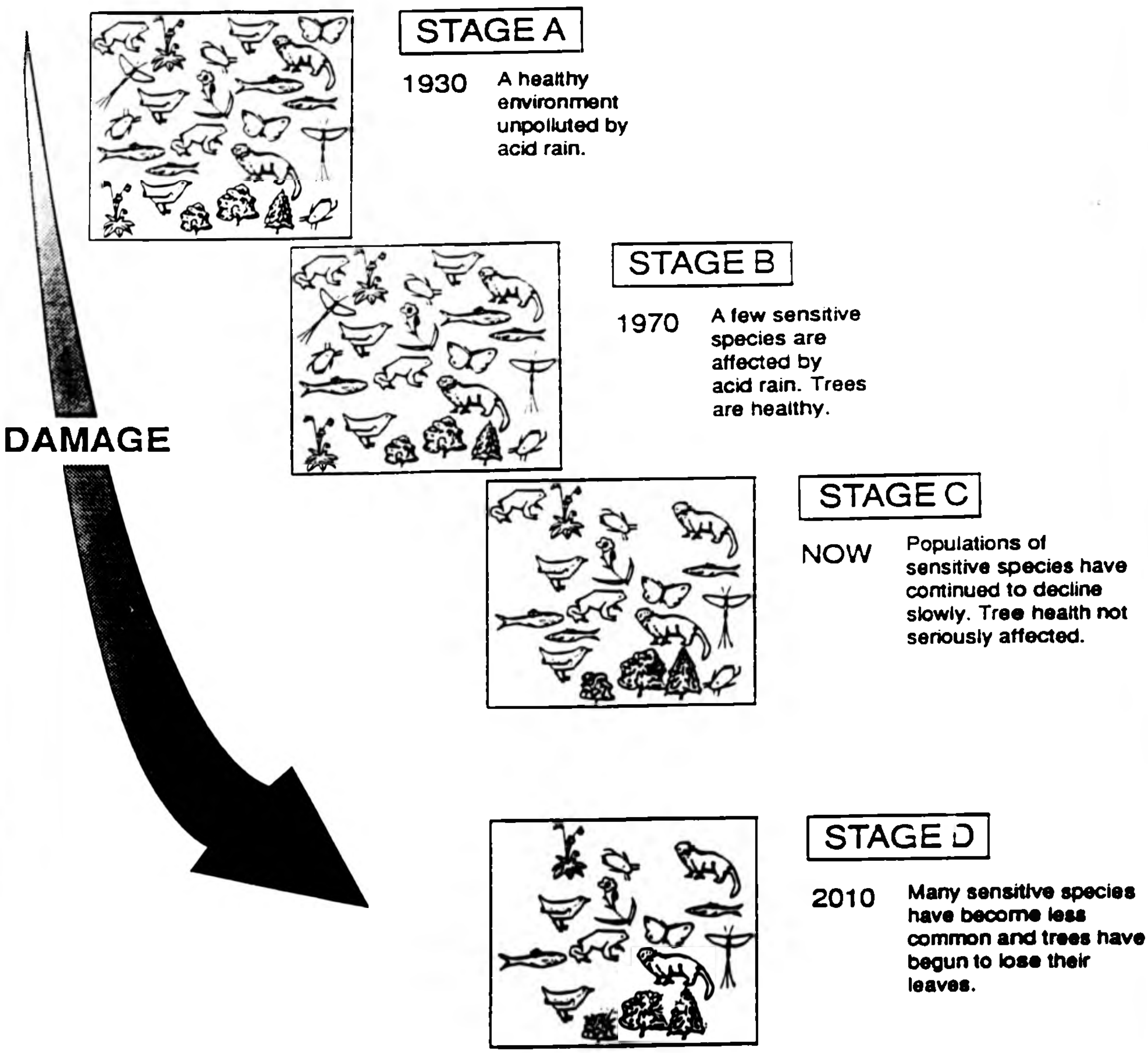

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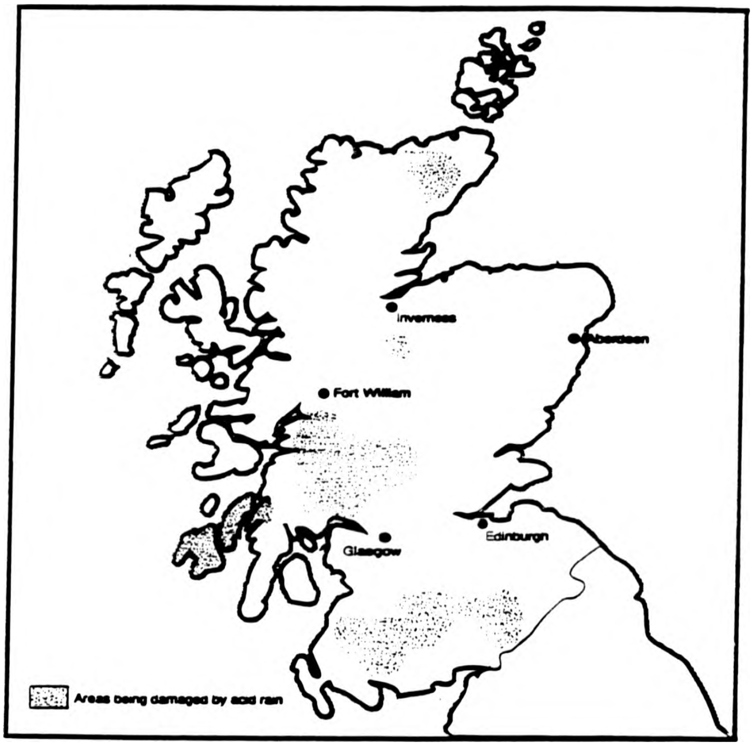

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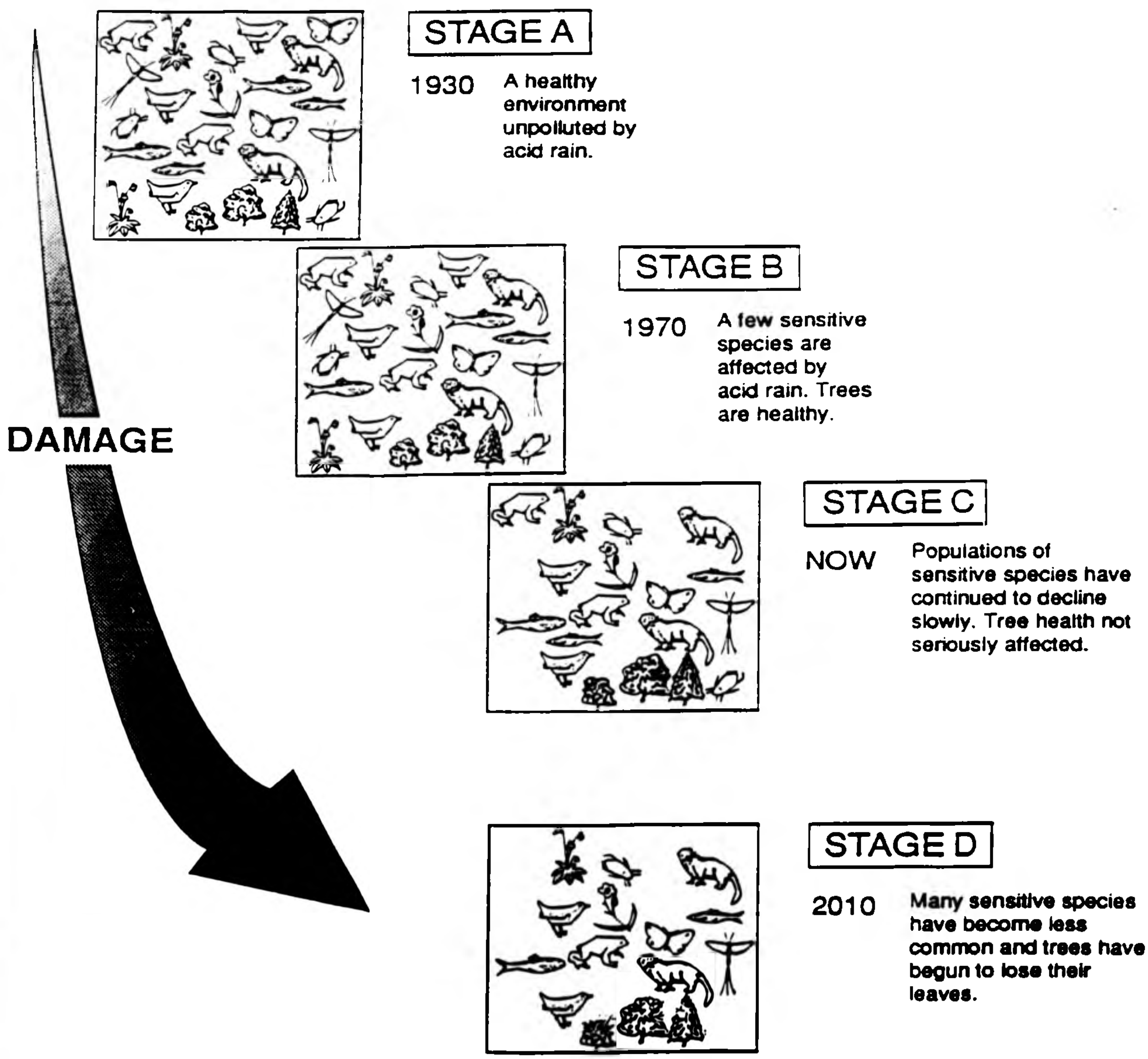

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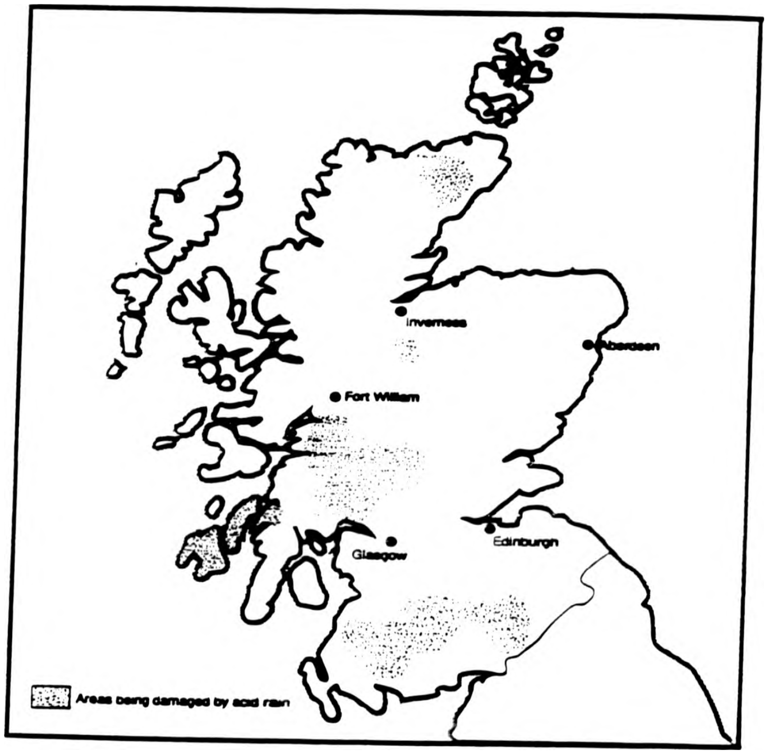

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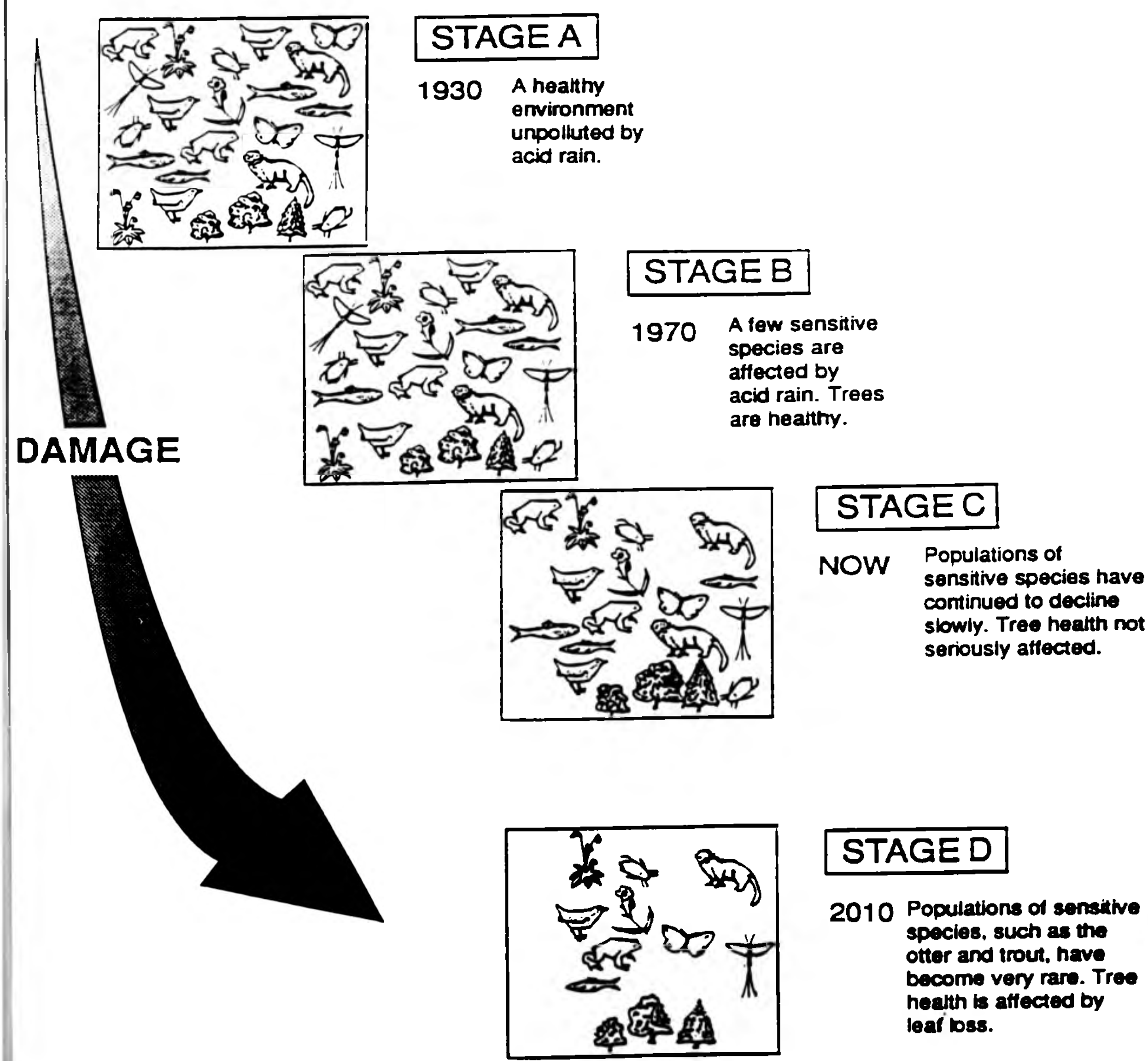

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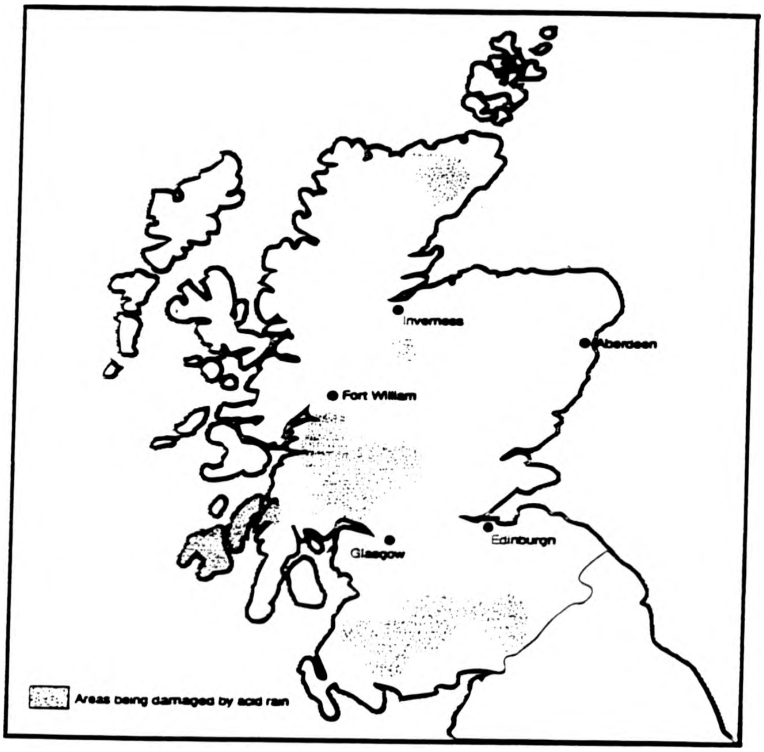

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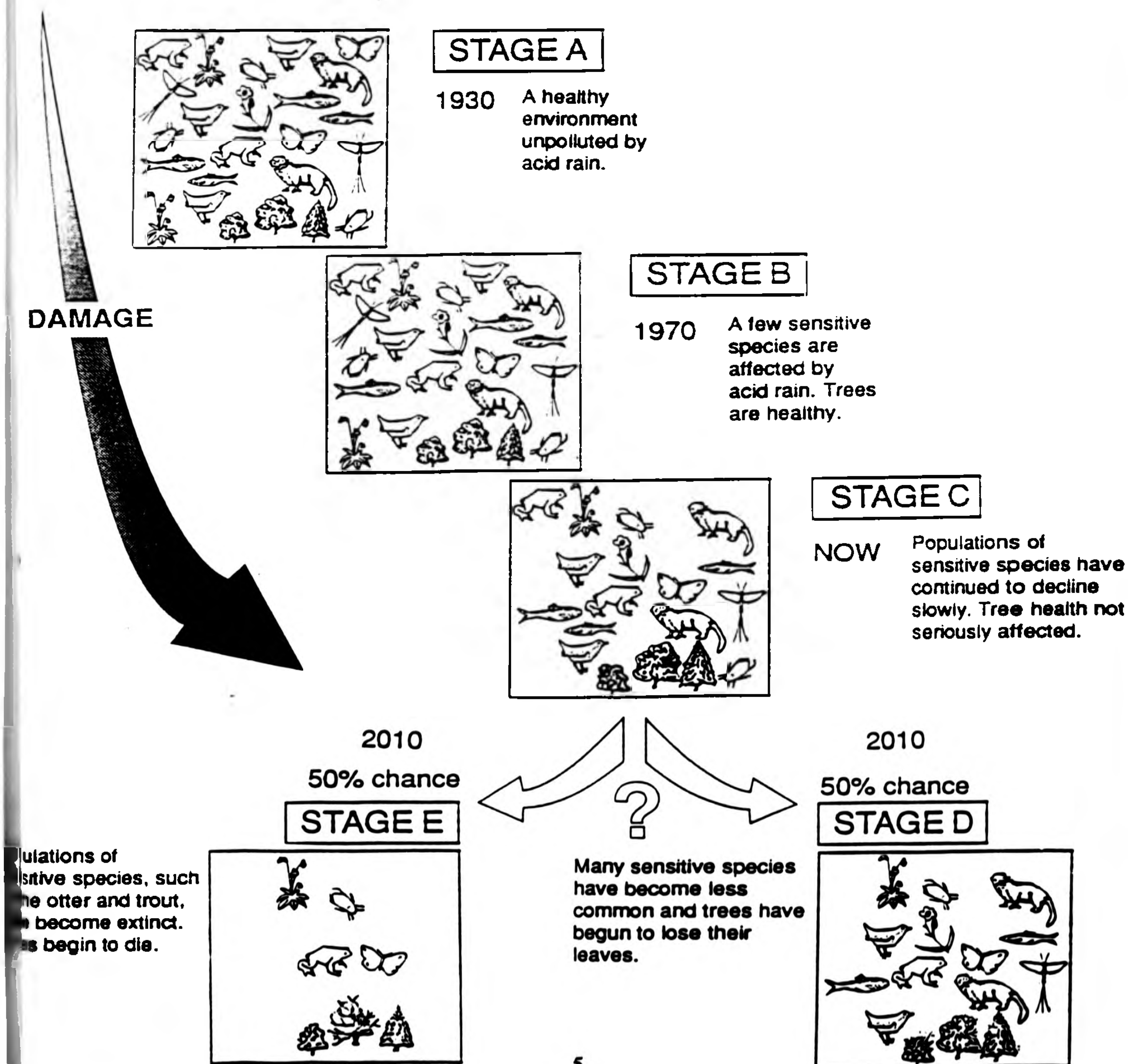
In some areas of the Scottish hills, scientists believe that the environment is being damaged. The chart below describes the damage which has occurred in these areas since 1930, and **the expected future damage, if nothing is done about Acid Rain**. As you can see there is some uncertainty over how bad future damage might be.



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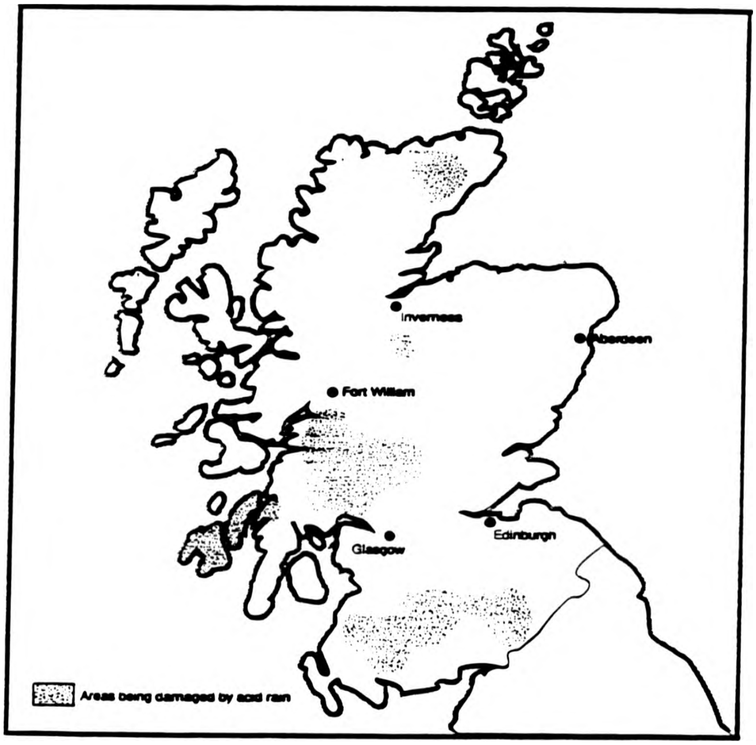


Acid Rain damage in Scotland is now at STAGE C. By 2010 we will have reached either STAGE D or E if nothing is done. We are uncertain about future damage. Some scientists predict STAGE D and others STAGE E.

However, by taking action now to reduce Acid Rain, scientists expect a full recovery to a healthy environment (*STAGE A*). This will take *20 years*.

Reducing Acid Rain will lead to higher costs for your household through higher prices for cars, electricity and household goods.

10. If you could be guaranteed that industry was forced to act and recovery was assured, would you be willing to pay anything (even a small amount) towards reducing acid rain?

Yes ☐ 1 No ☐ 2

If No, please say why ....

Appendix 2 Questionnaire and Covering Letter Used in the Survey

The Macaulay Land Use Research Institute
CRAIGIEBUCKLER, ABERDEEN AB9 2QJ
Telephone (0224) 318611 Fax (0224) 311556

16 February 1993

A Hughes
55 Hutchestown Court
Glasgow

Today, human impact on the natural environment of Scotland is greater than ever before. Land use practices, industrial activity and recreational developments all affect the quality of our natural environment. Protecting the environment costs money and the government has to make difficult decisions about how much to spend on the environment and on services such as health, education and national defence. However, nobody really knows what people think about the natural environment and the need to protect it.

Your household is one of a small number drawn randomly from the Scottish population. We would like to ask your opinion on some issues which affect the natural environment of Scotland. In order that our results will truly represent the views of the residents of Scotland it is important that you, or another adult member of your household, complete all of the questions. (The questionnaire should take approximately 15 minutes). Please return the questionnaire in the envelope provided (no stamp required). The survey is organised by the University of Stirling and the Macaulay Land Use Research Institute, Aberdeen.

You are assured of confidentiality. The questionnaire has an identification number on the front cover only so that we can check your name off our list when your questionnaire is returned. Your name will never be associated with any of the answers you give.

Your views are important and the aim of this survey is to help policy makers come to more informed decisions about projects which affect the natural environment.

If you have any questions about the survey please write or call me. Thank you very much for your help.

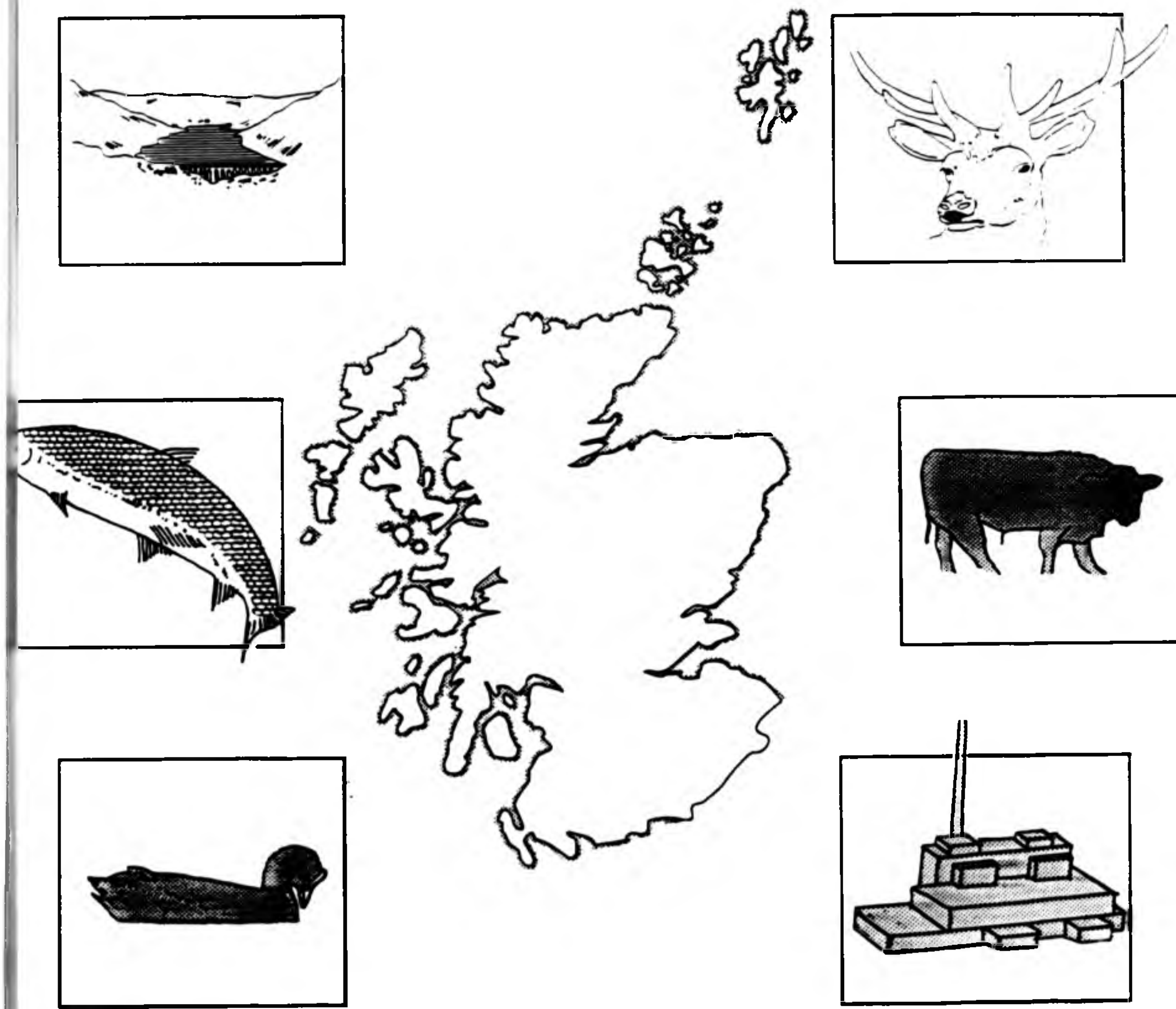
Yours sincerely

Douglas Macmillan

Douglas Macmillan
Project Director

Enc

The NATURAL ENVIRONMENT of SCOTLAND:
A national survey of Scottish residents' views and opinions about the environment



Please return this questionnaire to:

Douglas Macmillan
Environmental Economics Group
The Macaulay Land Use Research Institute
Craigiebuckler
Aberdeen, AB9 2QJ

In Scotland, most people live in cities and towns. We are fortunate, however, to have large areas of moorland, forest and mountains which offer many people the opportunity to participate in a range of outdoor activities.

1. Listed below are a number of different countryside activities and pastimes.

Please tick the relevant boxes to indicate how frequently you participated in each of the following countryside activities during the *last 12 months*.

	Main reason	Never	less than 3 days per year	3-12 days per year	more than 12 days per year
[a]	fishing	☐	☐	☐	☐
[b]	hill walking/rock climbing	☐	☐	☐	☐
[c]	scenic driving	☐	☐	☐	☐
[d]	skiing	☐	☐	☐	☐
[e]	bird-watching	☐	☐	☐	☐
[f]	gentle walks/picnics	☐	☐	☐	☐
[g]	cycling	☐	☐	☐	☐
[h-]	other (please specify)				
	....	☐	☐	☐	☐
		1	2	3	4

2. Did any of these visits involve staying away from home?

Yes ☐ 1 No ☐ 2

3. **If Yes, please tell us which type of accommodation you normally use *(tick no more than two)***

- Relatives/Friends ☐ 1
- Bed and Breakfast ☐ 2
- Hotel ☐ 3
- Tent ☐ 4
- Caravan ☐ 5
- Cottage/Self-catering ☐ 6
- Other (please specify)
  .... ☐ 7-9

4. Programmes about nature and the environment are often on TV and radio.
Please indicate which statement most accurately reflects your own opinion about these programmes *(tick one box only)*

- I make a special effort to watch/listen to them and I usually enjoy them ☐ 1
- I enjoy many of them but some hold my interest for a short while ☐ 2
- I'll watch/listen to them if there is nothing better on ☐ 3
- I never watch/listen to them ☐ 4

5. How frequently do you do the following:

	Never	Occasionally	Often	Does not apply
(a) Take bottles to a bottle bank	☐	☐	☐	☐
(b) Use pesticides in your garden	☐	☐	☐	☐
(c) Contribute paper for recycling	☐	☐	☐	☐
(d) Cut down the amount of car travel for environmental reasons	☐	☐	☐	☐
	1	2	3	4

8. Look at the environmental issues listed below.

Please score each one on a scale 1 to 4 according to how concerned you are about them (circle the appropriate number).

	Not Concerned	Slightly Concerned	Concerned	Very Concerned
[a] Global warming	1	2	3	4
[b] Ozone layer	1	2	3	4
[c] Loss of tropical forests	1	2	3	4
[d] Acid rain	1	2	3	4
[e] Sewage in the sea	1	2	3	4
[f] Conservation of the African rhino	1	2	3	4
[g-m] Other (please specify)	1	2	3	4

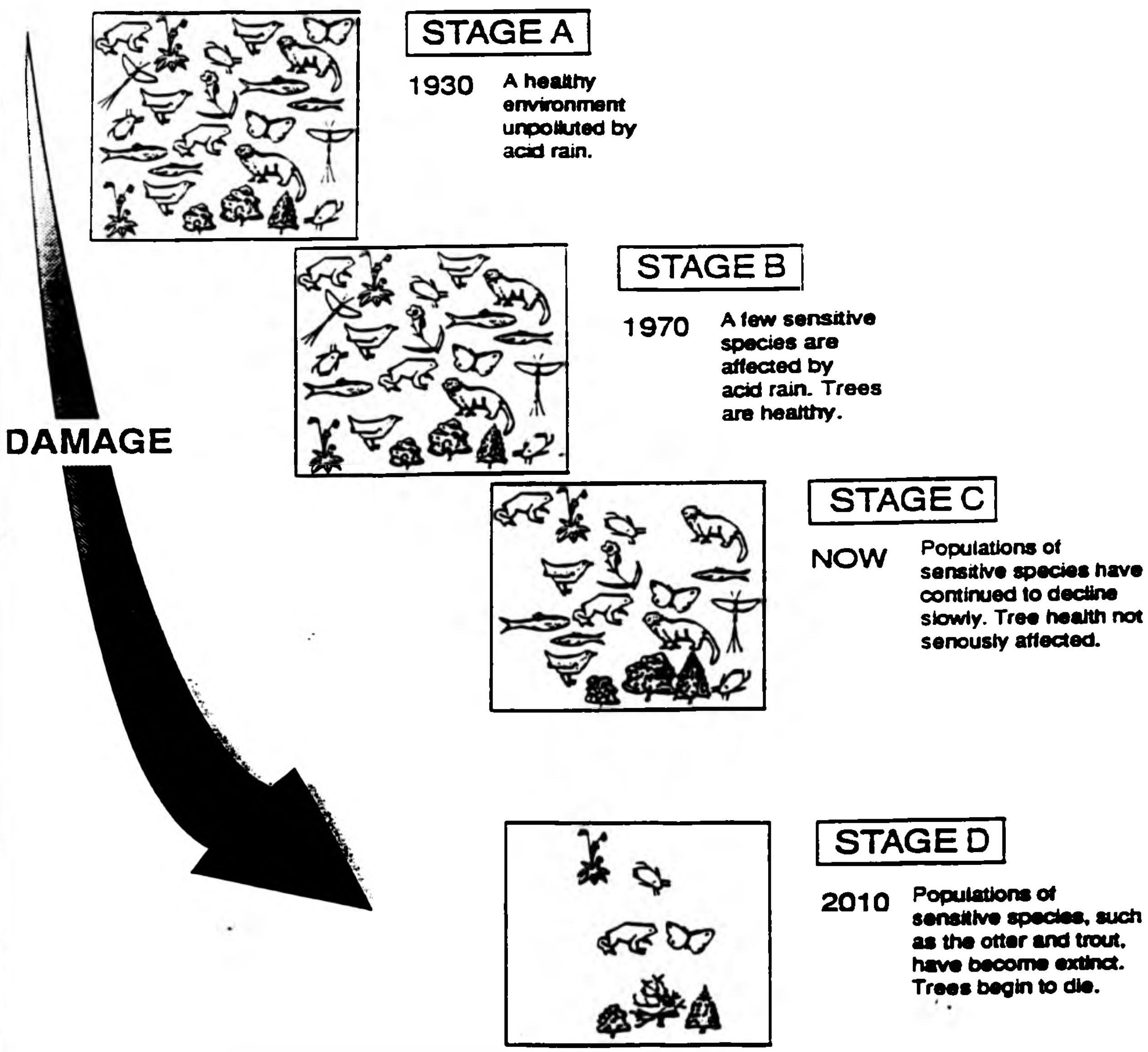
9. If you are a member of an environmental organisation/charity **please indicate which ones you belong to:**

- [a] Greenpeace ☐
- [b] Royal Society for the Protection of Birds ☐
- [c] The Scottish Wildlife Trust ☐
- [d] The National Trust for Scotland ☐
- [e] Friends of the Earth ☐
- [f-g] Other (please specify) .... ☐
- [h] None ☐

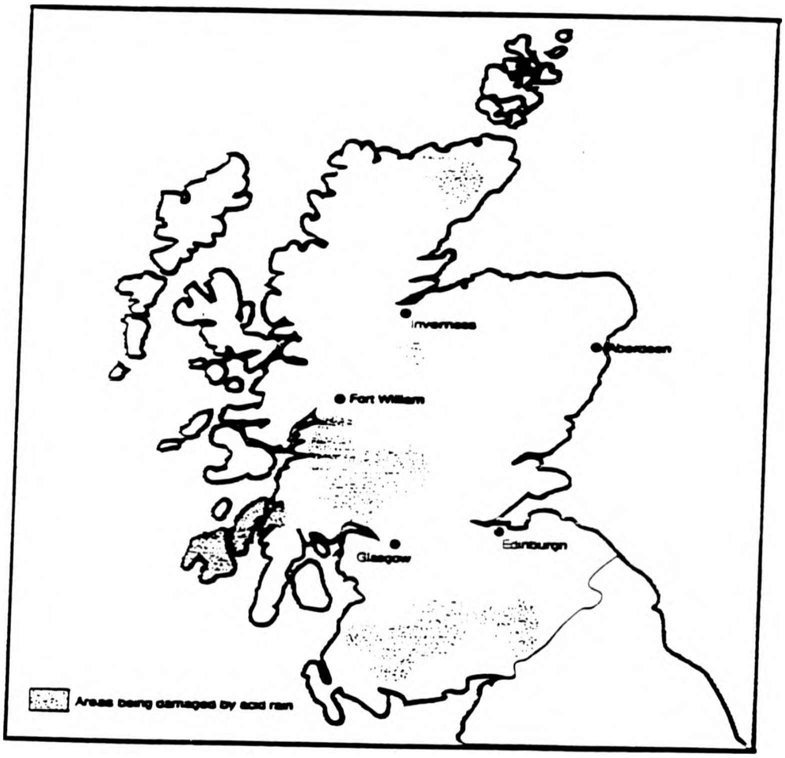
I would like to tell you a little about the effects of Acid Rain on the natural environment of Scotland.

The major cause of Acid Rain is sulphur and nitrogen gases produced by power stations, manufacturing industries and cars. These gases are picked up by water droplets in clouds and then falls as Acid Rain.

In some areas of the Scottish hills, scientists believe that the environment is being damaged. The chart below describes the damage which has occurred in these areas since 1930, and **the expected future damage, if nothing is done about Acid Rain.**



The areas where this Acid Rain damage is occurring are shown on the map below. (In other areas the environment and the species that live there are not affected).



Acid Rain damage in Scotland is now at STAGE C. By 2010 we will have reached STAGE D if nothing is done.

By taking action now to reduce Acid Rain, scientists expect a full recovery to a healthy environment (*STAGE A*). This will take *20 years*.

Reducing Acid Rain will lead to higher costs for your household through higher prices for cars, electricity and household goods.

10. If you could be guaranteed that industry was forced to act and recovery was assured, would you be willing to pay anything (even a small amount) towards reducing acid rain?

Yes ☐ 1 No ☐ 2

If No, please say why ....

11. Before reading this information were you aware that acid rain was damaging the environment?

Yes ☐ 1 No ☐ 2

12. How would you describe your response to this information on Acid Rain *(tick one box only)*.

- I understand it ☐ 1
- I understand but don't entirely agree ☐ 2
- I found it slightly confusing ☐ 3
- I really don't understand most of it ☐ 4

If the environment is to recover to STAGE A then a substantial reduction in Acid Rain pollution will have to be made.

An independant consultant's report to government has calculated that the annual cost to your household will be: £270

13. Would you be willing to pay this amount each year?

Before you answer, please consider:

- In order to make this payment every year, you may need to reduce the amount you spend on other things.
- Other environment problems which may be of concern to you will not be affected.
- If the total amount people are willing to pay is not enough it is almost certain the environment will reach STAGE D.

Yes ☐ 1 No ☐ 2 Don't know ☐ 3

If you answered *Yes* to Question 13 please answer questions 14 and 15. If you answered *No* please go to Question 16. If you answered *Don't know* go to Question 17.

14. If you answered *Yes* to Question 13:

What spending might you give up to make this payment?

(a)	Entertainment/recreation	☐	(f)	Alcohol/tobacco	☐
(b)	Holiday	☐	(g)	Savings	☐
(c)	Food	☐	(h)	Clothes	☐
(d)	Nothing: wouldn't notice it	☐	(i)	Donations to charity	☐
(e)	Don't know	☐			

15. Listed below are some reasons people might give for paying this amount.

Please indicate which one most closely reflects your own view by ticking the appropriate box *(tick one box only)*.

I care about animals and plants	☐	1
I would catch more fish	☐	2
Future generations should have an unpolluted environment	☐	3
All pollution is bad	☐	4
Own reason (please specify)	☐	5
....		
Don't know	☐	6

Please go to question 17

16. If you answered *No* to Question 13:

Listed below are some reasons *why* people might give for saying No.

Please indicate which one most closely reflects your own view by ticking the appropriate box. *(tick one box only)*.

I can't afford to pay this amount	☐	1
I don't think these cuts are worth this amount to me	☐	2
There are large areas of Scotland not affected by acid rain	☐	3
Own reason (please specify)	☐	4-8
....		
Don't know	☐	9

It would be helpful for our survey if you could give us a little background information about you and your household

17. Are you? Male ☐ 1 Female ☐ 2

18. Town of residence ....

(if you don't live in a town please give nearest one to your home)

19. What is your approximate age?

- 25 or less ☐ 1
- 26-35 ☐ 2
- 36-45 ☐ 3
- 46-55 ☐ 4
- 56-65 ☐ 5
- over 65 ☐ 6

20. What is your *household's* approximate annual *gross* (i.e. before tax) income.
(Please remember that all your responses will be treated in strictest confidence).

- less than £5000 ☐ 1
- £5001-£10000 ☐ 2
- £10001-£15000 ☐ 3
- £15001-£20000 ☐ 4
- £20001-£25000 ☐ 5
- £25001-£30000 ☐ 6
- £30001-£35000 ☐ 7
- £35001-£40000 ☐ 8
- over £40000 ☐ 9

21. Are you:

- working full-time ☐ 1
- working part-time ☐ 2
- a student ☐ 3
- currently unemployed ☐ 4
- house wife/husband ☐ 5
- don't work - disability/illness ☐ 6
- retired ☐ 7
- other ☐ 8

22. At what age did you leave Secondary School? ☐

23. Have you ever attended College or University? Yes ☐ 1 No ☐ 2

24. Do you have any children or grandchildren? Yes ☐ 1 No ☐ 2

If Yes, please indicate their age group(s)

	children		grandchildren	
0-5 years	☐	a	☐	e
6-10 years	☐	b	☐	f
11-15 years	☐	c	☐	g
16 and over	☐	d	☐	h

25. Do you read a newspaper regularly? Yes ☐ 1 No ☐ 2

If Yes, which one(s) ?

Daily			*Sunday*		
Record	☐	a	Sunday Post	☐	i
Express	☐	b	Scotland on Sunday	☐	j
The Herald	☐	c	Sunday Times	☐	k
Scotsman	☐	d	Sunday Express	☐	l
Sun	☐	e	Sunday Mail	☐	m
Other (please specify)			Other (please specify)		
....	☐	f-h	....	☐	n-p

THANK YOU VERY MUCH FOR ANSWERING THESE QUESTIONS

If you would like to add any comments please do so in the space below:

Appendix 3 Values of H generated by MAGIC for sample catchments under alternative abatement scenarios

Appendix 3a: Values of H generated by MAGIC for sample catchments under no abatement (0% reduction)

Catchment	area(km2)	Y1988	Y1993	Y1998	Y2003	Y2008	Y2013	Y2018	Y2023	Y2028	Y2033	Y2038
Arron	0.3	0.06	0.09	0.09	0.1	0.1	0.1	0.1	0.1	0.11	0.11	0.11
Enoch	2.4	0.05	0.07	0.08	0.09	0.09	0.09	0.09	0.09	0.09	0.09	0.09
Neldricken	4	0.14	0.16	0.17	0.18	0.19	0.19	0.19	0.19	0.19	0.2	0.2
Valley	7.1	0.08	0.1	0.11	0.11	0.12	0.12	0.12	0.13	0.13	0.13	0.13
Long of Glenhead	0.9	0.12	0.15	0.15	0.17	0.17	0.17	0.17	0.18	0.18	0.18	0.18
Round of Glenhead	0.8	0.1	0.12	0.13	0.14	0.15	0.15	0.16	0.16	0.16	0.16	0.16
Riecawr	12.7	0.28	0.27	0.26	0.25	0.24	0.25	0.26	0.25	0.22	0.18	0.14
Macaterick	9.1	0.15	0.15	0.14	0.14	0.14	0.14	0.14	0.14	0.13	0.12	0.1
Grannoch	12.6	0	0	0	0	0	0	0	0	0	0	0
Dornal	1.9	1	1	1	1	1	1	1	1	1	1	1
Maberry	14.7	0.35	0.33	0.31	0.29	0.26	0.27	0.27	0.27	0.25	0.21	0.17
Harrow	3.9	0.09	0.09	0.07	0.06	0.04	0.06	0.09	0.07	0.02	0	0
Dungeon	9.3	0	0	0	0	0	0	0	0	0	0	0
Minnoch	1.3	0.06	0.06	0.05	0.04	0.02	0.05	0.07	0.04	0	0	0
Finlas	10	0.58	0.58	0.58	0.58	0.57	0.57	0.56	0.56	0.55	0.55	0.55
Balloching	1.9	0.3	0.3	0.3	0.3	0.31	0.3	0.28	0.27	0.26	0.25	0.24
Goosie	0.8	0.59	0.59	0.59	0.59	0.59	0.58	0.58	0.57	0.57	0.57	0.57
Girvan Eye	0.4	0.37	0.39	0.39	0.41	0.41	0.41	0.42	0.42	0.42	0.42	0.43
Cornish	3	0.42	0.42	0.42	0.42	0.42	0.42	0.43	0.43	0.42	0.42	0.42
Skelloch	2	0.32	0.32	0.31	0.31	0.31	0.31	0.3	0.29	0.27	0.26	0.23
Brecbowie	0.5	0.46	0.46	0.46	0.46	0.45	0.45	0.44	0.44	0.44	0.44	0.43
Bradan	13.5	0.44	0.44	0.42	0.42	0.43	0.42	0.4	0.38	0.36	0.35	0.34
Round/Dungeon	0.9	0.17	0.18	0.18	0.18	0.18	0.18	0.18	0.18	0.17	0.17	0.17
Long/Dungeon	1.9	0.05	0.07	0.08	0.09	0.09	0.09	0.09	0.1	0.1	0.1	0.1
Narroch	0.7	0.06	0.08	0.08	0.08	0.09	0.09	0.1	0.1	0.1	0.1	0.1
Knocksting	0.8	1	1	1	1	1	1	1	1	1	1	1
Lochinvar	2.9	1	1	1	1	1	1	1	1	1	1	1
Howie	1.9	0.66	0.67	0.66	0.65	0.64	0.63	0.61	0.59	0.58	0.57	0.56
Brack	0.3	1	1	1	1	1	1	1	1	1	1	1
Barscobe	0.8	1	1	1	1	1	1	1	1	1	1	1
Moss Roddock	0.6	1	1	1	1	1	1	1	1	1	1	1
Loch of the Lowes	0.6	0.8	0.8	0.8	0.81	0.81	0.79	0.77	0.74	0.72	0.71	0.7
Gower	0.1	1	1	1	1	1	1	1	1	1	1	1
Drumlanford	0.6	1	1	1	1	1	1	1	1	1	1	1
Nahinie	0.7	1	1	1	1	1	1	1	1	1	1	1
Dow	1.3	0	0	0	0	0	0	0	0	0	0	0
	127.2											

Appendix 3b: Values of H generated by MAGIC for sample catchments under 60% abatement

Catchment	area(km2)	Y1988	Y1993	Y1998	Y2003	Y2008	Y2013	Y2018	Y2023	Y2028	Y2033	Y2038
Arron	0.3	0.06	0.13	0.18	0.22	0.24	0.25	0.26	0.27	0.28	0.28	0.29
Enoch	2.4	0.05	0.15	0.2	0.26	0.28	0.29	0.3	0.31	0.32	0.32	0.33
Neldricken	4	0.14	0.2	0.26	0.3	0.32	0.33	0.35	0.36	0.37	0.37	0.38
Valley	7.1	0.08	0.18	0.24	0.3	0.32	0.33	0.34	0.35	0.36	0.37	0.37
Long of Glenhead	0.9	0.12	0.19	0.24	0.29	0.31	0.32	0.33	0.34	0.35	0.36	0.36
Round of Glenhead	0.8	0.1	0.18	0.24	0.28	0.31	0.32	0.33	0.34	0.35	0.36	0.37
Riecawr	12.7	0.28	0.31	0.33	0.34	0.35	0.37	0.39	0.39	0.37	0.36	0.34
Macaterick	9.1	0.15	0.19	0.21	0.23	0.24	0.25	0.26	0.26	0.26	0.25	0.25
Grannoch	12.6	0	0.04	0.08	0.12	0.13	0.15	0.16	0.17	0.18	0.15	0.13
Dornal	1.9	1	1	1	1	1	1	1	1	1	1	1
Maberry	14.7	0.35	0.37	0.37	0.38	0.38	0.39	0.41	0.41	0.41	0.39	0.37
Harrow	3.9	0.09	0.15	0.2	0.23	0.25	0.28	0.3	0.29	0.27	0.25	0.23
Dungeon	9.3	0	0.08	0.14	0.19	0.21	0.22	0.23	0.24	0.24	0.25	0.25
Minnoch	1.3	0.06	0.13	0.17	0.21	0.22	0.25	0.27	0.26	0.24	0.22	0.2
Finlas	10	0.58	0.61	0.64	0.66	0.67	0.68	0.68	0.68	0.68	0.68	0.69
Balloching	1.9	0.3	0.33	0.36	0.39	0.4	0.4	0.4	0.39	0.38	0.38	0.38
Goosie	0.8	0.59	0.61	0.64	0.65	0.67	0.67	0.67	0.67	0.68	0.68	0.69
Girvan Eye	0.4	0.37	0.42	0.46	0.49	0.51	0.53	0.54	0.55	0.56	0.57	0.58
Cornish	3	0.42	0.44	0.47	0.49	0.51	0.52	0.53	0.53	0.54	0.55	0.56
Skelloch	2	0.32	0.34	0.36	0.37	0.38	0.38	0.38	0.38	0.37	0.36	0.34
Brecbowie	0.5	0.46	0.49	0.52	0.54	0.54	0.55	0.55	0.56	0.56	0.56	0.56
Bradan	13.5	0.44	0.47	0.49	0.51	0.52	0.52	0.51	0.5	0.5	0.49	0.49
Round/Dungeon	0.9	0.17	0.23	0.28	0.31	0.32	0.33	0.33	0.33	0.34	0.34	0.35
Long/Dungeon	1.9	0.05	0.14	0.2	0.24	0.27	0.28	0.3	0.3	0.31	0.32	0.33
Narroch	0.7	0.06	0.14	0.2	0.25	0.27	0.28	0.29	0.3	0.31	0.32	0.32
Knocksting	0.8	1	1	1	1	1	1	1	1	1	1	1
Lochinvar	2.9	1	1	1	1	1	1	1	1	1	1	1
Howie	1.9	0.66	0.77	0.85	0.92	0.95	0.97	0.98	1	1	1	1
Brack	0.3	1	1	1	1	1	1	1	1	1	1	1
Barscobe	0.8	1	1	1	1	1	1	1	1	1	1	1
Moss Roddock	0.6	1	1	1	1	1	1	1	1	1	1	1
Loch of the Lowes	0.6	0.8	0.85	0.9	0.95	0.98	0.98	0.97	0.97	0.97	0.97	0.98
Gower	0.1	1	1	1	1	1	1	1	1	1	1	1
Drumlanford	0.6	1	1	1	1	1	1	1	1	1	1	1
Nahinie	0.7	1	1	1	1	1	1	1	1	1	1	1
Dow	1.3	0	0.06	0.11	0.18	0.21	0.23	0.23	0.18	0.12	0.08	0.04
	127.2											

Appendix3c: Values of H generated by MAGIC for sample catchments under 90% abatement

Catchment	area(km2)	Y1988	Y1993	Y1998	Y2003	Y2008	Y2013	Y2018	Y2023	Y2028	Y2033	Y2038
Arron	0.3	0.06	0.12	0.16	0.2	0.24	0.28	0.3	0.31	0.32	0.33	0.33
Enoch	2.4	0.05	0.13	0.18	0.23	0.33	0.33	0.35	0.36	0.37	0.38	0.38
Neldricken	4	0.14	0.19	0.24	0.28	0.33	0.36	0.38	0.39	0.4	0.41	0.42
Valley	7.1	0.08	0.16	0.22	0.35	0.35	0.35	0.39	0.41	0.41	0.42	0.43
Long of Glenhead	0.9	0.12	0.18	0.23	0.27	0.32	0.35	0.36	0.38	0.39	0.4	0.41
Round of Glenhead	0.8	0.1	0.16	0.22	0.26	0.32	0.35	0.36	0.38	0.39	0.4	0.41
Riecawr	12.7	0.28	0.3	0.31	0.33	0.36	0.4	0.42	0.42	0.41	0.39	0.37
Macaterick	9.1	0.15	0.18	0.2	0.22	0.25	0.27	0.29	0.29	0.29	0.29	0.28
Grannoch	12.6	0	0.01	0.05	0.09	0.16	0.2	0.22	0.23	0.24	0.22	0.2
Dornal	1.9	1	1	1	1	1	1	1	1	1	1	1
Maberry	14.7	0.35	0.36	0.36	0.37	0.39	0.41	0.43	0.44	0.43	0.42	0.41
Harrow	3.9	0.09	0.14	0.17	0.21	0.27	0.31	0.35	0.34	0.33	0.31	0.29
Dungeon	9.3	0	0.06	0.11	0.16	0.23	0.27	0.28	0.29	0.3	0.31	0.31
Minnoch	1.3	0.06	0.11	0.15	0.18	0.24	0.29	0.32	0.31	0.29	0.27	0.26
Finlas	10	0.58	0.61	0.63	0.65	0.69	0.7	0.71	0.71	0.71	0.72	0.72
Balloching	1.9	0.3	0.32	0.35	0.37	0.41	0.42	0.42	0.42	0.42	0.42	0.42
Goosie	0.8	0.59	0.61	0.63	0.64	0.67	0.68	0.69	0.7	0.7	0.71	0.71
Girvan Eye	0.4	0.37	0.41	0.44	0.48	0.52	0.54	0.56	0.58	0.59	0.6	0.61
Cornish	3	0.42	0.44	0.46	0.48	0.51	0.53	0.54	0.56	0.57	0.57	0.58
Skelloch	2	0.32	0.33	0.34	0.36	0.38	0.39	0.4	0.39	0.39	0.38	0.37
Brecbowie	0.5	0.46	0.48	0.5	0.52	0.55	0.57	0.57	0.58	0.58	0.59	0.59
Bradan	13.5	0.44	0.45	0.47	0.49	0.53	0.54	0.54	0.53	0.52	0.52	0.52
Round/Dungeon	0.9	0.17	0.22	0.25	0.29	0.34	0.36	0.37	0.37	0.38	0.38	0.39
Long/Dungeon	1.9	0.05	0.12	0.17	0.22	0.29	0.32	0.34	0.35	0.36	0.37	0.38
Narroch	0.7	0.06	0.13	0.18	0.22	0.29	0.32	0.33	0.34	0.36	0.36	0.37
Knocksting	0.8	1	1	1	1	1	1	1	1	1	1	1
Lochinvar	2.9	1	1	1	1	1	1	1	1	1	1	1
Howie	1.9	0.66	0.74	0.81	0.88	0.99	1	1	1	1	1	1
Brack	0.3	1	1	1	1	1	1	1	1	1	1	1
Barscobe	0.8	1	1	1	1	1	1	1	1	1	1	1
Moss Roddock	0.6	1	1	1	1	1	1	1	1	1	1	1
Loch of the Lowes	0.6	0.8	0.84	0.88	0.92	0.99	1	1	1	1	1	1
Gower	0.1	1	1	1	1	1	1	1	1	1	1	1
Drumlanford	0.6	1	1	1	1	1	1	1	1	1	1	1
Nahinie	0.7	1	1	1	1	1	1	1	1	1	1	1
Dow	1.3	0	0.03	0.08	0.14	0.23	0.27	0.29	0.25	0.2	0.16	0.14
	127.2											