An Investigation of regulatory efficiency with reference to the EU Water Framework Directive: an application to Scottish Agriculture

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Declaration

I hereby declare that this thesis is my own composition, that the work described is my own except where assistance is explicitly acknowledged, and that it has not been submitted for any other degree or professional qualification.

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Abstract

The Water Framework Directive (WFD) has the stated objective of delivering good status (GS) for Europe's surface waters and groundwaters. But meeting GS is cost dependent, and in some water bodies pollution abatement costs may be high or judged as disproportionate. The definition and assessment of disproportionate costs is central for the justification of time-frame derogations and/or lowering the environmental objectives (standards) for compliance at a water body.

European official guidance is discretionary about the interpretation of disproportionate costs which consequently can be interpreted and applied differently across Member States. The aim of this research is to clarify the definition of disproportionality and to convey a consistent interpretation that is fully compliant with the economic requirements of the Directive, whilst also being mindful of the principles of pollution control and welfare economics theory. On this basis, standard-setting derogations should aim to reach socially optimal decisions and be judged with reference to a combination of explicit cost and benefit curves – an application of Cost-Benefits Analysis - and financial affordability tests. Arguably, these tools should be more influential in the development of derogation decisions across member states, including Scotland.

The WFD is expected to have extensive effects on Scottish agriculture, which is faced with the challenge of maintaining its competitiveness, while protecting water resources. Focusing the analysis on the socio-economic impacts of achieving water diffuse pollution targets for the sector, a series of independent tests for the assessment of disproportionate costs are proposed and evaluated. These are: i) development of abatement cost curves for agricultural Phosphorus (P) mitigation options for different farm systems; ii) a financial characterisation of farming in Scotland and impact on profits of achieving different P loads reductions at farm level are investigated in order to explore issues on "affordability" and "ability to pay" by the sector; and iii) an investigation of benefits assessment using discrete choice modelling

to explore public preferences for pollution control and measure non-market benefits of WFD water quality improvements in Scotland.

Results from these tests provide benchmarks for the definition of disproportionate costs and are relevant to other aspects of the economic analysis of water use in Scotland. This study helps to clarify the nature of agricultural water use and how it leads to social tradeoffs with other non agricultural users. Ultimately, this perspective adds to the debate of how and where water is best employed to maximize its value to society.

PREFACE AND AKNOWLEDGMENTS

This PhD thesis is the product of a journey of four years. An extraordinary journey which has been possible thanks to the Scottish Agricultural College (SAC), the School of GeoSciences of the University of Edinburgh, the Macaulay Institute in Aberdeen and the financial assistance of the Scottish Government Rural and Environment Research and Analysis Directorate (RERAD). I feel privileged to have been given the opportunity to undertake this project and to develop my skills and knowledge as a researcher in environmental economics.

It is difficult to summarise the fantastic experiences that I have had in the last four years but I consider it necessary to briefly describe the journey and to acknowledge those individuals who have kindly taught me and given their support and help at various stages of this study.

Overall I would like to express my gratitude to my supervisor Dominic Moran. He has been a mentor to me since I met him six years ago and instrumental in the development of this research. I thank him for his guidance, ideas, support and time.

I first began the journey in the Scottish Government where I worked for the first six months of the PhD as an assistant economist in the Environment Team of RERAD. This position gave me the opportunity to witness the realities of implementation of the Water Framework Directive in Scotland and to learn first hand about the real challenges of providing economic advice to water policy. As a consequence, I identified the assessment of disproportionality as the main topic for my research. I have fond memories of my time in Victoria Quay. In particular, I would like to thank to Andrew Moxey and Tom Harvie-Clark for the opportunity given.

The journey continued in the Land Economy and Environment Research Group of SAC, where I was based during the core part of this PhD during which time I obtained the case-study for my thesis - farming and diffuse sources of water pollution - and was given the opportunity to collaborate in other research projects. My thanks go to several members of staff and fellow students in the department and in the College, particularly to the library services and IS group for their computer assistance. Specifically, I would like to thank; Alan Renwick, the Head of the Department, for his treatment of PhD students and for providing us with the conditions and necessary support for undertaking our research; Cesar Revoredo-Giha, who was instrumental in the development of chapter 5 of this thesis, for sharing his vast knowledge of agricultural economics with me; Michael MacLeod for his advice and helpful comments throughout all these years; and to, Kostas Ververidis, Clare Hall, Sally Huband, Steven Boyne, Alistair McVittie, Donna Ewen, Davy McCracken, Andrew Barnes and Kate Heal (School of GeoSciences) for their help and support at various points during the last four years.

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I dedicate this work to my family;

To my wife Ailbhe, my parents Manuel and Pilar, my brothers Jose Maria and Ignacio, my sister Blanca and her family and my wife's family for their love and unconditional support.

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ABBREVIATIONS

AB:	Averting Behaviour Valuation Method
BFT:	Benefits Function Transfer
BMP:	Best Management Practices
BOD:	Biological Oxygen Demand
BT:	Benefits Transfer
BV:	Bequest Values
CA:	Conjoint Analysis Valuation Method
CAP:	EU Common Agricultural Policy
CBA:	Cost-Benefit Analysis
CE:	Choice Experiments Valuation Method
CEA:	Cost-Effectiveness Analysis
COI:	Cost-of-illness Valuation Method
CRP:	UK Collaborative Research Programme
CRP study:	Baker et al., 2007 (see references)
CVM:	Contingent Valuation Method
D/A:	Debt to asset ratio
DC:	Disproportionate Costs
DCCV:	Dichotomous Choice Contingent Valuation
DEFRA:	Department for Food and Rural Affairs
DO:	Dissolved Oxygen
DUV:	Direct Use Values
EA:	Environment Agency – England and Wales
EC:	European Commission
ESA:	Environmentally Sensitive Area
EU:	European Union
FAS:	Scottish Farm Accounts Survey
FIO:	Faecal Indicator Organism
FRS:	UK's Family Resources Survey
GCS:	Good Chemical Status
GEP:	Good Ecological Potential
GES:	Good Ecological Status
GS:	Good Status
GP:	Good Potential
HCA:	Human Capital Approach Valuation Method

HMWB:	Heavily Modified Water Body
HPLM:	Hedonic Pricing applied to Labour Market Valuation Method
HPPM:	Hedonic Pricing Applied to the Property Market Valuation Method
IIA:	Independence of Irrelevant Alternatives
IP:	Implicit Prices
IUV:	Indirect Use Values
LFA:	Less Favoured Areas
MA:	Meta-Analysis
MAC:	Marginal Abatement Costs
MC:	Marginal Costs
ML:	Mixed Logit Model
MLURI:	Macaulay Land-Use Research Institute
MNL:	Multinomial Logit Model
MSC:	Marginal Social Costs
N:	Nitrogen
NFI:	Net Farm Income
NVZ:	Nitrate Vulnerable Zones
OV:	Option Values
P:	Phosphorus
PCCV:	Payment Card Contingent Valuation
PEPFAAC:	Prevention of Environmental Pollution from Agricultural Activity Code
PoMs:	Programe of Measures
PPP:	Polluter Pays Principle
PV:	Present Value
RBD:	River Basin Districts
RBMP:	River Basin Management Plan
RC:	Replacement Cost Valuation Method
RERAD:	Scottish Government Rural and Environment Research and Analysis Directorate formerly known as SEERAD
RIA:	Regulatory Impact Assessment
ROA:	Return on assets
ROE:	Return on equity
SAC:	Scottish Agricultural College
SEERAD:	Scottish Executive Environment and Rural Affairs Department
SEPA:	Scottish Environment Protection Agency

SRP:	Soluble Reactive Phosphorus
TAC:	Total Abatement Costs
TCM:	Travel Cost Valuation Method
TIFF:	Total Income from Farming
UK-TAG:	United Kingdom Technical Advisory Group WFD
WEWS Act:	Water Environment and Water Services (2003) Scotland Act
WFD:	Water Framework Directive
WTA:	Willingness To Accept
WTP:	Willingness To Pay
XV:	Existence Values

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CHAPTER 1 INTRODUCTION

1.1. Policy background

In 2000 the European Commission introduced the Directive 2000/60/EC *Establishing a Framework for Community Action in the Field of Water Policy*, more commonly known as the *Water Framework Directive* (WFD). The WFD is possibly the most significant water legislation ever to emerge in Europe. It connects a number of existing fragmented legislation for different aspects of water conservation and protection, thus establishing a common framework for the management of the water environment within which Member States will work to achieve its objectives. The overall aim of the WFD is to protect water resources in the long term, by establishing and enforcing the framework for water management strategies that will ensure sustainable, efficient, and equitable management of European water resources. EU Member States have the responsibility to transpose the Directive into their own legislations and to implement it in a way to ensure that the objectives of the Directive are met. By rationalizing both the supply and demand for water, the Directive will increasingly circumscribe the way in which water is used by all economic sectors. In essence, it represents the end of an environmental free good for water users.

The key elements of the WFD could be summarised as follows (European Commission, 2000):

- Expand the scope of water protection to all waters (surface waters and groundwater)
- Achieve 'Good Status' of water resources by a certain deadline (good ecological and chemical status for surface waters, good chemical and quantitative status for groundwater)
- Integrated river basin management, managing water resources at the river basin scale
- Use of a 'combined approach' of emission limit values and quality standards, and phasing out of specific dangerous/hazardous substances
- Use of economic instruments, methods and tools for the development of sustainable water management policies
- Get citizens more closely involved through active involvement and participation of stakeholders and the public
- Streamline water related legislation

The Directive sets a stringent timetable for implementation which stipulates the main steps to be followed for achieving its objectives. One of the most important milestones is the establishment of a 'River Basin Management Plan' by 2009, which will provide detailed information on how the objectives set for the river basin will be reached by 2015. The River Basin Management Plan (RBMP) has to include a Programme of Measures (PoMs) to reach "Good Status" for each River Basin District. This may include actions such as: i) measures to manage pressures arising from specific activities such as agriculture, forestry, industry; ii) environmental permitting systems or abstraction and discharge control regimes; iii) measures of water demand management; iv) economic incentive measures such as taxes on fertilizers; v) river restoration strategies, etc. (Interviews et al., 2006).

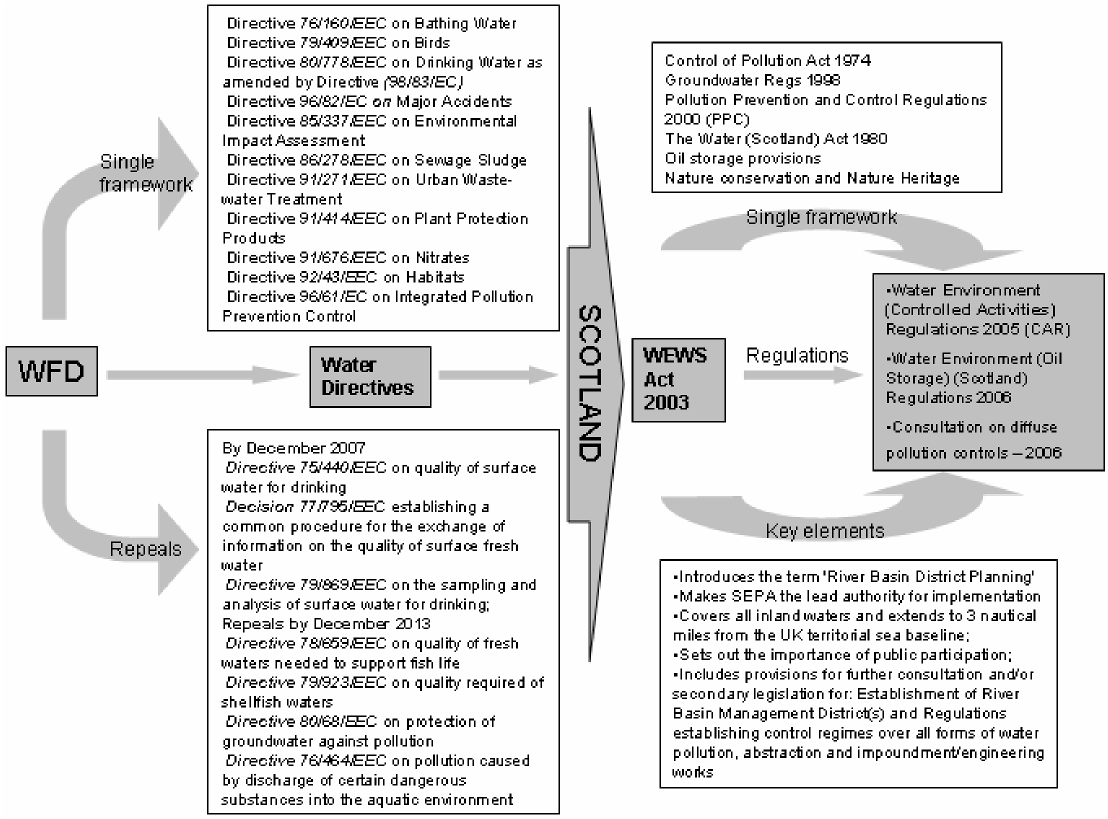
An important component of the WFD is the requirement to incorporate economic thinking into the formulation of water policy. In other words, the consideration of the costs and benefits associated with use. Such thinking is particularly relevant for: i) the selection of the most cost-effective PoMs for the River Basin Management Plans; ii) the use of specific economic instruments to support application of cost recovery, such as water pricing or taxes; iii) the considerations relating to the impact of water policies on the water using and polluting sectors; and iv) the assessment of disproportionate costs in order to justify exemptions to achieve Good Status at particular water bodies.

1.1.1. Recent water policy developments in Scotland

The WFD is a revolutionary piece of legislation, not only because of its ambitious objectives but also because it introduces the need to revise current ways of managing the water environment. One of its key features calls for the need to set a single legal framework to water policy in order to facilitate the achievement of its objectives and its implementation process. Figure 1.1 illustrates the main legislative and regulatory changes in water policy induced by the Directive in Scotland.

European Directives are legally binding and directly applicable by Member States. Directives have to be transposed integrally into national law within compulsory deadlines. If they are not properly transposed and applied, member states at fault are condemned and sanctioned by the European Court of Justice. In the past, cases of non-compliance with previous pieces of European water policy legislation have been frequently reported (Grimeaud, 2004).

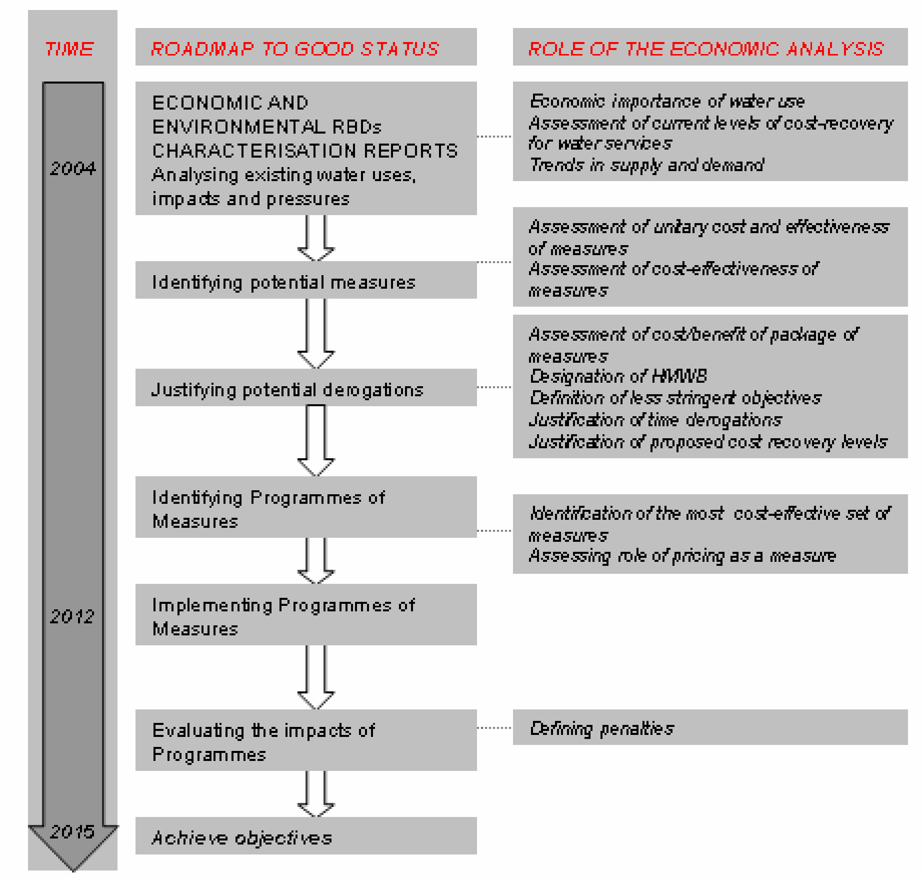
Figure 1.1 WFD induced changes to water policy in Scotland



Scotland presents a good illustrative example of the implementation process of the WFD at national level, which has been transposed by the Water Environment and Water Services (Scotland) (WEWS) Act 2003. This Act represents the first transposition of a European Community law instrument into Scots law by the Scottish Parliament via primary legislation. In this respect, the implementation of the Directive marked the beginning of the political devolution process in some administrative areas, environmental protection included, from Westminster to Scotland and the achievement of its objectives (i.e. Good Status) by 2015, can be interpreted (in due course) as one of the first tests of maturity for the recently established Scottish Government. In accordance, Scottish officials have not taken lightly the implementation of the WFD. Legal analysts have concluded that the WEWS Act is a well thought piece of legislation (Hendry, 2006; Clark, 2006), which, as the WFD mandates, has induced water use reform in Scotland by enabling all the necessary legal requirements to achieve the objectives of the Directive (the key elements of the WEWS Act are introduced in figure 1.1).

1.1.2. The economic analysis of water use

Figure 1.2 Integration of the economic analysis in the WFD timetable for implementation



The Directive places great importance on the economic analysis of water use at different geographical scales (per water body, catchment area, River Basin District or at least, at national level) in order to inform (aid) the decisions to be taken in the PoMs for the RBMPs to reach Good Status. This information would be of relevance for the assessment of the economic impact associated with PoMs and would provide basic information for the selection of the most appropriate set of measures (e.g. cost-effectiveness analysis), designation of protected areas, designation of Heavily Modified Water Bodies (HMWB) and finally, in the process of granting exemptions. Also, it would facilitate the analysis of the distributive impact of any actions taken (e.g. identification of winners and losers, help to resolve conflictive situation, etc) and more precisely, it would be essential in the designing of water pricing policies (implementation of which is mandatory by 2010). Figure 1.2

illustrates the timetable for implementation of the different elements of the WFD with a brief description of the required economic analysis.

1.1.3. Review of the standard setting process for the definition of "Good Status"

Subject to annex V of the Directive, Member States are required to define Good Status in terms of those environmental standards that will help to support the biology of the water environment. In Britain, The UK-TAG[1] is currently immersed in the process of defining Good Status (including the design of the environmental standards and the development of the classification schemes). As biological parameters are the key component of the definition of Good Status, the standards are being defined according to the relevant status class boundaries (high, good, moderate, poor and bad) which compare with different levels of biological quality elements (e.g. covering algae, fish plants, etc) for the different types of surface water bodies (e.g. rivers, lakes, etc). In consequence and following the Directive's definitions, the UK-TAG is designing (or updating in the case of existing legislation) the following environmental standards for the water quality of rivers in the UK (see table 1.1). This table also describes how different standards are being designed.

The designed standards will be for the whole of the UK (and fully compliant with the WFD requirements and other Directives). The approach to their implementation will be administration-specific, depending on different existing and proposed legislative and policy regimes, for each country within the UK (e.g. the ways in which abstraction is controlled in England & Wales, Northern Ireland and Scotland are different). Standards will be used to develop the classification schemes, as for example, each river in the UK will be assigned to one of five ecological status classes (high, good, moderate, poor and bad) or in the case of failing to meet them, to one of the five ecological potential classes (maximum, good, moderate, poor or bad). Additionally, there will be two surface water chemical status classes (good and not good). The "*one out-all out*" principle will decide their quality status; determined by the worst quality element, in the case of good ecological status, or the worst chemical element in reference to good chemical status. Furthermore, a surface water body will be classified also as "not good" if the standards for one or more priority substances

[1] The United Kingdom Technical Advisory Group (UKTAG). A group created to provide advice on the technical/scientific side of the implementation of the Directive, is a partnership of the UK environment and conservation agencies. http://www.wfduk.org/

(standards to be agreed at EU level) or dangerous substances (list Annex IX Directive) are exceeded.

Table 1.1 Environmental conditions, types and design of standards for rivers in the UK under the WFD

Environmental condition	Type of standard	Standards Design
I) General Water quality (Ecological status class boundaries: High, Good, Moderate, Poor and Bad)		
General physico-chemical quality elements	Biological Oxygen Demand (BOD) and dissolved oxygen demand (DOD), Ammonia	
	pH	Use of numeric values that have been referenced to ecology
	Nutrient: Phosphorus and other (not defined yet)	
	Temperature (not defined yet)	
	Salinity (not defined yet)	
Water flow and water levels	Change from natural flow conditions	Numeric values supported by hydrological modelling, based upon the best available understanding of links to ecology
Morphological quality elements	Type and degree of physical alteration (physical structure and condition of the bed, banks and shores)	Development of a decision framework based on best available knowledge supported by numeric thresholds
II) Chemical pollutants (Chemical status class boundaries: Good and Not Good)		
Toxic pollutants (called specific pollutants)	Standards for pollutants discharged in significant quantities	- Priority substances, Environmental Quality Standards (EQSs) design at European level - Dangerous substances: listed annex IX WFD

Source: (UK-TAG, 2006)

1.2. Problem statement - Assessment of disproportionality

As introduced in figure 1.1, new legislative and regulatory settings for implementation of the WFD in Scotland have already been designed (although to the authors knowledge the standards have not yet been set). The regulatory approach taken by the Scottish Environment Protection Agency (SEPA) will be a major factor influencing action for compliance and the total financial/administrative costs water users/polluters will have to bear as a result.

Obviously, delivering good status (GS) is cost dependent, and in some water bodies pollution abatement costs may be high or judged as disproportionate. The assessment of disproportionality, in the context of the WFD, makes reference to the justification of conceding exemptions for the achievement of environmental objectives; these include: granting time-frame derogations to achieve Good Status or allowing the lowering of environmental standards (from good status to good potential - HMWBs) when a water user finds the total costs of the most cost-effective PoMs too expensive or disproportionately expensive to undertake.

European official guidance is discretionary about the interpretation of disproportionate costs and simply states that its assessment has to be the outcome of a political decision informed by the economic analysis. EU Guidance only imposes the application of cost-effectiveness analysis for the selection of measures and merely outlines the application of economic decision-making tools (cost-benefit analysis, multi-criteria analysis, etc.) as valid approaches to inform the derogation decision-making process across Europe (European Commission, 2002a). Consequently, disproportionality is being interpreted and applied differently across Member States[2].

In Sweden, environmental regulators are currently undertaking the task of calculating costs and benefits for as many water bodies as possible, with cost-benefits ratios defining exemption cases. This approach seeks the achievement of economic efficiency and disproportionate costs are defined at the point where costs outweigh benefits.

In England and Wales, the Environment Agency (EA) also applies CBA theory in their approach to assess derogations under the WFD. Early guidance documents designed to develop a methodology for the assessment of exemptions recommended the application of CBA at different levels of detail and scale (i.e. sectoral versus individual water users) to be accompanied by a financial viability of the sector/water user to assess ability to pay for improvements (RPA, 2004). This guidance also recommends a full costing approach (economic versus financial costs accounting). Realities of implementation have led the EA into developing typical costs of remediation per sector and standard benefit estimates in order to transfer results across different parts of the country.

In Scotland, the definition of disproportionate costs is slightly different than in the previous two examples. SEPA addresses the issue by following a two steps method. The first step is an assessment of the technical feasibility and cost effectiveness of different options. The

[2] The following examples have been taken from the proceedings of the international workshop: "How can economics best support water policy decision making? Taking stock of the first years of WFD implementation" that the author attended in Ungersheim (France), May 2 -4, 2007

second step is a disproportionate cost screening test based on the sequential answer to four questions: (i) is it economically efficient or are benefits obviously not worth costs? (ii) Does it accord with the polluter pays principle? (iii) is it affordable? (iv) has some recent investment in environmental improvements been made? To address costs and benefits, SEPA uses a qualitative impact assessment method, accounting for social, economic and environmental factors. Positive or negative rankings are applied depending on the magnitude of the impact of the measure and the geographical level of the impact. Answers to these questions are based on expert judgment.

Differing definitions of disproportionate costs are a consequence of the lack of official EU guidance on this topic. Whilst the achievement of "Good Status" is a harmonised objective across Europe, Member States are left to their own devises in the search for tools to assess exemptions and forced to wonder about issues such as the scale of the analysis, the selection of indicators or the design of threshold values. In this respect only a succinct reply from the commision is offered: "*disproportionality should not begin at the point where costs simply exceed quantified benefits*" (European Commission, 2002a). This statement is intended to account for the uncertainties that surround benefits assessment (e.g. difficulties in accounting for all possible types of values derived from water use, which often results in an underestimation of the total economic value).

SEPA's pragmatic approach relies on expert opinion and avoids the undertaking of benefits assessment in their practical interpretation of disproportionate costs. This approach, which is in essence legitimate considering the succinct guidelines imposed by the EC, is not consistent with conventional economic theory and ignores any public preferences for the restoration of the water environment in Scotland. This approach has its benefits for the agency as it allows for resource use minimisation by basing any decisions on derogation on the judgement of its own officials. However, this system infers a high degree of subjectivism in the decision-making process which could favour some water users above others. In addition, decisions could be difficult to justify if a derogation claim dispute emerges for the resolution of conflictive cases with individual stakeholders.

The problem is that ultimately, the assessment of exemptions may prove to be one of the most controversial steps in the implementation process of the WFD. Decisions may reveal issues of competitiveness between water users or uneven distribution of the financial costs associated with the Directive (Pearce, 2004). Consequently, the choice of instruments and methodologies should be at the centre of the disproportionality debate. This would offer a much needed model of rationality to inform decision making processes across Europe.

1.2.1. Agriculture as a problem sector

Agriculture is the most significant and controversial water user in most EU countries, as it is associated with both water quality environmental concerns and problems of poor water use management. Across the EU, agriculture is seen as the sector that creates the biggest challenges to meeting the requirements of the WFD (Herbke et al., 2005). These challenges relate to the reduction of diffuse pollution from agricultural sources and to the regulation of agricultural water consumption. In the case of diffuse pollution, excessive or inappropriate use of fertilisers and pesticides contribute to water pollution through leaching and run-off, which can lead to eutrophication symptoms in rivers and lakes, high nitrogen fluxes to coastal waters, and increased nitrate concentrations in groundwater.

Furthermore, these problems create significant competition between farming and other water users. Contamination of raw waters by agricultural diffuse pollution forces urban water suppliers to increase investment and costs per unit of pollutant removed for drinking water provision. The protection of wetland and aquatic biodiversity or recreational uses are hampered by physical damage to ecosystems.

In Scotland, it is estimated that diffuse pollution currently results in up to 23% of the water bodies being at risk of not achieving Good Status, and it is now a more significant source of pollution than point sources in most water bodies (SEPA, 2005a,b). Furthermore, in these reports, SEPA has named agriculture as a significant cause of diffuse water pollution, and more specifically, as the dominant diffuse pollution pressure affecting rivers.

1.2.2. The polluter pays principle (PPP) and agriculture

The main rationale behind the introduction of the PPP in European environmental policy is the need to internalise negative externalities. Externality is purely an economic term that refers to those goods, products or services that are not typically reflected in market prices, and are therefore not assigned their real value or sometimes no value at all. Under certain assumptions, if a good is on the market, it will be correctly valued and therefore, correctly allocated. In the context of environmental protection, the PPP aims to internalise the costs of environmental damage derived from the polluting activity by demanding compensation for negative impacts on the environment. Polluters will accordingly reconsider their damaging uses.

The negative environmental and human health externalities of modern agriculture are well documented (Pretty et al., 2001; Herbke et al., 2005) and include:

i. The effects of pesticides contaminating water and harming wildlife and human health
ii. Nitrates and phosphate from fertilizers, livestock wastes, and silage effluents contaminating water and contributing to eutrophication. The impacts of eutrophication are as a result of changes in the nutrient balance in water courses, allowing for the rapid expansion of algal blooms, which induce deoxygenation, fish deaths and nuisance to other water users
iii. Farming practices may alter the soil composition; disrupting water courses, and making possible the run-off from eroded land, causing flooding and, as a side effect, damage to housing and natural resources.
iv. Release of microbiological pathogens and organic pollutants into waters (from animal manure, residues of veterinary preparations, etc.), as they could pose a serious threat and represent a long-term risk to human health
v. Harm to consumers exposed to damaging contaminants and bacterial organisms in foods; and,
vi. Contamination of the atmospheric environment by emissions of greenhouse gases; such as; methane, nitrous oxide and ammonia derived from livestock, their manures and fertilisers.

Several studies have attempted to estimate the economic costs of agricultural negative externalities for the US and different European countries, including the UK (see Pretty et al., 2001 for a detailed literature review on the estimation of negative agricultural externalities). Despite applying differing methodologies, Pretty et al., (2000) and Hartridge & Pearce (2001), suggest that agricultural external costs could be in the order of £1-2 billion per annum in the UK.

These figures are only indicative, as there are many gaps and contested methodological issues in both studies. But the numbers can be used to illustrate the extent of the problem and to understand the implications of the PPP for agriculture. If the estimates were to be accepted, they would reflect the "free lunch" that agriculture enjoys at the expense of other users. In relation to water resources management, this comes about because of prevailing water use patterns and ill-defined entitlements. Alternatively this is the sum that the sector would have to pay in order to compensate for the pollution/negative impacts produced.

In the context of the WFD and agriculture, the PPP aims to internalize the externalities arising from farming practices that have a negative impact on the water environment. As an example, Pretty et al., (2000) argue that pesticide externalities in drinking water alone average £8.6/kg of active ingredient used in UK agriculture[3]. If Britain were to internalise the total economic costs of pesticide used, assuming that this figure accounts for all the impacts produced, this figure would indicate the unit level of taxes or charges per Kg of applied product necessary to compensate others (e.g. society, the water industry) for the loss of welfare generated as a result of the application of pesticides.

1.2.3. Dealing with the problem

Previous regulatory approaches for the control of agricultural diffuse pollutants, which consisted of the application of voluntary measures at farm level, failed to achieve the expected improvements in water quality (European Commission, 1999, Aubin & Varone, 2002). This was proven by the lack of success in the implementation process of the Nitrates Directive in the late nineties. As an example, the UK failed to comply with all of its legal requirements, which resulted in a condemnation (in the form of fines) in the European Court of Justice in the year 2000 (European Commission, 2002b).

New legislative and regulatory settings for implementation of the WFD will greatly influence action at the farm level and will mark the extent of the total financial and administrative costs farmers will have to bear. In Scotland, new operating rules and controls to tackle diffuse sources of pollution have just recently been passed by the Scottish Parliament (i.e. The Water Environment (Diffuse Pollution) (Scotland) Regulations 2008). Standard-setting regulation will once again be the weapon of choice to reduce water pollution, even though the Directive encourages the application of new and innovative control instruments to achieve GS; including economic instruments (European Commission, 2002a); such as pollution trading permits.

Further controls to achieve mitigation of diffuse sources of water pollution add to one of the main concerns of modern UK agriculture, which is the increasing burden of regulations and the associated growth in compliance costs (NFUS, 2005). In this respect, the sector is already trying to come to terms with various environmental protection Directives (e.g. Nitrates and Bathing Waters Directives, etc.), different regulations (eg. Controlled Activities Regulations for water abstraction in Scotland, designation of Nitrate Vulnerable Zones - NVZs etc.), new and increasingly more complicated support schemes (CAP reform, agri-environmental schemes, etc) and the different codes of good agricultural practice available (such as the Prevention of Environmental Pollution From Agricultural Activity – PEPFAA – code, which sets out advice for reducing N losses from agricultural land in Scotland).

In theory, new regulations for the WFD are expected to be target-specific, proportional to the degree of environmental risk generated, and remediation measures cost-effective (SEPA, 2006). Although, there are no preliminary estimates available in Scotland to assess the extent of compliance costs by the sector, nonetheless it is assumed that the burden of regulatory and compliance costs is expected to make an impact (NFUS, 2005). As an illustrative example, DEFRA estimates for England suggest that increases in costs associated with future regulations and charging proposals in place for the agricultural sector could be in the region of £133m per annum by 2015 (with the implementation of the WFD accounting for £30m per annum). This would equate to a total increase in annual production costs for the agricultural sector of about 1.3% for the average farm business over the next decade (DEFRA, 2005a). Furthermore, DEFRA (2007a) has also estimated that WFD related policy options will impose costs of around £200 million on the English agricultural sector alone.

1.2.4. Public support for agriculture

To add to the problem, agriculture is an economic sector highly dependent on public support, much of which is in return for a range of positive externalities that agriculture offers to society (e.g. food supply and security, maintenance of rural livelihoods and landscapes, etc). Agricultural subsidies account for the majority of total farm income. In the UK, total subsidies (less levies paid) to the farming sector in 2004 amounted to £2.8 billion (DEFRA, 2005b). This total figure was made up of £2.3 billion directly linked with direct payments,

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[3] Based on British water companies reported annual operating costs for pesticide removal from water courses for compliance with environmental standards (Pretty et al., 2000)

and approximately £0.5 billion related to a range of schemes including, for example, animal disease compensation and a number of agri-environment schemes (such as Countryside Stewardship and Arable Stewardship schemes). In order to put these figures into context, it can be noted that the total value of farming output in 2004 (including subsidies which were directly related to products) was £16.9 billion, and that total income from farming (which also includes subsidies that were directly related to products) in 2003 was about £3 billion. As these figures indicate, subsidies are currently very material in relation to total farm income (Keyworth & Yarrow, 2005).

In summary, the implementation of the WFD is expected to have extensive effects on Scottish agriculture. On one hand, farming production is the main source of diffuse water pollution, which poses the greatest risk to the achievement of GS in Scotland. On the other hand, due to the multi-functionality aspect of agricultural goods and services, the sector delivers positive externalities to society. Thus agriculture is now faced with the challenge of maintaining its competitiveness, while protecting the environment. The economic viability of the sector and its ability to absorb the additional costs of protecting the water environment will determine farmers' efforts to achieve GS. In theory, there is substantial scope for the identification of ways to reconcile the WFD and public support (i.e. CAP objectives) and develop policies that will deliver benefits to farmers, the environment and other water users. In this sense, the objective is to create new types of incentives for farmers not to pollute, or to encourage the application of more environmentally sensitive practices.

1.3. Main research objective

The previous sections have briefly outlined the policy context underpinning this research and have introduced Scottish agriculture as the main case study for this thesis. In addition, we have identified the most significant challenges that surround the application of WFD targets to control agricultural diffuse pollution in Scotland.

Understanding the meaning of disproportionate costs, especially in the specific case of agriculture, is central for the likely success of implementing the Directive in Scotland. This poses a challenge for both policy makers and environmental regulators, as it requires finding the balance between reducing agriculture's impacts on the water environment for the achievement of Good Status and protecting the competitiveness of the sector. Water policy and regulation in this country (and in the rest of Europe) should strive to achieve efficient implementation. In this respect, the economic analysis of water use should play a pivotal role in informing future decisions in water resource management and influence the development of derogation decisions across member states, including Scotland.

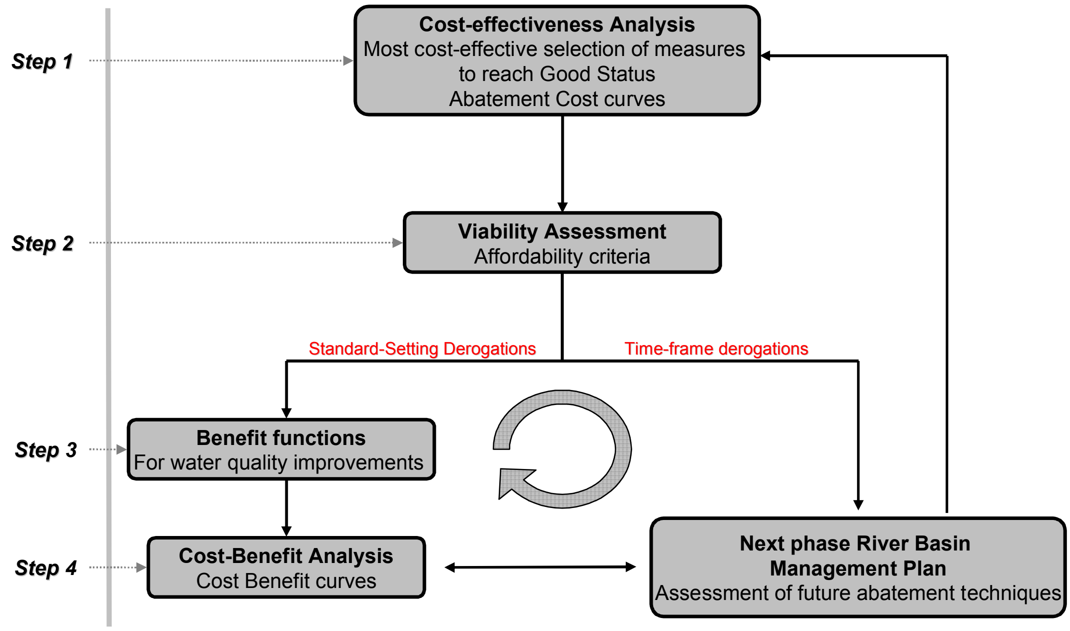
The main aim of this research is to clarify the definition of disproportionality under the WFD and to convey a consistent method for its analysis. The argument for contention in this thesis is that a rational model to inform decisions on derogations is needed and that economic theory provides a definition of disproportionate costs and the methodological tools that can inform its assessment (i.e. CBA). Arguably, these tools should be more influential in the development of derogation decisions across member states, including Scotland. This study aims to help clarify the nature of agricultural water use and how it leads to social tradeoffs with other non agricultural users. Ultimately this perspective adds to the debate of how and where water is best employed to maximize its value to society.

1.4. Research design - methods

Based on economic theory, standard-setting derogations should be judged with reference to cost and benefit curves – an application of the CBA method - combined with a financial viability assessment of the firm. For the justification of time-frame derogations, the assessment of benefits is not needed, as there is no need to justify changes in environmental objectives.

Figure 1.3 offers a guide to the main methodological steps proposed in this thesis for the assessment of exemptions under the WFD. This methodology is mindful of the following requisites: First, to be fully compliant with the economic requirements of the Directive (e.g. built upon the cost-effectiveness analysis for the selection of measures). Second, to be based on the principles of pollution control and welfare economics theory, as derogations should aim to reach socially optimal decisions. Third, to take into account implicit differences between the types of derogations being sought (e.g. time-frame versus standard-setting derogations). And finally, to be coherent with current guidelines for the economic appraisal of public policy and regulations impact assessments in the UK (e.g. UK Treasury Green Book). The justification for the need to include these requirements is given in chapter 3.

Figure 1.3 Methodological steps for the assessment of disproportionate costs



We now briefly discuss the implications of using such a model for a hypothetical farm, which will be used to introduce the different methodologies employed in this research. We aim to research the answer to the following question: what information would be required in order to judge if a farm should be granted exemptions under the WFD?

Independently of the types of derogations being sought, a preliminary cost-effectiveness analysis of all the measures available to the farmer to reduce water pollution, needs to be undertaken as a requirement for the design of PoMs to reach Good Status. Once all possible measures have been ranked in terms of their cost-effectiveness, the following information will be available: i) what measures are needed to reach Good Status; and ii) the extent of the total financial costs of reaching the stated objective. Chapter 4 illustrates an application of the abatement cost curve method for the economic appraisal of agricultural Phosphorus (P) mitigation options for different farm systems.

Once information about costs and effectiveness of measures has been collated, it can be compared with an assessment of the financial viability of the farm and the ability of the farmer to absorb the additional costs of protecting the water environment. Ultimately, compliance costs will determine the farmer’s efforts to achieve Good Status at particular water bodies (Lago et al., 2006). The use of financial indicators or income ratios provides a good option for assessing the costs of meeting the environmental requirements of the Directive at individual and sectoral level (Defra, 2006a). However, there is a need to distinguish between ability to pay and affordability. This distinction is more subjective and controversial. Two methods are outlined in chapter 5 in order to explore issues of "affordability" and "ability to pay" by the sector: 1) a financial characterisation of farming in Scotland and 2) impact on profits of achieving different P loads reductions at farm level are investigated.

If the viability assessment indicates that the application of the most cost effective selection of measures to achieve Good Status carry an unreasonable burden on the financial sustainability of the farm, regulators will then need to apply further derogation tests, which will differ depending on the type of derogations being sought.

For time-frame derogations, regulators can base their decision on the outcomes of the two tests introduced above. In practice, this would basically involve doing nothing until the beginning of the next river basin management cycle. This fundamentally means just waiting until there are new abatement techniques available to reduce the farmer's costs of compliance. Essentially, there would not be a need to lower the environmental standards. Once this is done, the whole cycle needs to be repeated for the next river management cycle – beginning again with CEA.

For standard-setting derogations, the analysis becomes more complex. The costs of reducing pollution at farm level need to be compared with the associated benefits of water quality improvements. The main rationale of applying benefit assessment of environmental quality improvement is that the lowering of the environmental standards needs to be: i) socially justifiable in the light of the WFD; and ii) following economic theory, the optimal point of pollution control (where costs equal benefits) is the only point when a satisfactory outcome for both society and the farmer can be found. Benefits assessment of water quality improvements is covered in chapter 6. This chapter illustrates applications of two different methods to explore public preferences for pollution control and measure non-market benefits of WFD water quality improvements in Scotland: 1) benefits function transfer from a recent valuation study in England and Wales and, 2) discrete choice modelling

1.5. Aims and objectives

In addition, the underlying objectives of this research relate to the critical evaluation of the proposed methodologies for the assessment of disproportionate costs. The specific research objectives of this thesis are outlined below in case study order:

Development of Cost Functions for Agricultural Best Management Practices

Policy objectives

i) To understand the economic implications of adopting different mitigation strategies at farm level (Best Management Practices - BMPs) in order to reduce farm diffuse pollution to water.

ii) To develop a criteria for the selection of BMPs at farm level, which is relevant in order to provide information on the most cost-effective selection of abatement techniques for PoMs to achieve GS.

iii) To assess the financial costs of reducing farm diffuse pollutants in order to achieve different target levels. Relevant for the assessment of disproportionate costs and for the analysis of the costs and benefits of farm level mitigation option strategies for Regulatory Impact Assessment (RIA) of River Basin Management Plans (RBMPs).

Methodological Objectives

iv) Explore the application of cost-effective ratios for the selection of BMPs at farm level.

v) Evaluate the suitability and limitations of the abatement cost curve method to estimate the extent of the financial costs associated with achieving different levels of diffuse pollutants loads reductions at farm level.

Financial Viability Assessment and Definitions of Affordability

Policy objectives

i) To investigate different definitions and measures of affordability for the practical definition of disproportionate costs.

ii) To undertake a financial characterisation of farming in Scotland.

iii) To assess and quantify the likely financial impacts for a typical Scottish farm of adopting different diffuse pollution mitigation strategies in order to achieve water quality improvements. This exercise would be relevant for the assessment of disproportionate costs and for the small business impact analysis of the costs and benefits of farm level mitigation option strategies for the RIA of RBMPs (Cabinet Office, 2003).

Methodological Objectives

iv) To develop a multidimensional financial criteria indicator to identify farms in poor financial condition and exploration of other issues to consider in this type of analysis for technology adoption at farm level.

v) To investigate the application to the Scottish dairy and arable sectors of an optimisation model to assess the likely changes to farm profits as a result of achieving different P loads reductions at farm level under two different scenarios: with and without government intervention (i.e. impact of regulations).

Benefits Functions for Water Quality Improvements

Policy objectives

i) Quantification of robust estimates of the overall benefits to society derived from the achievement of GS in Scotland.

ii) Evaluate the use for water policy analysis of the benefits transfer method

iii) Explore preferences and public perceptions about restoring river and loch quality to and beyond good status for the whole of Scotland, time preference for the improvements (2015 versus 2028) and whether regional differences exist within Scotland in preferences towards changes.

Methodological Objectives

iv) Review existing valuation approaches to the estimation of the non-market benefits of water quality improvements.

v) Practical application and evaluation of the Benefits transfer method. Validity assessment of the results.

vi) Practical application and evaluation of the choice experiments method. Assess the transferability of the estimates across locations.

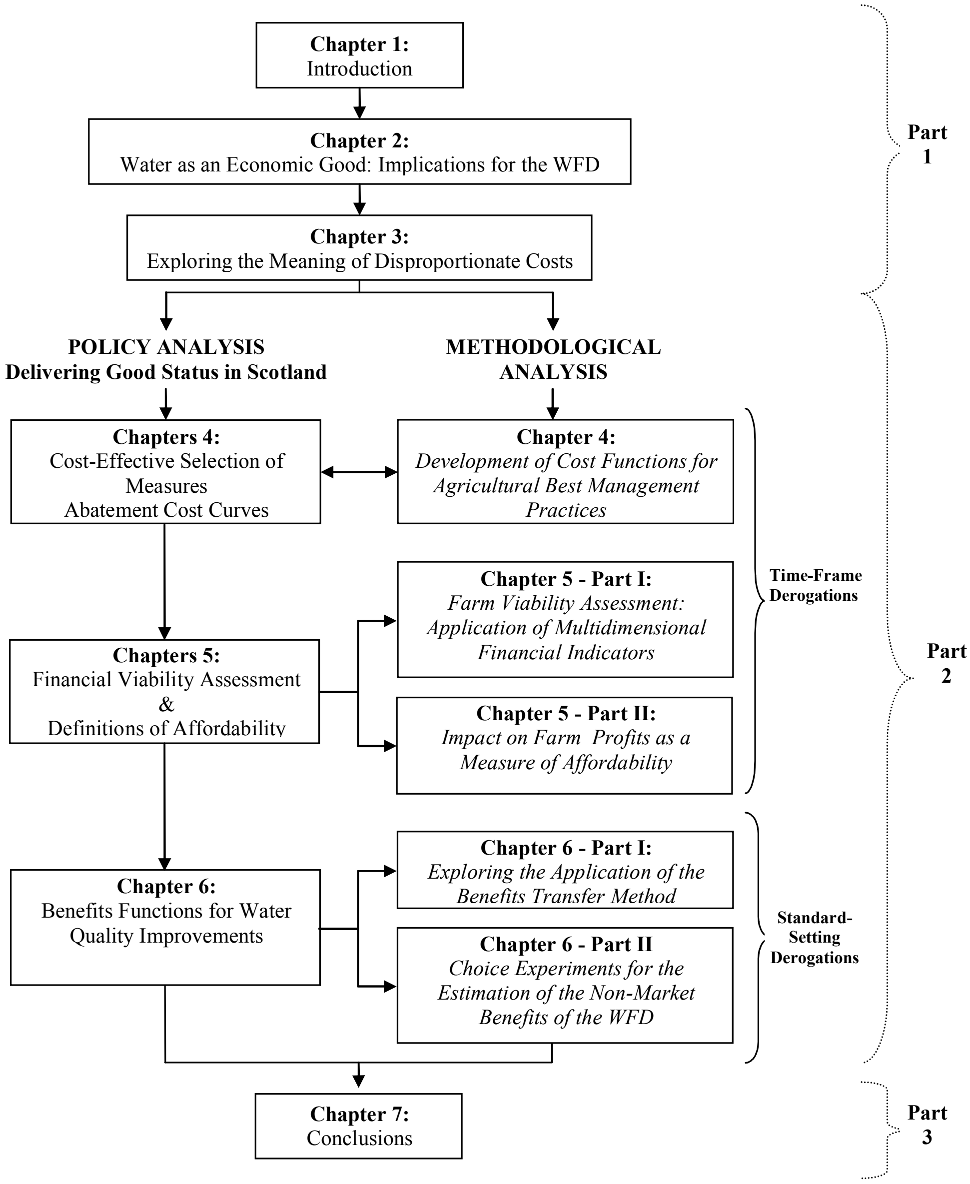
1.6. Thesis structure

The complete structure of the thesis is presented in Figure 1.4. It comprises three main parts. Part 1 provides an introduction to the main research topic and sets the scene for the empirical chapters of this thesis. This includes a review of the implications of treating water as an economic good for the WFD, which is followed by a detailed exploration of the assessment of derogations and the development of a theoretical definition of disproportionality. Part 1 provides a justification for the work undertaken in the case studies. Part 2 contains the core research chapters, comprising both policy and methodological analysis, presented for each of the proposed tests for the assessment of disproportionate costs. These chapters include detailed discussion of the policy problems and /or methodological challenges being addressed, the experiment design used, and the results and conclusions. Part 3 presents the overall conclusions for the thesis. An outline of the individual chapters is provided below.

Chapter 2 explores the role of the WFD's economic elements and how these relate to conventional water economic theory. Particular reference is paid to the determination of the economic value of water use, the cost recovery principle for the design of water pricing policies, cost-effectiveness analysis for the selection of measures to achieve GS and the assessment of exemptions. Furthermore, some of the main issues which surround the economic analysis of water use are identified.

Chapter 3 focuses on exploring in greater detail the economic interpretation of the meaning of disproportionate costs for the practical implementation of the Water Framework Directive (WFD). Implications for the agricultural sector are considered. The chapter is structured as follows. In the first section we set the question of disproportionality in the context of the basic economics of pollution control theory and the equi-marginal value principle. The next section considers the definition of GS and alternative definitions of disproportionate costs which are consistent with cost-effectiveness or cost benefit analysis principles. The final section reflects on the implications for a hypothetical farm, where theoretical exactitude may ultimately come second to a practical definition that regulators can employ quickly and practically when deciding on whether costs are disproportionate.

Figure 1.4 Thesis Structure



Chapter 4 explores the implications, limitations and possible applications of using the abatement cost curve method to estimate the extent of the financial costs associated with achieving different levels of nutrient loads reductions at farm level through the implementation of Best Management Practices (BMPs). The adoption of BMPs will be a critical component of the PoMs to achieve GS in Scotland, especially for the control of agricultural diffuse pollution. As a preliminary assessment of disproportionality, it is essential to understand the economic implications of adopting different mitigation strategies.

In this context, we have developed a detailed assessment of the financial costs and effectiveness of BMPs to reduce farm losses of main pollutants.

The aim of Chapter 5 is to assess the likely financial impacts for a typical Scottish farm of adopting different diffuse pollution mitigation strategies in order to achieve water quality improvements. In essence, this chapter uncovers how the cost of abatement would impact on the financial viability of the farm, with the underlying objective of investigating different definitions and measures of affordability for the definition of disproportionate costs. Chapter 5 offers an examination of two different practical definitions of affordability at farm level which are relevant to water policy: i) the use of farm financial indicators to assist in the decision-making process about derogations; and, ii) an assessment of impact on profits as a measurement unit of changes in farmers' welfare.

Chapter 6 addresses the third test of the proposed methodology for the assessment of disproportionality, which deals with the estimation of the overall benefits to society derived from the achievement of good status. The main aim of this chapter lies in the estimation of benefit values for WFD improvements in Scotland and the exploration of valuation methods to estimate non-market values for water quality improvements. Based on a brief literature review of available techniques for valuation of environmental improvements, the chapter is divided into two parts, which consist of empirical applications of two of these methods: Benefits transfer and discrete choice modelling. Part 1 of this chapter illustrates the results of a benefits function transfer exercise and evaluates the suitability of the method. We will show that even though the benefits transfer method may prove a valid alternative to answer our research question, and a "quick and inexpensive" way to inform policy decisions, its main weakness comes from the fact that it is impossible to assess the validity of the transferred values. This conclusion is needed to justify the undertaking of an original valuation study in Scotland. The second part of this chapter presents the results of a choice modeling exercise to elicit Scottish households' willingness to pay for improvements under the WFD.

Chapter 7 presents the main conclusions of the thesis and reviews the implications both for policy and methodology. The chapter concludes with a discussion of the main strengths and weaknesses of the thesis and recommendations for future research.

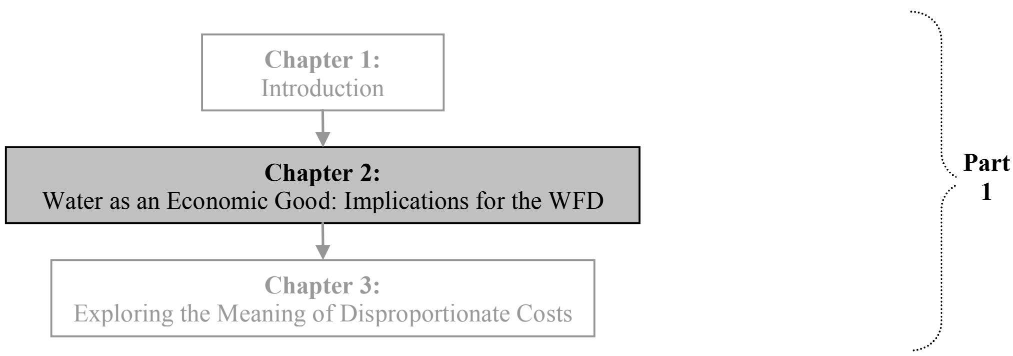
CHAPTER 2

WATER AS AN ECONOMIC GOOD: IMPLICATIONS FOR THE WFD

The conference on water and the environment, held in Dublin in 1992, formalised the definition of water as an economic good. This formalisation is regarded as a prerequisite for the achievement of sustainable uses of the resource. In the Water Framework Directive, this definition has been amended to justify its environmental objectives at all costs.

This chapter explores the role of the WFD's economic elements and how these relate to conventional water economic theory. Particular reference is paid to the determination of the economic value of water use, the cost recovery principle for water pricing policies, cost-effectiveness analysis for the selection of measures to achieve Good Status and the assessment for exemptions. Furthermore, some of the main issues concerning the economic analysis of water use are identified. This chapter provides a key reference point for the remainder of the research.

Figure 2.1 Thesis structure - Part 1



2.1. Background

The use of economics in water resource management was formalised in the *conference on water and the environment* held in Dublin in the year 1992. This conference fused the concepts of integrated water resource management and the definition of water as an economic good, a condition to achieve worldwide sustainable uses of the resource.

The influence of the Dublin conference in the formulation of the WFD is obvious; concepts developed there such as integrated water resource management, sustainable use of water, public participation, etc. have been directly applied in the formulation of the Directive. In the case of treating water as an economic good, the fourth principle of the Dublin statement reads "*Water has an economic value and should be recognized as an economic good, taking into account affordability and equity criteria*" (ICWE, 1992). However, in this case the Directive takes much more care in the translation of this principle and states that "… *water is not a commercial product like any other but, rather, a heritage which must be protected, defended and treated as such*" (European Commission, 2000).

In its definition of water, the WFD opts for defending the social dimension of the resource thereby adopting the Dublin statement aspect of "*taking into account affordability and equity criteria".* The definition purposefully omits to expressly state that water should be treated as an economic good. However, contrary to the definition given, water is clearly regarded as an economic commodity in the WFD (as an example, the word economics is mentioned 22 times throughout its legal text) and it directly borrows economic terms included in the Dublin statement; such as, cost-recovery or ability-to-pay.

It could be argued that there are two main reasons why the designers of the WFD decided not to specifically define water as an economic good. Firstly, the actual definition could be seen as means to justify the Directive's predefined environmental objectives as society's right to a clean water environment; and secondly, there may have been doubts about the role that economics could play in delivering its social objectives.

Green (2000) reports that following the Dublin conference, water practitioners looked to economists for an explanation of what the definition of water as an economic commodity would mean in practice. According to Green, this opened up a debate between those economists who applying the neoclassical principles of economics argued that water should be priced at its economic value and internalised into markets which would allocate the resource to its best uses; and those economists who being more flexible about the application of economic theory, understood that the core of water economics is about informed choices and how best achieve sustainable uses in broad societal contexts (Green, 2000).

Savenije and Van der Zaag (2002) observe that many policy-makers wrongly thought that the adoption of the conference's aforementioned fourth principle would lead to an economic pricing of water based on strict neo-classical economic theory, which many feared would not have due regard for social interests (i.e. provide water for the poor) or would make water use unfeasible for low-band value users (i.e irrigated agriculture). These policies would prove most unpopular, especially in countries with water scarcity problems. Consequently, it may be the case that the fathers of the WFD, explicitly decided to defend the social dimension of the resource.

The lack of agreement between economists on the use of economic principles is often evident in the formulation of environmental policies. Debates frequently focus on the different theoretical economic interpretations of the policies rather than on exploring and explaining to non-economists/practitioners, the practical applications of the use of economics to solve specific environmental problems. In this instance, the debate which opened up after the Dublin conference is a good example, as both sides of the debate differ in theory but mean the same in practice. Essentially, they failed to explain to non-economists the potential role of the economic analysis of water use to deliver social, economic and environmental objectives. As an example, water markets do not necessarily have to adversely affect the poor, once the market is/has been created, there are further instruments available which can be employed to avoid social inequalities, such as sponsoring low income users (i.e. through the use of subsidies – increasing block tariff systems for water services; Dalhuisen et al, 2001).

2.1.1. The problem with water

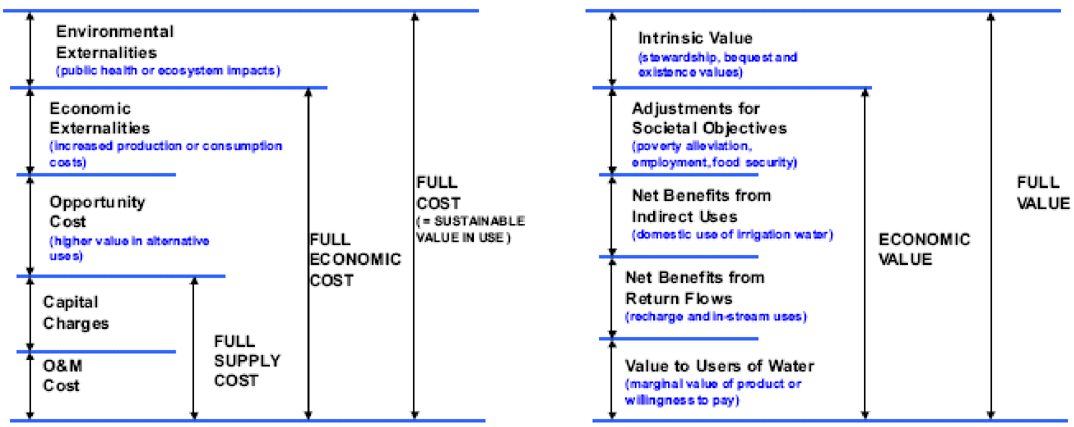
The main reason for the differences of interpretation in the use of economics in water resources management resides in the complex nature of water as a resource and the associated difficulties for its economic analysis. While it is well accepted that water has an economic value, the real problem for economists is that unfortunately, water cannot be treated as another common trading commodity due to its special characteristics and therefore, its economic value should not be directly equated with its price. Morris (2004) identifies the main reasons for this special treatment:

1. water is a fugitive, re-usable resource which is difficult to control and account for;
2. water is often a common property, with open access and ill defined property rights;
3. it provides public goods, such as the public health benefits of clean water;
4. water is used in many different ways which often result in external consequences to other water users and the environment (externalities);
5. it is essential for life, without close substitute: a need rather than a want
6. water is subject to uncertain supply associated with climate variations;

7. water has significant economies of scale associated with its managed supply; and,
8. water is an integral part of the functioning of ecosystems.

In theory, a sustainable use of water would be achieved when the full costs of supply equal its full price (value). The extraordinary characteristics of water as a resource make it difficult to establish markets for its right allocation amongst different users and for the correction of externalities. The resource has many different uses which can be assigned many different values. This complicates valuation or the measurements of demand. As an example, Rogers (1998) explored the notion of water use values and the general principles for the cost of water (figure 2.2).

Figure 2.2 General principles for cost and value of water



From: Savenije and Van der Zaag, 2002; modified from Rogers, 1998

Figure 2.2 highlights some of the problems economists face when attempting to estimate the full value and costs of water. On the costs side, the estimation of the opportunity costs (estimation of the higher value alternative use of water) and externalities (negative/positive non-marketable effects of water use) pose a challenge to optimal social allocation. In the demand side, the identification of different types of water users (e.g. indirect users of water such as recreation), the methodologies used for valuing use and the comparison of value estimates between uses is just as complicated.

Before analysing the different cost categories and value components relevant to the implementation of the WFD, it is important to point out that the two economic schools identified by Green (2000) differ mainly in the potential applicability of the economic analysis of water use. This analysis requires the accurate estimation of both the economic values of different uses and the costs of water supply, including its opportunity costs. One school defends that each cost/value category should be accurately estimated for the design of water markets, and the other school defends the importance of the economic analysis of use to inform decisions in water resource management, independently of the ultimate goal of establishing markets. Nevertheless, both schools advocate the purpose and importance of the economic analysis of water use for decision-making.

2.2. Economics and the WFD

Prior to the formulation of the WFD, EU policy-makers understood that defining water as an economic good could mislead the economic debate (or move it away from where they thought the debate should go). Previous pieces of European water legislation failed to achieve their targets because of the huge compliance costs associated with the required investments and the clear lack of consensus on the regulatory/economic instruments to be used to achieve the defined environmental objectives effectively (Kallis & Nijkamp, 2000; Aubin & Varone, 2002; Kaika, 2003).The WFD recognizes the economic dimensions of water in terms of scarcity, value and introduction of markets (prices which reflect the full cost of supply and the benefit in use). But it also clearly highlights that the interest now lies in the achievement of the Directive's objectives (good status, sustainable uses of water…) and the main role of the use of economic tools and principles is to effectively deliver these targets. We could argue that this follows Green's second school of water economics thought, but the vagueness in the Directive about the use of economics and its integration in water resource management leaves room for interpretation and debate.

Overall, the use of economics for the implementation of the WFD is prescribed in four main areas: i) the estimation of the demand for and the valuation of water in its alternative uses (Article 5); ii) the identification and the recovery of costs associated with water services having regard for the polluter pays principle and the efficient use of water (Article 9 and Annex III); iii) the use of economic appraisal methods to guide water resource management decisions (Article 11) and; iv) the use of economic instruments to achieve the objectives of the WFD, including the use of incentive pricing and market mechanisms (Article 11).

2.2.1. The economic analysis of water use

The WFD specifies a series of reporting dates for key tasks and activities aimed at the development and implementation of river basin management plans (see timetable for implementation in table 2.1). This applies to many elements of the Directive, including its economic requirements. However, the Directive's legal text is a prescriptive document and does not clearly specify how to implement or develop its requirements and key elements. Consequently, the Commission established informal working groups at European level to develop guidance documents to aid different aspects of the implementation process of the Directive, with the main objective of harmonising the implementation process across Europe and encouraging application (e.g. guidance documents have been produced on the analysis of pressures and impacts for the environmental characterisation documents or on the establishment of water quality standards).

Table 2.1 WFD detailed timetable for implementation

Year	Issue	Reference
2000	Directive entered into force	Art. 25
2003	Transposition in national legislation	Art. 23
	Identification of River Basin Districts and Authorities	Art. 3
2004	Characterisation of river basin: pressures, impacts and economic analysis	Art. 5
2006	Establishment of monitoring network	Art. 8
	Start public consultation (at the latest)	Art. 14
2008	Present draft river basin management plan	Art. 13
2009	Finalise river basin management plan including progamme of measures	Art. 13 & 11
2010	Introduce pricing policies	Art. 9
2012	Make operational programmes of measures	Art. 11
2015	Meet environmental objectives	Art. 4
2021	First management cycle ends	Art. 4 & 13
2027	Second management cycle ends, final deadline for meeting objectives	Art. 4 & 13

Source: European Commission (http://ec.europa.eu/environment/water/water-framework/timetable.html)

The working group dedicated to the attention of the Directive's economic issues was set up in December 2000 and named WATECO (for water economics). So far, this group has only produced one official guidance document (European Commission, 2002a), which covers general aspects of the economic analysis for the development of river basin management plans, paying special attention to the economic characterisation of river basin districts.

The WATECO group recognized that the economic analysis is a process of "*providing valuable information to aid decision-making and should be an essential part of the overall approach for supporting decisions*" (European Commission, 2002a). In theory, the objective of the analysis is to serve as an exercise in the elicitation of trade-offs and it is to be undertaken in co-ordination with other types of information and input, such as from the public participation processes (Kallis, 2005).

By the end of 2004, it was required that each Member State undertook an economic analysis of water use for each of their river basins (see timetable for implementation in table 2.1). This was produced together with a preliminary assessment of the balance of demand and supply of water services and the pressures and impacts on the water environment[4]. In other words, the economic analysis should provide information on what it costs, who pays, who gains and who suffers from the current situation and has to be integrated with other technical analyses such as the environmental analysis of pressures and impacts. This aims to ensure that a common description and characterisation of the river basin is obtained and used as the basis for the identification of the Programe of Measures (PoMs) and the development of the river basin management plans (RBMPs). The results of the economic analysis will be used to inform future WFD-related decisions.

2.2.2. The concepts of water use, value and costs

For the achievement of sustainable uses of water resources, the Directive goes beyond the concept of water demand management (an instrument traditionally applied in water resources management which aims to attain optimal uses of water to ensure the financial sustainability of the service) and promotes the introduction of water pricing policies, which also account for the recovery of environmental and resource costs of the different types of use. This could be seen as a way to attain a level of sustainability of water use, more in accordance with the environmental objectives of the Directive.

The theory behind demand management of water services is fairly simple. It aims to attain some sort of economic optimality in use by taking into account the value of water in relation to the financial costs of provision[5] (Winpenny, 1994). In the context of the WFD, the

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[4] For further information; the results of the analysis for each member state can be found at: http://forum.europa.eu.int/Public/irc/env/wfd/library?l=/framework_directive&vm=detailed&sb=Title

[5] In theory, this will be achieved when the marginal unit of water for each user has the same value.

objective is not only to achieve sustainable management of water resources but also sustainable uses. In consequence, one of the first steps in the economic analysis of water use is the identification of the different types of uses of water; each different use would imply a different economic value and in many instances may also incur costs. Unlike other commodities, the special characteristics of water as a resource, imply that the same good in theory has different economic uses in practice (which in the case of water, can also differ in levels of quantity and quality).

Table 2.2 Selected classifications of the value of water in economics

Turner, Georgiou, Clark and Brouwer (2004)	Rogers, Bhatia and Huber (1997)	Young (1996)	De Groot (1992)
Describe the components of the value of water using conventional categories of Total Economic Value, which is the sum of: ▪ Direct use values: Arise from direct interaction with water resources. They can be consumptive, e.g. irrigation or non-consumptive, e.g. recreational swimming ▪ Indirect use values: Services provided by water resources but that do not entail direct interaction, e.g. flood protection by wetlands ▪ Non-use values: Existence, bequest and philanthropic value. ▪ Option value: Satisfaction of knowing that the resource is available to future generations ▪ Quasi-option value: Derived from the potential benefits of delaying action until further information is available, e.g. value placed on conservation of a wetland until further information is available on the value of the species that are found within it.	Value of water use comprises economic and intrinsic value: ▪ Value to other users: Value of water in industrial and agricultural use and WTP for its domestic use ▪ Net benefits of return flows: Recognises the vital role played by return flows in many hydrological systems e.g. recharge of aquifers ▪ Net benefits from indirect use: Benefits associated with improvements in income and in health that can accompany schemes that provide water for irrigation, domestic and livestock use. ▪ Adjustments for social objectives: e.g. poverty alleviation, employment generation or food security Intrinsic value of water: Includes the stewardship, bequest, and pure existence value	Water related economic values are divided into the following classes: ▪ Commodity benefits: These are derived from personal drinking, cooking and sanitation, and from productive economic activity, e.g. agriculture ▪ Aesthetic and recreational values ▪ Waste assimilation benefits: These result from the sink function of waterbodies that carries away residuals from processes of human production and consumption. ▪ Dis-benefits or damages: These are found in connection with evaluations of foodplain and water quality management. ▪ Non-use values from knowing that a good exists, even though no direct experience is had of the good. Other possible values, include: intrinsic, ecosystem preservation and socio-cultural.	Value is categorised in terms of the nature of the contribution made to human welfare, categories: • Ecological value: Includes conservation and existence values. Usually only described qualitatively as valuation is limited, though it may be described using quantitative indicators (e.g. number of species) • Social value: Includes health and option values. It may be quantified through use of minimum standards for resource availability (e.g. to ensure sustainable harvesting) • Economic values: Includes consumptive use, productive use and employment value. It can be described in monetary units (e.g. value of the resource harvested), quantities (e.g. volume of a resource harvested) or the number of the people employed in the activity

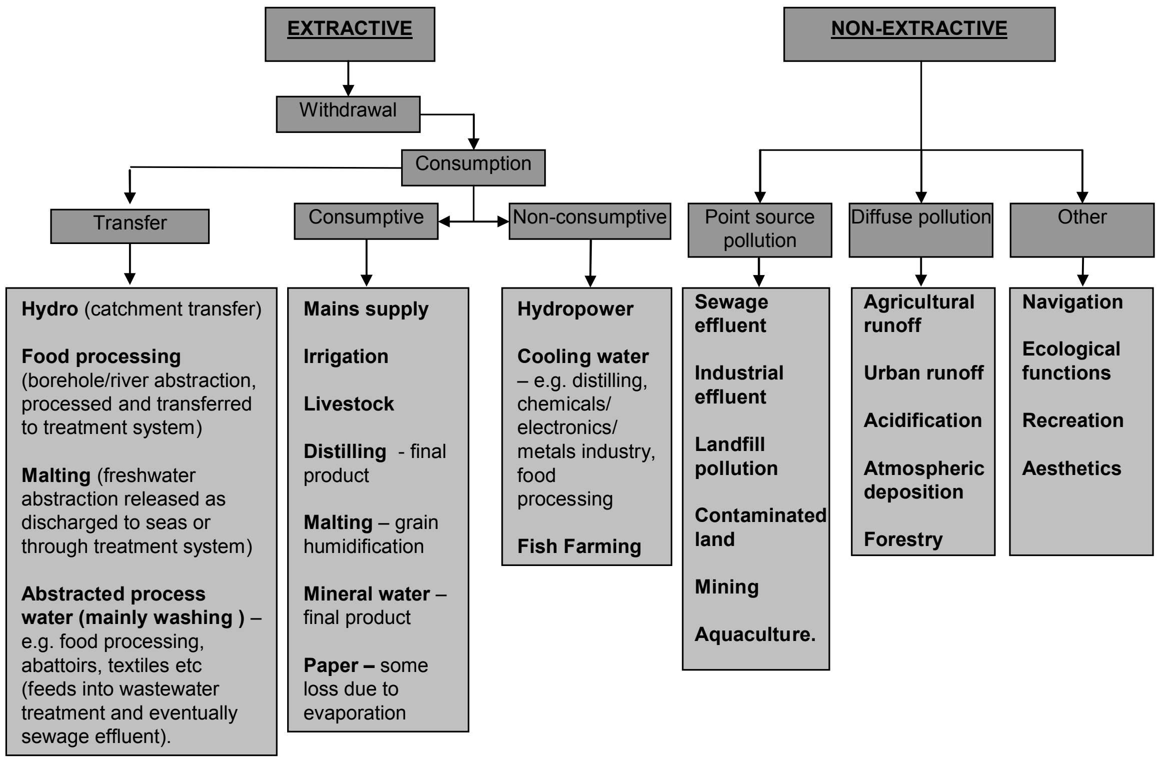
Modified from: Turner, Georgiou, Clark and Brouwer (2004)

The typology of the different uses of water is a contested issue in the water economics literature. As an example, table 2.2 introduces a selection of the various classifications used in water economics to describe the different types of values associated with the goods and services provided by water resources. Again, these differences are related to the versatility of the resource, which introduces different points of view.

Water uses are often divided, in relation to their economic values (or benefits derived from its use) and the nature of the use. As the most typical example used in the literature, the total economic value approach divides water use into two main types: use and non-use values (Turner et al., 2004).

The Directive is concerned mainly with use values, which can be classified as: i) direct uses, which are extractive and consumptive and have a direct impact on water quality and quantity. Some examples of direct uses are agricultural irrigation, water stored for hydroelectricity generation, drink production, etc... And ii) indirect values, which are related with recreational and aesthetic uses, and are typically non extractive and non-consumptive. Figure 2.3 offers some examples of the different definitions of water use according to the WFD.

Figure 2.3 Definition of uses of water for the WFD

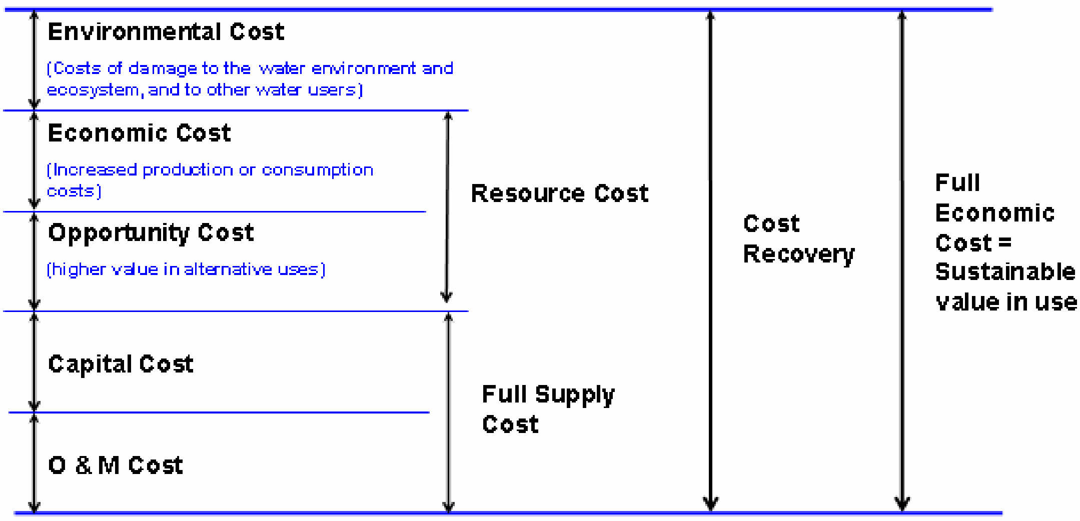


(Modified from: Moran ad Dann, 2007)

Normally, water use values are defined in terms of changes in quantity and quality. This poses one of the main problems in their estimation, which is that similar uses in theory may have different demands in terms of quantity and quality impacts in practice. For example, identical levels of in-stream abstractions may have different impact in quantity and quality depending on geographical and weather variations. Moran and Dann (2007) note that some texts of the Directive and related documents have used the term non-use to indicate non-abstractive uses (e.g. water for cooling purposes in hydropower and distilleries), as opposed to a passive activity totally unrelated to any direct exploitation. However, this is a disputed issue, even though water quality may not be altered as a result of a passive use, these types of abstraction have a temporal impact in water quantity. In light of the WFD, these types of use should also be included in the economic analysis.

The analysis of the full costs incurred by water users is also important under the WFD for the achievement of sustainable water pricing policies. The underlying aim is the rationalisation of water use in Europe and the drive to increase efficiency of resource use. The argument goes that paying for the full costs of the service will dissuade the unsustainable over-use of water courses beyond their assimilative capacity. While the estimation of supply/discharge costs faced by some users may be quite straightforward, this includes the assessment of the financial costs faced by the water user. The assessment of the external costs associated with use is another story, much more difficult to depict in practice. In theory, the full cost estimation of water use should also include: the costs associated with damage to the water environment, associated costs caused to other users and the opportunity costs of use (related with the estimation of the higher value of water in alternative uses). Under the WFD, the internalisation of the environmental costs and external economic and opportunity costs into prices for water services, named environmental and resource costs in its legal text, is the ultimate aim of the cost recovery principle. The Directive specifically states that the application of the recovery cost principle should go beyond the simple estimation of the costs of supply of water services (capital, operation and maintenance costs) and mandates its application for the three major users of water services; industry, agriculture and households. Figure 2.4 introduces a schematic representation of our interpretation of the types of costs of water use under the WFD.

Figure 2.4 Personal interpretation of the typology of costs of water use under the WFD



2.2.3. The polluter pays principle, the selection of measures and exemptions

The main rationale behind the introduction of the Polluter Pays Principle (PPP) in the WFD is the need to internalise negative externalities imposed on the water environment and other users of water. Externality is purely an economic term which refers to costs (or benefits) of production and consumption imposed on third parties who are not involved in the transaction and do not receive (or pay) compensation for these costs (benefits). Such external effects are common when goods or services which are not traded in the market are involved. This is typically the case with environmental goods and services. The absence of markets and well-defined property rights for such goods and services results in failure to assign them their true economic value and to safeguard their quantity and quality, leading to misallocation of environmental resources and environmental degradation.

The WFD divides impacts in terms of the source and cause of pollution into six main categories: point and diffuse sources of water pollution, morphological alterations to the water environment, the introduction of alien species, abstraction and flow regulation. Complementary to the recovery principle, which aims to recover the full costs of water services for the introduction of sustainable pricing policies, the introduction of the PPP to the Programme of Measures (PoMs) aims to ensure the achievement of the environmental objectives of the Directive, no deterioration and good status.

The approach emphasised by the WFD on achieving Good Status (GS) of water resources, highlights the choice of the most cost-effective combination of measures for bridging the gaps between the environmental objectives and the baseline scenario; in other words, the quality that would prevail as a counterfactual to the Directive being in place. This means that the POMs should provide the lowest-cost set of measures to reach a set environmental target for the river basin/water body.

Cost-effectiveness analysis is an optimisation method for finding the lowest cost means to reach a desired objective (Tietemberg, 1992). In this instance, the objective of the analysis is to achieve the desired environmental standards that define GS at least costs to society as a whole. The prescription of the use of economic instruments, such as Cost Effectiveness Analysis (CEA) for the selection of measures to achieve GS, is aimed at ensuring efficiency in the design of water resources policy/action and to avoid unnecessary economic and financial costs of compliance. By predefining the standards to be achieved based on parameters to protect the biology of the water environment, the directive avoids the issue of economic efficiency and the estimation of costs and benefits of action for the selection of measures. These are only applicable when there is a case for exemptions and some sort of optimal judgement is required.

Meeting GS in water bodies is cost dependent and in some cases costs may be high or judged as disproportionate. The assessment of disproportionality makes reference to the justification of conceding exemptions for the achievement of these objectives; such as: to grant time-frame derogations to achieve GS or even, allowing the lowering of environmental standards (from GS to good potential) when a water user finds the total costs of the most cost-effective programme of measures too expensive or disproportionately expensive to undertake (European Commission, 2000).

The WATECO guidance document on the use of economics for the WFD, states that disproportionality is to be decided by individual member states on a case-by-case basis and its assessment has to be the outcome of a political decision informed by the economic analysis (European commission, 2002a). However, this guidance does not set a clear course of action to assess disproportionality and only vaguely recommends the use of simple financial criteria for time-frame derogations and the application of cost-benefit analysis for seeking less stringent objectives. This aspect of the Directive needs further analysis and study, which is the main research theme of this thesis, as the assessment of exemptions may prove one of the most controversial steps on the implementation process. For example, poorly informed decisions about exemptions may unveil issues of competitiveness between water users or uneven distribution of the financial costs associated with the Directive (Pearce, 2004).

At the centre of the disproportionality debate is the choice of instruments and methodologies to offer a model of rationality to inform decision making processes. Applying CBA presents a challenge, requiring an assessment of the full social costs and benefits (social and environmental) associated with a proposed measure. Benefit assessment in particular raises some complex issues related to the process of valuation and the fact that some water bodies are more socially valuable in relative terms. Despite this, political decisions regarding exemptions should aim to achieve some sort of economic efficiency and coherence in the final decision. If not, decision-makers may face issues of conflicting rights between those who pay the costs of water quality improvements and those who benefit, as they may have overlooked the extent of the net social costs involved in complying with the Directive (Pearce, 2004). This is an issue which will be explored in further detail in the next chapter of this thesis.

2.2.4. Some practical issues concerning the economic analysis of water use

This section will look at some of the general issues identified in the academic literature that may reduce the accuracy of the preliminary economic analysis of water use for the implementation of the Directive. Overall, the definition of property rights, uncertainties surrounding the analysis and the issue of transaction costs may affect the design of sustainable water pricing policies or affect the appraisal of pollution reduction measures for WFD implementation purposes.

2.2.4.1. The issue of property rights in the WFD

One of the fundamental issues in the economic approach to resource and environmental issues is the precise definition of property rights. The role of markets and prices is central in the economic debate on the efficient allocation of environmental resources and a necessary condition is the establishment of well-defined and enforceable private property rights (Perman et al., 2003). In the case of water resources, Perry et al., (1997) set the establishment of property rights as the first condition for the introduction of market forces into the allocation of water. This condition states that the entitlements of all users under all levels of resource availability should be defined and include specified assignments to social and environmental issues (Perry et al, 1997).

In the case of the WFD, the issue of property rights has been purposely covered. One of its main features is that the water environment itself should be treated as another water user with all the consequences this labelling may entail. The application of the polluter pays principle to the WFD asks water users to pay for the environmental and social damage associated with their negative impact on the water environment (European Commission, 2000), aiming to internalise the damage to water ecosystems associated with water use and covering the issue of property rights. Technically, this represents the complete end of the "free lunches" era for private and institutionalised water resources management in Europe. For the first time in European water policy, all water users have been asked to pay/compensate for the full costs of using the resource, including environmental and social costs, to ensure that water resources are being sustainably managed.

To reinforce the PP principle and the cost recovery principle, the WFD also introduces a set of legal transposition requirements. These oblige each member state to incorporate the Directive into their national law (e.g. case of the WEWS (2003) Act in Scotland) and develop regulatory instruments for its enforcement (e.g case of the Controlled Activities Regulations (2005) in Scotland). The establishment of pricing policies and the PoMs for water pollution control and reduction options are also normative and the failure to meet the objectives of the Directive punishable.

As the objectives of the Directive are set and enforceable, what are the implications for the economic analysis? In terms of property rights, Pearce (2002) recognises that the WFD predefined objectives are independent of the costs to achieve them, as these goals do not acknowledge public preferences. In other words, the goals are completely independent of elicited human preferences. This is what Pearce calls the "public-trust" doctrine, which makes the goal of policy in the face of damages, the restoration of the pre-damage state of the environment. Under the WFD, "*Good Status*" reflects a legal judgement about the role of the Commission as a trustee of citizen's rights (Pearce, 2002). Accordingly, the Directive assigns property rights concerning the future state of the water environment to individuals, but their rights are inflexibly managed on their behalf by the Commission. In practical terms, benefits of action do not need to be estimated in this situation, and the value of the environmental damage would be equal to the costs to restore the water environment. Only when the costs are found prohibitive does the Directive allow for a relaxation of its goals, and costs benefit analysis of action is necessary. Derogations confer a new set of rights under the WFD, which ultimately have consideration for the extent of the total costs of restoration. In this situation, Pearce et al., (2006) identify the tax-payer as the rights holder, to the effect that they have a right not to have their taxes used in contexts where the costs of restoration are regarded as being, in some sense, disproportionate to benefits. We will explore the consequences in further detail in the next chapter.

2.2.4.2. Uncertainty

In a world of certainty, where all the required information is available, the economic analysis of water use would be quite straightforward, as it would come down to the application of many of the principles previously introduced in this chapter. However, in reality, the application of economic thinking to environmental policies is full of uncertainties that inevitably affect their optimal design and analysis (Pindyck, 2006). This section aims to briefly identify some of the various kinds of uncertainties that may affect the accuracy of the economic analysis of water use for the WFD; many of these will be analysed in further depth in later chapters of this thesis.

To summarize, the economic analysis under the WFD requires the consideration of the costs and benefits associated with use to be applied for: i) the use of specific economic instruments to support application of cost recovery for the implementation of sustainable water pricing policies; ii) the considerations relating to the impact of water policies on the water using and water polluting sectors; and iii) the assessment of disproportionate costs in order to justify exemptions to achieve GS at particular water bodies.

The first complication is that the shape of the cost and benefit functions is unknown. Furthermore, these functions are more likely to be non-linear, which would imply that the damage caused by any form of water pollution does not constantly increase with the level of pollution. This applies to the assessment of costs of measures and disproportionality under the WFD, raising the question of where would be the point at which exemptions should be granted. The Directive chooses to be precautionary in the use of CBA. It assumes that benefits of action are more likely to be underestimated in relation to the associated costs and points out that any decisions about derogations should not be simply taken when costs outweigh benefits and rather, when there is a large difference.

There is also uncertainty about the underlying physical and ecological processes, which, in light of the WFD, raise the question of irreversibilities and long time horizons associated with the restoration of the water environment and ecosystems. This is ultimately relevant for the assessment of the effectiveness of measures to improve water quality/quantity. As an example, even though remediation measures to tackle diffuse pollution may be in place, it is almost impossible to predict in practice how some measures will improve the quality of water. Nutrients may have been retained in the soil for generations and even if the activity is stopped, it is possible that nutrients may still leach to the nearest watercourse for many years to come. The same applies to restoration measures; a river which reaches GS is more likely to be able to sustain fish life, as the Directive defends. However, this does not imply that by cleaning up the river and reaching GS (after all the subsequent efforts), the river will be able to sustain fish life again, as the ecosystems which were sustaining fish before may have been damaged permanently.

The restoration of aquatic ecosystems may be subject to long time horizons. This issue also raises uncertainties about the methodology used for the estimation of cost and benefits for policy/options appraisal in water economics. Uncertainties about the discount rates to use in the analysis, the full economic impacts of environmental change and the rate of technological progress (e.g. which technology will be available in the future that might ameliorate the economic impacts and/or reduce the cost of abatement pollution in the first place) apply to the estimation of costs for the CEA and for both, the benefit and cost sides for the disproportionality analysis.

Finally, the economic analysis of water use is highly data dependant. Socio-economic descriptors and water-use data are not often available at river basin level and often have to be transformed or estimated from national figures with the aid of software packages. This adds a level of uncertainty in assessing current levels of cost recovery for the design of water pricing policies. For example, Mysiak and Sigel (2004) tested the accuracy of data transformation for the economic analysis of water use in the White Elster River basin in Germany. They discovered high levels of uncertainty when predicting demographic development and when restructuring national population data at the river basin district level.

In general, the WFD deals with uncertainty allowing for future iterations in its implementation process. This means that the implementation of the Directive is a learning process which should evolve conforming more is known about how best to achieve its targets. By 2015 the objectives set in the river basin management plans should be met, but the first implementation cycle does not conclude until 2021 and the following one is not due until 2027. By then, it is expected that some of the issues identified here will be dealt with and further investigated during the implementation process. In the meantime, it is extremely important that uncertainties in the economic analysis are identified and understood.

2.2.4.3. Transaction costs

One important aspect in the implementation process of the WFD, which has been so far overlooked in the economic analysis, is the role of transaction costs. Stavins (1996) defined transaction costs as inputs of resources or the difference between the buying and selling price of a commodity. In other words, when there are transfers of any property right, parties in the exchanges have to find one another, communicate and exchange information, which incurs costs. These are denominated transaction costs (Nguyen and Shortle, 2006).

These definitions of transaction costs are applicable to water quality trading schemes. The implementation of the Directive does not necessarily imply the creation of water markets but under its rule, transaction costs have a different definition but the same essence, which is that any additional costs associated with its implementation, should be borne by the water users.

In the absence of water markets, under the WFD, the definition of transaction costs compels all the costs to be incurred by the river basin competent authorities in enforcing and regulating the implementation process of the Directive. Some examples of these costs are: monitoring performance, enforcing compliance or establishing, implementing and revising the instrument employed to achieve the objectives of the Directive.This also includes the design, implementation and running up of pricing policies or the development of the economic analysis. The sustainability of water use could not be achieved if the financial sustainability of the competent water authorities is not also assured. Transaction costs are often overlooked in economic impact studies of the Directive and only used in practice, at government level, to finance the competent authorities. Nevertheless, independently of the regulator’s funding mechanisms effectively, these costs should be borne by the polluters and their estimation should be included in the economic analysis.

2.3. Conclusion

This chapter has introduced the economic articles and principles of the Water Framework Directive. We have attempted to contrast the economic provisions of the Directive with conventional water economics theory, in order to provide a clarification of their meaning, applicability and implications.

The potential for the use of economics in water resources management and policy design is considerable. The WFD indirectly recognizes the economic dimension of water through the prescription of a series of applications to the economic analysis of water use; such as pricing policies, the use of economic criteria for the selection of measures to achieve GS and to allow exemptions. This represents a step forward in the quest to rationalise water use in Europe.

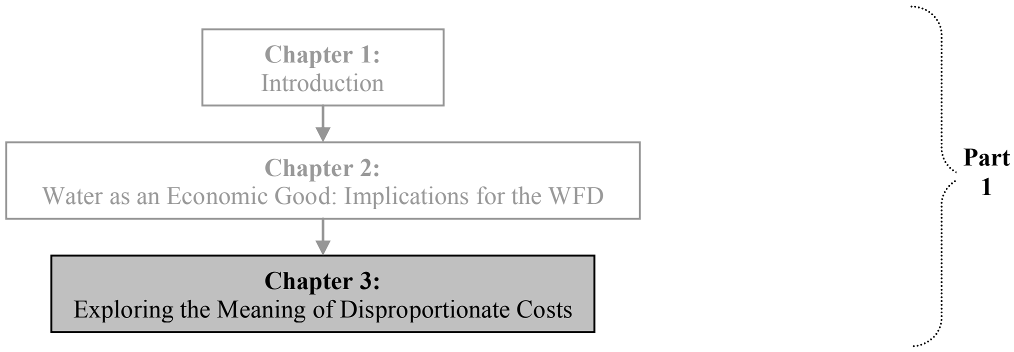
In practical terms, the challenge is enormous. The Directive and the subsequent economic guidance on economics have poorly assessed the practical implications of the economic interpretation of water use. This means that many of the economic aspects of the WFD are at risk of being overlooked or poorly applied, as a large amount of their interpretation has been left to the discretion of each member state. This fundamentally stresses the importance of further research on the economic theory and practicalities of implementation for each of the Directive's economic elements. The following chapter (chapter 3) is focused on exploring in greater detail the economic interpretation of disproportionality and considers a practical application to the agricultural sector.

CHAPTER 3

EXPLORING THE MEANING OF DISPROPORTIONATE COSTS FOR THE PRACTICAL IMPLEMENTATION OF THE WATER FRAMEWORK DIRECTIVE

Part 1 of this thesis (figure 3.1) is focused on introducing the main research topic and setting the overall theoretical background that would be applied in the subsequent empirical chapters. Chapter 2 reviewed the role of the WFD's economic elements and their overall implications for water resources management. Some of the main issues concerning the economic analysis of water use under the WFD were identified. In this chapter 3, we now focus on a detailed exploration of the assessment of exemptions, which is the main topic of this thesis.

Figure 3.1 Thesis structure – Part 1



The WFD consolidates existing water-related legislation and has the stated objective of delivering good status (GS) for Europe's surface waters and groundwaters. But meeting GS is cost dependent, and in some water bodies pollution abatements costs may be high or judged as disproportionate. The exact definition and assessment of disproportionate costs is central for the justification of time-frame derogations and/or lowering the environmental objectives (standards) for compliance at a water body. Official guidance is somewhat discretionary about the interpretation of disproportionate costs. Building on cost-benefit theory, this chapter clarifies the meaning of disproportionate costs, and conveys a consistent interpretation that should underlie the development of a practical derogation decision making methodology.

3.1. Introduction

The two main objectives of the WFD are (i) to restore good ecological and chemical status for all water bodies across the Community by 2015 and (ii) to integrate water management activities at the river basin level. To this end, Member States have identified river basin districts and designated the competent administrative authorities. The next step is to produce River Basin Management Plans, which is an ongoing process until 2009. The implementation of these management plans will then take place in three phases: 2009-2015, 2015-2021 and 2021-2027 (European Commission, 2000).

There is much to debate about the design and interpretation of the WFD, not least its economic underpinning and whether the Directive can be shown to confer net benefits. Irrespective of its aggregate economic efficiency, there is a question about how the designation of an ecological target translates into costs and benefits within different river basins. The incidence of costs is of particular interest to stakeholders with some industries inevitably being more implicated in the drive to cut pollution. This eventuality was foreseen in the design of the Directive, with a provision for conceding exemptions for the achievement of these objectives; such as to grant time-frame derogations to achieve Good Status, or permitting the lowering of environmental standards (from good status to good potential) when a water user finds the total costs of the most cost-effective programme of measures too expensive or disproportionately expensive to undertake (European Commission, 2000). Inevitably this provision is being invoked by some industries, with ensuing debate about the legitimacy of exemptions being claimed on this basis.

Existing guidance on the topic of disproportionality does not offer clear advice to implementing states on the definition of disproportional costs. The case is nominally to be decided by individual member states on a case-by-case basis. The European guidance states that its assessment has to be the outcome of a political decision informed by the economic analysis (European Commission, 2002a). However, this guidance only vaguely recommends the use of simple financial criteria for time-frame derogations and the application of cost-benefit analysis theory for seeking less stringent objectives.

The inevitable outcome is different definitions being applied across water bodies between different Member States. Accordingly, this chapter focuses on the economic interpretation of the meaning of disproportionate costs for the practical implementation of the Water Framework Directive (WFD). We consider the implications for the agricultural sector. The chapter has been structured as follows. In the first section we set the question of disproportionality in the context of the basic economics of pollution control theory and the equi-marginal value principle. The next section considers the definition of Good Status and alternative definitions of disproportional costs consistent with cost-effectiveness and cost benefit analysis principles. The final section reflects on the implications for a hypothetical farm, where theoretical exactitude may ultimately come second to a practical definition that regulators can employ quickly and practically when deciding on whether costs are disproportionate.

3.2. The WFD and the economics of pollution control

While the Directive has clear ecological objectives, for many their attainment is set in terms that are fundamentally economic. Costs of use, cost recovery, the recognition of the need to value benefits… are a few examples of the key elements of the Directive that emphasise the economic attributes of water use. But from the outset, there has been diverging views about the extent to which economic theory can be reconciled with administrative realities and limited regulatory capacity in many Member States. In this instance, economic theory does provide a useful reference point.

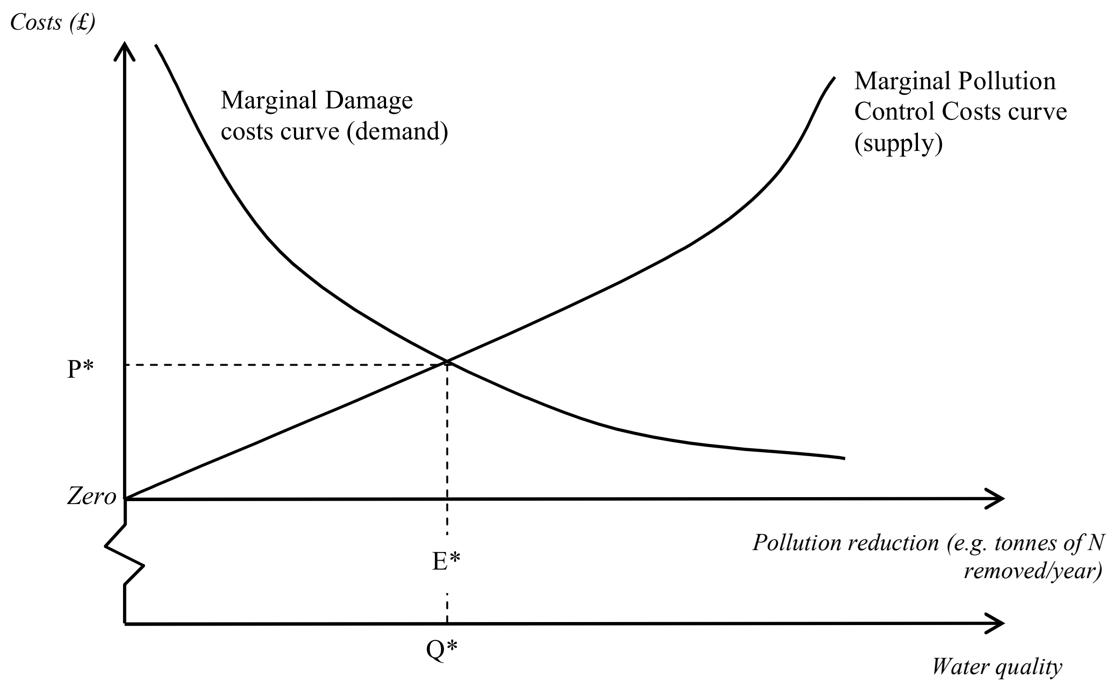
From a neo-classical welfare economics perspective, environmental degradation is depicted as a situation in which the activity of an economic agent (any economic agent: households, firms, governments) imposes external costs upon the rest of society in the form of pollution (Baumol and Oates, 1988). This damage may be mediated through for example pollution of a water body such as a lake. This is the perfect example of a market failure. Prices, or the lack of them, fail to produce an efficient allocation of resources, leaving polluters free use of the environment beyond its assimilative capacity. Pollution is then analysed, from an economic perspective, as a negative externality. Parties who suffer the consequences of the polluting activity experience a loss of welfare or utility (Pearce, 1974). Conversely, society benefits from mitigation or restoration programmes.

For the design of environmental protection policies, economists aim to find the appropriate set of prices to be paid by the polluter to compensate for the negative impact of their activities, in an attempt to internalise any loss of welfare that the "victims" of this activity may have suffered. The overall objective is to create competitive markets for the use of environmental assets, as they (in theory) would produce (in equilibrium) a pareto-efficient allocation of resources, where no economic agent (polluter or the victim) would be worse off as a result of any actions taken, implying no loss of welfare (Varian, 2003).

For the last 40 years, economists have applied the principles of microeconomic and welfare economic theory in the advocacy of efficient pollution control policies, with the underlying objective of using economic instruments to find the economically optimal level of pollution (Baumol and Oates, 1988). These instruments are designed to provide the necessary financial incentives for polluters to reduce the environmental degradation associated with their activities in order to achieve a (so called) “socially” desired environmental objective (Hanley et al., 1997). Some examples of these instruments are: pollution taxes/charges (piguvian taxes), pollution reduction subsidies, tradable emission permits.

Figure 3.2 introduces the basic economics of pollution control (adapted from Pearce and Turner, 1990; Varian, 2003 and ECO2, 2004). To simplify the analysis, the graph depicts one single factory which discharges nitrogen loads into the nearest river (one polluter, one water body), resulting in environmental damage.

Figure 3.2 The basic economics of pollution control



The figure depicts the marginal cost curves for pollution control and damage costs. The diagram mirrors economic demand and supply theory. The curve for pollution control (supply side) reflects the increasing abatement/private costs that the company may incur in order to reduce its nitrogen emissions into the river by one extra unit[6]. And the damage curve reflects the avoided (environmental and social) costs (demand side) associated with that environmental improvement. In other words, the higher the pollution levels, the more people (or society) are willing to pay for unit reductions. Assuming a direct dose-response relationship between the firm's output, environmental protection expenditure and environmental quality improvements, increasing pollution control means (in the graph) that damage costs decline (conversely the environmental/social benefits increase) meanwhile the firm's control costs go up. Alternatively, low pollution control costs imply higher damage costs.

In theory, if both curves are known, any policy responses based on this information would result in an efficient allocation of pollution control and the value Q* would represent the "socially" desired level of water quality/pollution, equivalent (under the assumptions made) to point E*, which illustrates the pareto-efficient level of control of pollution/emissions. These points can be found on the X-axis where the firm's marginal abatement costs, MAC, equal marginal social cost, MSC (Varian, 2003). As an example of the many applications of this 'equi-marginal value' theorem, the point P* can be used to set pollution charges (or piguvian taxes), assuming that the pollution control costs curve represents private costs of remediation measures for the firm (MAC) and the environmental damage costs curve represents social costs (MSC) under perfect competition (Hanley et al., 1997).

3.3. Application to WFD

The same basic framework can be used throughout to illustrate the economics of water resources pollution control applied to the implementation of the WFD. These concepts ultimately allow us to clarify disproportionate cost. Assume that we are dealing with nitrogen emissions of one single firm/polluter altering the water quality of a river. We begin the analysis by introducing the definition of Good Status, the environmental target of the Directive. This is followed by a graphic representation of Cost Effectiveness Analysis (CEA)

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[6] Note the important difference in economics between total and marginal costs. Marginal cost indicates the

and the firm's financial efforts to achieve the environmental requirements of the WFD. This leads to a more complete consideration of the role of Cost-Benefit Analysis (CBA).

3.3.1. Defining 'Good Status'

The definition of 'good status' as the objective of the WFD is clearly the driver of much of the subsequent cost analysis underlying the implementation of the Directive. A fixed ecological standard implies a degree of inflexibility in implementation, which in some circumstances will imply that costs of compliance can exceed benefits. The ability to modify or seek derogation from compliance means that ecological rigour has to be balanced against economic criteria. Effectively, the standard-setting process will determine whether the uptake of measures to reduce environmental pollution will be enough to achieve 'good status' by 2015, and if not, what will be the gap between the actual levels of water pollution and the target standard.

Subject to annex V of the Directive, each member state is required to define Good Status in terms of those environmental standards that will help to support the biology of the water environment. In Britain, The UK-TAG[7] is currently engaged in the definition of Good Status (including the design of the environmental standards and the development of the classification schemes) and has recently published for consultation the 1st phase of their programme: "*UK environmental standards and conditions*" (UK-TAG 2006).

As biological parameters are the key component of the definition of good status, the standards are being defined according to the relevant status class boundaries (high, good, moderate, poor and bad) that compare to different levels of biological quality elements (e.g. covering algae, fish plants, etc…) for the different types of surface water bodies (e.g. rivers, lakes…). In consequence and following the Directive's definitions, the UK-TAG is designing (or updating in case of existing legislation) the following environmental standards for the water quality of rivers in the UK (see table 3.1). The table also describes how different standards are being designed.

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change in costs as we consider reducing one more unit of pollution.

[7] The United Kingdom Technical Advisory Group (UKTAG), group created to provide advice on the technical/scientific side of the implementation of the Directive, is a partnership of the UK environment and conservation agencies. http://www.wfduk.org/

Table 3.1 Environmental conditions, types and design of standards for rivers in the UK under the WFD

Environmental condition	Type of standard	Standards Design
I) General Water quality (Ecological status class boundaries: High, Good, Moderate, Poor and Bad)		
General physico-chemical quality elements	Biological Oxygen Demand (BOD) and dissolved oxygen demand (DOD), Ammonia pH Nutrient: Phosphorus and other (not defined yet) Temperature (not defined yet) Salinity (not defined yet)	Use of numeric values that have been referenced to ecology
Water flow and water levels	Change from natural flow conditions	Numeric values supported by hydrological modelling, based upon the best available understanding of links to ecology
Morphological quality elements	Type and degree of physical alteration (physical structure and condition of the bed, banks and shores)	Development of a decision framework based on best available knowledge supported by numeric thresholds
II) Chemical pollutants (Chemical status class boundaries: Good and Not Good)		
Toxic pollutants (called specific pollutants)	Standards for pollutants discharged in significant quantities	- Priority substances, Environmental Quality Standards (EQSs) design at European level - Dangerous substances: listed annex IX WFD

Source: (UK-TAG, 2006)

The designed standards will be for the whole of the UK (and fully compliant with the WFD requirements and other Directives). The approach to their implementation will be administration-specific, depending on different existing and proposed legislative and policy regimes, for each country within the UK (e.g. the ways in which abstraction is controlled in England & Wales, Northern Ireland and Scotland are different). For the first river basin cycle (to be ready by 2015), where knowledge on the actual status of the water environment is more limited, these standards are being designed based on best currently available knowledge for managing the water environment. For later stages of the river basin planning cycle, to start after 2015, the standard-setting process will be subjected to scientific review.

These standards will be used to develop the classification schemes, as for example, each river in the UK will be assigned to one of five ecological status classes (high, good, moderate, poor and bad) or in the case of failing to meet them, to one of the five ecological potential classes (maximum, good, moderate, poor or bad). Additionally, there will be two surface water chemical status classes (Good and Not Good). The "*one out-all out*" principle will decide their quality status; determined by the worst quality element, in the case of good ecological status, or the worst chemical element in reference to good chemical status. Furthermore, a surface water body will be classified also as "not good" if the standards for one or more priority substances (standards to be agreed at EU level) or dangerous substances (list Annex IX Directive) are exceeded.

3.3.1.1. Standard-setting process: Implications for agriculture

These environmental standards will be used throughout the UK for the identification of generic (national level) and specific (catchment/water body level) relevant water management issues, which will come as a result of the classification exercise. In consequence, standards will be used to assess the gap between current water pollution levels and the desired objective, good status, and to provide the basis for a decision-framework to manage negative pressures to the water environment. These standards will influence the development/review of existing charging regimes (e.g. permit limits for nutrients emission, effluent charges for water abstraction) or any other regulatory tools required for the control of polluting activities and to monitor, relative to the standard, the effectiveness of any measures used to improve the water environment. In this section, we outline the possible implications of the WFD standard-setting process for the agricultural sector.

As introduced above, the UK-TAG is currently working on the different components of the definition of good status. To date, only proposals to support Good Ecological Status, mainly concentrated on the boundary between Good and Moderate Quality Status, have been subject to consultation (Phase 1 of the Programme "UK Environmental standards and conditions"). At this stage, it is very difficult to assess the likely impacts that the proposed standards will have for the agriculture sector in the UK, as regulatory agencies are obviously waiting until all the standards have been designed, to specify the rules by which these standards will be used to take decisions at river basin management level. However, it is possible to draw some conclusions from the comparison between proposed and current standards.

Table 3.2 illustrates the extent of river length in different areas of the UK that may be reported as worse than good as a result of the proposed standards (water quality standards for Biological Oxygen Demand (BOD), Dissolved Oxygen (DO), Ammonia and Phosphorus (P) levels in rivers). Values are compared with the results from current classification schemes. Overall, the proposed environmental standards to reach good ecological status, for the substances mentioned above, are slightly more stringent than current standards (in nine of the cases the standards have been risen, in six have been lowered and only in one case they remain the same). As for example, in Scotland, compliance with the directive would require tougher controls for dissolved oxygen, ammonia and phosphorus, but controls in BOD may be relaxed.

Table 3.2 Implications of existing thresholds and proposed standards

	Type of standard (Per cent of river length reported as less than good)							
	BOD		**Dissolved oxygen**		**Ammonia**		**Phosphorus**	
	Existing	*Proposed*	*Existing*	*Proposed*	*Existing*	*Proposed*	*Existing*	*Proposed*
England	25.6	18.7	30.8	24.6	14.6	17.3	65.0	63.3
Wales	3.7	3.7	2.4	4.1	1.4	2.7	11.4	12.8
Scotland	8.2	7.6	7.5	8.9	7	10.7	13.6	14.1
Northern Ireland	19.0	16.3	27.8	37.2	4.4	16.3	22.2	17.0

Source: (UK-TAG, 2006)

Overall, there is only a marginal difference between the proposed standards and existing ones. This may lead us to conclude that the achievement of the environmental objectives of the Directive would not bring dramatic changes in the way water resources have been managed to date. As standards remain more or less the same, the possible consequences for water operators may be in line with current water quality standards. However, the implementation of the WFD implies some reconsideration of how water is managed and it is expected that the overall situation will change with the introduction of new regulatory schemes and a sense of direction in terms of the environmental objectives to be achieved (Good Status). The table above implies that either current regulations are not working or that some pressures to the water environment have not been regulated for (e.g. case of diffuse pollution). This has major implications for agriculture, as the Directive is bringing new operating rules and controls to tackle diffuse sources of water pollution.

3.3.2. The Cost-Effectiveness Analysis (CEA)

The application of the polluter pays principle (PPP) to the WFD asks water users to pay for the environmental and social damage associated with their negative impact on the water environment (European Comission, 2000). This covers the issue of property rights and basically represents the end of the "free lunches" era for private and institutionalised water resources management in Europe. For the first time, users have been asked to pay for the full costs of using the resource, including environmental and social costs, to assure that water resources are being sustainably managed (introduction of sustainable water pricing policies, for example, justification for volumetric charge for water abstraction in Scotland).

CEA is an optimisation method for finding the lowest-costs means to reach an objective (Tietenberg, 1992). In the context of the WFD, the objective of the analysis is to achieve the desired environmental standards (Good Status) at the lowest possible costs to society as a whole. The prescription of the use of economic instruments, such as Cost Effectiveness Analysis for the selection of measures to achieve good status, is aimed at ensuring efficiency in policy/action design and to avoid unnecessary economic and financial costs. However, CEA does have limitations.

Figure 3.3 The Cost-Effectiveness Analysis of water quality improvements options

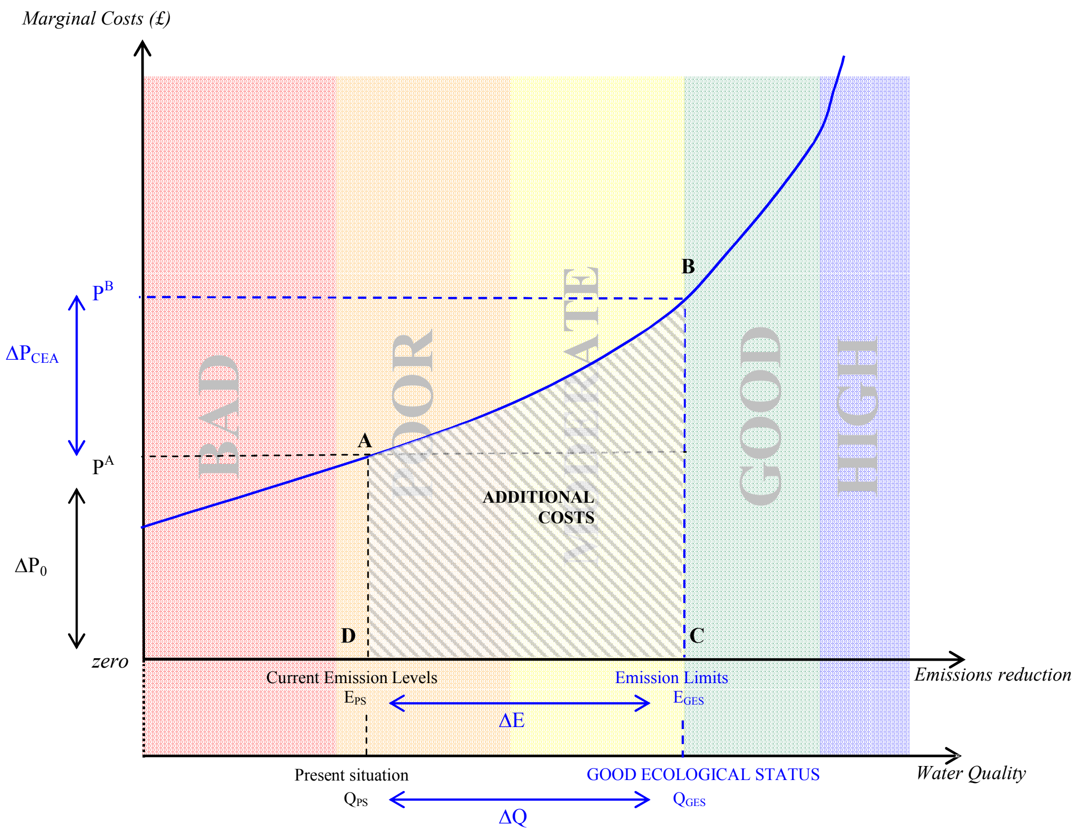


Figure 3.3 shows a graphical interpretation of CEA for our hypothetical firm, assuming that the MAC curve is defined on the lowest cost set of options available to the firm to reduce its emissions to water and a direct cause-effect relationship between the implementation of these options and water quality improvements. Accordingly, nitrogen emission reductions are shown on the horizontal axis, costs are shown on the vertical axis and the background reflects (for a case water body) the ecological status[8] class boundaries under the WFD (bad, poor, moderate, good and high), options are ranked in increasing order of their costs per emission reduction unit.

The overall objective of the CEA is to minimise the incremental marginal costs of pollution control for the firm (*min* ΔP_{CEA}) whilst achieving water quality improvements to at least the point where the desired water quality levels are achieved (Q_{GES}). In figure 3.3, ΔP_{CEA} is the difference between the future (hypothetical) marginal costs of remediation measures (P^B) and the marginal costs of current practices (P^A), which may well be zero if there are no remediation measures already in place. Additionally, change in water quality (ΔQ) is derived by estimating the extent of water quality improvements needed to achieve a specified environmental objective, Q_{GES} is the desired water quality situation, minus baseline water quality levels (Q_{PS}).

As this is an analysis at the margin, the area underneath the MAC curve is a total magnitude and it can be measured/estimated (Chiang, 1984). In consequence, the scale of the additional compliance costs to reach GES for the firm under the WFD (additional environmental protection costs excluding extra regulation costs) is represented by the area formed by the points ABCD (ΔP_{CEA} in figure 3.3).

As long as good ecological status is achieved (Q_{GES}), the objective is to find the set of remediation measures that would minimise this area. The extent of the total costs of compliance with the Directive would depend on the water quality improvements (level of standards) needed to reach GES and where the emission limits are set (E_{GES}) by regulators to reach these objectives[9]. Note that this analysis is described without reference to benefits other than the prescribed level of good water status.

[8] Good status is the combination of good ecological status (GES) and good chemical status (GCS), for simplicity we now focus our analysis in the achievement of good ecological status

[9] For this analysis, we are assuming a direct relationship between water quality and emission reductions. We imply that $E_{GES} = Q_{GES}$

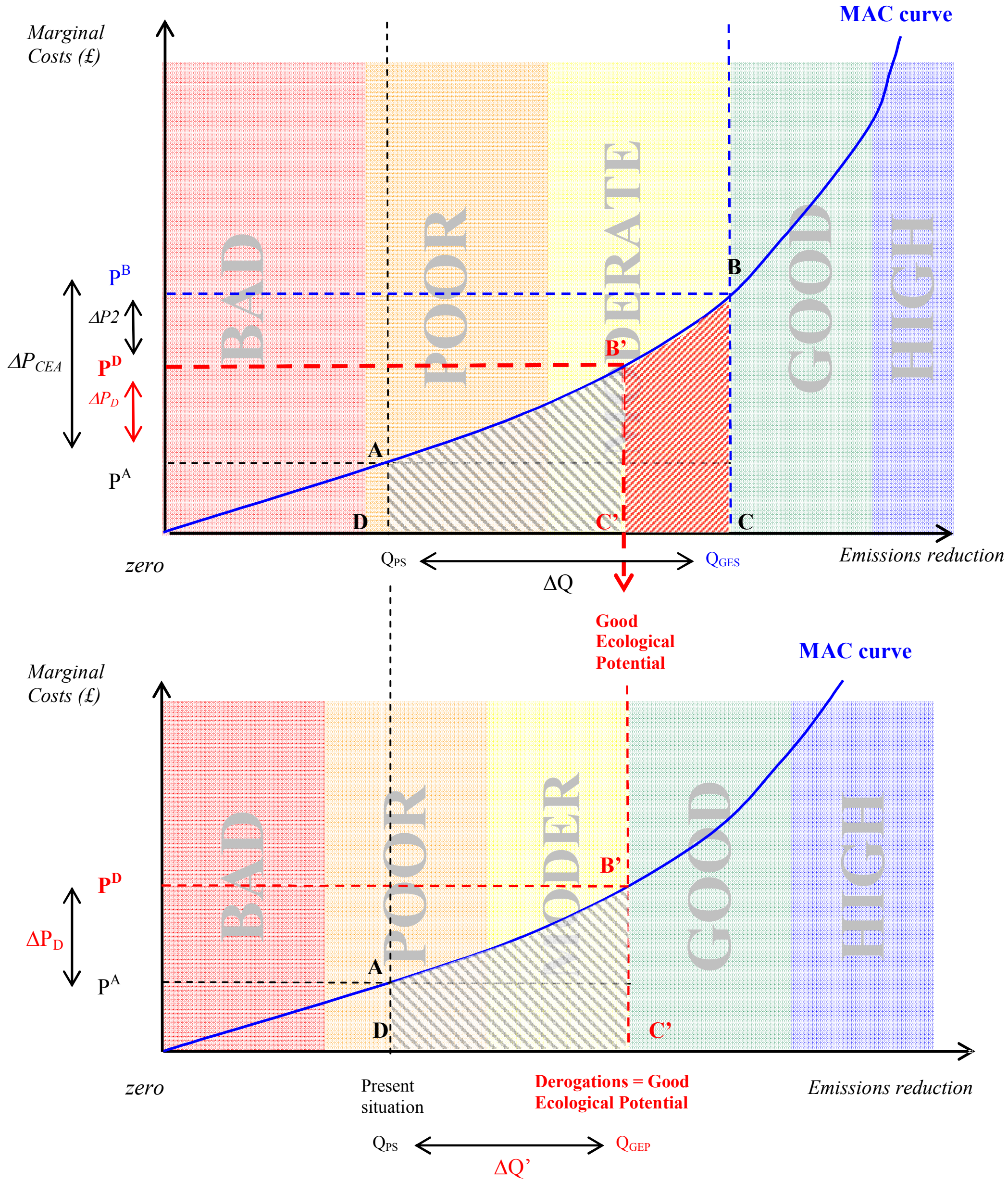
3.3.3. *Disproportionate costs: a first interpretation*

Consider now figure 3.4 in which the same firm finds it too costly to reach GES and claims that it can only afford to abate to the point P^D (Y–axis). This point represents the firm's maximum compliance effort with the Directive.

Under the WFD, this situation leaves the firm with two possible options. First, the firm may either seek to be granted time-framed derogations/exemptions. This would allow the firm to wait until new abatement technologies are available, which can reduce its overall marginal costs of compliance, and for the regulators there would be no need to lower the environmental standards. This means introducing flexibility in the speed of implementation, which the Directive already accounts for by allowing for different phases on the implementation of the River Basin Management Plans (2009-2015, 2015-2021 and 2021-2027). Alternatively, the firm may have a case to seek less stringent environmental objectives, and this would be represented at the point where P^D=MAC (B').

If the standard-setting derogation was allowed in this hypothetical case, based solely on the estimation of the point P^D, the additional costs for the firm would be represented by the Area AB'C'D (figure 3.4), and *good ecological potential* (GEP) could be found in theory by drawing a vertical line to the X-Axis, where P^D=MAC. The lower graph in figure 3.4 shows the situation under the new environmental objectives, as Ecological Potential would have different water quality status class boundaries to Ecological Status. The other conclusion is that the difference between P^D and P^B (i.e. the difference in the additional costs of achieving Good Ecological Status and Good Ecological Potential) represents *ΔP2* (area B'BCC', in figure 3.4) illustrating one interpretation of Disproportionate Costs. This situation could imply a re-design of the environmental standards for the specific water body and/or lowering the emissions limit previously set for the firm/water body.

Figure 3.4 Graphical representation of Disproportionate Costs



3.4. Assessment of disproportionality in theory

The estimation of the point P^D may prove sufficient to justify time-frame derogations based on an assessment of the economic viability of the firm[10]. This may be the simplest interpretation of disproportionality, but one which is based on cost-effectiveness alone. CEA is an optimisation tool but it does not provide optimal/efficient solutions as a whole. It does not try to maximise utility for all the economic agents involved, but to reach an objective at least costs for the firm. Arguably, this interpretation of the Directive is incomplete.

Ultimately, the change of objectives (from GES to GEP) needs to be socially justified under the WFD. As suggested by pollution control theory, a social optimal considers more than just abatement costs; it is necessary to consider the full range of social and environmental damage costs associated with the firm's polluting activities[11]. These costs in turn mirror the benefits derived from reducing pollution. In other words, as pollution is reduced in a water body, there is a notional function reflecting the increasing social benefits derived from whatever uses are made of the river.

This aspect is considered in the marginal social cost (MSC) curve (see both figures 3.5 and 3.6). This curve reflects a decrease in damage costs to society (or conversely reflects the social benefit). Initial low cost abatement delivers high social costs, which progressively fall as the firm's pollution control costs increase by one extra unit. In economic theory, a pareto-efficient level of pollution control (Q*) is found where MSC=MAC, and the optimal pollution control expenditure needed to internalise the damage produced by the firm should be set at P* (see figure 3.2).

Due to the uncertainties surrounding the monetary estimation of the damage costs functions, which are mainly associated with the economic valuation of environmental improvements[12], regulators normally set the standards using other criteria. In this case, GS is defined as a function of those environmental standards necessary to support the biology of the water environment, and therefore regulators have to presume that these standards would reflect to some extent society's demand for environmental quality – assuming the shape of the MSC

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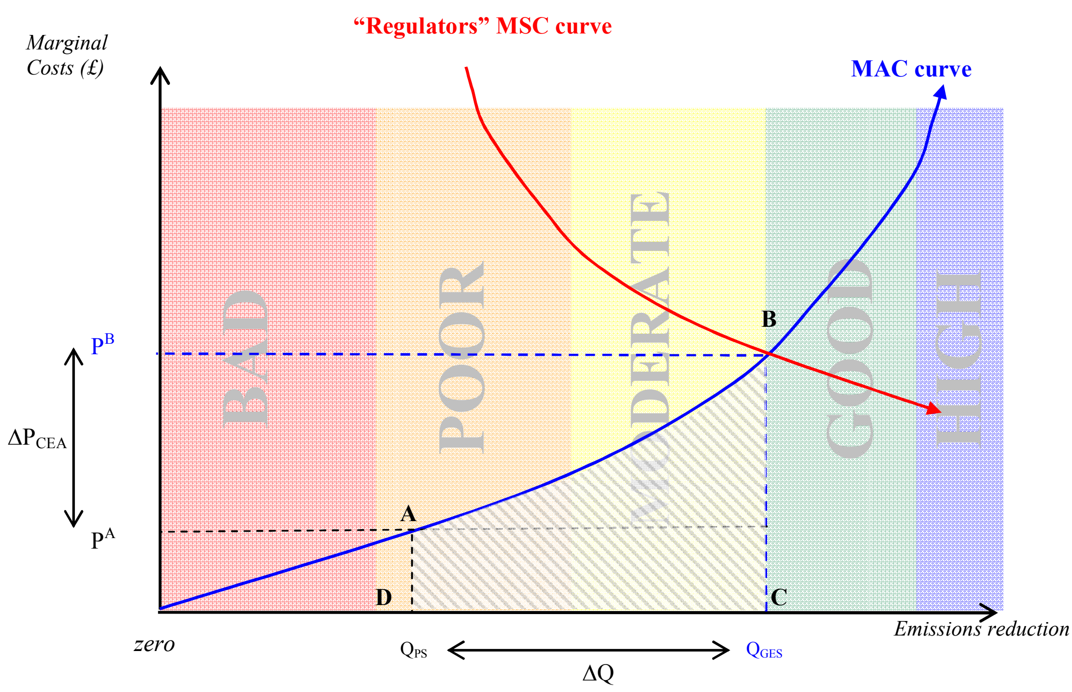
[10] For practical purposes, these decision making steps would be similar to those used in the determination of Best Available Techniques (BAT) and the determination of BAT based permits conditions within Directive 96/61/EC concerning integrated pollution prevention control.

[11] Note that this remains true even if the uses are passive or non use "existence" benefits.

[12] More information on the contested issue of the use of environmental valuation in decision-making can be found in the following report (Ecologic, 2005).

curve (figure 3.5). Regulators assume the shape of the MSC curve by drawing the MSC line anywhere as long as this curve cuts the MAC curve where the desired water quality levels are found (point B in figure 3.5)

Figure 3.5 The Cost-Benefit Analysis, assuming the shape of the benefits function



However, figure 3.6 shows the economic inefficiency of the standards-based system when the "real" MSC is introduced. In this hypothetical situation[13], the area BB'F (figure 3.6) represents the net loss to society as a whole, including the firm, of reducing pollution to Q_{GES} instead of Q*, which represents the "socially" desired level of water quality/pollution control. This introduces an economic justification for the firm to seek the lowering of environmental standards, and for the regulators to at least consider the claims on this basis. In this context then, disproportional should ideally be judged with reference to cost and benefit curves, and therefore an application of Cost-Benefit Analysis (CBA). CBA is a decision-making tool which is explicitly highlighted for the assessment of exemptions in the

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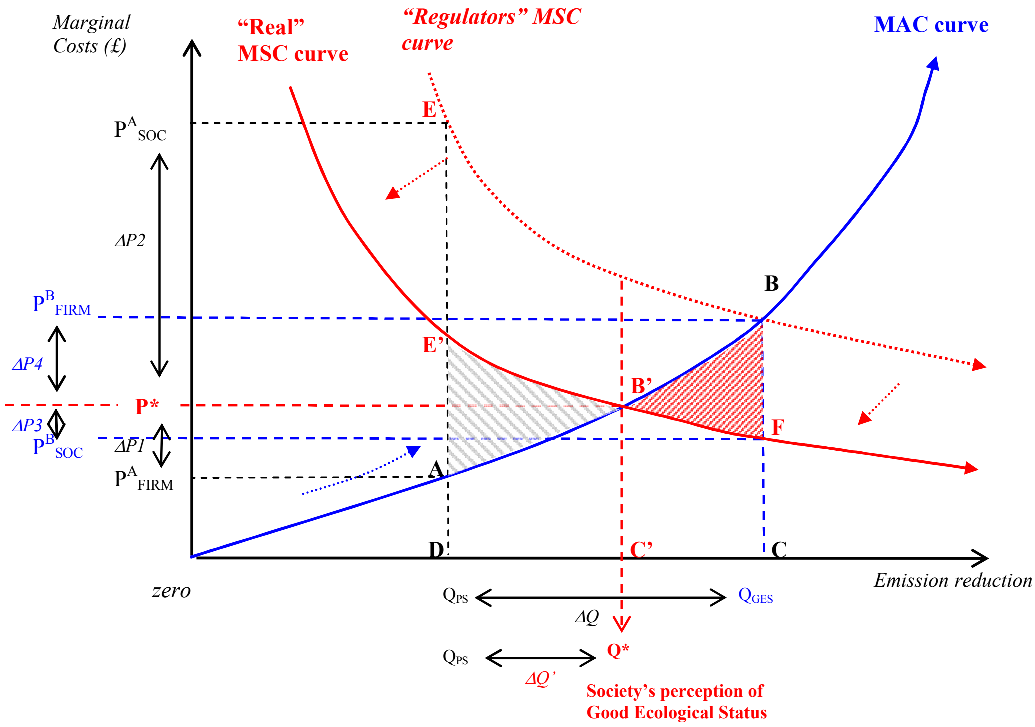
[13] Note that for this analysis the "real" MSC curve has been drawn below the "regulators" MSC curve to show the economic inefficiencies associated with assuming the shape of the benefits curve. However, the "real" MSC curve could be plotted anywhere in the graph or have any other shape. It may even be the case that society's perception of GES surpasses that of the scientific assessment.

WFD literature (European Commission 2000 and 2002a; RPA 2004; Hanley and Black, 2006a).

Figure 3.6 Economic inefficiencies associated with assuming the shape of the damage costs curve

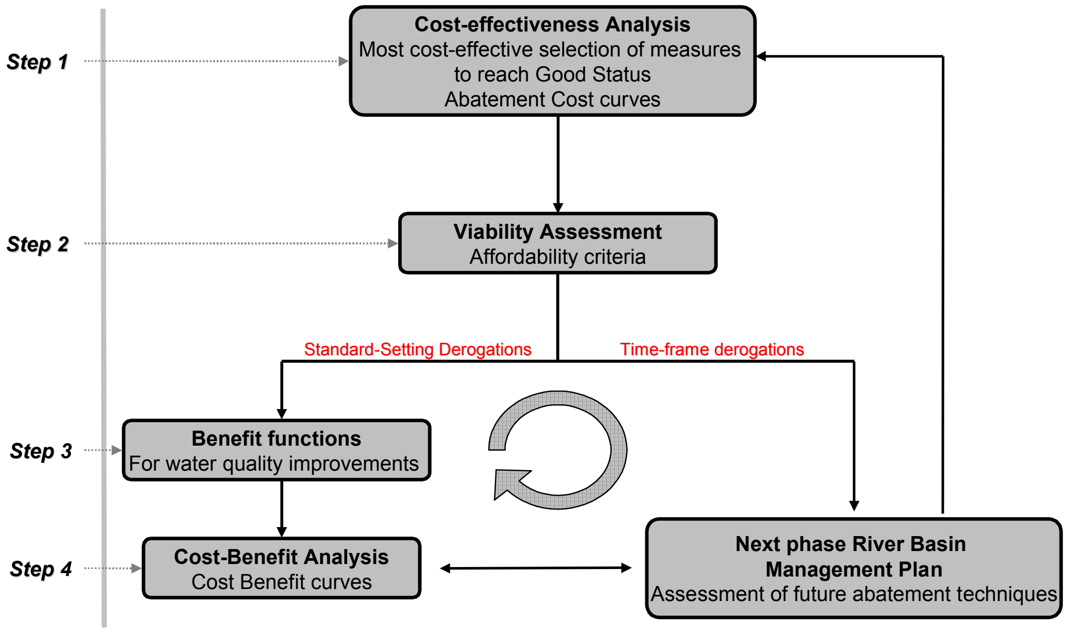


Lowering the environmental standards for a specific water body or allowing a polluter to maintain its emissions levels on the grounds of a disproportionality test, may prove one of the most controversial steps of the WFD implementation process. Decisions may reveal issues of competitiveness between water users or uneven distribution of the financial costs associated with the Directive (Pearce 2004). Applying CBA presents a challenge, but it is a rational model to inform decision making processes (Pearce et al., 2006). Benefit assessment in particular brings up some complex issues related to the process of valuation and the fact that some water bodies are more socially valuable in relative terms. Despite this, political decisions regarding exemptions or derogations to achieve GS should, if possible, be informed by the appraisal of the costs and benefits of options to improve water quality, with the underlying objective of achieving some sort of economic efficiency and coherence in the final decision. If not, decision-makers may face an issue of conflicting rights between those who pay the costs of water quality improvements and those who benefit, as they may have overlooked the extent of the net social costs (area BB'F in figure 3.6) involved in complying with the Directive.

3.5. Assessment of disproportionality in practice

In this thesis, we argue that a rational model to inform decisions on derogations is needed and that economic theory provides a definition of disproportionate costs and the methodological tools that can inform its assessment. Using economic theory, we conjecture that ideally standard-setting derogations should be judged with reference to cost and benefit curves – an application of the CBA method.

Figure 3.7 Methodological steps for the assessment of disproportionate costs



While instructive, the application of theoretical principles to water resource management can be constrained by the realities of data and administrative capacity. A major stumbling block in the theoretical story is whether sufficient reliable benefits assessment data are available. These constraints are evident across Member States, with differing levels of economic input for supporting decisions. The practical application of the basic principles outlined in this paper presents a challenge. Figure 3.7 offers a guide to the main methodological steps needed for the assessment of exemptions under the WFD. In order to better grasp the concept, we briefly introduce below the implications of using such a model for a hypothetical farm. We aim to answer the following question: what information would be needed to judge if a hypothetical farm should be granted exemptions?

First of all, we are dealing with the simplest possible case. Independently of the types of derogations being sought, a preliminary cost-effectiveness analysis of all available measures to the farmer to reduce water pollution needs to be undertaken as a requirement for the design of programme of measures (PoMs) to reach good status. Once all possible measures have been ranked in terms of their cost-effectiveness, the following information will be available: i) what measures are needed to reach good status; and ii) the extent of the total financial costs of reaching the stated objective. In this case, the use of abatement cost curves proves a valid and transparent management tool to support these types of decisions (Beaumont and Tinch 2004). This method provides an estimate of costs to reach a pre-defined level of abatement, and also reveals the most efficient path to this discharge level

Once information about costs and effectiveness of measures has been collated, this needs to be compared with an assessment of the financial viability of the farm and the ability by the farmer to absorb the additional costs of protecting the water environment. Ultimately, this will determine the farmer's efforts to achieve good status at particular water bodies (Lago et al., 2006). The use of financial indicators or income ratios provides a good option for assessing the costs of meeting the environmental requirements of the Directive at individual and sectoral level (DEFRA 2006a). However, there is a need to distinguish between ability to pay and affordability. This distinction is more subjective and controversial.

If the viability assessment indicates that the application of the most cost-effective selection of measures to achieve good status places an unreasonable burden on farm incomes, regulators will then need to apply derogation tests, which will differ depending on the type of derogations being sought.

For time-frame derogations, regulators can base their decision on the outcomes of the tests introduced above. In practice, this would basically involve doing nothing until the beginning of the next river basin management cycle. This fundamentally means just waiting until there are new abatement techniques available to reduce the farmer's costs of compliance. Essentially, there would not be a need to lower the environmental standards however, an appraisal of future pollution abatement options may prove useful at this stage. Once this is done, the whole cycle needs to be repeated for the next river management cycle – beginning again with CEA.

For standard-setting derogations, the analysis becomes more complex. The costs of reducing pollution at farm level need to be compared with the associated benefits of water quality improvements. The main rationale of applying benefit assessment of environmental quality improvement is that the lowering of the environmental standards needs to be: i) socially justifiable under the WFD; and ii) following economic theory, the optimal point of pollution control (where costs equal benefits) is the only point when a satisfactory outcome for both, society and the farmer can be found. As we have introduced in this paper, the rationale for the application of CBA to justify standard-setting derogations is to achieve economic efficiency in the exemptions decision making process.

There are evidently many uncertainties associated with this analysis, which are beyond the scope of this chapter and that will be analyse in further detail in Part 2 of this thesis. For example, in addition to the obvious challenges in benefits assessment, questions remain about the technical effectiveness of measures or best management practices to control diffuse pollution, and the attribution of responsibility to individual farmers. These uncertainties are the subject of extensive ongoing research in Member States.

3.6. Discussion

Overall, the Water Framework Directive sets a clear course of action for many of its key elements, including most of its economic components. For example, to achieve *Good Status* and to reinforce the *Polluter-Pays Principle* and the *Cost Recovery Principle*, the WFD introduces a set of legal transposition requirements. These oblige each member state to incorporate the Directive into their national law (e.g. case of the Water Environment Water Services (2003) Act in Scotland) and develop regulatory instruments for its enforcement (e.g case of the Controlled Activities Regulations (2005) in Scotland). The establishment of pricing policies and the Programme of Measures for water pollution control and reduction options are also normative and the failure to meet the objectives of the Directive punishable.

However, for the assessment of derogations, the lack of official EU guidance on the use of CBA clearly stands out compared with the prescribed choice of CEA for the selection of measures to achieve Good Status. This raises a question as to whether the objectives of the Directive are set and enforceable, and about the role of CBA in European water policy.

Ultimately the predefined objectives of the WFD, are independent of the costs of achieving them, as these goals do not acknowledge public preferences and are completely independent of elicited human values (as they are set by the regulator). This has been called the "public-trust" doctrine, which makes the goal of environmental policy in face of pollution, the restoration of the pre-damage state of the environment (Pearce 2002). Under the WFD, "*Good Status*" reflects a legal judgement about the role of the Commission as a trustee of citizen's rights for environmental improvements. In this instance, the achievement of Good Status does not need to be justified, the benefits of action do not need to be estimated, and the value of the damage would be equal to the costs to restore the water environment. Consequently, the application of CEA to the selection of measures will suffice to reach Good Status at least costs.

Nevertheless, the Directive "recommends" the use of CBA only to allow for a relaxation of its goals when costs are found prohibitive. This differs from the normal use of CBA in policy analysis, where it is used widely to justify policy choices. This clearly introduces discrepancies between the structure and ethos of the WFD and its implementation strategy.

When applying CBA for the assessment of individual/sectoral cases of disproportionality, member states may discover that they are implementing and enforcing a highly inefficient piece of legislation. If the costs of action outweigh the overall environmental benefits of the Directive, the question remains: is the Directive worth implementing? This is a dangerous road to take and definitely, an application of CBA not encouraged in the text of the WFD.

3.7. Conclusion

The WFD (European Commission, 2000) and subsequent guidance documents on the interpretation of its economic elements (European Commission, 2002) provide limited guidance on the meaning of disproportionate costs for the justification of exemptions in the achievement of Good Status. This chapter shows that economic theory provides a definition and the methodological tools that can inform its assessment.

Ideally disproportionate costs should be judged with reference to cost and benefit curves. But the pursuit of CBA opens the Directive to wider interrogation that questions its overall economic efficiency.

Cost-Effectivenes Analysis alone provides a partial tool to justify derogations. But the decision-making tools used for the assessment of disproportionality under the WFD should vary depending on the nature of the derogation being sought. These tools mainly differ in the use/non use of benefit curves. For time-frame derogations, simple decisions can be based on an economic viability test of the firm, compared with the financial costs of the most cost-effective set of measures available to reach GS (outcome of the CEA). For the justification of derogations on the basis of less stringent objectives, it would also be necessary to know what gains in environmental quality can be achieved compared to the abatement costs – a full economic costs approach (Cost-Benefit Analysis) – to reach a "socially" optimal decision. If both the MAC and MSC curves are known, any policy responses based on this information would result in an efficient allocation of pollution control.

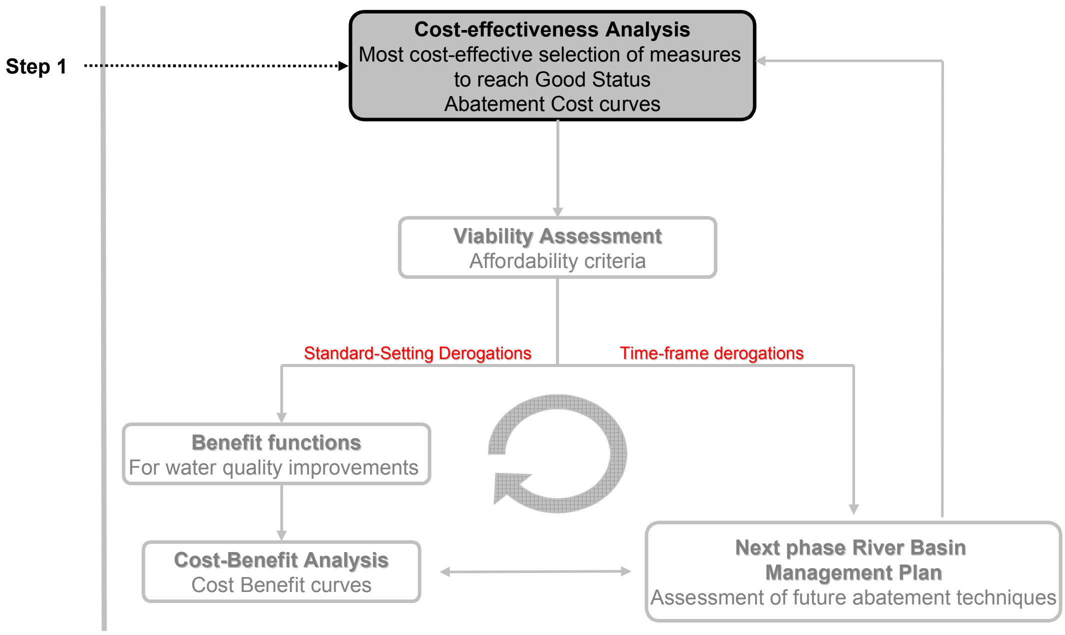
Overall, this chapter offers a welfare economics interpretation of disproportionate costs. It discusses what the theoretical meaning of "disproportionate" should be and how it should be translated into practice. Note that CEA generally just tell us the least cost ways or options to meet an objective. But the implementation of these options could be highly inefficient in the first place. This could be possibly the case in the WFD, as there is no overall cost-benefit proof and eventually, it will be needed to look at benefits. This thesis does so later. The following chapters of this thesis explore individual applications of different components of this interpretation of disproportionality (e.g. selection of measures, costs and benefits of remediation and implication on financial viability) and discusses the implications of applying such tests in policy analysis.

CHAPTER 4

ABATEMENT COST CURVES: METHODOLOGY TO DEVELOP COST FUNCTIONS FOR AGRICULTURAL BEST MANAGEMENT PRACTICES

The Water Framework Directive (WFD) has the stated objective of delivering Good Status (GS) for Europe's surface waters. But meeting GS is cost dependent and in some cases costs may be high or judged as disproportionate. The definition and assessment of disproportionate costs (DC) is relevant for the justification of time-frame derogations and/or lowering the environmental objectives (standards) when, in some individual cases, the most cost-effective Programme of Measures (PoMs) to achieve GS is unfeasible or found to be too costly.

Figure 4.1 Methodological steps for the assessment of disproportionate costs - focusing on the development of abatement costs curves



The overall aim of this thesis is to improve understanding of disproportionality and the development of a practical methodology to assist in the decision-making process of granting exemptions for the implementation of the Directive. Previous theoretical work has concluded that disproportionality should be judged with reference to cost and benefit curves of abatement options in conjunction with an economic viability test of the firm. Following the theoretical principles introduced in Chapter 3, the first practical step of the definition of disproportionality requires estimation of the financial costs associated with the implementation of the most cost-effective combination of measures available to reach Good Status (figure 4.1).

This chapter introduces a practical application to the Scottish agricultural sector. The adoption of Best Management Practices (BMPs) will be a critical component of PoMs to achieve GS in Scotland, especially relevant for the control of agricultural diffuse pollution. Agricultural BMPs, which are a set of measures or combination of measures for preventing agricultural non-point source pollution, are regarded as the most effective and practicable means of controlling farm nonpoint pollutants in order to reach good water quality standards (Hilliard et al, 2002; USEPA, 2007). Nevertheless, as a preliminary assessment of disproportionality, it is essential to understand the economic implications of adopting different mitigation strategies. In this context, we have developed a detailed assessment of the financial costs and effectiveness of BMPs to reduce farm losses of main pollutants using the Abatement Cost Curve methodology. The aim is to assess the extent of the financial costs associated with the introduction of BMPs at farm level in Scotland and appraise the suitability of economic methods to consider their cost-effectiveness.

The results offer a basis to grant time-frame derogations under the WFD and provide an indication for the selection of the most cost-effective combination of BMPs for different farm systems. Arguably, this framework could be used to assess the level of farm support to deliver water quality objectives under the new land management contracts in Scotland. Important methodological challenges remain, in particular with regard to some of the assumptions employed. For the justification of standard-setting derogations, further work needs to include the economic estimation of the environmental benefits derived from abatement options.

4.1. Introduction

In Scotland, diffuse pollution currently results in up to 23% of the water bodies being at risk of not achieving the environmental objectives of the Directive (SEPA, 2005a,b). Most of this pollution comes from the agricultural sector, which has been identified as the dominant diffuse pollution pressure affecting the water quality of rivers (SEPA, 2005a,b). The application of BMPs at farm level is seen as the most effective way for controlling diffuse water pollution from agricultural sources. However, the WFD calls for the implementation of the most cost-effective means to reach good status, making the assessment of their associated costs an imperative. The choice of BMPs and the appraisal of their cost-effectiveness will be a critical component of PoMs to achieve good status in Scotland, and will play a crucial role in the ability of the agricultural sector to cope with the WFD requirements.

This chapter begins by introducing a mathematical interpretation of the cost minimisation problem, which ultimately represents the application of the cost-effectiveness analysis (CEA) for the selection of BMPs at farm level. This is followed by a review of the methods available in the academic literature to solve this problem, which comprise mainly of the abatement cost method and dynamic optimisation modelling. The application, in this particular case, of a non-parametric method has been chosen mainly due to lack of information on the relevant bio-physical information required to run farm level optimisation models of options to reduce nutrient losses and the need to relax the underlying farm modelling assumption of profit maximisation, which as it is argued in this chapter, contradicts current evidence regarding farmers uptake of BMPs.

This chapter explores the implications, limitations and possible applications of using the abatement cost curve method to estimate the extent of the financial costs associated with achieving different levels of nutrient loads reductions at farm level through the implementation of BMPs. We then propose the following methodological steps in order to apply the abatement cost method: i) Identification of farm sources of diffuse pollution; ii) A literature review of available BMP abatement techniques for different farm systems; iii) Data sources and manipulation; iv) Cost-effectiveness ratios for the selection of measures; and v) Development of abatement cost curves.

4.2. The cost minimisation problem

Under the WFD, there is a need to carry out a CEA for the selection of measures to reach good water status. In respect to the agricultural sector and the role that Best Management Practices may play in reducing diffuse sources of pollution, the objective of the CEA would be to find the most cost-efficient selection of measures to reduce farm nutrient losses. The application of CEA represents the typical cost minimisation problem in economics (Tietemberg, 1992), as we need to find the cost function that minimises costs for the achievement of a selected/given level of output or as in this case, certain nutrient loads reductions. In this study, Nitrogen (N), Phosphorus (P) and Faecal Indicator Organism (FIO) load reductions at farm level are considered. Consequently, the objective function is to minimise costs as illustrated in equation 1 (this mathematical representation of the cost minimisation theorem follows that of Van der Veeren, 2002):

$$Min(C(P,N,FIO)) = Min\sum_i c_i x_i \tag{1}$$

Subject to the following constraints:

$$\begin{aligned} &\sum_i P_i x_i \geq R_p P_0 \\ &\sum_i N_i x_i \geq R_N N_0 \\ &\sum_i FIO_i x_i \geq R_{FIO} FIO_0 \\ &0 \leq x_i \leq 1 \end{aligned} \tag{2}$$

where; x_i relates to the abatement activity implemented (this is represented as a fraction with a value between 0 and 1, as shown in the last restriction). Each activity may target all, two or only one pollutant at the same time; c_i denotes the costs related to the implementation of nutrient abatement activity x_i (£/ha/year); P_i, N_i *and* FIO_i is the nutrient abatement associated with the implementation of activity x_i (reduction in kg of nutrient loss per hectare). The constraints in equations (2) state that the total nutrient abatement at source level should be more than or equal to a certain fraction *R* of the initial emission levels described in P_0, N_o *or* FIO_o. By changing this restriction on nutrient-emission reductions between 0 and 100%, it is possible to assign % reductions in relation to baseline levels in each individual measure.

4.2.1. Solving the cost minimisation problem

In general, two very different approaches can be used to solve this problem at farm level whist offering an indication of the total cost of abatement for the farmer; namely, the ad-hoc method and dynamic optimisation (Beaumont and Tinch, 2004). Both these methods focus upon technological detail and the impact upon individual enterprises of reducing sources of pollution. Due to their localised scale and application, these methods are often classed as bottom-up approaches as opposed to other methods which describe the economy wide impacts of abatement costs by top-down measures (e.g. Computable General Equilibrium)

Dynamic optimisation is more preferred amongst academics for the assessment of the cost-effectiveness of different measures/policy options to reduce loads of agricultural diffuse pollutants (Brady, 2003; Johansson, et al., 2004). Mathematical models can generally be divided into two main types depending on if they use a set of linear or non linear equations to solve the optimisation problem (non-linear include techniques such as positive mathematical programming or other "integrated" non-linear programmes). In recent years due to the advancement of computer technology, the field has seen the development of models with increasing levels of sophistication, which are able to depict a more accurate picture of the agricultural diffuse water pollution problem (see for example: Howitt ,1995; Aftab et al., 2003)

In essence, the central idea of environmental modeling is to include the bio-physical damage function in the net benefit function, and the use of welfare economic theory of production and consumption to represent relations between inputs and outputs (Greenberg, 1995). In relation to agricultural water pollution, models describe the complex interactions between economic, agronomic, hydrological systems, and even include the stochastic nature of some factors (e.g. climate, soil, topographic conditions; Aftab et al., 2007). Some models often also include an assessment of the uncertainty related to these stochastic factors as a probability function (Lacroix, et al., 2005).

An essential condition for the economic assessment of the most cost-effective selection of agricultural BMPs to reduce nutrient loads to water using mathematical programming is to link bio-physical models with farmer production functions. The most sophisticated bio-physical models predict crop yields, crop quality, water and nutrient flows in relation to different choices in field and management practices and assess their impact on farmers' production/welfare functions to determine the extent of the costs of abatement. However, the optimisation condition often collides with the definition of Best Management Practices, which can be defined as farming methods that minimize risk to the environment without sacrificing economic productivity (Hilliard et al., 2002), and the necessary conditions for the CEA, which does not seek an optimal solution but instead the least costly and most effective path to reach a water quality target (Lago et al., 2007). Both of these limit the application of optimisation models to our research question, which is the estimation of the total costs of abatement to achieve nutrient loads reduction targets.

Usually, the optimisation problem will find those BMPs that maximise a constrained profit function (e.g. constrained by assumed levels of inputs, outputs, area, etc. see chapter 5) whilst minimising nutrient loads. The first assumption in order to solve this optimisation problem using mathematical models, is that farmers are profit maximisers (Turpin et al., 2005). Often studies also include the assumptions that farmers are fully informed, perfectly competitive and/or risk-neutral or adverse (Johnsen, 1993). Van der Veeren (2002) explains that assuming profit maximisation for the economic analyses of farm measures to reduce water pollution, automatically implies that each measure applied to reduce or increase nutrient emissions would reduce profits. This not only assumes that all farmers are producing efficiently in the first place, but also that if these measures were to increase profits or reduce costs of production BMPs would have already been adopted in the initial situation. This collides with the fact that in practice many BMPs will deliver environmental benefits and at the same time, offer some efficiency savings to the farmer; such as using fertiliser recommendation systems or applying nutrient budgeting techniques. The possible savings available to farmers from following these types of practices have been found to be of around £14 per kg/ha from reduction in the application of N and P for the cereal sector in England and Wales (SAC/University of Cambridge, 2004) or a total average saving for the typical Scottish farm of around £1,500 a year (Frost et al, 2002). In reality, farmers are not fully informed and often practices are chosen not based on profit maximisation but on other factors; such as, the application of traditional farming practices (SAC/University of Cambridge, 2004). In essence, current farm practices contradict traditional firm behaviour theory.

In particular, for the initial equilibrium situation, in which profits are maximised, the assumption of profit maximisation implies that marginal abatement costs have to be zero. This assumption is needed in terms of controlling for the fact that if marginal costs are found to be different from zero in the initial situation, it would be attractive for certain activities to increase or lower their emissions. Both these situations would contradict the assumption that farmers start off from an optimal equilibrium situation. Accordingly, if these costs are based on emissions reductions, the potential benefits to production of increased nutrient emissions are often beyond the scope of calibration of the mathematical model. Furthermore, mathematical models also ignore that if marginal costs are negative in the initial situation, this illustrates that emissions reductions would be beneficial at low levels of abatement. We will establish later in this chapter that this is often the case with the application of BMPs to reduce farm nutrient loads. The main benefit associated with the application of the abatement cost method is that it does not require any initial assumption regarding the optimisation of profits.

4.2.2. The abatement cost curve method

In order to approach our research question, we still need to identify a method which would allow us to investigate the scale of the marginal abatement costs for different types of measures to reduce farm diffuse pollutants without having to assume profit maximisation at the same time. The application of ad hoc methods, which in this case involves the development of abatement cost curves, allows us to answer this question. This method can be used to provide an estimate of the financial costs to reach a required level of abatement, and also to reveal the most efficient route to achieve a defined discharge level or environmental standards. Basically, an application of cost-effectiveness analysis.

Ad-hoc methods are less sophisticated and impressive than dynamic optimisation models, especially if compared with integrated modeling. Their applications are scarce in the academic literature - as these methods mainly do not pose much of a challenge for researchers, they are very data intensive and they only account for uncertainties in a qualitative manner. Nevertheless, there are a few exceptions. Some examples of the application of this method can be found in the air quality literature - e.g. to assess the impact of implementing air emissions reduction strategies- but only a handful can be found on water quality (see Beaumont and Tinch, 2004).

This method does not come without its limitations; costs and effectiveness of options to reduce pollution are assessed separately, as opposed to optimisation which allows the inclusion of many variables into the function to be optimised. The method does not provide optimal solutions which limits its applicability at policy level (this would ultimately depend on the objective of the environmental policy to be assessed). Because of the large amounts of data needed, studies are often very speculative (location unspecific) and normally applied to the average business/firm of a determined economic sector which would be affected by the regulations. Abatement cost curves are mainly focused on the estimation of the financial costs to the polluter in order to reach an environmental target. Wider economic effects are difficult to include and studies often have to rely on transferring estimates of economic costs and effectiveness levels, which decrease the accuracy and relevance of the results.

Otherwise, the application of the abatement cost method to water quality problems is very popular in the grey literature (e.g. in governmental research reports for policy impact evaluation; MMA, 2002, Dutch Ministry of Water, 2005) and its use has been widely recommended by member states for the economic analyses of water quality improvements under the WFD; examples include: EU guidance (European Commission, 2002), UK (RPA,

2003) or Denmark (Jacobsen, 2007). The main reason for its popularity may well be that this method allows the ready evaluation of costs and the impact upon individual firms, aiding in the negotiation process between the government, regulators and industries when new environmental quality targets are introduced (Jung et al., 1996). Additionally, this method has been proven as a good management tool to identify win-win situations in light of environmental regulations. Beaumont and Tinch (2004) proved the validity of abatement cost curves to assess the extent of efficiency savings for industries in the Humber estuary (England), when the most cost-effective and efficient selection of methods to reduce copper discharges were applied in order to comply with water quality standards.

Following the theoretical principles introduced in Chapter 3 (figure 3.1), the first step in the definition of disporpotionality requires gauging the extent of the financial costs associated with the implementation of the most cost-effective combination of measures available to reach Good Status. The remainder of this chapter explores the implications, limitations and possible applications of using the abatement cost curve method to estimate the extent of the financial costs associated with achieving different levels of nutrient loads reductions at farm level through the implementation of Best Management Practices.

4.3. Methodology

An integrated stepwise methodology was applied, as adapted from Beaumont and Tinch (2004). This consists of the following steps:

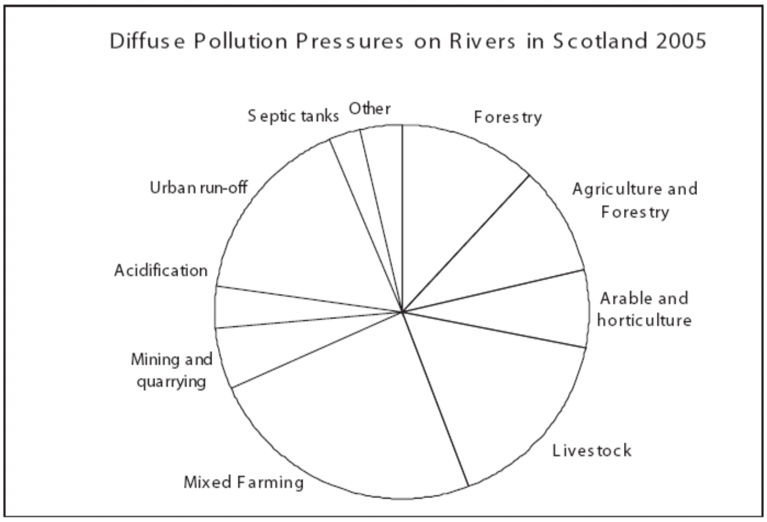
1. Identification of farm sources of diffuse pollution
2. A literature review of available BMPs abatement techniques for different farm systems
3. Identification of data sources and data manipulation
4. Development of cost-effectiveness ratios for the selection of measures
5. Development of abatement cost curves

4.3.1. Identification of farm sources of diffuse pollution to running waters

Diffuse pollution can be defined as *"Pollution arising from land-use activities (both urban and rural) that are dispersed across a catchment, or subcatchment, and do not arise as a process effluent, municipal sewage effluent, or an effluent discharge from farm buildings "* (D'Arcy, 2001). This definition includes any non-point source contamination (such as sheet run-off from fields), and contamination from many dispersed, often individually minor, point sources. Examples of these are field drains and urban and other surface water discharges. Diffuse pollutants include nutrients (such as nitrates and phosphates), faecal indicators, pesticides, metals and hydrocarbons from sources such as agriculture, industry and recreation (SEPA, 2005a). Alternatively, another definition regards pollution sources as "diffuse" if they are of a nature such that traditional monitored discharge consents cannot be applied (Frost et al., 2002).

It is estimated that diffuse pollution currently results in up to 23% of the water bodies in Scotland being at risk of not achieving good ecological status, and it is now a more significant source of pollution than point sources in most water bodies (SEPA, 2005a,b). SEPA has identified agriculture as a significant cause of diffuse water pollution in Scotland, and clearly, as the dominant diffuse pollution pressure affecting rivers. Figure 4.2 (below) shows the relative pressure per industry sector for diffuse pollution, placing rivers at risk of failing to meet water quality standards in Scotland. Agricultural related activities account for around 60% of all diffuse pollution pressures to Scottish rivers.

Figure 4.2 Diffuse pollution pressures on rivers in Scotland *by sector*



From: Morris et al., 2006

Farm sources of diffuse pollution in Scotland are well documented (Vinten et al, 2005). Frost et al, (2002) carried out an assessment of the environmental impacts of current agricultural management practices. This study surveyed sources of pollution deriving from six farms in Scotland - hill sheep, upland stock, mixed stock and arable, dairy, general arable and intensive arable with vegetable production. In terms of their impact on water resources, the study concluded that there is a strong relationship between water pollution and poor farm management practices. The following examples of sources of diffuse pollution were found: poor pesticide handling and usage; inappropriate fertiliser and manure handling and usage, including significant excesses of phosphates; soil erosion, which although unlikely to damage the productivity of the soil, was a source of pollution of surface water; inadequate water margin buffer strips; and run-off from farm steadings (Frost et al., 2002).

The resulting impacts of these practices to water quality is that nutrients such as nitrogen (N) and phosphorus (P), which are present in fertilisers, can contribute to the process of eutrophication, whereby increased levels of algae consume the available oxygen, lowering the ability of the water to support aquatic life and making it unsuitable for other potential users. Organic substances, such as milk and manure, can also directly affect oxygen levels when they find their way into water, with subsequent impacts on aquatic life. Pollution from bacteria in animal manures can cause human health concerns either directly (ingestion when swimming) or indirectly (through contamination of fish/shellfish). Additionally, soil erosion can also transport phosphorus and pesticides which are attached to soil particles to watercourses (Scottish Executive, 2005). Table 4.1 illustrates a detailed description of the main environmental impacts resulting from specific agricultural diffuse pollutants and management practices.

Table 4.1 Diffuse Pollution concerns from UK agriculture

Pollutant	*Example Sources*	*Impacts*
Farm chemicals including; • Oils & Hydrocarbons • Pesticides • Veterinary medicines	Machinery maintenance, accidents, spill storage disposal. Yard run-off in beef and dairy systems. Use of pesticides on arable, fodder crops and horticultural crops. Use of sheep dips and other medicines on livestock to stop spread of disease and maintain high levels of animal husbandry.	Toxicity to aquatic species. Contamination of streams sediments Groundwater contamination. Nuisance (surface water). Taste and toxicity (drinking water quality). Contamination of potable supplies
Silt	Run-off from arable land. Upland erosion by sheep farming. Lowland erosion by outdoor pig systems, dairy beef and beef herds.	Destruction of gravel riffles. Loss of breeding and rearing sites. Sedimentation of natural pools and ponds. Costs to abstractors (e.g. fish farms, drinking water supplies).
Organic wastes	Agricultural wastes - slurry, silage liquor and surplus crops - dairy and beef herds. High production of waste products from intensive livestock units. Waster for land application.	Lowering of oxygen levels. Toxicity to aquatic life. Loss of habitats. Nutrient enrichment.
Faecal pathogens	Application of livestock organic wastes to farmland. Use of dirty water for irrigation. Non separation of dirty and clean water.	Health risks to humans. Non compliance with recreational bathing water and shellfish standards.
Nitrogen	Agriculture fertilisers -used for horticulture, improved grassland, fodder and arable crops. Atmospheric deposition - emission of ammonia from livestock	Eutrophication Contamination of drinking water supplies (rivers and groundwaters) Accelerated plant growth Changes in community structures Acidification
Phosphorus	Agricultural fertilisers - addition to fodder, improved horticulture and arable crops. Soil erosion - loss of phosphorus combined with soil particles from livestock and arable farming activities	Eutrophication Changes in community structures Accelerated plant growth Increased filtration of potable supplies

Modified From; DEFRA, 2002

4.3.2. Literature review of available BMPs abatement techniques for different farm systems

As introduced above, there is a fundamental need in Scotland to reduce water pollution by diffuse sources from agriculture in order to achieve the objectives of the WFD. However, this poses two separate challenges: i) the need for further research; the identification and development of suitable technical and technological methods for reducing losses from agriculture, and ii) acceptability; to take measures or apply instruments which will encourage farmers to adopt these methods (ADAS, 2002).

In theory, a best management practice (BMP) is a measure or combination of measures for preventing or reducing agricultural non-point source pollution. The definition often includes economic considerations. The US EPA classifies agricultural BMPs as the most effective and practicable (including technological, economic, and institutional considerations) means of controlling agricultural nonpoint pollutants at levels compatible with environmental objectives (USEPA, 2007). An alternative definition describes BMPs as a farming method that minimizes risk to the environment without sacrificing economic productivity (Hilliard et al, 2002).

Based on these definitions, BMPs are seen as the most effective way for controlling water pollution from agricultural diffuse sources. However, the WFD calls for the implementation of the most cost-effective means to reach good status, making the assessment of the extent of their associated costs an imperative. Therefore, the choice of BMPs based on their economic appraisal will be a critical component of PoMs to achieve good status in Scotland. Additionally, implementation of mitigation strategies at farm level are expected to play a crucial role in the ability of the agricultural sector to cope with the WFD requirements. The overall objective of this chapter is to assess the extent of the financial costs associated with the introduction of BMPs at farm level in Scotland and appraise the suitability of economic methods to consider their cost-effectiveness.

The remainder of this section has been divided as follows; first, we provide a brief description of the different stages at which BMPs can remove diffuse pollutants at farm level. This will help us understand when nutrients are no longer a farm input/output and become an environmental problem and how BMPs work in practice. This is followed by a literature review of available studies which contain information about BMPs, their associated costs and effectiveness; in order to assess their suitability for a CEA analysis, and the development of cost curves.

4.3.2.1. How BMPs work at farm level

Agricultural diffuse pollutants are difficult to mitigate at farm level due to the complex interactions between their functional behaviour and the associated processes and pathways that take place before watercourses are reached (Chadwick et al, 2006). Three general ways have been identified for the control of water diffuse pollutants at farm level, pollutants may be targeted at the source, mobilisation and delivery stages of their flux pathways (IGER/ADAS, 2005). These controls can be applied by the farmer: as planning/general farm measures, in-field, field margin or riparian, in-stream, and steading measures (Vinten et al, 2005).

Source controls target the available quantity of potential pollutants which are vulnerable to mobilisation and delivery to water. In this instance, the amount of pollutant source can be measured at the source area - this is the place of production or mobilisation. Among source controls it is important to distinguish; i) external sources, which are the primary sources of potential pollutants that are not notionally imported across the farm gate (i.e. mineral fertilisers, feedstuffs, etc); ii) Recycled sources, which make reference to those secondary sources of pollution that are generated internally within a farm as part of the production system (these cover for example; farm manures, slurries and dirty water, excreta from grazing animals) and iii) internal sources; these are tertiary sources of pollution which are generated within the soil profile (such as: decomposition of organic materials, soil formation, etc).

The second main control affects the mobilisation of diffuse pollutants. These controls target the efficiency of the different mechanisms by which pollutants enter watercourses from agricultural land. Mobilisation controls are divided into three main categories: i) Solubilitation, which is the chemical and biological release of potential pollutants from soil sources into soil water; ii) Detachment, making reference to the removal of soil particles; and iii) Contingent control, which targets any accidental mobilisation which may be generated by management failures (such as slurry store leaking directly into watercourses) and also targets any incidental mobilisation of pollutants, described by the processes that occur when rain and run-off interact directly with fresh applications of pollutant to the soil surface (including when codes of good practice have been applied).

Finally, delivery controls target the run-off of diffuse pollutants from agricultural land by changing the relative importance or reducing the efficiencies of the different pathways that water can take between the farm land and a receiving water body. Delivery controls can be separated, in terms of the type of hydrological flow targeted, into: i) surface flow, movement of water over the soil surface which is due to either infiltration or saturation excess, allows surface run-off to make a rapid and direct connection to a water body; ii) preferential flow, which describes the rapid vertical movement of water in the soil, it occurs via cracks, macro-pores or artificial drainage channels; and iii) Through flow, making reference to the opposite, slow, vertical and lateral movements that can occur in the soil (such as vertical drainage to the groundwater table).

In general, BMPs would aim to reduce the amount of pollutants which could potentially escape the farm. Some practical examples: i) by reducing inputs of products such as fertilisers and pesticides to land and reducing pollution risks by changing the pattern/timing of use; ii) by maximising retention of these products within the field and the crops to which they are applied; iii) by slowing or reducing transportation into watercourses; iv) management of soils to reduce capping and compaction and increase the organic content; and v) by changing land use to less intensive systems or lower risk purposes.

4.3.2.2. Review of BMPs studies

The academic literature covering information about BMPs is extensive (Shepherd et al, 2006). There are numerous scientific studies which assess the effectiveness of individual options at farm level (e.g. installation of buffer strips or ponds for nutrient retention/removal) or the mitigation potential of options to remove specific diffuse pollutants (such as Nitrates or Total Phosphorus). See for example; Yin et al., (2006) and Heal et al., (2006). On the contrary, studies which also present detailed information about their associated costs are more rare, and mostly belong to the grey literature.

A non-exhaustive literature review of studies that assessed BMPs was conducted in order to identify possible measures and data sources to include in our analysis. Overall, their individual suitability for a cost effectiveness analysis was assessed. Table 4.2 provides a summary with the findings of our investigation, which had the following search criteria: i) The main criteria for the selection is that detailed information about agricultural BMPs to reduce farm diffuse pollution to water was presented, alongside an assessment (qualitative or quantitative) of costs and effectiveness of the mitigation options; ii) even though our main concern was the review of Scottish studies, it was considered that the list should also include studies from other locations in order to compare different practices in the collection of information of agricultural mitigation measures; iii) this review was open to all types of studies, either of an academic nature or grey literature (reports and documents..); and iv) units presented for costs and effects of each measure were assessed to check their suitability for a CEA for the selection of measures. Detailed information about each specific study is available in the annexes at the end of this thesis (see Annex I).

Some of the main conclusions which would be relevant for the cost effectiveness analysis of agricultural BMPs, relate to the interactions between the following issues: scale of the analysis, types and methods used for costing the measures, description and ways to quantify the amount of reductions and assessments of uncertainty.

Results of effectiveness are location specific, varying if the analysis is performed at water body, river basin and country level. This is due to the complex processes that affect diffuse pollutants on their pathway from the farm to watercourses. All of these studies offer a more or less detailed description of both measure (what behaviour is changed) and the mechanism (what it is actually done) required in order to calculate effect and costs. Different mechanisms affect the interaction between measures, however, none of the studies have assessed quantitatively how these interactions may affect the combined effect of measures.

Levels of effectiveness can be calculated either by estimating the effect at the edge of the farm or by calculating their effect at the edge of a river basin/water body. The scope of the analysis also influences the estimation of costs; for example, associated economic costs (or alternatively, non-water related costs) are usually calculated if the study in question covers other sectors as well as agriculture. At farm level, the geographical position of measures within the farm might change both effect and costs. This is not normally included.

Table 4.2 Literature review of BMPs studies

Reference /year	Scope	Suitability for CEA
Dickson et al (2005)	Determine to what extent improved farm practices and BMPs could contribute to improve water quality in a small rural cathment in Scotland.	Bad. Lack of cost data (no other types of costs than capital investment), measures of effectiveness in different units.
Frost et al, (2002)	identification and assessment of BMPs to reduce the impact of agricultural environmental management. Not only focussed in water quality. Other environmental media covered: land and air.	Poor, different cost categories used (difficult aggregation), no quantitative assessment of effectiveness.
McTaggart et al, (2002)	Regulatory Impact Assessment (of proposed options) for the Action Programme regulations of Nitrate Vulnerable Zones in Scotland. Estimation of costs and benefits for the measures contained in each of the options for each farming sector within each NVZ, together with an assessment of their relative importance for reductions in leaching of nitrates	Good but only covers BMPs for Nitrogen. Difficult to disaggregate cost information
IGER/ADAS, 2007	User manual that presents costs-effectiveness estimates of integrated diffuse mitigation measures at farm level. Latest output of an on-going DEFRA research programme to mitigate farm pollution. Summarises the results of many projects	Very good. Effectiveness and baseline levels estimates in same units. Possible to calculate % reductions for each measure.
ENTEC/ADAS, 2006	Benchmark costings under the WFD leading to a cost proforma for use by governmental departments or any other agencies involved in the implementation process of the Directive in the UK. Part of the Collaborative Research project on economics of the WFD	Good. But excel database does not provide links between effectiveness and baseline levels.
SAC/ University of Cambridge, 2004	Increase understanding of the potential role that voluntary and regulatory instruments may have on reducing farm loads of N and P. Method: data envelopment analysis (modelling technique) to find out efficiency savings in nutrient input for different agricultural sectors	N/A

DEFRA, HM treasury Concultation, 2004	Consultation document on approaches and possible measures to improve water quality through catchment sensitive farming	Bad. Qualitative assessment of diffuse sources of pollution.
RPA, 2003	new cost estimates for the agricultural sector to update Regulatory Impact Assessment of introducing the WFD in England and Wales	Poor. good indication of costs. This work was updated in ADAS, 2007
ADAS, 2002	appraise the nature and effectiveness of approaches taken in other countries to minimise diffuse pollution of water from agriculture and the policy options available to control the problem	Poor. Good source of information about BMPs and international experiences in their application
Dutch Ministry of Water, 2005	Economic frameworks for a consistent application of the cost-effectiveness analysis under the WFD across the different regions in Holland. Uses examples/case studies. Different sectors/pressures analysed.	Good. Only covers four measures for one agricultural sector (pig farms).
Ecologic, 2003	general approach for the selection of the most cost-effective combination of measures to achieve the objectives of the Directive. Uses examples/case studies. Different sectors/pressures analysed.	Poor. Different units costs estimates. Difficult to make comparisons for CEA. Effect of some measures only assessed qualitatively
Ministerio del Medio Ambiente, 2002	Application of Cost-effectiveness Analysis for the selection of measures to improve water quality/quantity in the Cidacos river basin (Spain). Different sectors/pressures analysed, including agricultural pressures	Good, this study is an application of CEA for the selection of measures in Spain. Specific for Spanish issues.
Hilliard et al, 2002	literature review of existing BMPs to reduce agricultural diffuse sources of water pollution	Poor. Good source of information, understanding BMPs. It could be used for a qualitative assessment of the cost-effectiveness of measures

The majority of studies reviewed had some degree of inconsistency in the costs and effectiveness units presented. The condition for CEA is that units of costs and effect should be of a comparable magnitude within the measures. Types of costs are not often described. A total cost figure for each measure for the whole agricultural sector or for the whole farm is sometimes presented which makes it impossible to discern between recurring or non-recurring costs. Almost all studies identify measures which offer costs savings. Estimates often do not offer any reference about the lifetime of the measures or the time scale of the estimate, which makes it impossible to calculate net present values. Additionally, cost units are often mixed (e.g. £/farm, £/ha or £/m^3).

Assessments of effectiveness most commonly cover measures that target nitrates, phosphates and FIO. However, different units of effectiveness are used which make it very difficult to compare effects for the same measures between studies. For example, for phosphorus some studies use kg of total P removed or bio-available P removed. Also they use different units of scale (per farm or per ha). Only some studies include some assessment of baseline losses of diffuse pollutants at farm level. This is important for the CEA, as the different levels of effect to reduce loads have to be comparable with baseline losses at farm level for the specific pollutant.

Usually, no systematic assessment of uncertainty is carried out. Only one of the studies reviewed (ENTEC/ADAS, 2006) has assessed the degree of uncertainty that surrounds the effectiveness of measures. This was addressed by presenting lower, average and upper bound ranges of effectiveness, though there is no information about how these ranges were assessed.

In conclusion, the evaluation of costs and environmental benefits of BMPs to reduce diffuse pollution presents many challenges. CEA is easier to apply to some environmental problems than others. This tool has proven a degree of success in dealing with point source pollution issues (see Beaumont and Tinch, 2004). However, its application to the control of agricultural diffuse sources of water pollution still needs further investigation, in the meantime, it is a mistake to assume that diffuse sources would behave as point sources of water pollution. Many factors influence the effectiveness of measures, this is specifically related with the nature of diffuse pollutants: how the pollutants behave, transport and interact need to be realistically included in the analysis. From a methodological perspective, these studies prove that there is a need for standardisation of information for the CEA, not only in the way costs and measures of effect are collected but also, in the way they are presented.

For the transfer of estimates, units of costs and effectiveness have to be easy to disaggregate. There is a need for information in units which are easily adjusted between farm area or hectares, which is relevant for the transfer of estimates between different locations.

4.3.3. Data sources and manipulation

The effectiveness and costs estimates for different BMPs to reduce farm losses of N, P and FIO have been sourced from the following report:

"*An inventory of methods to control diffuse water pollution from agriculture (DWPA) – User Manual*" IGER/ADAS, January 2007. Prepared as part of DEFRA project ES0203

From all the studies reviewed in the previous section and reported in table 4.2, this diffuse pollution manual illustrates detailed information on cost and effectiveness estimates of 44 BMPs to reduce farm nutrient losses and therefore is the most suitable for undertaking a cost-effectiveness analysis. In addition, the information presented in this report offers scope to introduce some degree of variability in the analysis; as the data is separated by farm types for two different types of soils; sandy and clay loam soils. This introduces the possibility of accounting for farm and soil heterogeneity in the CEA by being able to assess two very extreme soils conditions that affect the effectiveness of each BMP (sandy and clay loam soils are at either end of the soil type spectrum of soil conditions found in Scotland).

This report builds on previous work undertaken by ADAS/IGER for DEFRA on understanding and identifying the costs and effectiveness of options at farm level to control different agricultural diffuse sources of water pollution. Previous reports separately addressed nitrates (Scholefield, 2005), phosphorus (Haygarth, 2003) and FIO (Haygarth, 2005). The manual identifies, from these projects, a range of methods which could be adopted by farmers to reduce nutrient losses. It provides detailed information about each measure, including; description, applicability, costs, effectiveness to reduce pollution, etc. The report concentrates on the so called three main farm diffuse pollutants of concern: nitrate (N), phosphorus (P) and faecal indicator organisms (FIOs) and illustrates 44 measures for their control, which have been grouped into the following categories:

- Land Use
- Soil Management
- Livestock Management
- Fertiliser Management

- Manure Management
- Farm infrastructure

Table 4.3 offers a summary of the different measures and the pollutants they target for different farm systems. Detailed information about each specific measure is available in the annexes at the end of this thesis (see Annex II).

Table 4.3 Summary of IGER/ADAS's 44 BMPs to reduce farm diffuse pollutants

			Farm types																				
			Arable			**Arable plus manure**			**Dairy**			**Suckler Beef**			**Broilers**			**Breeding pigs (indoors)**			**Breeding pigs (outdoors)**		
			type 1			type 2			type 3			type 4			type 5			type 6			type 7		
ID	**Category**	**BMP Measures**	N	P	FIO	N	P	FIO	N	P	FIO	N	P	FIO	N	P	FIO	N	P	FIO	N	P	FIO
1	Land Use	Convert arable land to extensive grassland	x	x		x	x																
2	Soil Managament	Establish cover crops in the autumn	x	x		x	x								x	x		x	x				
3	Soil Managament	Cultivate land for crop establishment in spring rather than autumn	x	x		x	x								x	x		x	x				
4	Soil Managament	Adopt minimal cultivation systems	x	x		x	x								x	x		x	x				
5	Soil Managament	Cultivate compacted tillage soils		x			x									x			x				
6	Soil Managament	Cultivate and drill across the slope		x			x									x			x				
7	Soil Managament	Leave autumn seedbeds rough		x			x									x			x				
8	Soil Managament	Avoid tramlines over winter		x			x									x			x				
9	Soil Managament	Establish in-field grass buffer strips	x	x	x	x	x	x							x	x	x	x	x	x	x	x	x
10	Soil Managament	Loosen compacted soil layers in grassland fields								x			x										
11	Soil Managament	Maintain and enhance soil organic matter levels	x	x		x	x																
12	Soil Managament	Allow field drainage systems to deteriorate	x	x		x	x		x	x		x	x		x	x		x	x		x	x	
13	Livestock Management	Reduce overall stocking rates on livestock farms							x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
14	Livestock Management	Reduce the length of the grazing day or grazing season							x	x	x	x	x	x									
15	Livestock Management	Reduce field stocking rates when soils are wet							x	x	x	x	x	x									
16	Livestock Management	Move feed and water troughs at regular intervals							x	x	x	x	x	x							x	x	x
17	Livestock Management	Reduce dietary N and P intakes							x	x					x	x		x	x		x	x	
18	Livestock Management	Adopt phase feeding of livestock							x	x								x	x		x	x	
19	Fertiliser Management	Use a fertiliser recommendation system	x	x		x	x		x	x		x	x		x	x		x	x				
20	Fertiliser Management	Integrate fertiliser and manure nutrient supply				x	x		x	x		x	x		x	x		x	x				
21	Fertiliser Management	Reduce fertiliser application rates	x	x		x	x	x	x	x		x	x		x	x		x	x				
22	Fertiliser Management	Do not apply P fertilisers to high P index soils		x			x			x			x			x			x				
23	Fertiliser Management	Do not apply fertiliser to high risk areas	x	x		x	x		x	x		x	x		x	x		x	x				
24	Fertiliser Management	Avoid spreading fertiliser to fields at high-risk times	x	x		x	x		x	x		x	x		x	x		x	x				
25	Manure Management	Increase the capacity of farm manure (slurry) stores							x	x	x							x	x	x			
26	Manure Management	Minimise the volume of dirty water produced							x	x	x								x	x			
27	Manure Management	Adopt batch storage of slurry									x									x			
28	Manure Management	Adopt batch storage of solid manure										x			x								
29	Manure Management	Compost solid manure										x			x								
30	Manure Management	Change from slurry to a solid manure handling system							x	x	x				x	x	x						
31	Manure Management	Site solid manure heaps away from watercourses and field drains				x	x	x				x	x	x	x	x	x						
32	Manure Management	Site solid manure heaps on concrete and collect the effluent				x	x					x	x	x	x	x							
33	Manure Management	Do not apply manure to high-risk areas				x	x		x	x	x	x	x		x	x		x	x	x			
34	Manure Management	Do not spread farmyard manure to fields at high-risk times				x	x					x	x										
35	Manure Management	Do not spread slurry or poultry manure to fields at high-risk times				x	x		x	x	x				x	x		x	x	x			
36	Manure Management	Incorporate manure into the soil				x	x			x					x	x		x	x				
37	Manure Management	Transport manure to neighbouring farms							x	x	x	x	x		x	x		x	x	x			
38	Manure Management	Incinerate poultry litter													x	x							
39	Farm infrastructure	Fence off rivers and streams from livestock							x	x	x	x	x	x									
40	Farm infrastructure	Construct bridges for livestock crossing rivers and streams							x			x											
41	Farm infrastructure	Re-site gateways away from high-risk areas		x			x			x			x			x			x			x	
42	Farm infrastructure	Establish new hedges		x			x			x			x			x			x			x	
43	Farm infrastructure	Establish riparian buffer strips	x	x		x	x		x	x	x	x	x	x	x	x		x	x	x	x	x	x
44	Farm infrastructure	Establish and maintain artificial (constructed) wetlands	x	x		x	x		x	x	x	x	x	x	x	x		x	x	x			

Source: IGER/ADAS, 2007

In order to determine costs and effectiveness of the 44 measures identified in the manual, seven farm systems were assessed in practice. The assessment included the following types of farms: arable, arable plus manure, dairy, beef, broilers, breeding pigs (indoors) and breeding pigs (outdoors). Table 4.4 introduce their characteristics.

Table 4.4 Summary of the model farm systems used by IGER/ADAS for estimating the cost and effectiveness of mitigation methods

#	Farm system	Animal count	Excreta (t/year)	Managed as Manure (%)	Field area (ha)	Average fertiliser	
						Kg N/ha	Kg P2O5/ha
1	Arable	0	0	n/a	300	165	60
2	Arable plus manure	0	2700	100	300	140	58
3	Dairy	270	5040	60	150	190	35
4	Suckler Beef	220	1850	50	100	80	30
5	Broilers	150000	2550	100	437	145	48
6	Breeding pigs (indoors)	1330	2125	100	71	145	48
7	Breeding pigs (outdoors)	2536	3568	0	24	0	0

Source: IGER/ADAS, 2007

4.3.3.1. About the costs and effectiveness estimates

For the cost side of the 44 measures, the report provides information on how much it would cost to implement each of the abatement techniques in terms of the financial costs to the farmer (including investment and operational costs). Costs are expressed per hectare and at farm level for the model farms identified in table 4.4. The following types of costs are assessed as appropriate for the different methods and for the different types of farms; one-off costs, annual cash costs, annualised capital costs (amortised over a given period of time) or annual and amortised costs.

Where the report identifies cost per hectare, these were estimated for the whole farm and not only for the area affected by the method. For example, cultivating compacted tillage soils affects 20% of the farm at a cost per ha treated of £20. Applying this measure to the whole of the farm would give a cost of £4/ha, assuming a total farm area of 100 ha. Appendix II of the report lays down the assumptions used in calculating the costs of each method (for further information see IGER/ADAS, 2007).

Some cost data modifications were needed to reflect the Scottish scenario. In general, we have transferred the costs of IGER/ADAS farms to the typical Scottish farms, and the following farm systems (which are more common in the Scottish rural landscape) are included in our analysis: arable, arable plus manure, dairy and beef. Due to lack of original Scottish data, it is assumed that some types of costs, especially those related with investment and operational costs of most of the measures would be the same independently of the location of the farm. Nevertheless, the costs of those measures that are identified in the manual as having an impact on farm's gross margins, output, etc. have been adjusted to reflect those of the Scottish typical farm. Below, we explain data modifications employed in order to adjust cost estimates to Scottish conditions for measures 1, 12 and 13 of the IGER/ADAS manual.

For our study, the Scottish "typical" farms have been defined from average farms presented in the Farm Incomes Report (Scottish Executive, 2002-2005). These representative farms do not differ much in size from those model farms used in the IGER/ADAS report. However, as we are defining a "typical" farm, which is defined from averages from a survey on many agricultural financial indicators, we do not hold information about some of their physical inputs/outputs which are relevant for this study; for example: excreta produced per year, % managed as manure, fertiliser applications. Accordingly, it is impossible to assess how these farms relate to those used by IGER/ADAS and it was only possible to assume that these farms share the same problems.

Averages for the years 2002-05 have been used to estimate these farms' financial indicators (i.e. level of output, subsidies, gross/net margins - see next chapter for more information on how these estimates have been derived from the Farm Incomes Report published by the Scottish Executive). These estimates have been used to modify the cost estimates of the measures listed below, we also offer a brief description of the measures and the assumptions used for the estimation of their financial costs (the assumptions for the costs estimates are the same as used by IGER/ADAS, 2007):

Measure 1: Convert arable land to extensive grassland. This measure is only applicable to arable farms and can be divided into: i) Measure 1a, conversion to ungrazed grassland; the application of this measure means that the land is left ungrazed after conversion and no livestock is purchased. In order to estimate the associated costs to the farmer, the annual loss of net farm income for the typical arable Scottish farm was calculated. ii) Measure 1b, conversion to extensive grazing; this would mean a change in farm systems: from cropping to livestock. For the estimation of the total costs associated with this measure, the fall in gross margins for the typical Scottish arable farm was calculated. Other considerations for the application of this measure include livestock purchase (including depreciation) and any other capital and annual costs associated with the conversion to livestock production (fencing, hedges, water provision, etc.).

Measure 12: Allow field drainage systems to deteriorate. Allowing field drainage systems to deteriorate would incur increasing annual costs for all the farm systems analysed (especially for arable farms). The costs for this measure have been estimated as the associated % loss of output. For arable farms it was assumed an 0.5% increasing annual loss of output for the first 5 years and 1.5% for the following five (IGER/ADAS, 2007).

Measure 13: Reduce overall stocking rates on livestock farms. Costs for a 50% reduction in livestock numbers on the typical dairy and beef Scottish farms were estimated by halving their respective gross margins.

Nevertheless, the IGER/ADAS cost estimates need to be considered carefully. More information is needed about the annual costs of some of the measures. For example, this report does not provide specific information on annual costs (i.e. maintenance costs) of those measures that need capital investment. They do take into account annual costs but also including amortised capital over a number of years. This creates a serious risk of double counting (the way these costs are presented in the report make it impossible to differentiate specific annual costs). At this stage, these costs have not been included in the analysis, which implies that costs for some measures may have been underestimated.

The IGER/ADAS report presents estimates of the effectiveness of each measure in reducing losses of each of the main farm diffuse pollutants; N, P and FIO. Baseline losses, in the absence of any mitigation measures, were estimated for the farm model systems presented in table 4.4. These were divided between components originating from the soil, from manure/excreta and from fertiliser use. Environmental models were used to assist in the estimation of nitrate and P losses. The NITCAT, NCYCLE and MANNER models were used for nitrate losses at farm level and the PSYCHIC model for P. Expert judgement was used to estimate baseline losses of FIO and the effectiveness of mitigation measures. Estimates of effectiveness for each measure were averaged over the whole farm area for each system on a clay loam and a sandy loam soil, assuming a medium rainfall of 850 mm/year.

The suitability of applying the aforementioned environmental models to Scottish conditions has already been tested for the estimation of total sediment and phosphorus loads lost from agricultural and forestry areas. This study formed the basis of a recent national scale exercise that led to the development of a screening tool to identify water bodies vulnerable to specific pressures in Scotland (see: Anthony et al., 2005).

4.3.4. Cost-effectiveness ratios for the selection of measures

Prior to the estimation of the abatement cost curves, it is necessary to establish a ranking of measures, as there is a need under the WFD to sort mitigation options depending on their increasing levels of cost-effectiveness to reach good status. Cost-effectiveness ratios, an application of CEA, can be applied when all the benefits of abatement effort can be measured in one consistent dimensional unit (Johnsen, 1993); e.g. units such as, kg of pollutant loads reductions or % loads reductions per measure.

This type of analysis only compares two parameters: costs opposed to a single physical unit of benefits. One important requirement for the application of CE ratios is the necessity to obtain a high degree of consistency between units for both cost and effectiveness estimates. For the estimation of costs, the minimum requirement is to hold detailed information on the financial costs (recurring and non-recurring costs) for each measure and select a discount rate to be able to compare options in terms of their present values. For the assessment of effectiveness, comparable units between different sources of pollution are needed for their aggregation into one single measure of effectiveness. Our analysis covers N, P and FIO load reductions at farm level, each BMP can target one, two or three of these pollutants at the same time. Applying CE ratios leaves two options i) to aggregate the effectiveness of all diffuse pollutants into one single measure of effectiveness for each measure or ii) to base the analysis in only one pollutant.

One possible solution which can be applied in the case of measures that have an effect on several nutrients aimed at the same objective (e.g. reduction in eutrophication) is to weigh these effects together and measure them in equivalent units e.g. N-equivalents or eutrophication equivalents (Jacobsen, 2007). However, in order to do this, the effects of both N, P and FIO losses would need to be calculated and the trade-off between them estimated.

In some cases when this trade-off has not been calculated, some economic studies assume a 1:1 relationship between pollutants, where, for example, a reduction of 1kg of N is equivalent to 1 kg of P (NIRAS, 2006). However, scientific research indicates that this relationship could be well above 10:1 between N and P. In this instance, the implications of assuming a 1:1 ratio would make measures aimed at reducing P-losses less cost-effective in comparison to measures aimed at reducing N-losses.

There is no information available about the trade-offs between P, N and FIO in Scottish waters. Under the risk of underestimating these trade-offs, the second option was preferred, and we opted to focus the cost-effectiveness analysis on those measures that would reduce nutrient loads at farm level for one single pollutant. In this case, options to reduce Phosphorus loads were considered for our analysis, as it has become the key limiting factor controlling the degree of eutrophication of rivers in Europe (Smith et al, 2005) and Scotland (Aukerman, 2004).

Evidence suggests that of the major plant nutrients, P is typically in shortest supply in rivers and other freshwaters and so generally has the greatest potential to limit plant growth (Kronvang et al, 2005). Mainstone and Parr (2002) provided the evidence for this when they studied which nutrient N or P could limit plant growth to a greater extent. After carrying out an assessment of N and P relative and absolute availabilities for over 5000 rivers across England and Wales, they observed that at SRP (Soluble Reactive Phosphorus) concentrations at which is known that P is limiting plant growth, the ratio between N and P were always above 8. For this reason, they concluded that N is invariably surplus to plant requirements relative to P in such situations, confirming the view that P is the most likely nutrient to be limiting plant growth.

Accordingly, CE ratios for the cost-effectiveness analysis of BMPs to reduce agricultural P for the selection criteria were estimated following this expression:

$$CE_r = PV/E$$

Where; CE_r stands for Cost-effectiveness ratio. *PV* reflects the Present Value over the lifetime of each measure or as in this case, over the time period given to achieve the objectives of the Directive (the year 2015 was chosen at it reflects the end of the first river basin management plans); and *E,* represents the reduction in P loads for each measure.

For this type of analysis, we hold information on effectiveness estimates for each measure in terms of abatement costs and reduction in pollutant loss for N, P and FIO, when applicable for each measure, per type of farm and for two types of soils; sandy and clay loam. This attempts to reflect the variability on effectiveness levels associated with different soil conditions. Units of effectiveness for nitrates and P loads reductions are expressed in terms of Kg of total P or N reductions per hectare. Reductions in FIO losses for each measure are represented as % load reduction per ha. Abatement costs are represented in unit costs (£/ha) and reflect Present Values over an 8-year period, up to 2015, and an annual discount rate of 3.5% has been used, following UK Treasury guidance (HM Treasury, 2003).

This information is used to obtain cost-effectiveness indicators to reduce agricultural P loads (£/Kg P removed per ha or the equivalent £/% reduction) for each individual measure - see tables 4.5, 4.6, 4.7 and 4.8 below. Detailed information about the CEA results for each specific measure is available in the annexes at the end of this thesis (see Annex III). Measures are sorted in increasing order, in this way, measures are classified in cost effectiveness terms (as prescribed in the Directive) and the selection order is defined in a precise and consistent manner. This allows us to gauge the gap between the baseline scenario and the environmental objectives to be achieved by the year 2015. Implementing a measure with a lower CE ratio would be more cost-efficient than implementing first another measure with a higher ratio. These indicators could be used for the development of cost effectiveness ladder based graphs in order to compare the different measures per type of soil and type of farm system, which could be a way of detailing the information for the PoMs to reach good status. These graphs have been included in the appendix (see annex IV).

For the estimation of the cumulative impacts of implementing successive measures, it was assumed that the implementation of each specific measure is on/off, meaning that we are not considering that the farmer can for example use a fertiliser recommendation system (measure 19) in half the field, instead of all the recommended field area. For this analysis, it is also assumed that the cumulative effectiveness levels of measures, which are implemented in a sequential order in terms of their cost-efficiency, are purely additive. The analysis also takes compounding into consideration, which means that the effectiveness of the next most cost-effective measure to be implemented varies depending on the overall effect of the previous measure applied, as the baseline loss of pollutants at farm level has been modified. Finally, even though the selection criteria for the ranking of measures is based exclusively on P loads reductions, additional N and FIO reductions for each measure are shown in the tables as alternative benefits.

The CEA results illustrated in tables 4.5 to 4.8 suggest that the implementation of BMPs offer scope to reduce P losses at farm level at no or little cost to farmers. Some measures, especially relevant to farms in clay loam soils, even illustrate negative CE ratios which would deliver efficiency savings to the farmer and therefore, increase farm profits. A further analysis of the information presented in these tables by types of farms suggests that different techniques would be more suitable to some farm types than others. Arable and Arable plus manure types of farms would obtain larger benefit from the implementation of those measures that target farm infrastructure (establishment of riparian buffer strips) and soil management techniques (cover cereal crops in the autumn). In comparison, for those farms with animal activity (dairy and beef types of farms), the CEA analysis suggests that reduction of P losses at farm level would be most cost-effectively achieved by targeting the management of fertilisers and manures (i.e. integrate fertiliser and manure nutrient supply and targeting the timings of spreading farmyard manure in the field).

Table 4.5 Cost-effectiveness indicators for BMPs to reduce agricultural P loads (£/Kg P removed per ha or the equivalent £/% reduction) for Arable farms

Sandy loam soil		**CE ratio***	**P Loss****	**N Loss****
ID	BMP Measures		Cumulative % Reduction	
43	Establish riparian buffer strips	1.55	33.33	2.94
2a	Establish cover crops in the autumn: cereals	1.97	40.00	16.26
41	Re-site gateways away from high-risk areas	2.85	42.00	16.26
6	Cultivate and drill across the slope	3.54	45.87	16.26
5	Cultivate compacted tillage soils	4.72	49.48	17.90
42	Establish new hedges	5.00	59.58	17.90
8	Avoid tramlines over winter	5.31	62.28	17.90
9	Establish in-field grass buffer strips	5.33	79.88	25.79
3	Cultivate land for crop establishment in spring rather than autumn	12.99	81.22	27.25
2b	Establish cover crops in the autumn: other crops	13.39	83.10	37.23
1a	Convert arable land to extensive grassland: conversion to ungrazed grassland	15.29	93.24	97.54
1b	Convert arable land to extensive grassland: conversion to extensive grazing	19.02	96.62	99.03
7	Leave autumn seedbeds rough	47.24	96.85	99.03
Clay loam soil				
4	Adopt minimal cultivation systems	-65.85	4.78	5.32
22	Do not apply P fertilisers to high P index soils	-6.04	6.02	5.32
44	Establish and maintain artificial (constructed) wetlands	3.01	43.61	25.46
2a	Establish cover crops in the autumn: cereals	3.48	46.80	36.57
41	Re-site gateways away from high-risk areas	3.64	48.19	36.57
6	Cultivate and drill across the slope	4.94	50.67	36.57
5	Cultivate compacted tillage soils	6.59	53.03	36.57
8	Avoid tramlines over winter	7.41	55.27	36.57
19	Use a fertiliser recommendation system	12.07	55.86	39.26
21b	Reduce fertiliser application rates: 20% reduction in P	13.88	56.43	39.26
3	Cultivate land for crop establishment in spring rather than autumn	18.11	58.52	40.56
1a	Convert arable land to extensive grassland: conversion to ungrazed grassland	18.19	79.44	97.47
1b	Convert arable land to extensive grassland: conversion to extensive grazing	22.55	88.11	98.92
2b	Establish cover crops in the autumn: other crops	23.68	88.78	99.08
43	Establish riparian buffer strips	23.80	89.03	99.10
42	Establish new hedges	28.75	89.41	99.10
23	Do not apply fertiliser to high risk areas	28.98	89.64	99.11
24	Avoid spreading fertiliser to fields at high-risk times	37.57	89.77	99.11
9	Establish in-field grass buffer strips	63.59	90.17	99.19
7	Leave autumn seedbeds rough	65.85	90.64	99.19
12	Allow field drainage systems to deteriorate	1023.88	90.68	99.22

* £/% Reduction in P loss/ha, NPV/ha over an 8 year period (up to 2015). Discount rate 3.5%

**Farm level per ha

Table 4.6 Cost-effectiveness indicators for BMPs to reduce agricultural P loads (£/Kg P removed per ha or the equivalent £/% reduction) for Arable plus Manure farms

Sandy loam soil		**CE ratio***	**P Loss****	**N Loss****
ID	BMP Measures		Cumulative % Reduction	
43	Establish riparian buffer strips	1.88	27.50	2.63
2a	Establish cover crops in the autumn: cereals	2.62	32.94	21.42
41	Re-site gateways away from high-risk areas	3.80	34.61	21.42
6	Cultivate and drill across the slope	4.72	37.88	21.42
5	Cultivate compacted tillage soils	6.30	40.99	21.42
34	Do not spread farmyard manure to fields at high-risk times	6.30	42.46	22.80
42	Establish new hedges	6.67	51.09	22.80
8	Avoid tramlines over winter	7.09	53.54	22.80
9	Establish in-field grass buffer strips	7.11	69.80	30.25
3	Cultivate land for crop establishment in spring rather than autumn	17.32	71.31	32.09
2b	Establish cover crops in the autumn: other crops	17.85	73.46	45.19
1a	Convert arable land to extensive grassland: conversion to ungrazed grassland	20.39	85.40	98.08
1b	Convert arable land to extensive grassland: conversion to extensive grazing	25.36	90.88	99.33
7	Leave autumn seedbeds rough	62.99	91.33	99.33
Clay loam soil				
4	Adopt minimal cultivation systems	-71.58	4.40	6.86
20	Integrate fertiliser and manure nutrient supply	-59.05	5.16	9.60
22	Do not apply P fertilisers to high P index soils	-6.56	6.30	9.60
36	Incorporate manure into the soil	0.00	7.05	7.83
44	Establish and maintain artificial (constructed) wetlands	3.07	43.49	29.52
2a	Establish cover crops in the autumn: cereals	3.79	46.43	44.72
41	Re-site gateways away from high-risk areas	3.96	47.71	44.72
6	Cultivate and drill across the slope	5.37	50.01	44.72
35	Do not spread slurry or poultry manure to fields at high-risk times	6.56	51.21	46.89
5	Cultivate compacted tillage soils	7.16	53.36	46.89
8	Avoid tramlines over winter	8.05	55.41	46.89
19	Use a fertiliser recommendation system	13.12	55.95	48.45
33	Do not apply manure to high-risk areas	13.12	56.48	48.95
34	Do not spread farmyard manure to fields at high-risk times	13.12	57.00	49.45
21b	Reduce fertiliser application rates: 20% reduction in P	15.09	57.51	49.45
1a	Convert arable land to extensive grassland: conversion to ungrazed grassland	18.50	78.59	98.02
3	Cultivate land for crop establishment in spring rather than autumn	19.68	79.53	98.08
43	Establish riparian buffer strips	21.55	80.02	98.11
1b	Convert arable land to extensive grassland: conversion to extensive grazing	23.08	88.25	99.26
2b	Establish cover crops in the autumn: other crops	25.74	88.86	99.42
42	Establish new hedges	31.25	89.22	99.42
23	Do not apply fertiliser to high risk areas	31.50	89.43	99.42
31	Site solid manure heaps away from watercourses and field drains	39.37	89.48	99.42
24	Avoid spreading fertiliser to fields at high-risk times	40.83	89.60	99.42
9	Establish in-field grass buffer strips	62.20	90.02	99.48
7	Leave autumn seedbeds rough	71.58	90.46	99.48
32	Site solid manure heaps on concrete and collect the effluent	81.50	90.50	99.48
12	Allow field drainage systems to deteriorate	1112.92	90.53	99.50

* £/% Reduction in P loss/ha, NPV/ha over an 8 year period (up to 2015). Discount rate 3.5%

**Farm level per ha

Table 4.7 Cost-effectiveness indicators for BMPs to reduce agricultural P loads (£/Kg P removed per ha or the equivalent £/% reduction) for Dairy farms

Sandy loam soil		**CE ratio***	**P Loss****	**N Loss****	**FIO Loss****
ID	BMP Measures			Cumulative % Reduction	
20	Integrate fertiliser and manure nutrient supply	-18.90	5.00	7.38	0.00
35	Do not spread slurry or poultry manure to fields at high-risk times	0.79	24.00	18.01	10.00
33	Do not apply manure to high-risk areas	1.57	31.60	20.02	10.00
43	Establish riparian buffer strips	1.83	55.54	21.99	19.00
41	Re-site gateways away from high-risk areas	1.90	57.76	21.99	19.00
42	Establish new hedges	5.00	66.21	21.99	19.00
16	Move feed and water troughs at regular intervals	5.93	71.28	23.27	27.10
25	Increase the capacity of farm manure (slurry) stores	7.10	77.02	29.56	41.68
30	Change from slurry to a solid manure handling system	11.75	86.21	37.64	65.01
37a	Transport manure to neighbouring farms 5km	13.70	93.11	43.77	72.01
39	Fence off rivers and streams from livestock	15.36	93.45	44.24	74.81
10	Loosen compacted soil layers in grassland fields	17.01	93.78	44.24	74.81
18	Adopt phase feeding of livestock	25.67	94.09	46.98	74.81
37b	Transport manure to neighbouring farms 20km	28.42	97.05	52.19	79.84
15	Reduce field stocking rates when soils are wet	41.34	97.34	52.19	81.86
17	Reduce dietary N and P intakes	66.93	97.47	54.54	81.86
13	Reduce overall stocking rates on livestock farms	126.50	98.48	73.17	90.93
Clay loam soil					
20	Integrate fertiliser and manure nutrient supply	-29.40	3.21	5.88	0.00
21b	Reduce fertiliser application rates (reduction in P by a 20%)	-7.35	5.29	5.88	0.00
22	Do not apply P fertilisers to high P index soils	-3.67	7.32	5.88	0.00
35	Do not spread slurry or poultry manure to fields at high-risk times	0.64	30.16	19.72	10.00
33	Do not apply manure to high-risk areas	2.32	34.90	20.90	19.00
44	Establish and maintain artificial (constructed) wetlands	3.01	60.94	48.82	35.20
41	Re-site gateways away from high-risk areas	3.80	61.91	48.82	35.20
16	Move feed and water troughs at regular intervals	6.73	66.95	49.57	41.68
19	Use a fertiliser recommendation system	7.35	67.66	52.54	41.68
25	Increase the capacity of farm manure (slurry) stores	8.11	73.32	59.52	53.34
37a	Transport manure to neighbouring farms 5km	16.54	84.37	63.09	62.68
23	Do not apply fertiliser to high risk areas	19.29	84.82	63.20	62.68
18	Adopt phase feeding of livestock	21.14	85.74	65.36	62.68
43	Establish riparian buffer strips	29.87	86.04	66.38	66.41
42	Establish new hedges	31.11	86.49	66.38	66.41
30	Change from slurry to a solid manure handling system	33.74	88.37	71.33	79.84
37b	Transport manure to neighbouring farms 20km	34.31	93.19	73.86	83.88
39	Fence off rivers and streams from livestock	35.84	93.34	74.24	85.49
15	Reduce field stocking rates when soils are wet	46.30	93.93	74.62	86.94
36	Incorporate manure into the soil	49.24	94.00	74.62	86.94
17	Reduce dietary N and P intakes	55.12	94.36	76.11	86.94
24	Avoid spreading fertiliser to fields at high-risk times	107.54	94.44	76.18	86.94
10	Loosen compacted soil layers in grassland fields	119.05	94.48	76.18	86.94
13	Reduce overall stocking rates on livestock farms	144.58	96.41	83.19	93.47
26	Minimise the volume of dirty water produced	285.60	96.43	83.24	94.12
14	Reduce the length of the grazing day or grazing season	385.82	96.46	85.70	94.71
12	Allow field drainage systems to deteriorate	1397.96	96.48	86.12	94.71

* £/% Reduction in P loss/ha, NPV/ha over an 8 year period (up to 2015). Discount rate 3.5%

**Farm level per ha

Table 4.8 Cost-effectiveness indicators for BMPs to reduce agricultural P loads (£/Kg P removed per ha or the equivalent £/% reduction) for Beef farms

Sandy loam soil		**CE ratio***	**P Loss****	**N Loss****	**FIO Loss****
ID	BMP Measures			Cumulative % Reduction	
34	Do not spread farmyard manure to fields at high-risk times	0.45	35.00	1.67	0.00
43	Establish riparian buffer strips	2.56	54.50	4.40	10.00
37a	Transport manure to neighbouring farms 5km	3.15	70.43	15.02	10.00
31	Site solid manure heaps away from watercourses and field drains	3.15	71.90	15.02	10.00
33	Do not apply manure to high-risk areas	3.15	73.31	15.96	10.00
16	Move feed and water troughs at regular intervals	5.46	77.31	18.30	19.00
37b	Transport manure to neighbouring farms 20km	6.52	85.25	27.38	19.00
32	Site solid manure heaps on concrete and collect the effluent	13.72	85.99	27.78	27.10
10	Loosen compacted soil layers in grassland fields	17.01	86.69	27.78	27.10
15	Reduce field stocking rates when soils are wet	25.59	88.02	27.78	34.39
14	Reduce the length of the grazing day or grazing season	51.18	88.62	33.80	40.95
13	Reduce overall stocking rates on livestock farms	66.67	92.03	48.51	70.48
Clay loam soil					
20	Integrate fertiliser and manure nutrient supply	-472.44	0.10	2.50	0.00
21b	Reduce fertiliser application rates; 20% Reduction P	-5.25	3.10	2.50	0.00
22	Do not apply P fertilisers to high P index soils	-2.62	6.00	2.50	0.00
34	Do not spread farmyard manure to fields at high-risk times	0.98	21.04	3.31	0.00
33	Do not apply manure to high-risk areas	2.25	26.57	4.12	0.00
37a	Transport manure to neighbouring farms 5km	2.69	56.68	12.11	0.00
44	Establish and maintain artificial (constructed) wetlands	2.86	74.87	85.35	20.00
41	Re-site gateways away from high-risk areas	3.17	75.63	85.35	20.00
19	Use a fertiliser recommendation system	5.25	76.36	85.72	20.00
31	Site solid manure heaps away from watercourses and field drains	5.25	77.07	85.84	28.00
37b	Transport manure to neighbouring farms 20km	5.57	86.47	87.02	28.00
16	Move feed and water troughs at regular intervals	5.85	88.36	87.23	35.20
23	Do not apply fertiliser to high risk areas	7.87	88.83	87.34	35.20
32	Site solid manure heaps on concrete and collect the effluent	22.87	89.16	87.45	41.68
24	Avoid spreading fertiliser to fields at high-risk times	27.36	89.38	87.55	41.68
15	Reduce field stocking rates when soils are wet	28.43	90.34	88.79	47.51
39	Fence off rivers and streams from livestock	38.40	90.53	88.98	52.76
43	Establish riparian buffer strips	38.40	90.72	89.17	57.48
13	Reduce overall stocking rates on livestock farms	54.05	94.15	90.97	78.74
10	Loosen compacted soil layers in grassland fields	85.04	94.21	90.97	78.74
12	Allow field drainage systems to deteriorate	177.85	94.27	91.35	78.74
14	Reduce the length of the grazing day or grazing season	255.90	94.33	91.71	80.87

* £/% Reduction in P loss/ha, NPV/ha over an 8 year period (up to 2015). Discount rate 3.5%

**Farm level per ha

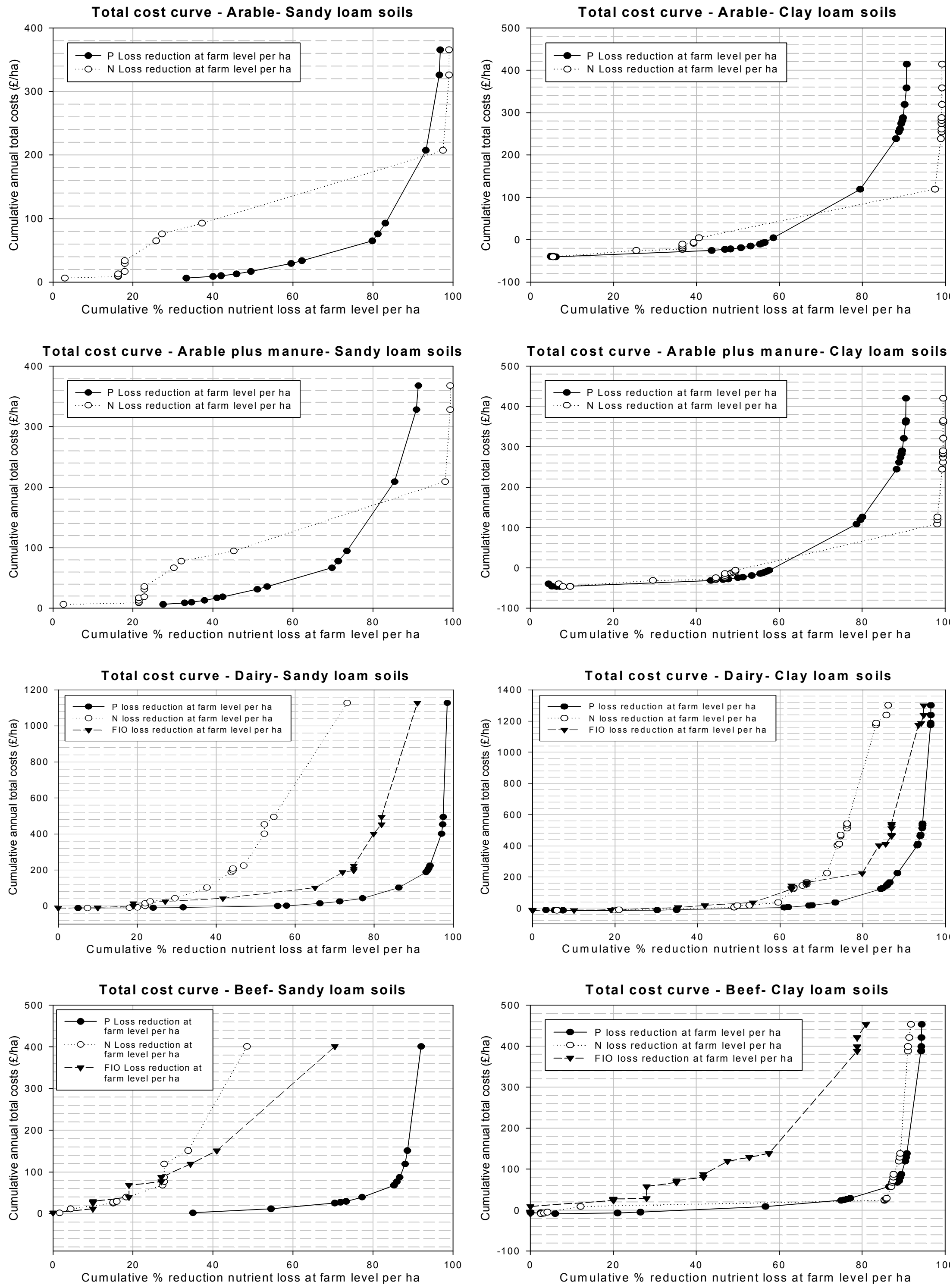
4.3.5. Development of abatement cost curves

The cost effectiveness analysis introduced in the previous section provides a basis for the selection of measures under the WFD; and assists in the identification and filtering of relevant measures for different farm models. However, for the assessment of the total abatement costs associated with reaching different loads reductions, which is important for the assessment of disproportionate costs, we still need to solve the cost minimisation problem introduced earlier.

In this section, the information about costs and effectiveness of measures (previously introduced) is used as an input to develop cost functions. The objective is to establish a mathematical description of the relation between costs and nutrient emission reductions at farm level. This is essential in order to assess the shape of the marginal abatement cost function which will be necessary for the economic analysis of the measures and to establish a link with the marginal benefits of reducing loads of diffuse pollutants.

Following the results of the cost-effectiveness analysis, total cost curves for P mitigation were developed for all types of farms for sandy and clay loam soils. The curves were obtained by plotting the respective cumulative annual total abatement costs (£/ha/year) with the associated % reduction in P loss at farm level for each relevant measure. These curves are represented in figure 4.3. These graphs also show the additional % reductions in N and FIO. Total abatement costs (TAC), which are the total costs of abating nutrient loads by a certain amount per year, in this case reflect annual equivalent costs for each measure (average PVs for each measure per year).

Figure 4.3 Total Abatement Cost curves for P mitigation options at farm level for different types of farm systems for sandy and clay loam soils. The additional % reductions in N and FIO are also shown



Total abatement cost curves can be represented in two ways: constantly increasing (i.e. linear form) or increasing at the margin (i.e. polynomial or exponential growth form). It is obvious by looking at the plotted data in figure 4.3 that the relation between total abatement costs and P loads reductions at farm level is not linear. In this case, assuming a linear relationship would over estimate both the costs and effects of improvements associated with the measures selected at low levels of abatement. Alternatively, at high levels of abatement a linear relationship would fail to reflect the high costs needed to achieve small reductions. In practical terms, assuming a linear relationship between costs and environmental improvements has been normal practice for the management of freshwater resources in Europe and the US for the last 30 years and, as Statzner et al., (1997) illustrate, this assumption was responsible for much of the over spending and lack of success in achieving water quality targets in fresh water management. As water managers failed to see that at the margins, costs increase dramatically whilst further environmental improvement become more difficult to achieve.

Using the least square method, it is possible to fit a polynomial form function to the data plotted in figure 4.3. Curve fitting is normal practice in the abatement costs literature (NERI, 2006; Martinez and Albiac, 2006). The goal of non-linear regression is to fit a model (function) to a set of data (x,y). The least square method finds the best-fit values of the variables in the model which can be used to derive the function (rate, constants, affinities, receptor numbers, etc). The method finds the line that minimises the sum of the square of the vertical distances of the points to the line. However, finding a fit is not an easy task and sometimes many different functional forms can be fitted to the same data.

Following goodness of fit best practice in economic analysis (Krueger and Lewis-Beck, 2007)[14], table 4.9 presents function fitting test results for a cubic polynomial form for the estimation of annual TAC curves for different types of farms for sandy and clay loam soils. This table also illustrates the regression coefficients and their respective statistical significance. The curve fitting tests show that cubic form functions offer a very good fit for arable and arable plus manure types of farms, in order to describe the relation between TAC and % P load reductions (with adjusted R^2 very close to 1, low standard errors and high statistical significance of the regression coefficients). Curve fits for dairy and beef are not as

[14] Krueger and Lewis-Beck (2007) have recently reviewed best practice in reporting goodness of fit regression statistics in economic analyses. They conclude that best practice actually consists in the presentation of adjusted R^2 and standard error of the estimate measures.

good, with adjusted R^2 of around 70% and no statistical significance ($\alpha < 10\%$) of some of the estimates.

Table 4.9 Coefficients and curve fitting statistical significance for the TAC curve cubic polynomial equations

	Adj. Rsqr	(SEE)	Regression Coefficients							
			y0		a		b		c	
Arable – Sandy	0.98	21.97	-774.876	**	45.197	***	-0.836	***	0.005	***
Arable- Clay	0.97	28.79	-93.042	**	11.509	**	-0.352	**	0.003	***
Arable + manure -Sandy	0.99	12.51	-365.369	***	24.741	***	-0.525	***	0.004	***
Arable + manure -Clay	0.97	26.23	-102.817	***	12.292	***	-0.375	***	0.003	***
Dairy- Sandy	0.56	194.42	-170.827		23.655		-0.674		0.005	*
Dairy- Clay	0.73	220.08	-225.069		42.667	**	-1.284	***	0.010	***
Beef- Sandy	0.68	61.33	-2,299.554	*	129.090	*	-2.261	*	0.013	**
Beef- Clay	0.67	87.92	-44.678		11.094		-0.402	**	0.003	***

Statistical significance of the coefficients:"*" at 10% level, "**" at 5%level and "***" at 1% level

In this case, the goodness of fit exercise reported in table 4.9 was developed as a result of the need to obtain a mathematical relationship between costs and effectiveness of measures for different types of farms and soils for the subsequent analyses carried out in this thesis (i.e. marginal analysis and profit optimisation in chapter 5). The accuracy of the results is reliant on the information available which drives the shape of the total abatement cost curves presented in figure 4.3. Unfortunately, curve fitting results presented in table 4.9 cannot be used to draw conclusions about which factors are driving the cost-effectiveness of measures for the different farm types analysed. Nevertheless, it can be used to argue the case for the need of more information on the costs and effectiveness of options to reduce P losses for dairy and beef farm types.

The selection of a functional form is driven by the need to find a suitable mathematical expression that best describes the data (i.e. cubic polynomial) whilst facilitating the aforementioned subsequent analyses. The selection of a cubic polynomial functional form is desirable in this case to other functional forms as it fulfils the necessary assumptions for optimisation and analysis at the margin. This type of function will always offer a maxima or minima and its second derivative differs from zero. In addition, the popularity of this functional form within these types of analyses is due to simplicity in its mathematical manipulation (NERI, 2006).

TAC curves presented in figure 4.3 confirm that clay loam soils offer higher P loads reductions than sandy loam soils. This type of soil is also more prone to offer overall efficiency savings to the farmer when the most cost-effective measures are applied in sequential order. Our results indicate that for clay loam soils, P load reductions of around 60% (for all the farm types analysed) can be obtained at no extra total costs to the farmer. In this instance, the need to increase farmers' awareness and education about the potential benefits associated with the implementation of different BMPs could prove fundamental for the achievement of good status, as there is a significant potential to achieve a win-win situation.

Ideally, the criteria for the selection of measures to target N, P and FIO losses at farm level, should include all these three types of farm nutrients. However, this analysis proves that if environmental trade-offs between pollutants are not known, it is possible (and probably more accurate) to target losses reductions for one of the nutrients for the CEA of the selection of measures. This method still allows us to analyse the remaining pollutants as alternative reductions/benefits.

4.3.5.1. Marginal abatement costs curves

A feature of this model is that with the coefficients of the TAC curve, we can derive the marginal abatement cost (MAC) curves. The first derivative of the total abatement cost function is the marginal abatement cost function (Varian, 2003). Using a cubic polynomial form for the (TAC) curve implies that the mathematical relationship between TAC and MAC can be described as follows:

$$TAC = y_0 + aX + bX^2 + cX^3 \quad \leftrightarrow \quad TAC' = MAC = a + 2bX + 3cX^2$$

Figure 4.4 Marginal Abatement Cost (MAC) curves for options to reduce agricultural P loads.

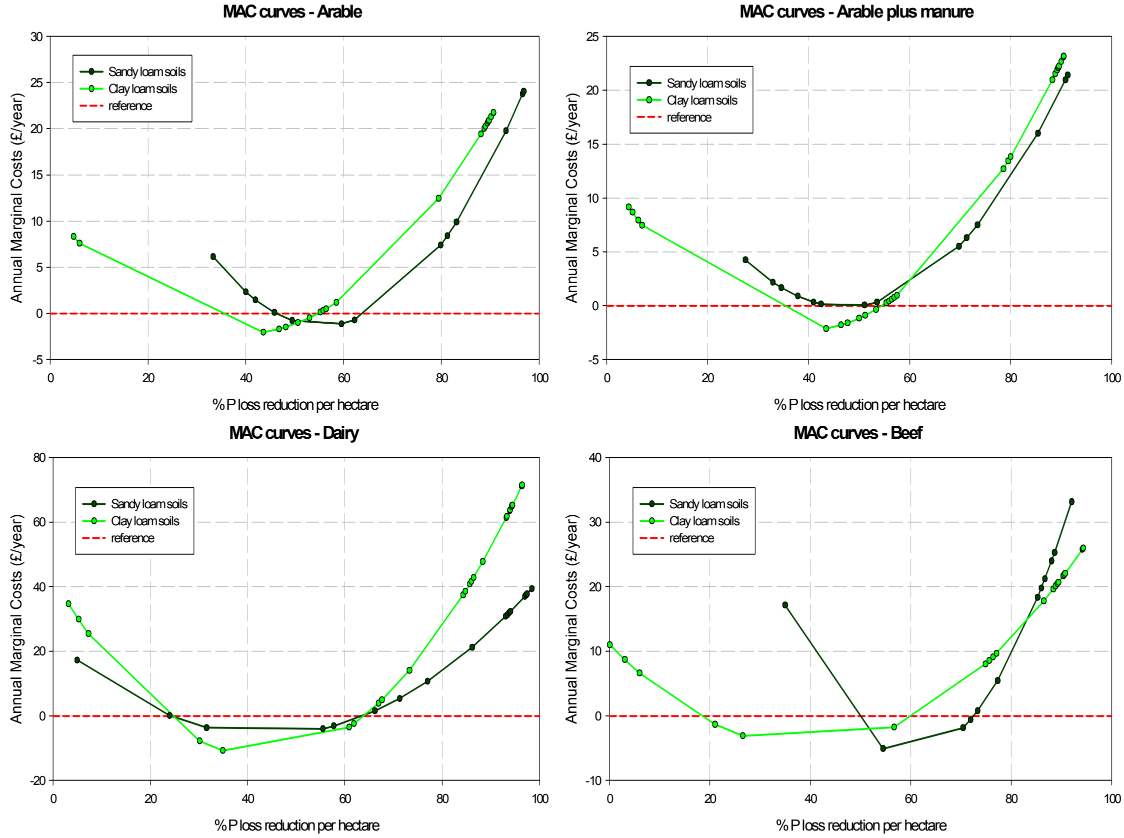


Figure 4.4 illustrates annual marginal abatement costs curves for different P reduction levels for different farm systems on sandy and clay loam soils. All marginal costs initially decrease with increasing targeted P load reductions, until reaching the minimum point that represents a discrete, lowest possible cost-efficient load reduction level. Overall, we would expect least cost P load reductions in sandy rather than in clay loam soils[15].

This analysis suggests that in the initial situation marginal costs for individual farms are not zero and that marginal costs are expected to decrease at initial stages of implementing abatement measures at farm level. Furthermore, if zero MCs are assumed for the initial situation, as is common practice in dynamic optimisation, the analysis would already be

[15] Arguably, this may well be because of the functional form imposed in order to establish a mathematical relationship between costs and effectiveness of BMPs. A quadratic polynomial form is expected to have a maximum or a minimum.

assuming that farmers are abating between 50 to 70% of the total P load and may ignore any possible cost savings which are associated with low levels of abatement. In practical terms, if MC decrease at early stages of abatement, there is a reason for farmers to abate up to the point where the costs of the next measure becomes more expensive than the measure previously installed. It would make sense to abate if the next % unit of P load reductions is cheaper than the previous one, as long as it is cheaper than the penalty imposed by the regulators.

One of the possible applications of marginal abatement costs analysis is that it can provide an indication of the possible limitation/success of applying certain economic instruments for the control of environmental pollution, which under the WFD is very relevant for the identification of delivery mechanisms of measures for the achievement of good status.

In this study, marginal abatement costs change for different farm systems and different types of soils. One of the requirements for the application of economic instruments (such as pollution trading schemes) is that marginal costs of abatement should differ between polluters (Tietenberg, 1985). Under this circumstance, farmers with low abatement costs may choose to abate more and trade part of their permit to higher abatement costs farmers. For this analysis we have assumed cost homogeneity within each class of farm systems. Under this assumption, our results indicate that a degree of trading would be possible between different farm systems or between the same systems, under different types of soils.

However, under normal circumstances, we would expect to find cost heterogeneity between farms of the same category (as no two farms are the same or are managed in the same way). Under this situation, the design of a trading system usually depends on the nature of the pollutant being regulated and traded. In environmental economics, pollutants are normally categorised into three different classes; uniformly mixed assimilative, uniformly mixed accumulative and non-uniformly mixed assimilative pollutants (Tietenberg, 1985). For the first two classes, a simple trading scheme on a one to one basis would prove efficient by simply equalising marginal abatement costs across polluters. For non-uniformly mixed pollutants, which is clearly the case of agricultural P loads, the design of a pollution trading scheme is more complicated, depending not only on the level of discharges at farm level, but also upon the intrinsic characteristics of each farm, location and transfer characteristics of the pollutants (Hung and Shaw, 2005). This is a consideration for further research.

4.4. Uncertainty/sensitivity analysis

The findings of our study need to be taken with caution. Our cost-curves can be misleading as they suggest that the effects of measures are independent of each other and that reductions in Phosphorus leaching are additive. Both of these are relevant for the combination of measures. Additionally, due to data constraints, cost curves of mitigation options have been developed for different types of Scottish farms based on estimates of costs and effects transferred from English farms.

The validity of our results depends to a large extent on the accuracy related to the transfer of data used for the cost and effectiveness estimates of BMPs at farm level. The main question is how relevant are our results to Scottish conditions. In theory, a quantitative sensitivity analysis to assess the validity of the transferred database should have been performed, including margins of errors of the estimates. However, this was impossible in this case, due to lack of original Scottish data for comparison. As an alternative solution, a qualitative assessment of the data was carried out instead. In order to evaluate the validity of the IGER/ADAS data for Scotland, an expert focus group was conducted (this was done as part of WP 3.5 SEERAD research programme on the management of water quality). Members of the group included farm experts and soils and water scientists with relevant experience in agricultural diffuse pollution issues in Scotland[16].

In summary, the group concluded that there is a need to alter estimates of N and P loss for Scotland. For example, estimates of P losses from erodible sandy loam soils in the IGER/ADAS report were too low according to their professional knowledge on Scottish types of soils. Additionally, it was identified that costs of mitigation do not consider interdependency between measures. This refers to those measures which are inter-connected, for example; for measure 20 (integrate fertilisers and manure nutrient supply) to be effective on a dairy farm system, measure 25 (increase the capacity of farm manure slurry stores) is very likely to be needed first. Furthermore, our data does not include the possible economies of scale (cost savings) associated with implementing some measures at the same period of time.

The issue of interdependency between measures also affects the assessment of the combined effectiveness of BMPs to reduce farm P loadings. Our study assumes that the cumulative

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[16] members of the group included: Andy Vinten, Philippa Booth, Martyn Futter (MLURI), Michael Macleod, Alex Sinclair, Bill Crooks (SAC)

effectiveness levels of measures, which are implemented in a sequential order in terms of their respective cost-effect ratios, are purely additive. As a matter of fact, this may not be very true when some measures are combined. For example, in the case of Phosphorus, this is related with the mixture of types of P (including inorganic soluble forms, organic non and bioavailable forms and attached to particulates - see Mainstone, et al., 2000, for more information) and the controlled mechanisms by which the nutrient is targeted by each measure. In consequence, the following alternative scenarios which could affect the selection of measures can be observed: i) combined effectiveness of measures could be classified as intermediate whereas their combined effectiveness levels may not be felt fully, because part of the potential pollutant load has been mitigated by a most cost-effective measure implemented prior to the measure under consideration; and ii) measures may be alternative, meaning that their potential to reduce P loadings at farm level can be reduced to zero by a measure which has been already implemented.

Furthermore, our cost curves are static comparative presenting the net effects of changing from one environmental load to another. Thus, the method does not include time aspects and this is important when interpreting the results (NERI, 2006). Some BMPs may result in a more or less instant reduction in P loads whereas others may have a significant time lag between the time of implementation and the time of the resulting effect. This is an important factor that has not been included in our analysis and that may affect the selection of measures. Ultimately, this information could be included. However, it would be worth to assess its relative importance to decision making.

Our validation exercise strongly highlights the need for further research, especially on the assessment of combined effectiveness of BMPs. Additionally, there is a need to collect Scottish relevant estimates of costs and effectiveness of BMPs to reduce water diffuse pollution at farm level, including estimates which are readily applicable to a CEA. Some of the issues identified above (and throughout this chapter) should be considered so as to increase the applicability of the results. If all these issues are investigated and accounted for, the application of the abatement cost method for the selection of agricultural BMPs is a powerful tool to aid water policy in the implementation process of the WFD and in the detection of win-win situations.

Another factor of uncertainty surrounding the analysis which is very relevant for the achievement of water quality improvements under the WFD, is associated with the fact that nutrient loads reductions at farm level do not necessarily may deliver water quality improvements. There are many external factors that influence catchments responses to agricultural diffuse pollutants mitigation strategies; including climate change impacts. Kronvang et al., (2005) identified these possible sources of uncertainty, which have been summarised in table 4.10 below:

Table 4.10 Factors and mechanisms that may influence catchment responses following adoption of mitigation measures to combat P losses from agricultural areas

Factor or mechanism	***Impact***
Climate change impacts	Increases in temperature, precipitation and run-off will counteract and thereby mask catchments responses in P export due to increases in P mobilisation and P transport from source areas.
Reducing P input to agricultural areas implementing general measures	Delay in P leaching responses due to present P status and P saturation in topsoils and subsoils.
Changing in farming practice such as soil tillage changes	The P loss from fields low in erosion risks may increase due to releases of dissolved P as a consequence of no autumn tillage.
Riparian buffer zones	If not harvested, freezing of P from dead plant material may be a source of dissolved P. Storage of P in buffer zones may become a P source on the longer term through stream bank erosion.
Restoration of natural sinuosity in river channels	Increased stream bed and bank erosion for shorter or longer time periods and hence input of particulate P.
Restoration of wetlands, inundated riparian floodplains, irrigated riparian areas, etc.	Release of P from sediments enriched in P from former agricultural inputs may counteract the benefits from deposition of particulate P for shorter or longer time periods.
Retention in rivers, lakes and reservoirs	Increased temperature and precipitation as a consequence of climate change will reduce the natural P buffering capacity of lakes and reservoirs. Reductions in P loading to lakes and reservoirs may increase the P release from sediments being formerly enriched in P.

Our study does not take into account the impact of most of these external factors, as they are exogenous to the farmer (this is one of the main drawbacks of the abatement cost method). Only a degree of variability on soil conditions has been included, as we are interested in analysing the extent of the costs of implementing measures to reduce P loads at farm level. However, from a water policy perspective, there is a need to consider, identify and quantify the different mechanisms counteracting the possible benefits of BMPs adopted at farm level. These mechanisms can delay or mask the final effect at water body/catchment scale for indefinite (shorter and longer) time periods and would obviously affect the design and success of any policy/regulatory instruments aimed at reducing agricultural pollution in order to obtain good status before the end of the first river basin management plan in Scotland (i.e. design of emissions limits at farm level, etc). Farmers' abatement efforts to reduce diffuse pollution are indeed highly dependant on how regulatory agencies would associate agricultural nutrient losses with the achievement of the directive's objectives. Accordingly, this raises the following questions; how environmental standards to achieve good status will translate into enforceable farm loads reductions or how far on the abatement cost curve farmers will be asked to go. From an economic perspective, the question remains as to whether it should be the farmer who bears the mitigation costs of reducing the impacts that these external factors may have on water quality.

4.5. Conclusions

The adoption of Best Management Practices (BMPs) will be a critical component of programme of measures to achieve GS in Scotland, especially for the control of agricultural diffuse sources of water pollution. As a preliminary assessment of disproportionality, it is essential to understand the economic implications of adopting different mitigation strategies.

The abatement cost method is a management tool which offers a way to present detailed and transparent information on the financial costs and effectiveness of multiple available options for reaching environmental targets. Despite the existence of important methodological challenges, in particular with regard to some of the assumptions employed, our study proves that this method can be applied to the economic analysis of mitigation options to reduce farm losses of main diffuse pollutants. However, some of the challenges ahead mainly relate to the collection of Scottish primary data (which can be easily applicable to cost curves), the need for further research on the behaviour of different diffuse pollutants and the assessment of the combined effectiveness of mitigation options for their selection.

Our results suggest that P load reductions can be obtained at no extra total costs to the farmer. In this instance, the need to increase farmers’ awareness and education about the potential benefits associated with the implementation of different BMPs could prove fundamental for the achievement of good status and the ability of the Scottish agricultural sector to cope with the WFD requirements, as there is significant scope for the achievement of win-win situations.

The possible implications for policy is that these results, compared with a financial viability assessment of the farm, may offer a basis to grant time-frame derogations under the WFD by encouraging negotiations between farmers and regulators. Different definitions and measures of affordability are investigated in the following chapter of this thesis (i.e. chapter 5), which is concerned with the second test of disproportionality. Overall, chapter 5 investigates how different levels of abatement costs would impact the financial viability of the farm.

In addition, the study presented in this chapter provides an indication for the selection of the most cost-effective combination of BMPs for different farm systems, which ultimately will be relevant for Programme of Measures to achieve good status. Finally, it can be argued that this framework could be used as a starting point to assess the level of farm support to deliver water quality objectives under the new land management contracts in Scotland. However, for the justification of standard-setting derogations, further research work needs to include the economic estimation of the environmental benefits derived from abatement options.

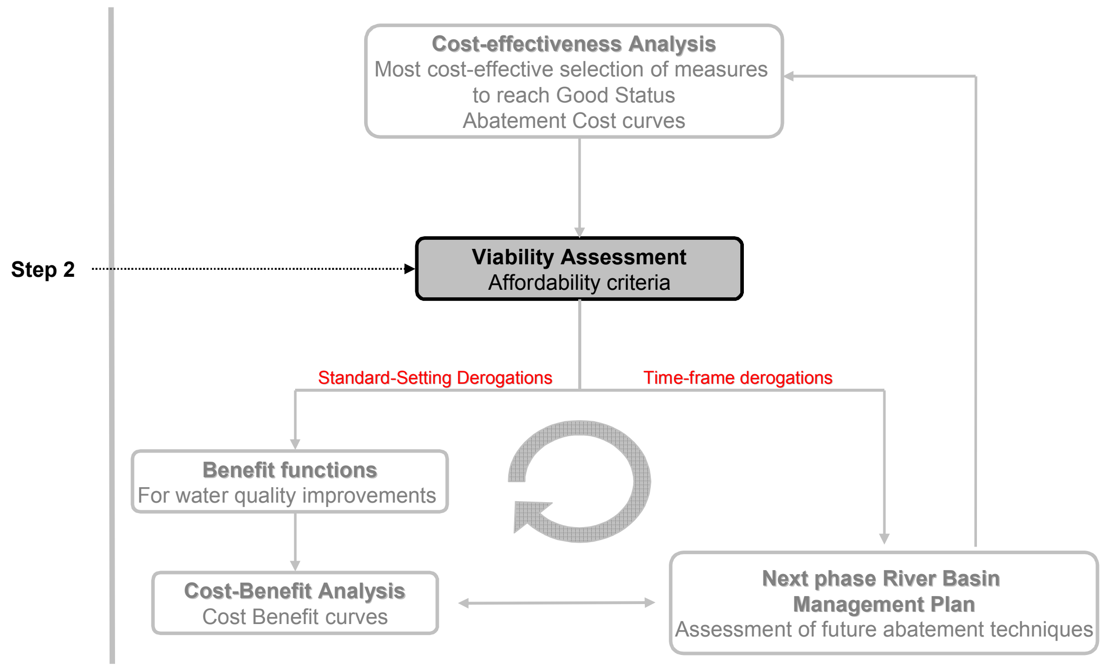
CHAPTER 5

FARM VIABILITY ASSESSMENT AND DEFINITIONS OF AFFORDABILITY

The purpose of this chapter is to assess the likely financial impacts for a typical Scottish farm of adopting different diffuse pollution mitigation strategies in order to achieve water quality improvements. This analysis is relevant for the development of a practical methodology to assess derogations for the implementation of the WFD in Scotland.

The present chapter explores in practice the second test for the assessment of disproportionate costs, following the theoretical methodological steps as identified in Lago et al. (2007) and chapter 3 of this thesis, in order to aid water policy in the decision making process of granting exceptions under the WFD. Figure 5.1 illustrates a summary of the methodology proposed, and highlights the topic covered in this chapter.

Figure 5.1 Methodological steps for the assessment of disproportionate costs - focusing on the viability assessment



The first step of this methodology was covered in the previous chapter, whereby information was provided about the most cost-effective selection of measures to achieve good status and the financial costs to the farmer of achieving different nutrient mitigation levels. Cost-

Effectiveness ratios for the selection of measures and abatement cost curves for P loads reductions at farm level were developed.

The second step of this methodology is concerned with the assessment of how these costs would impact the financial viability of the farm, with the underlying objective of investigating different definitions and measures of affordability for the practical definition of disproportionate costs.

5.1. Background

Agriculture is the most significant and controversial water user in most EU countries, as it is associated with both water quality environmental concerns and problems of poor water use management (Lago et al., 2006). Across the EU, agriculture is seen as the sector that creates the biggest challenges to meeting the requirements of the recently enacted Water Framework Directive - WFD (Herbke et al., 2005). In Scotland, for example, the environment protection agency (SEPA) has identified agriculture as a significant cause of diffuse water pollution, and clearly, as the dominant diffuse pollution pressure affecting rivers. It is estimated that diffuse pollution currently results in up to 23% of the water bodies in Scotland being of poor ecological quality, and it is now a more significant source of pollution than point sources in most water bodies (SEPA, 2005a,b).

Certainly, the WFD was enacted to answer public concerns about the water environment and aims to deliver social benefits by bringing water ecology and chemical conditions across Europe to Good Status. However, there is increasing anxiety amongst water users about the overall costs of reaching those environmental targets described in the Directive. This is especially true of those economic sectors, like agriculture, that believe themselves to be at both ends of the carrot and stick approach used by the Commission. On one hand, public support provided to agriculture in Europe dominates most of the EC's annual budget (European Commission, 2006a) and on the other hand, agriculture is one of the most regulated sectors of the economy, in terms of, for example; environmental protection (i.e. Water Framework Directive, Nitrates Directive, Groundwater Directive, Integrated Pollution Prevention & Control (IPPC) Directive...) or animal welfare (i.e. EU Meat Chicken Welfare Directive), etc.

Industry concerns about compliance costs were also present in the formulation of the WFD whereas, alongside its ambitious environmental objectives, exemptions or derogations to their realisation were included. Exemptions ought to be based on an economic criteria and should only be granted if costs to reach Good Status are judged to be too high or disproportionate (European Commission, 2002). Inevitably this provision may be invoked by some industries, with ensuing debate about the legitimacy of exemptions being claimed on this basis. As a matter of fact, required changes in farming practices in order to achieve the Directive's objectives are expected to be substantial. In England, DEFRA (2007a) has estimated that WFD related policy options will impose costs of around £200 million on the agricultural sector alone.

At the centre of the disproportionality debate under the WFD is the choice of tools for the analysis of the widespread socio-economic and financial impacts of delivering water quality improvements. Whilst some academic attention has been given to the comparison of the financial costs and social benefits of reaching good status (Bateman et al., 2006; Hanley and Black, 2006; De Nocker et al., 2007), little attention has been paid to the question of affordability - or the evaluation of the financial impacts to individual water users - and to the degree to which costs to reach Good Status may damage their financial viability or sustainability.

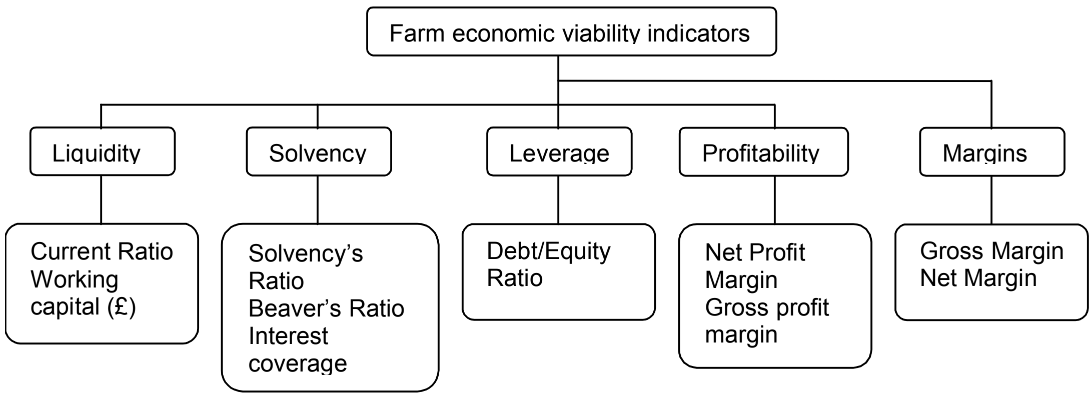
This chapter covers this issue, its purpose is to evaluate different methods of assessing the likely financial impacts of adopting different pollution mitigation strategies to achieve water quality improvements. Accordingly, we undertake an examination of two different practical definitions of affordability at farm level which are relevant to European water policy: i) the use of farm financial indicators to assist in the decisions-making process about derogations; and, ii) an assessment of impact on profits as a measurement unit of changes in farmers' welfare.

5.2. Part I - Farm viability assessment[17]

Policy guidance documents on the economic tools and methods to assess whether individual exemptions to the achievement of environmental targets should be granted, often highlight the use of financial indicators as a measure of affordability (USEPA, 1995; DEFRA, 2005c; European Commission, 2006b). This section aims to evaluate the suitability of such information for the economic analyses which have been proposed (in chapter 3) in order to inform policy choices for the application of Article 4 on exemptions to achieve Good Status under the WFD. In particular, we consider an application to the Scottish agricultural sector.

In the agricultural economics literature, financial indicators and evaluation criteria have been used for the development of prediction models of farm financial performance; in order to: i) assess the economic viability of farms; ii) explain and predict various degrees of financial health and; iii) for loan repayment purposes (Argiles, 2001). Figure 5.2 illustrates an outline of the most common financial indicators used in the literature.

Figure 5.2 Farm financial indicators used in the literature



Modified from: Bright and Florey, 2005

[17] Note that this analysis is at the farm level. In order to assess the financial viability of the sector as a whole (e.g. for comparison with other sectors) as a result of compliance with WFD targets other tests would also need to be considered to depict a more complete picture. These tests would basically involve an assessment of the market structure in agriculture (including a description of the extent of the market, an estimation of elasticity in prices for agricultural products and assessment of competition amongst farms and different types of farm systems; For more information see EC, 2006b). Regulators need to be aware that the structure of the market can further influence the ability of the sector/farm operator to pass on compliance costs to customers and/or suppliers. Nevertheless, evidence suggest that farm businesses have no market power in terms of price setting and that in general, the sector is not characterised by rapid technological change (DEFRA, 2007b). It is also expected that farm measures to reduce farm diffuse pollutants would not have a negative impact on innovation or competition between farms (DEFRA, 2007b).

The main indicators can be briefly defined: i) Liquidity, which measures the ability of a farm business to meet its current financial obligations, without disrupting its on-going business operations; ii) Solvency, which measures the amount of debt and other expenses obligations relative to the amount of equity invested in the farm, thus, providing an indication of the ability to repay all financial obligations if assets were sold; iii) Leverage, which provides an indication of the extent to which a farm business already has fixed financial commitments and therefore, provides a proxy of how much money the business can borrow. Finally, iv) Profitability measures the extent to which a farm business generates a profit from the use of its resources.

For the assessment of disproportionality under the WFD, we can consider a definition of viability (in the broader sense) as the ability of a farm to exist on a profitable basis over a long period of time (European Commission, 1991; Saunders and Cumberworth, 1992). However, profitability may not tell the whole story about the financial health of farm businesses, i.e. profit tests may indicate that the farm will continue to maintain profit levels typical for the sector after compliance with regulations, but the Debt/equity ratio could indicate that the farm may have problems raising the required capital through debt, to pay for those investments which would therefore affect profits sooner or later. Widespread financial impacts of compliance are often difficult to identify and assess. Accordingly, any investment in environmental protection may be considered unaffordable when a farm is in poor financial condition. We aim to explore if this statement is demonstrable and applicable to the agricultural sector.

The first part of this chapter is concerned with exploring the suitability of applying farm financial performance indicators as one of the data requirements for the assessment of what constitutes disproportional under the WFD. Accordingly; a financial characterisation of farming in Scotland, applying a multidimensional financial criteria to identify farms in poor financial conditions, is first undertaken. The application of the method and the lessons for water policy are critically appraised. Secondly, other issues to consider for technology adoption at farm level which are relevant for the implementation of BMPs at farm level and which are normally excluded in financial assessments are investigated. (e.g. technical efficiency, subsidies and other sources of off-farm income).

5.2.1. Methodology

There are a few main methodological problems associated with classifying farms according to different categories of financial health based on the use of single indicators. First, the precise measures of the financial condition of the farm sector are difficult to identify (Shonkwiler and Moss, 1993). This is mainly due to the different types of farm systems and their different characteristics, which complicates their aggregation. Secondly, classifications often only use one parameter as a descriptor. As an example, Santarossa (2003) used the deviation from the mean long-term debt for the financial pre-classification of Scottish farms. If a farm was either consistently below the mean or gradually moving away from the mean, then it would be classified as failing while the opposite would result in a farm being classified as financially healthy. This type of approach has been regarded as arbitrary and incomplete (Argiles, 2001). Ultimately, a farm's financial position depends on many factors, as for example; farm profits (net or gross) do not capture changing real assets values, making any financial classification based solely on these two indicators incomplete and difficult to interpret. Thirdly, further criticism relates to the usual reliance on one-year agricultural data, which does not account for the marked variability that comes from the pronounced random effects that affect farm activity (Argiles,2001) .

To overcome these issues, we have applied a multidimensional financial criteria, which allows us to classify Scottish farms according to their financial performance over a period of eight years. The main advantage of using a multidimensional criteria is that we are able to assess financial performance based on a series of indicators grouped together as opposed to conventional farm accounting techniques which independently assess each financial indicator (e.g. liquidity, solvency, leverage...). Thus, individual indicators may give a clear description of the sector on specific financial factors, but are not suitable for drawing conclusions about overall farm financial performance. Furthermore, by analysing financial performance over eight years, we aim to reduce the variability of random effects that characterises the analysis of farm financial data.

Melichar's criteria (1985)[18] is used in this study to determine the financial position of each of the farms surveyed in the FAS accounts. The main benefit of applying this procedure is that it combines a series of financial tests to classify farms into four financial position categories.

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[18] Emanuel Melichar, a former chief agricultural economist for the US Federal Reserve, developed a multi-dimensional financial classification system to identify those farms that were under financial stress and therefore, unable to repay agricultural loans during the farm debt crisis of the 80s in the US.

i) good; ii) fair; iii) stressed; and iv) vulnerable (see annex V for further details). According to Melichar (1985), farms which are classified as vulnerable are currently experiencing financial trouble and may not survive in the short term, while those in the stressed group are heading for trouble unless returns improve or management practices are changed. Farms classified as fair may not be able to sustain their equity or fully service their debt in the long term, but they are not in serious trouble presently; and finally, those falling in the good position are not experiencing financial distress.

The definitions of the financial ratios applied in this study in order to implement Melichar's scheme are as follows:

D/A	=	*Debt to asset ratio* equals total debt as a percentage of total farm assets.
ROA	=	*Return on assets* equates to net farm income before interest payments minus the value of unpaid labour as a percentage of total farm assets.
ROE	=	*Return on equity* equals net farm income minus the value of unpaid labour and interest payments as a percentage of equity.
Equity	=	Total farm assets minus debt.

5.2.2. Data and Results

Data from the farm agricultural accounts (FAS) survey in Scotland, between the years 1997 to 2004, was used to classify each individual farm operation according to their financial performance based on a multidimensional criteria (RERAD, several years).

FAS, is the Scottish equivalent of the Farm Business Survey in England and Wales. It is an annual survey undertaken by the Scottish Agricultural College (SAC) on behalf of the Scottish Government under which a range of management accounting information on all aspects of farmer's and grower's businesses is collected. The survey uses a sample of farms - a varying sample of up to 500 accounts are collected each year - that is representative of the national population of farms in terms of type, size and regional location. In this chapter, a financial characterisation according to Melichar's criteria of all the farms included in the FAS survey between 1997 and 2004 is presented. Later on in this section, we will prove that statistically it does not matter if we analysed only those farms that are surveyed each year on a regular basis or all the farms in the sample. We will prove that if new entrants to the sample are randomly selected, the distribution of their financial position is equivalent to that of unique entries for the whole time period and on an annual basis.

Table 5.1 below introduces descriptive statistics for the variables used in the financial position classification for the different types of farms used in the FAS accounts. The mean D/A, ROA and ROE ratios computed for all the types of farms in the sample are 6.21, -7.01 and -7.73%, respectively. The average farm in the sample has £441,124 in equity. We observe low mean values for ROA and ROE in the sample. This is mainly due to the fact that without accounting for subsidies, on average, most of the farms surveyed have negative net income. In principle, Net Farm Income (NFI) tracks financial viability, so that if financial returns are consistently negative, any farming system would be unsustainable (OECD, 2001).

Table 5.1 Descriptive statistics for the variables used in the financial position classification for the sampling period (1997-2004)

		FARM TYPES								
VARIABLE		Cereals	General Cropping	Dairy	LFA Specialist Sheep	LFA Cattle	LFA Cattle & Sheep	Lowground Cattle & Sheep	Mixed	Totals
Debt to Asset Ratio (%)	Mean	8.65	6.64	5.12	4.95	5.74	6.71	11.86	6.58	6.29
	StdDev	12.17	9.84	6.09	5.99	6.48	7.24	19.23	8.48	8.26
	Max	116.14	61.08	55.44	68.31	53.63	58.31	81.62	59.05	116.14
	Min	0.01	0.06	0.20	0.01	0.02	0.04	0.02	0.06	0.01
Returns on Assets (%)	Mean	-8.13	-5.50	-7.22	-7.41	-6.19	-7.36	-5.53	-8.10	-7.01
	StdDev	15.09	11.17	9.25	10.42	8.82	9.24	7.02	10.45	10.25
	Max	58.52	72.75	17.53	34.62	21.26	24.12	15.47	14.52	72.75
	Min	-83.48	-150.15	-68.25	-56.17	-45.53	-49.56	-22.70	-63.12	-150.15
Return on Equity (%)	Mean	-9.33	-6.70	-7.68	-7.84	-6.71	-7.95	-8.44	-8.88	-7.73
	StdDev	20.93	20.92	9.83	11.22	9.83	9.99	16.80	11.58	13.16
	Max	125.11	105.34	19.33	35.56	23.15	25.35	16.71	17.78	125.11
	Min	-134.24	-369.47	-72.48	-60.38	-75.05	-49.83	-123.52	-72.98	-369.47
Equity (£)	Mean	541,447	631,314	624,528	296,768	326,118	349,849	380,223	472,059	441,124
	StdDev	510,460	520,535	347,576	202,841	215,725	245,679	289,339	441,312	371,262
	Max	2,444,778	2,679,517	1,953,813	1,374,931	1,273,199	1,705,056	1,139,563	2,791,996	2,791,996
	Min	-70,020	695	68,479	25,089	42,562	29,610	19,137	22,467	-70,020
Sample Size		313	423	559	428	880	702	70	548	3923

Table 5.2 summarises financial performance positions by type of the 3,923 farms included in the sample between 1997 and 2004 in Scotland. Of the 3,923 farms surveyed in the FAS accounts in this period, 640 (16.3%) fall in the good financial position category. The majority of farms, 2,255 (57.5%) fall into the fair category, 387 (9.9%) are in the stressed category and the remaining 641 entries (16.3%) are described as vulnerable. Adding together those farms in stressed and vulnerable financial condition, a striking quarter of all the farms surveyed are having or have had financial difficulties over the analysed period. For different farm systems, a third of the Cereals and Lowground cattle and sheep types of farms are in poor (i.e. stressed or vulnerable) financial condition.

Table 5.2 Financial position classification based on Melichar's criteria for all farms by type during the sampling period (1997-2004)

	Financial Position								
	Good		Fair		Stressed		Vulnerable		Total
Farm type	No. Farms	%	No. Farms	%	No. Farms	%	No. Farms	%	No. Farms
Cereals	47	15.0%	172	55.0%	9	2.9%	85	27.2%	313
General Cropping	72	17.0%	264	62.4%	21	5.0%	66	15.6%	423
Dairy	76	13.6%	348	62.3%	59	10.6%	76	13.6%	559
LFA Specialist Sheep	59	13.8%	257	60.0%	63	14.7%	49	11.4%	428
LFA Cattle	185	21.0%	491	55.8%	98	11.1%	106	12.0%	880
LFA Cattle & Sheep	122	17.4%	368	52.4%	81	11.5%	131	18.7%	702
Lowground Cattle & Sheep	13	18.6%	33	47.1%	3	4.3%	21	30.0%	70
Mixed	66	12.0%	322	58.8%	53	9.7%	107	19.5%	548
TOTAL	640	16.3%	2255	57.5%	387	9.9%	641	16.3%	3923

5.2.2.1. *Results by size*

Figure 5.3 Percentage of farms in poor financial condition by type and size

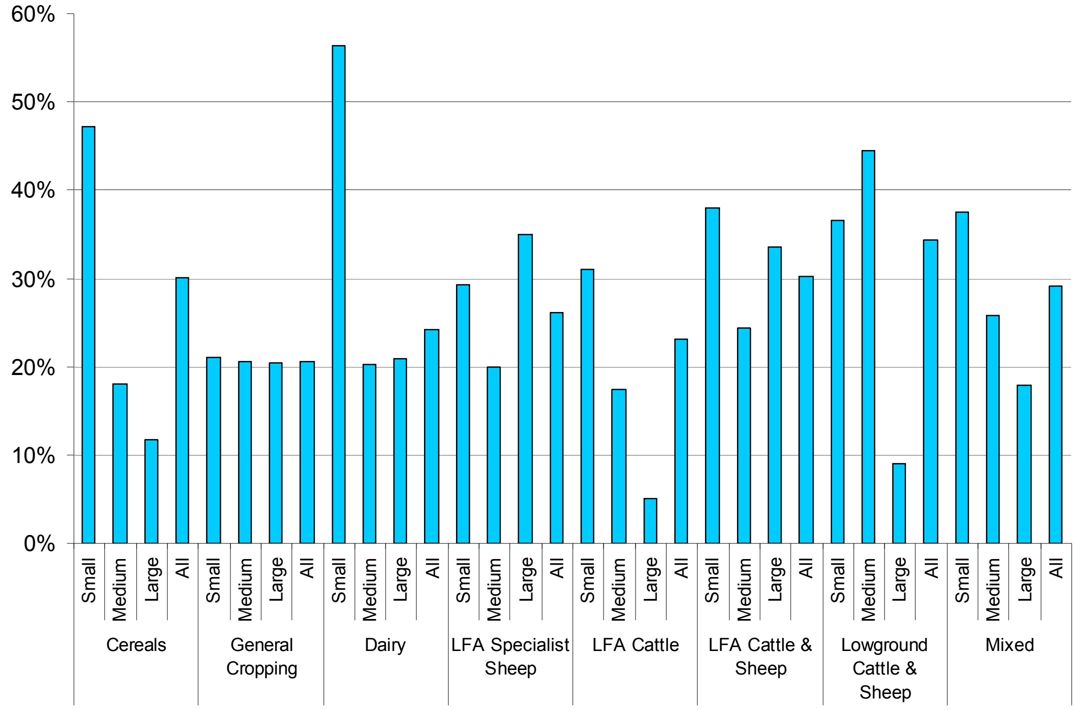


Figure 5.3 illustrates the percentage of farms classified as being in stressed and vulnerable financial health by size and type of farm, according to Melichar's criteria. These results suggest that there is a relationship between the size of the farm and overall financial health; with a larger number of smaller farms being in an unhealthy financial condition. An exception to this rule applies to the less favoured areas (LFA) specialist sheep farms, as within this type of farming, a bigger proportion of larger holdings have financial troubles in

comparison with smaller farms. Alternatively, in the general cropping sector there does not appear to be a clear relationship between size and financial performance.

5.2.2.2. Testing the distribution

One common problem with the statistical analysis of panel data, which is relevant for assessing the financial condition of each specific farm per year using the FAS accounts, is that these types of datasets do not include a unique sample of farms over the whole of the analysed period (Durguner, 2007), thus leaving an unbalanced panel dataset (cross sectional/time series). The FAS survey consistently attempts to keep records on the same farms every year, but sometimes, sample numbers vary and need to be updated for diverse reasons, such as; some farms exit the survey meanwhile others entered the sample to cover for those that left or failed to give their financial information on time for the annual results. Accordingly, aggregation of results from different years needs to be considered carefully, as the results may be biased by the sample selection of farms for each given period. To check for this type of bias, we have compared the statistical distribution of financial position classification based on Melichar's criteria for all the farms included in the farm accounts versus those records that only include a unique same sample of farms between 1997-2004. A total of 128 farms were found to be included in FAS for each single year between 1997-2004, compared to 490 farms which is the average number of farms covered by the survey in any given year.

A chi-square test, as introduced by Reza-Hoshmand (1998), was used to test the hypothesis that the statistical distribution between both samples was equivalent. The test confirms that the null hypothesis cannot be rejected at the 0.01 level of significance (table 5.3 below). This indicates that for the purposes of the financial analysis using FAS datasets, it does not matter if we analyse those farms that are surveyed each year on a regular basis or all the farms in the sample, as if new entrants to the survey are randomly selected, the distribution of their financial position would be equivalent. The same procedure was used to compare the distribution between same types of farms and between years. All of the results suggest that the null hypothesis cannot be rejected, apart from LFA cattle, which failed the test for similarity of distribution between same types of farms. However, this may be due to the small sample size of farms in this category (only 10 farms were present every year between 1997-2004 in the unique farms sample).

Table 5.3 Financial distribution of farms: Unique farms represented in the FAS dataset from 1997 to 2004 versus total sample

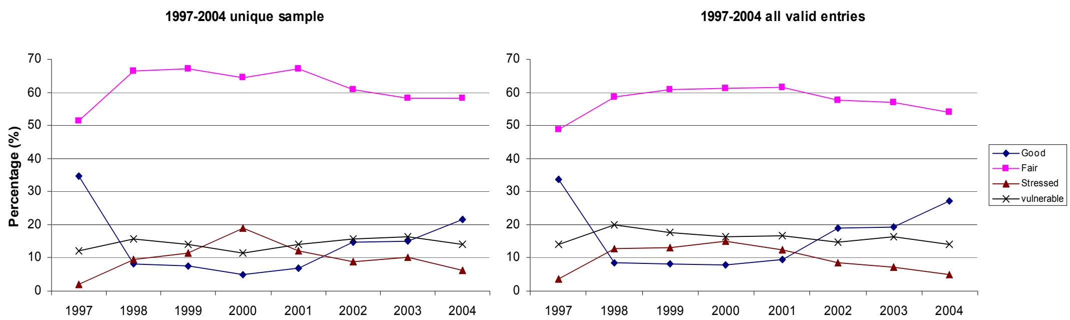
	All valid entries (1997 -2004)		Identical sample of farms (1997 -2004)	
Financial Position	No. Farms	Percentage*	No. Farms	Percentage
Good	640	16.3%	180	14.2%
Fair	2255	57.5%	780	61.7%
Stressed	387	9.9%	125	9.9%
Vulnerable	641	16.3%	179	14.2%
Total	3923	100.0%	1264	100.0%

* Based on a chi-square test, the hypothesis that the distribution of financial performance from the two samples is the same cannot be rejected at the 0.01 level of significance.

5.2.2.3. *Financial performance over time*

The similarity of the distributions becomes clear when we analyse the distributions of overall financial performance for all types of farms over time. See Figure 5.4 below. An analysis of the trends over time would also help us to briefly examine the factors that have affected the financial health of the Scottish agricultural sector over this period.

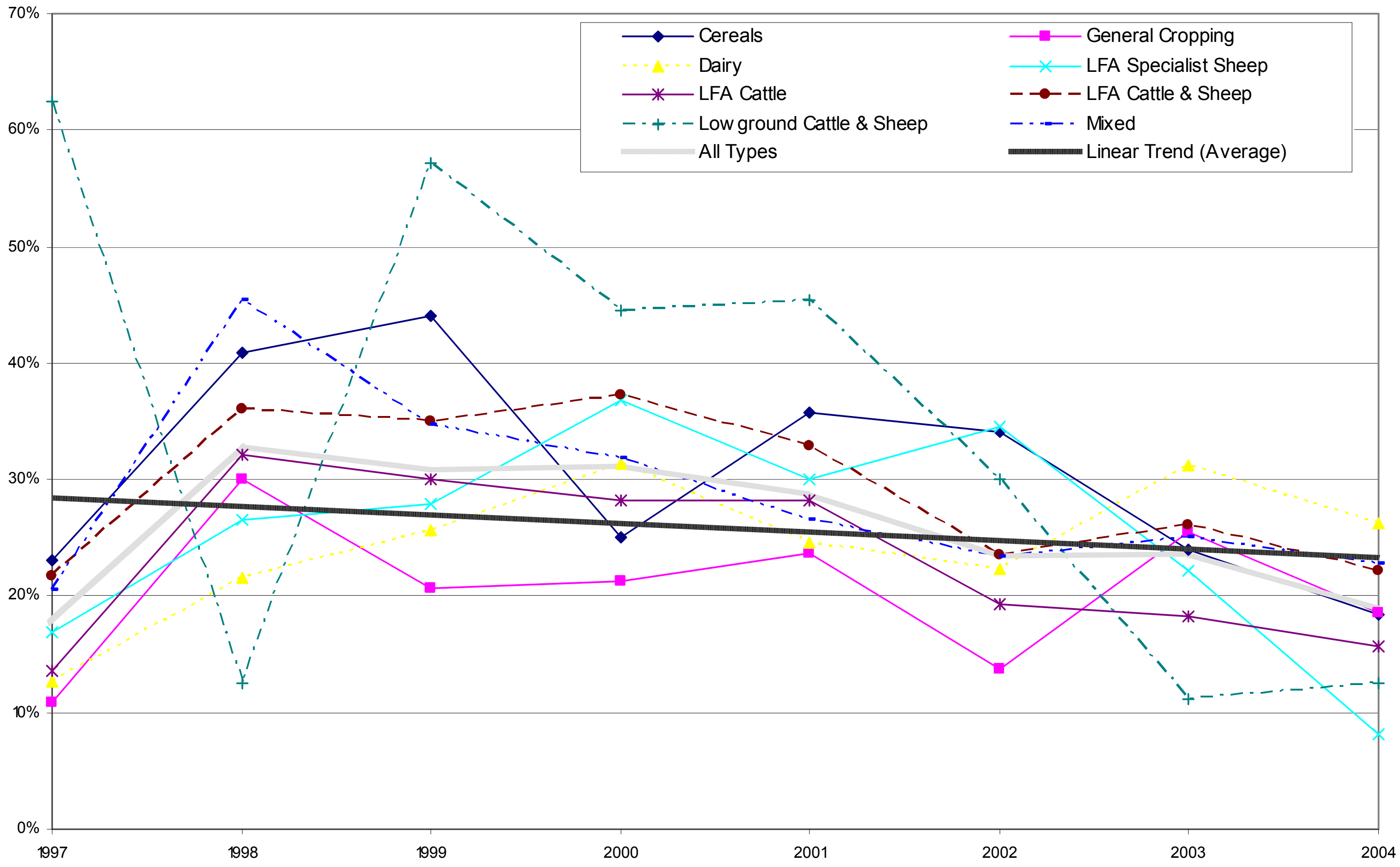
Figure 5.4 Financial criteria distribution over time for unique samples compared to all valid entries in the sample (1997-2004)



The number of farms in vulnerable condition has remained more or less constant over the period of time analysed. This may come from the fact that it is difficult for farmers to move out of this category. Alternatively, the number of farms under financial stress has been declining since the year 2000, suggesting that there seems to be an inverse relationship between farms in fair and good condition. Overall, the proportion of farms in good financial condition has been steadily rising since the year 2000.

Figure 5.5 shows the overall proportion of farms considered in poor financial position by type of farm system between the years 1997 to 2004. Variations in lowground cattle and sheep can be explained due to a small sample size. A linear trend line to the average farm confirms that overall numbers of farms in poor financial condition have been slowly declining from 1997 to 2004. As an example, the total number of farms in poor financial position has experienced a steady decline from around 35% of all farms ion the sample in 1998 to around 20% in 2004.

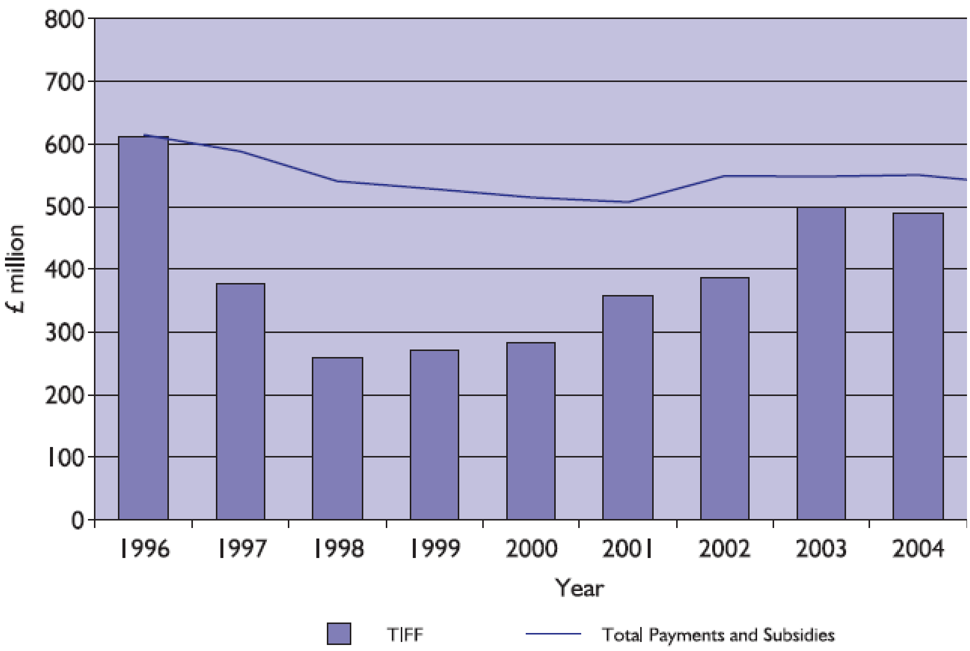
Figure 5.5 Percentage of farms by type classified as being in poor financial condition (1997-2004)



One of the reasons for the falling number of farms in poor financial condition may well be explained by the fact that some of these farms may have gone out of business during the time period under consideration. Unfortunately, the way information is collected through the census makes this hard to confirm. This is because the census records the existence of a holding and not a business. Therefore, a business could have failed and a more successful business taken over the holding but as far as the census is concerned this would not show up as a change in the number of holdings in Scotland.

In addition, another explanation for our results might be that the agricultural sector as a whole is currently recovering from two recent shocks; the crisis of 1997 and the foot and mouth outbreak in the year 2001. The former is more relevant for our analysis, as it marks the baseline year for this exercise.

Figure 5.6 Total income from farming and direct subsidies: 1996-2004



From: RERAD, 2007

Total Income from Farming (TIFF) in Scotland is estimated to have fallen by 42 per cent in 1998 compared to 1997, as indicated in figure 5.6 below. This figure represents annual estimates of direct subsidies received by farmers compared with Total Income from Farming

(TIFF) in Scotland between the years 1996 and 2004. TIFF[19] is an aggregated measure of farming income which represents the income to farmers, partners, directors, spouses and family workers. RERAD (1998) indicates that the fall in TIFF in 1997 was due to lower prices received by farmers for all major commodities, with weak world markets, the relative strength of sterling, compounded by poor weather conditions being major factors underlying the reductions.

The estimates presented in Figure 5.6 agree with the results of our analysis. This figure is also useful to illustrate the role of subsidies in farm income.

Thus far in this section, we have explored the suitability of applying farm financial performance indicators as one of the data requirements for the assessment of what constitutes disproportional under the WFD. In the US, financial impact analysis based on the application of individual indicators is recommended as the main analytical tool in order to demonstrate derogations to environmental objectives under the Federal Water Regulations (USEPA, 1995).

The application of a multidimensional financial criteria to assess farm financial viability provides a better understanding of affordability than individual indicators. It is a useful tool to identify those farms that might experience trouble as a result of tighter controls and increasing compliance costs (not only applicable to water policy but other regulations). Nevertheless, this type of test alone should be carefully considered at water policy level and not be given exclusivity when deciding about derogations/exemptions. The financial health of a farm describes performance but fails to tell the whole story about exactly how much a farm can "afford" to pay or whether it is "able" to pay for water quality improvements. More importantly, it does not provide information about the likely adoption at farm level of Best Management Practices to mitigate water pollution.

As an initial assessment criteria for disproportionate costs, it could be assumed that those farms regarded as being in poor financial condition would find it difficult to change management practices or invest in mitigation technology, as they are presently under financial stress. However, with the adoption of BMPs, there is also scope to increase

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[19] TIFF is business profits plus income to workers with an entrepreneurial interest. This measure is preferred over Net Income from Farming as an income aggregate. TIFF conforms to internationally agreed accounting principles, required by both UK national accounts and by Eurostat. TIFF and its underlying estimates of outputs and inputs feed into the national accounts and ultimately the national estimate of GDP (SEERAD, 2001).

production efficiency whilst lowering production costs (Valentin et al, 2004). Unfortunately, financial tests alone do not provide information on the efficiency of current production; or the impact of other sources of income on production levels and efficiency. This leads to the issue of how to treat technical efficiency and other sources of income to the farmer (i.e. off-farm activities/income and subsidies). We now briefly explore how the literature considers these factors and their possible implications when using financial performance indicators as a rule of thumb to grant derogations for the WFD.

5.2.3. Other issues to consider for technology adoption at farm level

Economic efficiency at farm level is often subdivided into technical efficiency and allocative efficiency (Fernandez-Cornejo, 2007). A farm is technically efficient if it uses the minimum possible levels of inputs to produce a given level of output, given the technology available (Kumbhakar and Lovell, 2000). Opposed to allocative efficiency, where the farm produces a given output using the best (minimum cost) input proportions given prevailing input prices (Fernandez-Cornejo, 2007).

Agricultural economists tend to focus on the analysis of the factors that affect technical efficiency. As an example, a technically inefficient farm would fail to produce the maximum attainable output with the amount of inputs used and the best technology at the farmer's disposal. Accordingly, there would be some scope to increase revenues and improve overall financial performance by changing management practices or investing in new technology. In this example, if the water quality regulator uses this farm's current financial performance results to grant exemptions, arguably, they would be rewarding its inefficient production, which inevitably would raise inequality issues and concerns with other farmers.

Generally, the literature associates technical efficiency with farmer's managerial ability and experience and it is defined relative to an "efficient frontier"(Fernandez-Cornejo, 2007). All farms operating on the efficient frontier are classified as 100 percent efficient with an efficiency score equal to 1. Therefore, farms using more inputs to produce a given output level than those on the efficiency frontier are inefficient and their efficiency score is less than one.

What follows is a review of those studies that have covered issues of technical efficiency for Scottish farms. Their results and findings are relevant to our own study, as all the papers reviewed have used the FAS accounts data.

Santarossa, (2003), assessed the technical efficiency of Scottish farms in order to determine possible sources of inefficiencies and identify those factors which could be targeted by agricultural policy in order to improve efficiency in the future. Using a stochastic parametric approach to measure the distance that would represent the deviation from optimum for each of the sampled farms, Santarossa found a mean value of technical efficiency of 63.2% for all types of farm systems, where 60% of the farms had an efficiency score of over 60%. He also observed variability in technical efficiency values depending on type and size of farms (Santarossa, 2003). This paper mixed the results of the prediction of financial distress with the findings from the technical efficiency analysis and concluded that those farms that are not financially healthy in Scotland, tend to operate below capacity thereby reducing efficiency.

Additionally, Barnes (2006) used the stochastic production frontier approach to assess the technical efficiency of three types of farming systems in Scotland (cereals, dairy and sheep) over the period 1989 to 2004, using again the FAS data. He observed mean technical efficiency values between the analysed period of 75%, 71% and 80% for cereals, dairy and the sheep sectors respectively. Furthermore, technical efficiency of the two livestock sectors decreased over this period, whereas cereals grew. The paper also investigated those factors which would impact on technical efficiency by sector in Scotland; i) For cereals, type of tenure and area were found to be strongly significant negative factors on technical efficiency; ii) For dairy farms, a move to LFA (Less Favoured Area) and an increase in area were found to have a negative effect on efficiency. Alternatively, a movement to ESA (Environmentally Sensitive Area) status, an increase in the debt ratio and experience were found to have contributed to improvements in efficiency throughout this period. Finally, iii) for the Beef sector, a move from non-LFA to LFA status was regarded as having a strong negative effect on efficiency, as does an increase in farm area. A positive factor to technical efficiency was farmer's experience.

The last study reviewed, Revoredo Giha, et al. (2007), focussed on the assessment of farm performance as a means to identify drivers for improvement for the achievement of farm sustainability in the context of recent CAP reforms. The authors took a different methodological approach from the previous two papers and focused instead on cost-efficiency of Scottish farms and the analysis of those factors that are important in its explanation. They used a distance to a stochastic multi-output cost frontier approach to derive cost-efficiency indices for different Scottish farm systems. Data was drawn from the FAS survey for the period (1997-2004). Results found mean values for cost-efficiency indicators of 58%, 49%, 46%, 39% and 31% for dairy, mixed farms, cattle and sheep, specialist sheep and cereal and general cropping farms respectively. Amongst the factors that strongly influence cost-efficiency, farm size and location were identified. Inconclusive results were offered by land quality (based on production criteria), type of tenancy, product diversification and financial situation variables. Finally, the authors suggest that there seems to be a link between high levels of direct subsidies and levels of inefficiencies.

Unfortunately, FAS data does not provide information about off-farm income. Consequently, we have to resort to an American study to assess the possible impact that other sources of off-farm income would have on overall farm efficiency and technology adoption. Fernandez-Cornejo (2007) studied this relationship using USDA's Agricultural Resource Management Survey (ARMS) data between the years 1996 to 2001. This study found that small farms improve their economic performance by compensating for the scale disadvantages of their farm businesses with more off-farm activities. Additionally, it was found that off-farm work reduces farm level technical efficiency but surprisingly in this case, they found that it increases technology adoption (only if new technology is associated with savings on farm managerial time).

The three Scottish studies used a variety of methodologies to assess the technical efficiency of Scottish farms. However, a common important conclusion shared by all of them is that technical efficiency varies between farm systems in Scotland, as does the different factors for inefficiencies. Additionally, there is considerable scope for technical (cost-efficiency) improvement across the industry. This is a fundamental conclusion for the assessment of disproportionate costs under the WFD, whether a farm is in poor financial condition or not, regulators also need to assess if there is scope to increase its technical efficiency before granting exceptions to achieve water quality targets. Furthermore, factors that affect farm efficiency need to be evaluated and regulations tailored accordingly. Additionally, there is potential scope to use different policy instruments (varying levels of farm support or encouragement of off-farm activities), in order to change attitudes towards adoption of new management practices. These issues need to be further investigated.

5.3. Part II - Impact on profits as a measure of affordability

In this section, impacts on profits as a general measure of affordability for the assessment of disproportionate costs for the implementation of the WFD are explored. A simple optimisation model to assess the likely changes to farm profits as a result of implementing the most cost-effective selection of BMPs to reduce P loadings at farm level is applied to the Scottish dairy and arable sectors[20].

5.3.1. Background

The welfare economics literature uses changes in profits as a substitute for measurements in welfare for a competitive firm (Just et al, 2004), always assuming that the ultimate goal of the firm is profit maximisation. Additionally, profitability usually plays a key role in technology adoption (MacDonald et al., 2007). In this respect, an affordability criteria for the WFD, from a producer perspective, could be set at the point where the firm's profits would fall below a certain "acceptable" level.

In practice, the academic literature is unclear as to where to set thresholds for numerical measurements of producer's welfare or how to define what constitutes "acceptable" in terms of profit losses. Gorlach and Pielen (2007) explored the suitability of applying different welfare indicators for the assessment of disproportionate costs for the practical implementation of the Directive. They regarded the analysis of costs in relation to a percentage loss in profits for an individual firm in comparison with the sector's average profits as only a "partially suitable" criterion for assessing a firm's ability-to-pay for water quality improvements, mainly because it is not known what exact percentage in profit losses could be regarded as disproportional.

However, a criteria which has not been explored in the water policy literature for the implementation of the WFD is not what constitutes "acceptable" but instead, what is "unreasonable". By re-formulating the question it is possible to find in the literature

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[20] The selection of these two farm types as examples for the simulation exercise has mainly been driven by two factors: 1) as shown in table 4.4, arable and dairy farms illustrate the highest fertiliser application rates in the baseline scenarios for the more general main cultivation systems and animal farming categories. 2) as this exercise is highly dependent on the accuracy of the goodness of fit models illustrated in table 4.9, which show good fit results for arable and arable plus manure and bad fits for dairy and beef types of farms, it was decided that for the simulation two examples covering good and bad fits were needed for illustrative purposes.

theoretical limits of what definitively constitutes disproportional under some circumstances[21]. In general, if net margins (revenue minus costs of production) or gross margins (revenue minus variable costs of production) become negative as a result of a planned investment, it can be assumed that the new investment would not be affordable, as the firm would no longer be profitable. In the case of the WFD, this represents a reason to apply for derogations.

Two complementary agricultural decision-making techniques on proposed investments can be found in the production economics literature; breakeven analysis and partial budgeting. These techniques are widely popular with farm managers and economists (Barnard and Nix, 1979). Using breakeven analysis, it is possible to assess the possible financial impact of BMPs by treating mitigation options as a series of investments, and assessing the impact that their costs would have in farm profits. The methodology is quite simple in principle, by keeping all other factors of production fixed, including prices, output and fixed and variable costs, and allowing profits to only be influenced by the costs of the proposed investment, it is possible to assess if profits would become negative as a result (Dillon, 1993). As an application to our case study, a measure of affordability for the WFD, it would be possible to predict when as a result of the sequential implementation of the most-cost effective selection of BMPs (identified in chapter 4), farm profits would fall to/below zero.

Other authors suggest that net margins would not be the ultimate measure of farmers wellbeing and that it is changes in gross margins, also referred to as producer surplus or Quasirent (Just et al 2004), which would need to be assessed instead. This has also been distinguished as the opportunity costs of the land (Wossink and Osmond, 2002). Applying this measure would imply that an investment would be justified as long as the farmer can still cover the variable costs of production. This is because fixed costs associated with fixed factors of production are sunken, meaning that a fixed expenditure could not be avoided even if the farm goes out of business.

These definitions are not mutually exclusive, and it is normal practice to assume that if a firm cannot cover all its costs of production, its financial sustainability in the long run may be at risk. Alternatively, if a firm cannot cover its variable costs of production, it may not be

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[21] Note that in this analysis we are not dealing with issues of technical efficiency or off-farm income, it can be argued that farm profits (or net farm income from farming) may not be the ultimate measure of farmer's welfare and others sources of income should be considered. However, other sources of income do not come from farming practices and questionably should not be considered.

viable in the short run, as the enterprise is definitely no longer profitable (Barnard and Nix, 1979). It is important to consider in the analysis that the farm is not given any credits and that economies of scale do not apply, e.g. that the farmer cannot live off non-farm income or that the farm does not form part of a wider enterprise, when farming may not be profitable in isolation but cost-effective in combination with other enterprises. Nonetheless, economic profit can be measured on a unit-of-output basis by referring to the gross and net margins; both definitions can be used to delimit thresholds by which policy can identify and prioritise action over those farms that would need to be further investigated for exemptions under the WFD.

The underlying objective of this study is to simulate and analyse how the associated financial costs of implementing a sequential combination of the most cost-effective P mitigation measures (BMPs) at farm level would impact on farm profits. Ultimately, we are concerned with the exploration of changes in profits as a suitable indicator of affordability for the assessment of disproportionate costs under the WFD.

Accordingly, we have developed a static optimisation model with the core objective of farm profit maximisation. This model would allow the exploration of the relationship between farm profits, optimal levels of P abatement and farm factors of production. The model considers farm pollution as a negative factor of production and the implementation of BMPs as a short term farm investment which would increase production and at the same time reduce nutrient emissions both to certain limits. Two different scenarios are considered with and without government intervention (i.e. regulations). The first part of this section introduces the mathematical description of the model, which is followed by an examination of the results and the sensitivity analysis.

5.3.2. The model: optimal levels of production and abatement

Let us consider the following profit maximisation problem for a farmer who has a pollution problem and has to decide the optimal level of pollution to abate whilst optimising levels of production.

Furthermore, let us suppose that the presence of pollution affects the production of the farmer (e.g., reducing his productivity) but also higher production levels result in a greater level of pollution in the farm. Thus, linking pollution abatement techniques with agricultural production, assuming that pollution can impact on the environment but also hinder farm production. This is understood to be the case in reality and it was a necessary assumption, in order to consider the potential scope of BMPs to offer efficiency savings. We shall consider that it is possible to reduce the level of pollution by making an investment (paid in several equal instalments and which will be part of the farm fixed cost, i.e., it will not affect the production marginal conditions and therefore the level of productivity). Thus, we shall hypothesise that the level of pollution is given by the function H, which depends positively on the production Q and negatively with respect to the level of abatement A, such as:

$$H = \left(\frac{Q}{A}\right) \tag{1)}$$

Where The "net" production, Q^*, i.e., incorporating the effect of the pollution is given by:

$$Q^* = \left(\frac{1}{1+\alpha \cdot H}\right) \cdot Q \tag{2)}$$

The level of pollution affects the productivity of the farm (i.e., the greater the level of pollution (*H*), the lower the net output (*Q**). It should be noted that *H* is affected by a parameter α that takes values from 0 to ∞. When $\alpha=0$, pollution does not affect production and net production is equal to the "gross" production, and no investment should be made to reduce its level or incur any cost. However, if $\alpha>0$, then pollution affects productivity and a cost to reduce it might be justified on profit maximisation grounds.

Let us suppose that the cost of abating pollution is a function $h(A)$, which satisfies the following properties: $h(0)=0,\ h'(A)>0, h''(A)\geq 0$. In addition, the farmer faces a variable cost function that depends on the level of gross output $c(Q)$ and satisfies the following properties: $c(0)=0,\ c'(Q)>0, c''(Q)\geq 0$ and has a fixed cost equal to *F* (i.e., this is a short term problem[22]).

Thus, the profit maximisation problem of the farmer can be considered in two different ways depending on whether the level of abatement is chosen by the farmer or is exogenously imposed by the government.

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[22] Note that for a long term optimisation problem fixed costs become variable costs.

5.3.2.1. Self-imposed level of abatement

There are two main assumptions in production economics relevant to our analysis. The first is that the primary goal is to maximise profits. The second is that a risk-return trade-off exists whereas a higher expected profit is often accompanied with greater risk (Gondonou et al, 2001). The latter is not very relevant to the implementation of Best Management Practices to reduce farm loads of diffuse nutrients. Under the WFD, GS is not a choice and BMPs are not a potentially risky investment decision but rather a least cost way to comply with water quality standards.

A simplified version of the standard profit function is illustrated below. Farm profits (π) are a consequence of total revenues minus the total costs of production (Just et al., 2004). Total Revenues are a function of the net output (Q) and Price (P). Total costs of production can be disaggregated into i) Total Variable Costs of Production *(c(Q)),* which are a function of the output produced and ii) F, which illustrates total fixed costs.

$$\pi = PQ - c(Q) - F \tag{3}$$

If the level of abatement is chosen by the farmer then replacing (1) in (2) and then into equation 3. We have that the profit maximisation problem if the farmer decides to implement the different BMPs in order to reduce P loads, is such as:

$$\begin{aligned}
&\underset{Q,A}{Max} \quad \pi = P \cdot \left(\frac{1}{1 + \alpha \cdot \frac{Q}{A}} \right) Q - c(Q) - h(A) - F \\
&s.t. \\
&Q, A \geq 0
\end{aligned} \tag{4}$$

The first order conditions for the profit maximisation of the farm are given by equation 5:

$$\begin{aligned}
\frac{\partial \pi}{\partial Q} &= \frac{P}{1 + \alpha \frac{Q}{A}} - \frac{\alpha P Q}{A\left(1 + \alpha \frac{Q}{A}\right)^2} - c'(Q) \equiv 0 \\
\frac{\partial \pi}{\partial A} &= \frac{\alpha P Q^2}{A^2\left(1 + \alpha \frac{Q}{A}\right)^2} - h'(A) \equiv 0
\end{aligned} \tag{5}$$

and the optimal values for A and Q are given by (assuming that the marginal cost function is constant equal to $\mathrm{c}'(\mathrm{Q}) = \mathrm{c}$ and the marginal cost of abatement is linear equal to $\mathrm{h}'(\mathrm{A}) = \mathrm{h} \cdot \mathrm{A}$:

$$Q^{*} = \frac{\left(\sqrt{P} - \sqrt{c}\right)^{3}}{\alpha^{2} h \sqrt{c}}$$

$$A^{*} = \frac{\left(\sqrt{P} - \sqrt{c}\right)^{2}}{\alpha h} \qquad 6)$$

Note from the expressions for Q^{*} and A^{*} (where the asterisk indicates optimal value) that as far as the price is above the marginal cost, their values are positive. The second order sufficient condition for a maximisation of the profit function can be expressed as a function of two variables in terms of the discriminant D (i.e., eq. 7)

$$D = \pi_{QQ} \pi_{AA} - \pi_{QA}^{2}$$

$$D(Q^{*}, A^{*}) > 0 \qquad 7)$$

$$\pi_{QQ}(Q^{*}, A^{*}) < 0$$

Replacing the values of Q and A into the profit function (eq. 4), we can obtain the optimal value of profits for the farmer when he decides over the two variables:

$$\pi^{*} = P \cdot \left(\frac{1}{1 + \alpha \cdot \frac{Q^{*}}{A^{*}}} \right) Q^{*} - c \cdot Q^{*} - \frac{h}{2} \cdot \left(A^{*} \right)^{2} - F \qquad 8)$$

5.3.2.2. Profit maximisation under government influence

If the level of abatement is set exogenously by the government, then the farmer's problem is to find the output that maximises his profits given the level of $\overline{A}$. This is:

$$\begin{aligned} &\underset{Q}{Max} \quad \pi = P \cdot \left(\frac{1}{1+\alpha \cdot \frac{Q}{\overline{A}}} \right) Q - c(Q) - h(\overline{A}) - F \\ &s.t. \\ &Q \geq 0 \end{aligned} \tag{9)}$$

The first order condition of the problem is given by equation 10:

$$\frac{\partial \pi}{\partial Q} = \frac{P}{1+\alpha \frac{Q}{\overline{A}}} - \frac{\alpha P Q}{\overline{A}\left(1+\alpha \frac{Q}{\overline{A}}\right)^2} - c'(Q) \equiv 0 \tag{10)}$$

The optimal level of output is then given by:

$$Q^* = \frac{\overline{A}\left(\sqrt{P}-\sqrt{c}\right)}{\alpha\sqrt{c}} \tag{11)}$$

The second order sufficient condition for a maximum is:

$$\frac{\partial^2 \pi}{\partial Q^2} \equiv -\frac{2A^2 P\alpha}{(A+Q\alpha)^3} \leq 0 \tag{12)}$$

Replacing the optimal output into the profit, we obtain the profit function for different levels of abatement imposed by the government. This is given by:

$$\pi^* = P \cdot \left(\frac{1}{1+\alpha \cdot \frac{Q^*}{\overline{A}}} \right) Q^* - c \cdot Q^* - \frac{h}{2} \cdot \left(\overline{A}\right)^2 - F \tag{13)}$$

$$\pi^* = \overline{A}\frac{\left(\sqrt{P}-\sqrt{c}\right)^2}{\alpha} - \frac{h}{2} \cdot \left(\overline{A}\right)^2 - F \tag{13')}$$

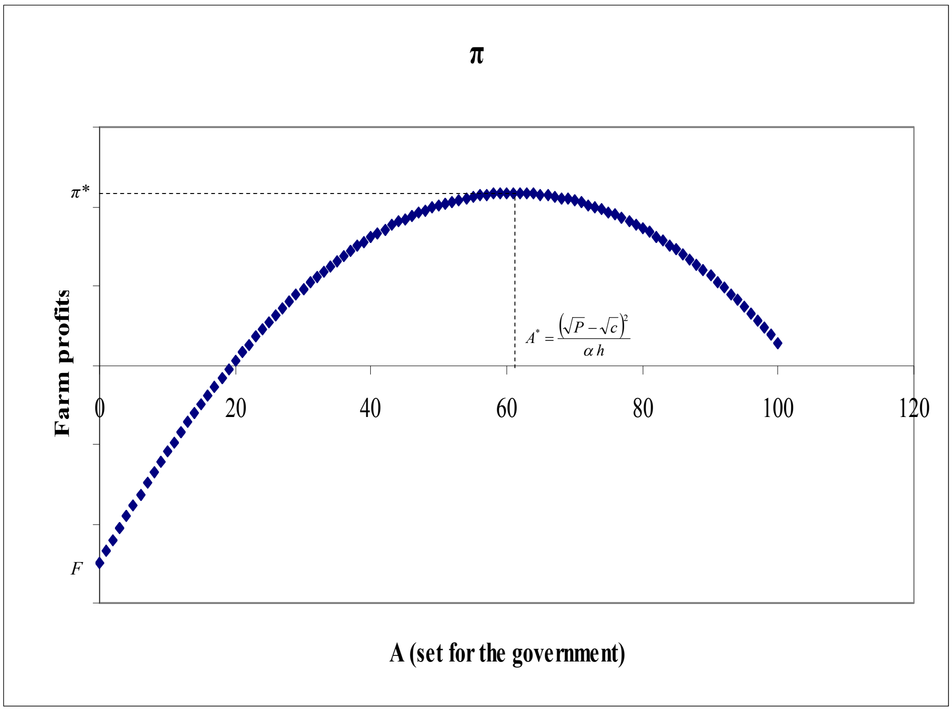
This mathematical explanation can be used to introduce a definition of disproportionate costs based on welfare economics theory. If profits are considered as the ultimate measure of farmer's welfare, there would be a case for derogations if profits fall below a certain threshold. In the agricultural economics literature, this threshold is often identified at the breakeven point, which is the point where profits would fall to zero as a result of new investments. In this case, the long-term breakeven value of $\overline{\mathrm{A}}$ is given by (14):

$$\widetilde{A} = \frac{\left(\sqrt{P}-\sqrt{c}\right)^2 - \sqrt{\left(\sqrt{P}-\sqrt{c}\right)^4 - 2Fh\alpha^2}}{\alpha h} \qquad 14)$$

5.3.3. Discussion

The following figure (figure 5.7) presents the relationship between the level of abatement and profit for specific values for the cost of abatement, the variable cost function and fixed costs.

Figure 5.7 Relationship between farm profits (π) and levels of abatement (A)



The line in the graph represents different farm profit maximisation levels in response to different levels of abatement, which are exogenously imposed by the government. In theory farm profits are expected to rise as a consequence of a cost-effective sequential implementation of Best Management Practices, as they may reduce nutrient loads at farm level whilst offering efficiency savings through better informed and more efficient farm management decisions (Valentin et al., 2004). Accordingly, as the government sets standards (or in this case different levels of A), farmers' assumed response would be to adjust production to those levels of output that maximise farm profits for any given level of Abatement set by the government A_G (x-axis in the graph). However, there is a fundamental difference between levels of A imposed by the government and what would be the optimal level of abatement (A^*) if farmers freely chose whether to reduce nutrient loads. Assuming that farm management decisions are perfectly informed and that farmers' are profit maximisers, farmers would chose those levels of A that maximise profits by matching optimal levels of production (Q^*) with optimal levels of abatement (A^*).

Under this optimality framework, we can simulate the inefficiencies of setting different levels of A outwith the optimal level A^* for the farmer. If $A_G<A^*$, the farmer initial reaction would be to move along the curve until they maximise profits (π^*) by producing and abating at optimal levels Q^* and A^* respectively. Accordingly, at points below A^*, it does not really matter where the standards are set (level of A_G), as in theory, farmers would be expected to seek profit maximisation and move production to match A^* and Q^* nonetheless. The problem arises when $A_G>A^*$, at these levels of abatement the farmer would be asked to abate beyond optimal levels of production. Thus, at any points of A beyond A^*, the farmer would be asked to produce outwith his/her efficiency point.

5.3.4. Data sources

IGER/ADAS (2007) data on the financial cost and effectiveness of each individual BMPs to reduce P loads at farm level for English and Welsh arable and dairy farms for two types of soils (e.g sandy and clay loam soils) was adjusted to reflect Scottish conditions (procedures for data adjustments are illustrated in chapter 4). Mitigation options were ranked in terms of their cost-effectiveness ratios. Annex VI offers an outline of the BMPs considered.

Solving the following cost minimisation problem (chapter 4):

$$Min(C(P)) = Min\sum_i c_i x_i \qquad 15)$$

Subject to the following constraints:

$$\sum_i P_i x_i \geq R_p P_0$$

16)

$$0 \leq x_i \leq 1$$

where; x_i relates to the abatement activity implemented (this is represented as a fraction with a value between 0 and 1, as shown in the last restriction); c_i denotes the costs related to the implementation of nutrient abatement activity x_i (£/ha/year); $P_{i,}$, is P abatement related with the implementation of activity x_i (reduction in kg of P loss at farm level per hectare). The constraints in equations (16) state that P abatement at source level should be more than or equal to a certain fraction R of the initial emissions described in P_0. By changing this restriction on nutrient-emission reductions between 0 and 100%, it is possible to assign % reductions in relation with baseline levels to each individual measure. Cubic polynomial abatement cost curves were fitted using the OLS method. Table 5.4 (below) presents the results of the best fit regression model for the following functional form (eq. 17).

$$f(y)=y_0+aA+bA^2+cA^3 \qquad 17)$$

Variable cost functions for different sizes of arable and dairy farms were estimated assuming a linear relationship between variable costs of production with respect to one aggregated variable of output. The main output from arable farms includes an aggregate of the production vectors for cereals, potatoes and other cash crops. For dairy farms, milk production has been considered. Costs and outputs by farm type were computed directly from the FAS data (RERAD, several years). The information covered the years 1997 to 2004, which resulted in an unbalanced panel data set (cross sectional/time series) of 358 individual farms. The farms were selected on the basis that they were included in the 03/04 survey and included in the sample for at least five years between 1997 to 2004 (similar approach as used in Revoredo et al, 2007). DEFRA's input price indices for the UK were used for agricultural materials, services and capital, as an estimate of those prices paid by FAS farmers (DEFRA, 2006b).

Table 5.4 BMPs abatement total cost functions regression coefficients and standard errors (£ equivalent per year/% reduction in P loadings at farm level)

ARABLE

Sandy Loam Soils:					
Adjusted R^2 =	*0.9768*	*F statistics=*	*197.7116*	*p of F statistics*	*<0.0001*
Parameters	Coefficient	SE	T	P	
y0	-5.6341	18.0104	-0.3128	0.7603	
a	4.7284	1.4159	3.3394	0.0066	
b	-0.1826	0.0369	-4.9444	0.0004	
c	0.0019	0.0003	7.0905	<0.0001	
Clay Loam Soils:					
Adjusted R^2 =	*0.962*	*F statistics=*	*178.0996*	*p of F statistics*	*<0.0001*
Parameters	Coefficient	SE	T	P	
y0	-37.8754	24.0875	-1.5724	0.1333	
a	5.0454	4.2166	1.1966	0.247	
b	-0.1957	0.1166	-1.6778	0.1107	
c	0.002	0.0008	2.5157	0.0216	

DAIRY

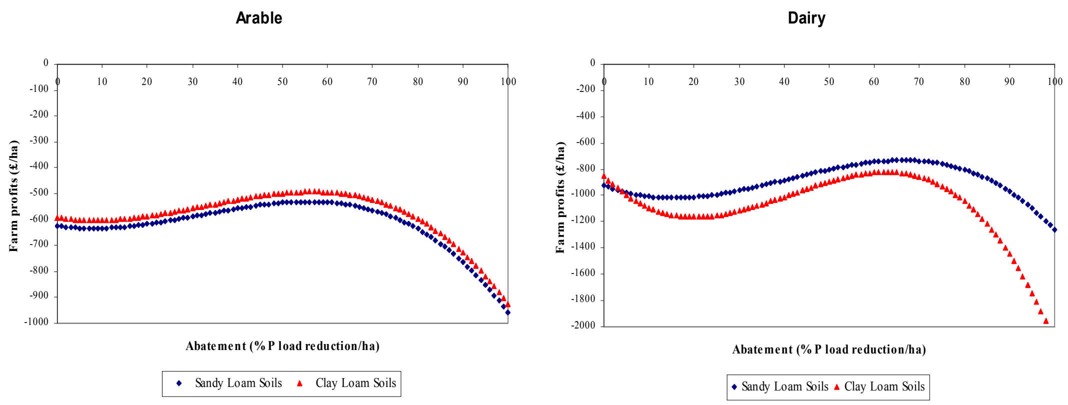
Sandy Loam Soils					
Adjusted R^2 =	*0.5654*	*F statistics=*	*8.3733*	*p of F statistics=*	*0.002*
Parameters	Coefficient	SE	T	P	
y0	-58.6709	153.3433	-0.3826	0.7078	
a	15.8459	15.3216	1.0342	0.3186	
b	-0.5285	0.3628	-1.4567	0.1673	
c	0.0043	0.0023	1.8893	0.0797	
Clay Loam Soils					
Adjusted R^2 =	*0.7295*	*F statistics=*	*25.2758*	*p of F statistics*	*<0.0001*
Parameters	Coefficient	SE	T	P	
y0	-131.236	140.9395	-0.9312	0.361	
a	34.8919	16.7248	2.0862	0.0478	
b	-1.1304	0.3976	-2.8433	0.009	
c	0.0091	0.0025	3.5967	0.0014	

Values for the constant alpha of 10 and 1 for arable and dairy farms respectively, were given. This aimed to take into consideration that pollution (through excess inputs and poor management practices) has a greater negative impact on arable farms production than in dairy farms. The sensitivity of the model to different values of alpha is analysed in the last section of this paper.

5.3.5. Results

The following graph (Figure 5.8) illustrates a simulation of the impact on profits (in relation to fixed costs) of achieving different P loads reductions per hectare for arable and dairy farms under two types of soils. According to our results, the impact of different levels of abatement set by the government on arable farms would be more or less similar, independently of the type of soil (e.g. sandy and clay loam soils). For dairy farmers, we can expect some variability between types of soils, as P abatement on clay loam soils seems to have a greater impact on farm profits compared to sandy loam soils.

Figure 5.8 Simulated impact of different levels of P abatement on farm profits (in relation with fixed costs) for all types of Arable and Dairy farms under soil heterogeneity per hectare



Bearing in mind the assumptions imposed in this simulation, results suggest that for initial levels of P abatement at farm level, the most cost-effective selection of BMPs would increase farmers' profits. This is because reducing excess inputs (i.e. reduce fertiliser applications) or protecting livestock access to water courses, just to mention a few examples, can increase productivity through a more efficient use of farm resources and subsequently, reduce farm nutrient losses. Though, need to bear in mind that for greater levels of P abatement, profits would fall dramatically for both arable and dairy farms. Therefore, it is important for water policy in light of the WFD, to consider the following: i) where is the optimal level of P abatement and ii) where it would be the farm's breakeven point.

Table 5.5 introduces the results of our model. An estimation of the optimal levels of abatement without government intervention for different sizes of arable and dairy farms is presented. It is important to consider that under profit maximisation, farmers aim to produce at optimal levels, which is the point where production and abatement are maximised, $Q*$ and $A*$ respectively. This point has its importance when compared with exogenously set levels of abatement and potentially, can reflect the possible success of enforcing P loads reductions at farm level.

As expected, our results show variability between different farm systems. For arable farms, optimal levels of abatement can be found at around 55% P loads reductions from baseline levels. For dairy farms this level is found to be around 65%. Surprisingly, our results do not offer much variability in optimal levels of P abatement for different sizes of farms and different types of soils.

Table 5.5 Optimal levels of P loads abatement for different sizes of Arable and Dairy farms under soil heterogeneity

								A*+		Breakeven Analysis+	
								SLS	CLS	SLS	CLS
		π_1^+	**R+**	**P+**	**C'(Q)+**	**FC+**	**α+**	(% P loads reductions)			
Arable											
	Small	-212.65	791.51	107.99	26.83	841.80	10	58.0	58.6	83.4	84.0
	Medium	-41.31	871.24	107.51	28.54	686.53	10	57.3	58.0	82.0	82.6
	Large	10.53	1101.52	108.44	40.37	727.17	10	54.1	54.9	74.3	75.3
	All	-77.13	922.04	107.93	33.44	740.08	10	55.9	56.6	78.7	79.4
Dairy											
	Small	-698.31	1436.21	18.73	9.04	1468.23	1	65.2	63.8	83.7	76.8
	Medium	-158.73	1524.47	18.78	7.65	1040.02	1	66.3	64.4	87.2	79.2
	Large	59.83	1848.13	18.79	7.15	1059.37	1	66.7	64.6	88.5	80.2
	All	-112.90	1681.03	18.78	7.42	1081.57	1	66.5	64.5	87.8	79.7

(+) π_1: Baseline Profits (£/ha); R: Value of production (£/ha); P: Price (£/unit**); C'(Q): Marginal Variable Costs; FC: Fixed Costs (£/ha); α:Constant; A*: Optimal level of Abatement (% P loads reductions); SLS: Sandy Loam Soils; CLS: Clay Loam Soils

(**) Price Units: Arable: Tonnes; Dairy: '00 litres of milk

For the definition of what constitutes unreasonable or disproportionate, it is important to evaluate what happens beyond optimal levels of abatement (right side of point A^* in figure 5.7). Table 5.5 also introduces the results of the breakeven analysis when in our simulation fixed costs are set equal to zero and profits fall to zero (e.g. gross margins per ha), as a result of investing in the most cost-effective mitigation technology to reduce P loads. Our results illustrate breakeven points on average for Scottish arable and dairy farms to be around 80% of P mitigation levels (table 5.5). Values vary (up to 10% differences) between types of farms, sizes and different soils, which is relevant if a fixed definition of disproportionate costs is applied across the sector, as we would expect different impacts. It is worth mentioning that breakeven points for arable farms seem to be lower for bigger holdings, which may well suggest that larger arable farms have greater scope for producing effectively than smaller ones, as levels of pollution seem to increase with arable productivity.

5.3.6. Sensitivity analysis

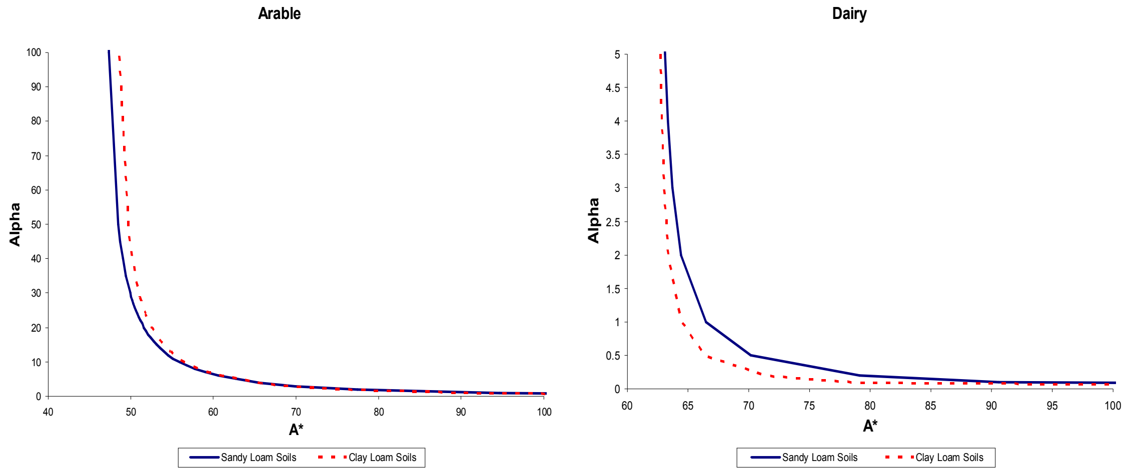
The model links pollution abatement techniques with agricultural production, assuming that pollution can impact on the environment but also hinder farm production. This is understood to be the case in reality and it was a necessary assumption, in order to consider the potential scope of BMPs to offer efficiency savings. Accordingly, for the mathematical formulation a constant named alpha was used to reflect the connection between levels of pollution, abatement techniques and production.

It was proposed in the theoretical explanation of our model that the level of pollution affects the productivity of the farm; for example, the greater the level of pollution H, the lower the net output Q^*. We also stated that H is affected by a constant α, which takes values from 0 to ∞. Accordingly, for very small values of α, pollution does not affect production and net production is equal to the "gross" production, and no investment should be made to reduce its level or incur any cost. However, if $\alpha > 0$, then pollution affects productivity.

It was impossible to find references in the literature that could provide an indication of real values for alpha, which highlights the need for further research on effective production in agriculture - in order to better understand the relationship between inputs (nutrients) and outputs (production and diffuse pollution). Nevertheless, values of α=10 and α=1 for arable and dairy farms, respectively, were used for our analysis. These values were chosen according to our own judgement and the information gathered in the previous chapter for the development of the cost curves.

As our results are highly sensitive to the value of alpha, it is necessary to evaluate the impact that different values of this constant would have on optimal levels of abatement. These are introduced in figure 5.9.

Figure 5.9 Sensitivity analysis for different values of alpha



As expected, small values of alpha offer greater optimal levels of abatement. This is because at low values of alpha, our model estimates that there are hardly any interactions between pollution and "gross" production. Alternatively, as values of alpha increase, optimal levels of A^* decrease up to a minimum level of optimal abatement. At this point, two main conclusions can drawn from figure 5.9: i) the impact of pollution in farm "gross" production is different for arable than dairy farms. For arable farms, A* levels stabilise in the 50% region for values of alpha beyond 30. For dairy farms, A^* takes levels of around 60 to 65% at alpha values beyond 2.5. And ii) there is little difference under soil heterogeneity.

5.4. Chapter conclusions

This chapter has presented an exploration of two different definitions of affordability; one which is based on an application of a multidimensional criteria of farm financial performance indicators and the other definition, which uses loss of farm profits as a result of compliance effort.

Essentially, farm financial performance indicators provide an incomplete picture of affordability. First, financial tests need to be aggregated in order to be able to draw conclusions about overall farm financial performance. Secondly, the fact that a farm might be in poor financial condition at the time of the analysis does not imply that it should be automatically be granted derogations. Other issues which are normally overlooked in the development of financial indicators include technical efficiency and other sources of income. These issues need to be accounted for before reaching a decision. Finally, farm financial indicators do not make any statement about the likely adoption of mitigation measures (i.e. BMPs) by the farmer or the possible financial benefits from their implementation (i.e. efficiency savings).

In practice, the approach outlined in the first part of this chapter could serve to identify those farms that are more likely to financially suffer from compliance. Nonetheless, the suggested framework could be useful in future ex-post assessments of the impact of regulations once the costs of compliance have been realised and accounted for in the FAS surveys. The main benefit of this system is that it exemplifies a detailed financial performance snapshot of the whole sector.

The analysis of the impact on profits as a measure of affordability opens up the possibility of linking profits with increasing levels of abatement (e.g. cost curves) for the assessment of disproportionate costs. The simple optimisation framework applied in the second part of this chapter simulates the relationship between farm profits, optimal levels of P abatement and farm factors of production. Results are in relation to the fixed costs of production under profit maximisation rules. The analysis suggests that there is scope for BMPs to increase farmers' profits at initial levels of abatement. In addition, this methodology offers some interesting applications for policy, such as; the assessment of optimal levels of abatement in relation to profits or allowing for the exploration of different definitions of affordability in terms of farm profit losses. In this analysis, affordability was set at the breakeven point for the farmer.

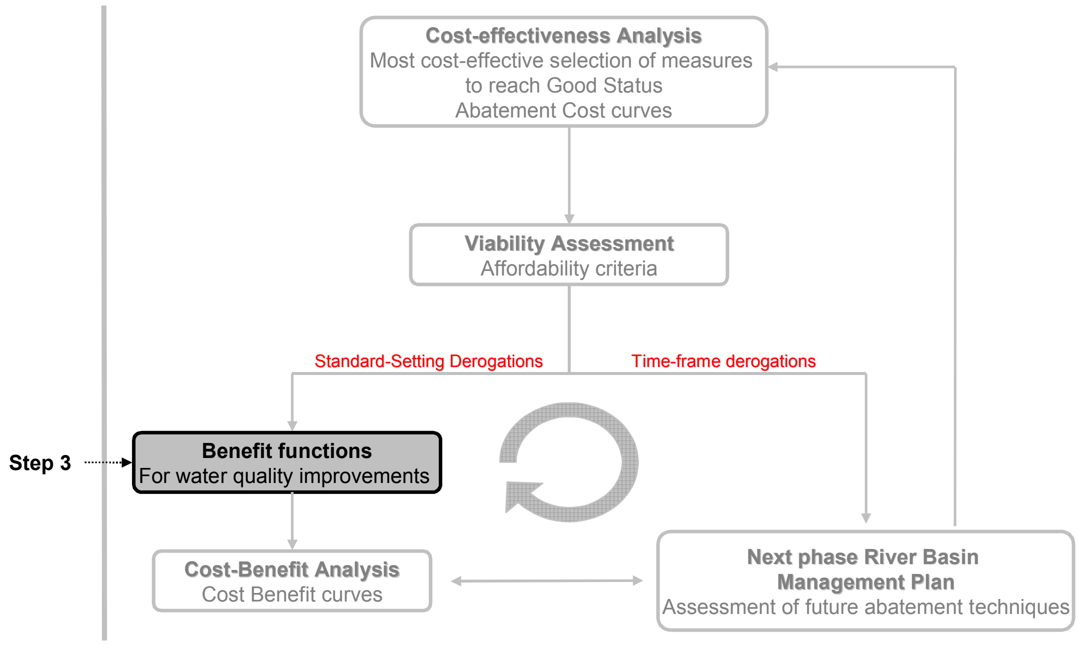
This chapter illustrates a very basic modeling approach in which most components of the analysis have been kept constant or oversimplified (e.g. variable cost functions of production for one aggregated variable of output, average quantities and prices for the selected types of farms). Nonetheless, this analysis can be used to highlight the potential application of more sophisticated dynamic optimisation approaches to assess disproportionate costs under the WFD (as long as the method is not applied for the selection of measures to achieve Good Status as introduced in chapter 4). This is clearly a topic for further research.

CHAPTER 6 DELIVERING GOOD STATUS IN SCOTLAND: METHODOLOGY TO DEVELOP BENEFIT FUNCTIONS FOR WATER QUALITY IMPROVEMENTS

Following on from the methodological steps for the practical assessment of disproportionate costs that were identified in chapter 3, we now come to an exploration of step 3, which is focused on the development of benefit functions for water quality improvements (figure 6.1).

Previous steps have explored issues in relation to the assessment of the financial costs of abatement (step 1 in figure 4.1) and in relation to the different ways to judge affordability problems in compliance with pollution reduction targets (step 2 in figure 5.1). Both these steps have clarified to some extent the theoretical and methodological considerations necessary for granting time-frame derogations, but as it has previously been shown, these tests alone depict an incomplete picture if standard setting derogations are considered.

Figure 6.1 Methodological steps for the assessment of disproportionate costs - Step 3: Developing benefit functions for water quality improvements



As previously introduced in chapter 3, standard setting derogations require the comparison of the financial costs of compliance with society's benefits for improvements in water quality. This is necessary if water regulators and practitioners are seeking economic efficiency in the decision-making process of granting standard based exemptions under the WFD (which is the argument defended in this thesis).

This chapter deals with the assessment of the overall benefits to society derived from the achievement of Good Status (GS). Estimates of the overall benefits of the WFD Programme of Measures (PoMs) to achieve GS are needed for the following reasons: First, from an economic efficiency point of view; the justification of derogations on the basis of less stringent objectives (which makes it necessary to know what gains in environmental quality can be achieved compared to the costs of abatement) needs a rational and comprehensive method to individually grant exemptions (i.e. CBA). Secondly, to assess at national level the extent of the economic impacts of implementing the Directive (i.e. RIA of implementing the Directive in Scotland). And thirdly, to prioritise action for environmental regulators.

Monetary valuation of non-market environmental goods is at the heart of research in environmental economics. This field has boomed in recent years with considerable advances in valuation methods and their application. However, whilst application of the results are becoming increasingly more important for the design and analysis of environmental policy (e.g. UK Treasury greenbook), detractors highlight limitations (e.g. costs of original studies, validity of the results, ethical and moral issues...) hindering widespread application in policy analysis. This is a debate not covered within the scope of this thesis.

In this chapter, we focus on the exploration of valuation methods to estimate benefit values for water quality improvements. These are briefly outlined in the introduction. The chapter has been divided into two separate parts. We begin part 1 by presenting the results of a benefits function transfer exercise which has been developed using original data from a recent water valuation study conducted in England and Wales (Baker et al, 2007). The applicability of benefits transfer is evaluated. We will show that even though the benefits transfer method may prove to be a valid alternative to answer our research question, and a "*quick*" and "*inexpensive*" way to inform policy decisions, its main weakness stems from the fact that it is impossible to assess the validity of the transferred values. This conclusion is the main argument employed in this thesis to justify the undertaking of an original valuation study in Scotland. The last part of this chapter presents the results of a choice modeling exercise, which slightly modifies the design applied in Baker et al, (2007), to elicit Scottish households' willingness to pay for improvements under the WFD. A justification for replication of Baker's et al (2007) approach for the benefits function transfer exercise and the choice experiment study design is offered in the introduction.

6.1. Introduction/background

The monetary valuation of changes in environmental quality is regarded as a difficult duty, mainly for three reasons; Firstly, environmental assets are often not directly reflected in markets due to their “externality” and “public good” nature. Secondly, there are frequently no natural units of measurement to classify these changes, and even when physical indices are available, these have to be related to individuals’ perceptions, which is often complicated and regarded as controversial. Thirdly, the forecasting of environmental quality changes is complicated by the fact that they are measured using bio-chemical and bio-physical classifications which are prone to change because scientific procedures are constantly evolving and being improved upon. This is not a problem for valuation per se, but poses a challenge for the generation of value estimates that are useful for policy purposes.

The overall aim of the proposed valuation exercise for this chapter is to find the non-market economic value of a change in environmental quality, which in this case consists of an improvement in water quality related to the definition of “Good Status”. Non-market valuation is based on people’s preferences for those changes, and values are measured either by a direct elicitation procedure or indirectly by analysing transactions in markets where preferences for an environmental good are assumed to influence the price of the marketed good. The value for the entire affected population is established by an exchange transaction reflected in the sum of each person's value for environmental improvements or in other words, the area under the demand curve of the environmental good that is improved.

There are two very well differentiated groups of non-market valuation methods: those based on elicitation of revealed or stated preferences. Revealed preference methods can be divided into Hedonic Price Method (HPM) and household production function approach.. Stated preference methods include the Contingent Valuation Method (CVM) and choice experiments (CE).

Table 6.1 below briefly outlines the main characteristics of those economic valuation methods which have been most widely used for the monetary estimation of benefits derived from water quality improvements[23]. This table illustrates the types of values covered, data requirements and main limitations for each method.

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[23] For a detailed review of these methodologies; see for example: Garrod and Willis (1999), Smith (2004). and Bateman et al. (2002) for CVM; Markandya et al., (2002); Garrod and Willis (1999) or Parsons (2003) for TCM;

Table 6.1 Summary of economic valuation methods for water quality improvements

	Method	Approach	Water Service Appropriate for method	Data needs	Limitations	Relevant examples - literature
Stated preference techniques	Contingent Valuation Method	People are asked to state WTP either in an open question or in yes/no question formats	All use values and non-use values (e.g. drinking water, fishing, protecting species)	Survey with scenario description and questions about WTP for specific services	Potential biases, e.g. due to hypothetical nature of scenarios or WTP elicitation format	Georgiou et al, 1998 Loomis et al, 2000 Green and Tunstall, 1991 Hanley and Kristom, 2002 Hanley et al, 2006 Baker et al, 2007 Bateman et al, 2006
	Choice experiments	WTP based on trade-offs between environmental attributes and cost expressed in choices in a hypothetical market	All use values and non-use values	Survey with scenario description and choice questions about alternative options that differ in their water quality levels and cost	Like CVM, needs correction of possible biases	Hanley et al, 2006 Baker et al, 2007 Johnston, 2007 Morrison and Bennett, 2004
Revealed preference techniques	Travel cost method	Estimate demand curve from data on travel expenditures	Recreation; boating, fishing, swimming	Survey on expenditures of time and money to travel to specific sites	Only captures recreational benefits; difficult to apply for multiple destination trips	Hanley et al, 2002 Groothuis, 2005
	Hedonic Pricing	Direct link assumed to exist between quality of an environmental good and a marketed good (e.g. property values)	Water quality, wetland services	Property values and characteristics including environmental quality	Requires extensive information on house prices and their characteristics and about the level of ecosystem service provision at hundreds of specific sites	Hitzhusen et al, 2007
	Change in productivity method	Assess the impact of change in water service on produced goods	Commercial fisheries, agricultural uses	Impact of change in water service on production; net value of produced goods	Information on biological impacts of changes in ecosystems services often unavailable	Pretty et al, 2001
	Benefits Transfer	No valuation method per se.	Depends on the scope of the original study/ies	Suitable original valuation study	Conditions of similarity between studies, including policy/objective context. Solution: test of convergent validity	Hanley, 2001 Johnson et al, 2007

Melichar and Ščasný (2004) or (Viscusi, 1993) for HPM. For a detailed explanation of the Benefits Transfer Method and Discrete choice modeling see the first and second parts of this chapter respectively.

Table 6.1 outlines a brief description of those methods that have been widely applied for the valuation of water quality improvements. Examples of their application are also illustrated in table 6.1.

Amongst these methods, the travel cost method is limited as to the types of values the analyst can cover in the study, as this method can only cover direct use values (e.g. recreation...). Arguably, CVM and Choice experiments (CE) are the most popular methods amongst the academic community, both methods cover use and non-use type of values (though separation between these types of values is regarded as difficult and often depends on the survey design employed for the elicitation exercise). These methods are often contested as a result of possible biases associated with the use of surveys to elicit people's preferences and the hypothetical market on which respondents face a hypothetical situation and make consumer choices without real money (e.g. protest responses, embedding effects...). However, as they have become increasingly popular in recent years, the literature has discovered ways to test for some of these problems (e.g. testing scope, ancillary questions; Alvarez-Farizo et al., 1999).

Special mention is given to benefits transfer. It is not a method per se which is highlighted by the fact that these studies are not popular amongst academics. However, the application of original TCM, CE and CVM results to other policy sites is widely used in the grey literature as it offers a quick and non-expensive way of assigning benefits estimates to economic policy assessments. Recent academic research in environmental economics has been focused in developing reliable tests for the validity of benefit transfer estimates. Initially for transferring values from original CVM and more recently for original CE studies (Barton, 2002; Hanley et al., 2006b; Johnston, 2007)

Both Choice experiments (CE) and Contingent Valuation (CV) are suitable tools for the elicitation of the non-market benefits of national water quality improvements. Maximum WTP can be estimated for improved policy alternatives relative to a counterfactual (which is often a 'status quo' alternative).

The application of CV is particularly useful when there is a specific interest in benefit estimates for a discrete change in environmental quality and when the policy as a whole is the focus of interest rather than aspects of it. CV is suited well to situations where estimates of total benefits of an environmental programme, as this case dictates, are needed. However, the main advantage of CE over CV is that CE allows for the exploration of marginal changes in different attributes of the environmental good being explored, in addition to offering value estimates for the 'whole' of the policy programme[24]. This feature of CE is in theory, particularly well suited for the development of benefits curves for the achievement of Good Status though, highly dependable on the experimental design being used.

The remainder of this chapter consists of empirical applications of two of these methods. Part 1 presents the results of a benefits function transfer exercise followed by an evaluation of the applicability of this approach. Whilst Part 2 of this chapter presents the results of a choice modeling exercise to elicit Scottish households' willingness to pay for improvements under the WFD.

6.1.1. Selection of a suitable study

The Benefit transfer and the choice experiment study presented in this chapter have been largely influenced by a recent valuation study conducted in England and Wales: Baker et al (2007). We now proceed to briefly introduce this study and offer a justification for its replication in this thesis. Further specific details about this study are introduced throughout this chapter.

Baker et al (2007) illustrates detailed results for three different willingness to pay (WTP) elicitation methods for national improvements in water quality in the same survey instrument: CV using both payment card (PC) and dichotomous choice (DC) as payment mechanisms and CE. This report presents WTP values for a permanent increase in real annual payments (increase water bills and other expenses) that the average household in England and Wales is willing to pay for different improvement scenarios under the WFD quality status labels by 2015 and 2028 for all surface water bodies in England and Wales.

The importance of this study resides in the expected policy applications of its results. WTP estimates are currently being used to inform the policy review process for the implementation of the WFD in England and Wales by Defra and the Environment Agency (EA). This includes the economic appraisal of options to reach GS for the river basin management plans due to be published in 2010. As introduced in chapter 1, the current approach to the use of valuation to inform WFD policy decisions in Scotland means that public perceptions for water quality improvements are being completely ignored and

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[24] This implies that the value of the whole can be defined as the sum of its parts – an assumption that has been contested in the literature (see for example; Bateman et al., 1997).

therefore, the economic assessment of derogations under the WFD and its implementation process is being conducted without any estimation of benefits. Fundamentally, the main reason for the replication of Baker et al (2007) in this thesis is the need to set Scotland on an equal footing with the rest of the UK in the application of benefits assessment for policy analysis under the WFD. In this instance, our results provide an alternative point of view to current implementation and provide WTP estimates that can easily be applied and compared with the rest of the UK should current views towards the use of valuation change in the near future.

In addition, due to the high policy relevance of the WTP estimates of Baker *et al* (2007), there is a need to critically appraise their design and results, which is covered in subsequent parts of this chapter. In part I, the transferability of the CV results to other locations is assessed as part of a benefits transfer exercise to Scotland. In addition, part II of this chapter reviews their CE design and the applicability of the results.

6.2. Part I - Exploring the application of the benefits transfer method

The previous section has briefly introduced the most widely used valuation methods that could hypothetically be applied to the monetary estimation of the non-market benefits provided by improving water quality to good status under the WFD. This section now focuses on the introduction and exploration of applications of the Benefits Transfer (BT) method.

BT is an inexpensive method, when compared with commissioning new research projects, and its applicability for policy impact analysis is recognised. BT is highlighted in the UK government's green book as a valid tool for the appraisal of benefits derived from public policies (HM Treasury, 2003; page 21). It is because of its popularity and the fact that the use of BT is generally advocated on the grounds of resource constraints by policy makers, that we have decided to focus this section on exploring its application and suitability for deriving values for the development of benefit functions for the assessment of disproportionality in Scotland (see figure 6.1).

BT, as it was introduced in the previous section, is not a valuation method per se. In summary, the method consists of taking values, demand and benefits functions from original studies conducted elsewhere and using them in another policy relevant site. There are three different variations of the BT method: i) Unadjusted or adjusted direct value transfer from a single study; ii) Unadjusted or adjusted benefit or demand function transfer from a single study; and iii) meta-analysis as a synthesis of the results of several valuation studies. All variations of the BT method share the same limitation, which is the identification of suitable original valuation studies that are relevant to the policy site to which BT is applied.

The academic literature ventures that it is actually very difficult to reliably estimate non-market benefits using the BT method; criticism is mainly focused on the assumptions employed, the lack of reliable data, and the validity and possible use of the transferred estimates. In the following sections, we introduce and explore the application, suitability and limitations of applying different variations of the BT method. The layout has been divided as follows: First, the application of meta-analysis is explored. This section will also introduce a review of relevant valuation studies of water quality changes that have recently been undertaken in the UK. Secondly, an application using an adjusted benefits function transfer from a recent contingent valuation study is explored. Finally, we illustrate our conclusions and introduce recommendations for future research.

6.2.1. Meta-analysis (MA)

We begin our exploration of suitable methods to obtain "transferred" monetary value estimates for water quality improvements due to WFD requirements in Scotland by assessing the application of the most complex variation of the BT method: Meta-Analysis of original valuation studies.

6.2.1.1. Introduction

Bergstrom and Taylor (2006) define MA ("the study of studies") as a method for summarising results of existing valuation studies by estimating statistical relationships between reported original values to a set of explanatory variables which would capture the heterogeneity between and across these studies. MA has a wider purpose from that of benefits transfer by not only offering a format for predicting monetary values but it is also regarded as a very useful technique for the synthesis of relevant literature on a particular valuation topic and to test hypotheses with respect to the explanatory variables on the value construct of interest. MA as statistical analysis of summary finding of prior empirical valuation studies offers a transparent structure by which to quantitatively assess results from other studies. In addition, the analyst can explore and understand patterns of assumptions, relations and causalities between studies (Bateman et al, 2000).

Meta-regression analysis is the statistical tool normally applied in MA (Van Houtven et al, 2007). The first methodological step, as in any other benefits transfer exercise, consists of collecting a set of primary studies that contain a common empirical outcome, e.g. willingness-to-pay (WTP) for improvements in freshwater quality. The dependent variable is a common summary statistic or “effect-size”, such as a regression coefficient for a predicted WTP value. Secondly, one or more values of this statistic are drawn from each primary study (e.g. either mean WTP estimates of all the values presented in the original study or value ranges which are then adjusted to reflect current prices). The explanatory variables in the regression include characteristics of the primary data (such as reference to the water quality change in relation to baseline levels or types of uses valued), study design, valuation method, sample size, model specification, econometric methods, and other “quality” variables such as place and date of publication. Most regressors are specified as binary dummies and most of these variables are also drawn from the primary studies (which often also includes unpublished studies in the "grey literature", working papers, government reports, student dissertations). In some analyses, the identity or characteristics of the primary investigators are used as regressors and even tests for author bias have been designed (Brouwer and Rolfe, 2008).

Applications of MA to the field of environmental economics for benefits transfer has expanded rapidly in recent years covering a wide range of subject areas, running parallel to the rapid growth in the use of environmental valuation for policy appraisal. For example just to mention a few areas/case studies; MA has been applied in urban pollution valuation (Smith and Huang , 1995) , woodland recreation values (Bateman et al, 2002) or valuation of wetland functions (Brouwer et al. 1999). Focusing in the water valuation field, four studies have been found that applied meta-analysis for the estimation of values for water quality improvements. Three conducted in the US (Johnston et al, 2003 and 2005; Van Houtven et al, 2007) and one in Australia (Brouwer and Rolfe, 2008). We now proceed to provide further information about two of these studies.

Van Houtven et al (2007) applied MA for the estimation of national WTP values of water quality changes in the US for required improvements under the implementation of the Clean Water Act. The authors of this study reviewed over 300 original studies related to water quality valuation; but only selected those values that were sufficiently comparable for inclusion in a meta-analysis. They limited the search to studies conducted in the US and that applied stated preferences techniques. Only those studies that described water quality in terms that could be converted to a common 10-point water quality scale, were used to provide predicted values for WTP for three levels of freshwater quality improvements under the Clean Water Act. For their analysis, they finally reviewed, 18 studies conducted between 1977 and 2003 which gave them 131 value estimates.

Alternatively, Brouwer and Rolfe (2008) applied this method to a national valuation exercise of water quality improvements in rivers and wetlands in Australia. The authors reviewed a total of 8 Australian discrete choice modeling studies with 93 observations in total (implicit prices). All the surveyed studies were quite similar in their design; e.g. application of CE as a valuation method, study cases located in the east coast of Australia, four of the studies conducted by the same team of researchers. They nevertheless concluded that the accuracy of their results were conditioned by the low number of observations included in the meta-analysis (which was related to the small number of reviewed studies), the different concepts of water quality values included; e.g. WTP for improvement in river flows, waterway restoration, healthy rivers, water dependent wildlife, water quality (recreational use), the wide variety of attributes used in the formulation of choice experiments and different measurement units (some imprecise) which conditioned their aggregation for the meta-analysis.

The review of these two studies uncovers two fundamental issues in relation to the application of meta-analysis for benefits transfer: i) the minimum number of original studies necessary to carry out a proper analysis; and ii) meta-analysis is not aggregation bias free. Even though metaregression is labelled as a transparent method to aggregate values and account for methodological differences between these studies, it is a method highly subject to inference by the analyst. This does not only refer to clear bias associated with the selection of original studies (e.g. search criteria) but in the case of complex valuation exercises, such as the valuation of overall improvements to water quality, the aggregation of different concepts of value under one flag requires a lot of manipulation. This issue, of course, has an impact in the applicability of the results.

To the authors' knowledge there has not been a single MA of water quality original valuation studies conducted in Scotland nor elsewhere in the UK. Unfortunately, the weaker side of MA is that in order to establish meaningful statistical relationships there is the need for a large database of relevant original studies. Availability of suitable original studies is fundamental for its application. However, this is very often not the case depending on the environmental aspect to be evaluated or the location or purpose of the meta-analysis. For example, Brouwer and Rolfe (2008) did review only 8 studies, but even though they all applied the same elicitation method (choice experiments) and were located in the same area (east of Australia), the authors concluded that their results were conditioned by the low number of observations used.

In the present chapter, we assess the possibility of conducting a MA, as an application of benefits transfer. We continue this section by reviewing a selection of water quality improvements valuation studies that have been conducted in mainland Britain since the year 2000. This exercise will also be used to introduce and review the relevant literature on valuation which will also be useful when we explore applications of another benefits transfer method; benefits function transfer.

6.2.1.2. The standard theoretical model

A recent survey by Van Houtven et al, 2007 provides the following general statement of the metaregression model:

$$WTP=f(Q,X,R,T,L)+e \tag{1}$$

where *WTP* is the effect-size; e.g. predicted mean annual WTP for water quality improvements scenarios, *Q* is a set of causes of the outcome (e.g., change in water quality), *X* are characteristics of the set of objects affected by *Q* in order to determine the outcome (e.g. average household income in each study), *R* is a set of characteristics of the research methods in each primary study (e.g. elicitation method, type of interview, number of respondents, percent users), *T* indicates the time period of the primary study, *L* is the location of the objects (e.g. where the study was set), and *e* is the error term. Equation 1 follows the typical grouping of explanatory variables in any meta-regression analysis.

6.2.1.3. Search for original studies

The first natural step in any meta-regression analysis is the identification of original studies. In this case, we conducted a search of the literature and selected a range of original valuation studies that have used monetary valuation techniques for the estimation of water quality changes. Our search had the following selection criteria:

Valuation studies of water quality changes carried out in mainland Britain in the last 8 years were identified (report findings, working papers and students' theses were also included in the search). The aim of this search criteria is to take into account to a certain degree the transferability of the original study site to Scotland by limiting their proximity[25]. The introduction of a time limit in our search criteria was also considered necessary in order to only include those studies that had undertaken valuation of water quality changes under the influence of the WFD.

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[25] Arguably, there is no empirical evidence that a water quality valuation study carried out in England would have more in common to Scotland than one undertaken, for example in Finland. While this is not known in terms of the physical and chemical characteristics of their water bodies, Scotland, as part of the UK, would undeniably share more of its socio-economic characteristics with England and Wales than with any other country in the EU.

Table 6.2 illustrates the findings of our search. Information for each study has been summarised following the same layout used by Bateman et al, (2000) and Van Houtven et al, (2007) for presentation of data sources for a meta-regression analysis. Original valuation studies identified have been classified by their location, type of water resources, water quality changes evaluated, elicitation methods employed, value types and mean WTP values found and units employed. In addition, the name of the publication's main author is shown (complete references for each study are provided in the bibliography).

Table 6.2 summarises the findings of thirteen studies that fulfilled our search criteria, of which five were undertaken in Scotland, seven in England and one covered England and Wales. The studies found include a wide range of types of water resources that are relevant under the WFD, for example; 5 studies covered rivers, 2 studies coastal water bodies, 1 study the valuation of lake water quality improvements and another wetland resources. Three of these studies used a wider scope and covered different types of water bodies simultaneously (rivers, lakes, coastal and canals).

In addition, these studies applied a wide array of different elicitation methods for the valuation of improvements in water quality (some studies employed more than one method); CVM is the most popular method and has been applied in 8 occasions. It is followed by CE, which has been employed 5 times and finally, a couple of studies used the travel cost method. In addition, there are also differences with types of values measured in these studies. The majority elicit WTP values for use and non-use values, whilst some CVM and TCM studies only cover use values.

Table 6.2 Summary characteristics of water quality improvements valuation studies carried out in mainland Britain since 2008

ID	Reference	Location	Resource type	Water Quality Change Evaluated	Elicitation Method	Value type	Mean WTP estimates[1]	Labels
1	Garrod et al. (2000)	England	Wetland/ floodplains	Environmental impact (river flows, bird numbers and biodiversity) caused by increased water abstraction from Hardham aquifer in summer and recharge of aquifer with water abstracted from River Rother in winter	CE	Use/ non use	£21.24	WTP per HH per year. Number of birds and diversity of plants found at wetlands.
2	Day et al. (2001)	Scotland	Coastal	Value of coastal water quality improvements from current levels to meet EU bathing waters quality standards, where standards are not met	CVM	Use	£6.36/10.43	Mean WTP per HH per year to improve bathing water quality Irvine/Ayr beaches to meet EU standards
3	Spurgeon et al. (2001)	England	river, canal and lakes	Maintaining or improving different levels of water quality and angling opportunities in the respondents' most frequently used water body	CV and revealed preference data	Use	£4.79/£7.64/ £10.19	WTP per HH per year to increase water quality from poor to reasonable/from reasonable to good and maintaining good respectively
4	McMahon (2001)	England	river, canal and lakes	Water quality and water services provision. Two surveys were conducted: 1) User survey; non-mains connected household WTP for provision of a sewer mains and household connection and 2) non-user survey; WTP to avoid the environmental and amenity values associated with non-mains sewerage or inadequate private drainage systems	CVM	Use/non use	£0.37/ £3,433	Mean WTP one-off payment per HH to avoid damage/ Mean WTP one-off payment per household for provision of sewer mains and household connection
5	Jacobs (2002)	England	River	Water flows. WTP for improvements in water levels	CVM	Use/non use	£0.74-£9.84	WTP £/HH/year values for the river Minram (full recovery) in 5 years. Results depend on distance from the river/users/non users
6	Georgiou et al. (2002)	England	River	Three water quality schemes evaluated (ranged from small, medium to large improvements). Attributes range from ability to boat in river, increase in wildlife and ability to swim in water. RFF index scores and water quality improvement levels considered derived from the UK Environment Agency's River Ecosystem Classification scheme	Contingent ranking/ CVM	Use/non use	£5.18	Mean WTP per HH per year for a unit increase in RFF index

7	Hanley et al. (2003)	Scotland	Coastal	Value of coastal water quality improvements from current levels to meet EU bathing waters quality standards	CVM	Use	£7.81	Per person per year. Mean increase in consumer surplus for coastal water quality improvements
					Travel cost method	Use	£0.48	Per person per visit. Mean increase in consumer surplus for coastal water quality improvements
8	Johnstone (2004)	England	River	Marginal changes in river water quality. two models used: one to predict number of trips and another to predict angling site choice	Travel cost method	Use	£0.04-3.93	Consumer surplus values per trip for a 10% change in river attributes
9	Bateman et al. (2005)	Scotland	Lakes	Valuation of policies for reducing the acidity in remote mountain lakes. HH annual WTP for improvements and no deterioration	CVM	Use/non use	£2.35-3.66	Per year improvement per lake
							£3.278-4.058	Per year no deterioration per lake
10	Hanley et al.(2006c)	Scotland	River	WFD water quality improvements in the River Clyde. Attributes: river ecology, aesthetics, bankside restoration	CE	Use/non use	£38.70	Mean WTP estimates for HH and year for improvements from fair to good
11	Bateman et al. (2006)	England	River and Lakes	Valuation of the prevention of euthrophication of rivers and lakes in EastAnglia caused by domestic sewage as a result of future climate and population changes.	CVM	Use/non use	£75.41	Mean WTP (£/HH/year) to reduce eutrophication impacts
12	Hanley et al. (2006b)	Scotland	River	WTP for improving water quality in two agricultural catchments. Attributes: Jobs, water flows and ecology. Expert judgement used to define policy scenarios. Also value how people perceive impacts of improvements (e.g. number of jobs)	CE	Use/non use	£25.91	WTP HH per year for a "big improvement" in river ecology. Non correlated pooled model
13	Baker et al. (2007)	England & Wales	water bodies	WTP for overall improvements in water quality as specified in the WFD (ecology)	CE	Use/non use	£0.79/0.86	CE mixed logit model -WTP per HH per year for a 1% improvement in the total number of water bodies labelled "good status" at local/national level
					CVM	Use/non use	£44.5/ £167.9	PC/DCCV Mean WTP values per HH per year in E&W for 95% water bodies in "good status"

[1] WTP values in year of study
CE=Choice Experiments, CVM=Contingent Valuation Method, PC=Payment Card, DC=Dichotomous Choice, HH=Household, WTP=Willingness To Pay

6.2.1.4. Issues in the application of meta-analysis

The selection of original studies for a meta-analysis needs to carefully consider the different elicitation methods employed in the original studies. For example, the aggregation of values which were found by applying revealed preference techniques (e.g. TCM) with those that applied stated preferences methods (e.g. CVM, CE) is not recommended. This aims to ensure a common value concept in the analysis which is translated from the different concepts of value applied in stated and revealed preferences techniques. Value measures from TCM and CV/CE are fundamentally different in their nature (i.e. Marshallian consumer surplus versus Hicksian compensating surplus, respectively). Accordingly these values, if pooled together, would introduce conceptual inconsistencies.

Along these lines on the selection of elicitation method, the magnitude of the value estimates for water quality changes also depends on the way in which estimates are derived. Even among those studies that applied the same elicitation techniques there are methodological differences; for example, when a CVM has been applied, we would expect differences between elicitation formats (whether an open ended or dichotomous choice format has been used). Additionally, sample sizes and survey instruments introduce contextual differences between studies that could a priori be regarded as "comparable"; such as the way the interviews have been conducted (e.g. face-to-face, over the telephone...) or payment mechanisms employed (differences between using increments in income or local council taxes to cover costs of hypothetical schemes to improve water quality).

Careful consideration also needs to be given to the scope and geographical coverage of original studies. In table 6.2, only one study has a national scope (Baker et al. 2007 provide value estimates for the whole of England and Wales). The remainder of studies seek to estimate values in small areas in Scotland or England (either individual rivers, Hanley et al. 2006b,c; or improving water quality conditions in rivers and lakes in specific areas of East Anglia, Bateman et al. 2006). Extrapolating values beyond their spatial boundaries inevitably would add to uncertainty. This is a considerable limitation when conducting an hypothetical MA for the valuation of improvements due to the WFD. Obviously, relevance for policy resides in national value estimates. Arguably, minimum geographical coverage with some relevance for policy would be catchment scale. Nevertheless, benefits need to cover the policy scope behind the water quality improvements; which in this case is mandated by the WFD (national, river basin district, catchment management plans).

Aggregation needs to account for contextual heterogeneity across different valuation studies. For example, different types of water improvements (e.g. reducing acidity in remote lakes; Bateman et al, 2005 or improving water flows; Jacobs, 2002), types of water bodies (coastal, rivers, lakes...), types of uses (e.g. use and non-use) and unit values of WTP (mean WTP values per year per household versus WTP per person per improvement, marginal WTP).

In addition, water quality descriptors are different across all studies identified in table 6.2. This is probably the main barrier to the application of meta-analysis to the aggregation of WTP values for overall water quality improvements. As introduced in chapter 2, water is a complex resource with many facets. Beyond the basic differences between quality and quantity, the economic valuation of water quality improvements in isolation is marked by differing definitions of quality and how people perceive those changes. In order to extract value estimates for water quality changes that can be expressed in comparable terms for a MA, it is fundamental that studies can be linked to a commonly defined water quality metric. This also relates to the different policy goals of each of the studies, which in this example, many collide with the overall policy objective of obtaining monetary values for WFD improvements or an overall achievement of "good status". Arguably, almost all of the reviewed studies cover some aspects of "good status" to a certain degree.

There are ways around these issues, one solution is to provide within-sample predicted values of the dependent variable under a particular set of conditions. For example, Van Houtven et al. (2007) provide predicted values for WTP for three levels of freshwater quality after scaling each individual study using a 10-point water quality index which translated to improvements required under the US Clean Water Act. The WFD water status labels provide grounds for adjusting individual studies values. However, there is a considerable degree of subjectivity surrounding such exercise, which would require further consideration by water quality experts in order to identify which water improvements would fall under each category. This is an issue for further research. However, a recommendation can be made for future valuation studies. Any adjustments that would be necessary for a meta-analysis of water quality improvements would greatly benefit if the original study had used an already defined water quality index or if the study relates to improvements to freshwater resources in Europe, apply the WFD prescribed quality labels. This would also increase the policy relevance and transferability of such studies. Under this framework, only one study in table 6.2 -Baker et al, 2007- applies WFD water quality status labels to value improvements.

Finally, other common issues to consider in a MA of valuation studies relate to the overall quality of the original study. Many MA papers introduce a quality of the study variable, dummy coding is used to reflect study quality differences, for example if it has been published in a peer-reviewed journal or presented in a conference or its results form part of the grey literature. This has been found to have an important weight in the regression functions that characterise the MA final results (Bateman et al., 2000). Our selection of studies in table 6.2, probably suffers from author bias. Four out of five Scottish water valuation studies have been undertaken by Nick Hanley (Professor in environmental economics at Stirling university) and his team of researchers. In addition, Ian Bateman (Professor in environmental economics in University of East Anglia) and his team signed three.

In conclusion, due to the limited number of relevant studies, different contexts covered, methodological heterogeneity and the impossibility of consolidating them under WFD related objectives, we believe that the application of meta-analysis at this stage would raise more questions than answers. At this moment, the aggregation of mean WTP values from original studies would most certainly suffer many limitations. Most importantly, results would be conditioned to the high degree of subjectivity and manipulation which would be necessary in order to find ways to pool these studies together.

Nevertheless, the implementation of the WFD and the more than possible future need in water policy to shed more light into the quantification of the non-market benefits of the Directive in order to justify investment, will guide future research in water valuation in this country and the rest of Europe. Consequently, researchers will tailor their research methods to the specific goal of achieving good status. We could expect a similar reaction to that in the US once the Clean Water Act was enacted nearly more than three decades ago.

6.2.2. Transferring benefits functions for water quality improvements for the implementation of the WFD in Scotland

The Benefits Function Transfer (BFT) approach is a method that allows for the incorporation of differing socio-economic and site quality characteristics between the original study site and the policy site under evaluation. In this type of benefits transfer only one original valuation study is selected[26]. The main assumption being that in BFT exercises the statistical relationship between WTP for improvements and subsequent independent variables are the same for both the study and policy site.

The accuracy of the BFT method is under constant scrutiny in the literature. Research in this topic focuses on testing whether this method offers better results than direct benefit estimate transfer in situations where demographic or environmental quality factors (for example) at the study site differ from those at the policy site. However, empirical results concerning the superiority of BFT are mixed. In a study of Wisconsin lake recreation, Parsons and Kealy (1994) found that benefit function transfer estimates were within 4% of the original model estimates, while unadjusted benefit estimate transfers were within 34%. Brouwer and Spaninks (1999) also found that benefit function transfer was more accurate (within 22%) than benefit estimate transfer. In contrast, Barton (2002) encountered that benefit estimate transfer, with transfer errors of 20% and 30%, outperformed benefit function transfer in the case of water quality improvements in Costa Rica.

In this section, we will undertake a BFT exercise to Scotland from a CVM valuation study recently undertaken in England and Wales. The suitability of the method to derive robust WTP values for water quality improvements under the WFD in Scotland and the accuracy of the results will be investigated.

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[26] Note that this statement is only true when the BFT approach is based on an original CV study. Transferable benefit functions for recreation values are frequently based upon multi-site Travel Cost studies (Bateman et al. 2000). Loomis (1992) proved that it is possible to derive a stylised zonal Travel Cost Method demand function from three sets of multisite TC functions.

6.2.2.1. *Benefit Function Transfer via contingent valuation - standard formulation*

Unlike unadjusted BT exercises where mean WTP at the policy site it is assumed to be equal to mean WTP values at the original site ($WTP_S = WTP_P$), BFT exercises attempt to adjust values by accounting for any possible differences (e.g. socio-economic and environmental quality variables included in the aggregated benefits function) between both sites. The conceptual model for conducting BFT between different locations involves defining the relevant Hicksian measure (in the case of a CVM study) of utility for the good or policy change being measured[27]. Equation 2 offers a conceptual representation of the benefits function transfer approach:

$$\begin{gathered}\textit{Survey site: } WTP_S = \alpha_s + \beta_{s1}X_{s1} + \beta_{s2}X_{s2} \\ \updownarrow \\ \textit{Policy site: } WTP_P = \alpha_s + \beta_{s1}X_{p1} + \beta_{s2}X_{p2}\end{gathered} \qquad 2)$$

Where *s* denotes the survey site, *p* the policy site and X_1, X_2 vectors of specific good characteristics and population characteristics for each site (e.g. income and education levels, baseline water quality levels...).

BFT is regarded as a suitable tool for the adjusted transfer of WTP estimates between different locations when the vector of attributes and socio-economic characteristics (X_1, X_2) that determine the similarities and differences between the policy and the survey site can be established. Where these differences exist and their magnitudes are known, it is possible to substitute those known variables into the survey site's original aggregated benefits function to provide valid BT estimates.

6.2.2.2. *Selection of a suitable study*

As with MA, the benefits function transfer method is highly dependant on the identification of suitable potential studies from which benefit functions can be transferred. Boyle and Bergstrom (1992) have laid out a technical criteria for the selection of original studies:

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[27] Note that the application of this method to revealed preference data is different. These differences have not been covered here - for more information: see Bateman et al. (2000) or Garrod and Willis (1999)

1. the non-market good to be valued at the policy site must be identical to that already valued at the study site;
2. the characteristics of the populations affected by the non-market good must be identical at each site;
3. the same welfare measure should be theoretically appropriate at each site.

From the selection of recent valuation studies illustrated in table 6.2 and following the strict selection criteria by Boyle and Bergstrom, we have decided to apply BFT to the results found by Baker et al. (2007). This study, which was undertaken by NERA consulting and ACCENT and commissioned by DEFRA and the Environment Agency, offers willingness to pay estimates for overall improvements in water quality in England and Wales as a result of implementing the Directive. Results are for national improvements in water quality per year per household. From all the studies reviewed in table 6.2, this is the only valuation study found that did apply the standard WFD ecological-based water quality metrics for description of baseline levels and improvements.

The application of BFT is also subject to some degree of subjectivity (e.g. choice of studies, understanding of the main differences between the original study and the policy site and the subsequent transformation of environmental and socio-economic variables). In this case and faced with a selection of 13 original valuation studies, location has not influenced our selection. All the "Scottish” studies identified needed some sort of transformation prior to transferring their benefits functions in order to adjust their values from local to national estimates. In addition, it would also have been necessary to adjust measurements of water quality to meet the status labels of the Directive.

Baker et al. (2007) offers detailed results for three different elicitation methods: contingent valuation using both payment card and dichotomous choice as payment mechanisms and choice modeling. The following section introduces a practical application of the benefits function transfer approach from their CV models for aggregation, the advantages and limitations of this method will be highlighted and explored. In addition, this report did also apply choice experiments. The following section of this chapter will be exclusively devoted to exploring the application of this elicitation method to the valuation of water quality improvements under the WFD in Scotland. In addition, some issues related to the application of benefits transfer from choice experiments WTP estimates will become clearer once the method is illustrated in the next part of this chapter.

6.2.2.3. *Introducing the selected study: Baker et al (2007)*

This study estimated the economic value placed by English and Welsh households for water quality improvements at local and national level. Three different elicitation methods (CVPC-CVDC and CE) were tested within the same survey instrument (which raises issues about how WTP values may be affected by concentration and awareness levels of the respondents when faced with different elicitation formats at once and the order in which the formats are presented to respondents). In total 1,487 interviews were achieved against a target of 1,500 in different locations spread across the study area. Table 6.3 illustrates their main findings; WTP values are presented in this table as the permanent increase in real annual payments (increase in water bills and other expenses) that a household in England and Wales is willing to pay for a scenario of 95% improvement to High Quality Status by 2015.

Table 6.3 Annual WTP values in Baker et al. (2006) for water Environment Improvements (95% by 2015) in England and Wales by elicitation method

	England		Wales		England and Wales	
Elicitation method / Model	**Mean WTP £/hh/yr**	**Median WTP £/hh/year**	**Mean WTP £/hh/yr**	**Median WTP £/hh/year**	**Mean WTP £/hh/yr**	**Median WTP £/hh/year**
PCCV Sample statistics	49.2	30.0	62.6	50.0	50.4	30.0
PCCV OLS Model	44.8	25.3	40.1	22.7	44.5	25.1
DCCV Logit model	167.0	167.0	181.4	181.4	167.9	167.9
CE Logit model	293.7	293.7	508.0	508.0	299.9	299.9

Source: Baker et al. (2007)

These results offer a wide range of mean WTP values. For England and Wales, WTP ranges between £44.5 per household per year for the PCCV OLS model to almost £300 for the CE Logit model. The authors clarify that the main rationale behind the use of three different elicitation methods is the need to offer robust ranges of WTP estimates that are useful for policy, as well as sensitivity analysis, and conclude that the true WTP value should lie somewhere between the PCCV and the DCCV results. The report does not illustrate any confidence intervals of the offered estimates.

Table 6.3 confirms that WTP estimates are highly sensitive to the elicitation method chosen. The payment card CV method, where respondents must freely select a monetary sum from a wide range presented on a card, usually produces lower answers than the dichotomous choice format or the CE method, where respondents accept or reject choices of payments for given monetary sums (Baker et al., 2007). It could be argued that in this case payment attributes for DC and CE were not properly calibrated and respondents were asked to overstate their true WTP. However, the valuation literature confirms that PCCV method normally leads to conservative estimates, which as a result of the free choice of a WTP figure, this means results are often downwardly bias (Mogas et al., 2005; McVittie and Moran, 2008).

6.2.2.4. Econometric models: PCCV and DCCV

Table 6.4 Variables description and coding employed in baker et al (2007)

Name	Variable description	Variable significance for CV models
ln_delta_hl	= ln(1+*delta_hl*); *delta_hl*= Variation in the high quality level, local area	PCCV*
ln_inc	= ln(1+*income*) (where income = 0 if missing)	PCCV***, DCCV***
income_miss	= 1 if income not reported, 0 otherwise	PCCV***, DCCV***
children	= 1 if household contains children	PCCV*
use	= 1 if contact, fishing or otheract	PCCV***
pol_control	= 1 if wishing to continue improvements for pollution control	PCCV***, DCCV***
sex	= 1 if respondent is a male	PCCV***, DCCV**
edu_12	= 1 if level of education between primary and O levels	PCCV*
edu_35	= 1 if level of education above A levels	PCCV***, DCCV**
wales	= 1 if country = Wales	PCCV, DCCV
cv_first	= 1 if order of the questionnaire is CV then CE	PCCV***
understood	= 1 if respondent understood "completely" or "a great deal"	PCCV**
dc_bill	= DC payment options (divided by a 100)	DCCV***
club	= 1 if member of a water related club	DCCV**
concentration	= 1 if respondent maintained concentration throughout	DCCV***
cvfirst_dcposition_0_1	= 1 if CE first and DC position = beginning	DCCV***
int_sex	= 1 if interviewer was a male	DCCV*
constant		PCCV*, DCCV**

Source: Baker et al, 2008

We now focus on the exploration of the PCCV and DCCV econometric models for aggregation that were presented in this report (Baker et al., 2007). The set of variables for inclusion within their analysis are summarised in table 6.4 alongside the coding used and their significance under each model. Based on comparisons with other valuation studies, the variables employed in this study are a correct mixture of those which theory suggests should have a direct influence in the econometric estimation of WTP (e.g. water quality levels, income and education levels, sex, use of the resource, attitude towards pollution control). In addition, it is worth mentioning the inclusion of variables that control the quality of the study, which (as this study reports) often also have an influence in the estimation of WTP values (e.g. understanding of the questionnaire, respondent's concentration and order of elicitation method are significant variables in these models).

Table 6.5 below introduces the OLS and Logit models for aggregation employed in Baker et al. (2007) for the econometric estimation of WTP values under the PCCV and DCCV methods respectively. The dependent variables are; *ln_wtp* for the PCCV OLS regression model and *dc_choice* for the Logit DCCV model. It is worth noting that in none of these models the variable *wales* is significant, which raises questions about its inclusion. One has to wonder why if the authors thought that location was important in the models for aggregation why dummies for other parts of England were not included as well. Besides, if there was a need to report different WTP estimates for England and Wales, this could have been solved by developing different models for each country.

Overall significance levels for both models are between acceptable limits, with an R^2= 0.16 for the OLS model (meaning that the model explains around 16% of the possible variance surrounding the estimation of WTP) and *Pseudo* R^2= 0.16 for the Logit model (which can only be interpreted as a rough goodness of fit of the model; Bateman et al. 2000)[28].

The results in table 6.5 can be used to derive aggregate values for WFD environmental improvements scenarios by attaching values to each of the variables employed in Baker et al (2007) PCCV and DCCV models and multiplying through by the respective coefficients. The valuation scenarios used in this study for the PCCV and DCCV methods illustrate an overall achievement in water quality of 95% "high quality" by 2015 for all surface water bodies in England and Wales. This translated to the WFD normative levels for the classification of surface water bodies comprises the high and good status classes. In general, all coefficients in both regression equations present the theoretically expected signs.

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[28] As already introduced in chapter 4, the assessment of goodness of fit, or levels of accuracy of econometric models, is not an easy task and the application of suitable measures/tests is a contested issue in the literature (Krueger and Lewis-Beck, 2007). There are different tests that can be used to assess the goodness of fit of logit models (chi-square, pseudo R^2 and Mcfadden's R^2; see Habb and MacConnell, 2002, for their description). The Pseudo R^2 statistic, which was employed in Baker et al, 2007 to illustrate the overall significance level of their logit model, is a goodness of fit measure based on the analysis of the likelihood ratios in relation with total sample size. This statistic attempts to recreate an analogue to the traditional R^2 measure used in OLS (Habb and

Table 6.5 Baker et al., (2007) adopted PCCV and DCCV Regression and logit models for aggregation

	PCCV		DCCV	
Variable	**coef**	**S.E. (sig)**	**coef**	**S.E. (sig)**
ln_delta_hl	0.671	0.487*	-	
ln_inc	0.258	0.045***	0.395	0.094***
income_miss	1.411	0.28***	2.066	0.556***
children	0.126	0.074*	-	
use	0.266	0.103***	-	
pol_control	0.445	0.112***	0.538	0.182***
sex	-0.181	0.069***	-0.284	0.14**
int_sex	-0.218	0.069***	-0.295	0.161*
edu_12	0.186	0.1*	-	
edu_35	0.511	0.104***	0.4	0.156**
wales	0.041	0.13	0.268	0.26
cv_first	-0.408	0.065***	-	
understood	0.258	0.119**	-	
dc_bill	-		-1.236	0.102***
club	-		0.437	0.175**
cvfirst_dcposition_0_1	-		0.562	0.192**
concentration	-		0.562	0.185***
constant	0.762	0.408*	-1.314	0.568**
Observations	1389		1389	
R2	0.158		-	
Pseudo R2	-		0.16	

*PCCV (Payment card contingent valuation): dependent variable = ln_wtp; OLS estimator used. n=1389, R2=0.32. t-test p values (one/two sided): *p < 0.10, **p < 0.05, ***p <0.01*
*DCCV (Dichotomous choice contingent valuation): dependent variable = dc_choice; Logit model used for aggregation. n=1389, Pseudo R2= 0.16. t-test p values (one/two sided): *p < 0.10, **p < 0.05, ***p <0.01*

In the OLS model presented above (table 6.5), the sum of the coefficients multiplied by the values of the variables they correspond to, yields the conditional mean of the log of WTP, as this is the dependent variable in the model (the authors found that the model with *ln_wtp* as the dependent variable fits the data substantially better than the model with WTP as the dependent variable). The exponential of this value, as illustrated by Goldberger (1968), gives the conditional median rather then the mean WTP estimate. Alternatively, the conditional mean can be derived by applying an adjustment equal to the mean of the exponential of the residuals from the model (Baker et al. 2007).

MacConnell, 2002). Furthermore, according to Hensher and Johnson (1981) a good fit is indicated by a Pseudo R^2 between 0.2- 0.4 and an exceptionally good fit should be considered beyond 0.4 levels.

For the Logit DCCV model, the results of table 6.5 (above) can be used to derive mean WTP estimates for the "95% overall improvement scenario" by applying formula (3), which is the linear random utility expression of mean WTP for parametric logit models (Haab and McConnell, 2002). In this expression, β is the coefficient of the bid variable (*dc_bill*), which was divided by a 100. α is a vector of the means of all variables and z_j is a vector of all estimated coefficients. In linear random utility, preference uncertainty (ε) is symmetric (assumes homogeneity in the population) with mean zero which means that mean and median WTP values with respect to random preferences are equal (Haab and McConnell, 2002).

$$E_{\varepsilon}(WTPj|\alpha,\beta,z_j) = 100 * \frac{-\alpha z_j}{\beta_{dc\_bill}} \quad 3)$$

6.2.2.5. *Variable transformations and data sources*

Table 6.5 above outlines the PCCV and DCCV model functions from the original study. Those variables that could be adjusted to transfer these functions to a Scottish context are discussed below.

ln_delta_hl is employed in the valuation exercise to describe variations in the high water quality levels in local areas. This variable reflects quantities of improvements in relation to baseline water quality levels by river basin district. Data comes from the Environment Agency's risk assessment data and the unit adopted to measure quantity of each status level is hectares of catchment area for rivers, and hectares of surface water area for lakes, estuaries and coastal areas. This variable is used in the original study to provide a contextual reference to respondents for the valuation exercise. One downside of this approach is that it does not distinguish relative amounts or values of improvements between different water bodies, assuming that respondents would express no preference for improvements between a river and a lake for example. Arguably, area units employed are not of the same magnitude and WTP estimates should have ideally been differentiated between running and standing waters. For our transferred model, data for this variable has been taken from SEPA's recently updated environmental characterisation report of the Scottish River Basin District (SEPA, 2007a,b).

Adjusted data for the transformation of socio-economic variables (*ln_inc, children, edu_12, edu_35)* come from: i) an analysis of the UK Family Resources Survey (FRS), 2005/06 (DWP, 2007) for income levels in Scotland and ii) education levels and number of children per household figures have been taken from the Scottish Household Survey 2005/06 (Scottish Government, 2007).

In addition, it was also possible to transform some of the original attitude variables. The Scottish government has recently published the results of a survey of Scottish public attitudes towards the water environment (Mori, 2006). This exercise interviewed 1,011 adults across the country. Results from this report were used to transform the attitudes towards pollution control variable (*pol_control)* in the original model and to obtain information on the proportion of people that make use of water resources in Scotland *(use).*

Finally, it was decided not to modify those variables that were kept fixed in the original model. These mainly comprise survey control instruments for the valuation exercise. In addition, there are other variables for which it has been impossible to find data for their transformation (e.g. *club* - membership to water related organisations in Scotland) or that they cannot be adjusted as they are intrinsic to the CV method employed (e.g. *dc_bill).*

6.2.2.6. Results

Our results of the benefits function transfer exercise are illustrated in table 6.6. Scottish transferred estimates from Baker's et al (2007) PCCV OLS and DCCV Logit models are presented by income group. For comparison, Table 6.6 also illustrates original estimates for England and Wales.

Accordingly, mean WTP values for the 95% overall water quality improvement scenario in Scotland ranges between £30 and £149 per year per household depending on the original elicitation method. Putting these figures into perspective, according to the General Register Office for Scotland around a total of 2.3 million households were registered in 2006 in the country (GROS, 2007), which would give a total benefits figure for WFD related water quality improvements in the range £69-342 million per year. The lower end of the range representing mean values of the PCCV format and the upper-bound range from the DCCV model.

Table 6.6 PCCV and DCCV original WTP results and adjusted values for Scotland by country and income group

		Annual WTP per household (£/hh/year) for 95% "good" water quality			
		All	Income Group		
			Low (<£300 p/w)	Med	High (>£1,000 p/w)
PCCV OLS Model	England & Wales	44.5	33.5	42.2	56.3
(mean values)	Wales	40.1	31.7	40.1	53.2
	England	44.8	33.6	42.6	56.5
Adjusted transfer values	**Scotland**	**30.1**	**23.4**	**30.8**	**39.5**
DCCV Logit Model	England & Wales	167.9	132.6	161.9	197
	Wales	181.4	152	181.3	216.4
	England	167	131.5	160.8	195.9
Adjusted transfer values	**Scotland**	**148.6**	**117.5**	**151.5**	**182.4**

6.2.2.7. *Sensitivity analysis*

The question remains as to the accuracy and policy relevance of transferred WTP estimates. Are these estimates reliable? What do they mean? Should values be transferred without any adjustment in the first place?

The reliability of transferred WTP values has been at the centre of the environmental economics research. However, research has focused primarily on the evaluation of the benefits transfer method per se and not on the interpretation of the values. Academic studies in this topic often perform the following experiment: i) estimation of original WTP values at two (or more) different sites, ii) attempt to predict the value of one site from the observations of the other, and iii) report transfer errors (Barton, 2002; Hanley et al., 2006b,c). This is mainly explained by the fact that it is assumed in function transfer exercises that the statistical relationship between the dependent and independent variables are the same for both study as well as policy site and therefore, the reliability of the transfer exercise would suffice if we cannot reject this assumption.

As a rule of thumb, if the values obtained from benefit transfer are not statistically different from those obtained through site-specific estimation, convergent validity is established. The application of different econometric tests have been investigated and proposed for convergent validity in order to assess the accuracy of benefits transfer exercises (Bateman et al., 2002).

In addition to the question of statistical equality of site-specific and benefit transfer estimates of WTP, Loomis (1992) argues that the percentage errors from benefit transfer are of interest to policy makers. Since the results of the hypotheses tests depend on the width of the confidence intervals, percentage errors from benefit transfer can be fairly large even when two estimates are not statistically different.

Table 6.7 below illustrates transfer errors between WTP values estimated for Scotland obtained through the benefits function transfer exercise and the original WTP values for England & Wales individually and then in combination. Transfer errors, which in table 6.7 represent the percentage differences between the transferred welfare estimates and the original estimates for both the PC and DC models, can be used to evaluate differences between applying unadjusted or adjusted WTP values from these sites to Scotland.

Table 6.7 - Benefits transfer errors

Elicitation Method/Model		**PCCV OLS Model**	**DCCV Logit Model**	**Transfer errors (%)**	
				PC Model	**DC Model**
England	Mean WTP £/hh/year	44.8	167.0	-32.8%	-11.0%
	Median WTP £/hh/year	25.3		-32.8%	
Wales	Mean WTP £/hh/year	40.1	181.4	-24.9%	-18.1%
	Median WTP £/hh/year	22.7		-25.1%	
England & Wales	Mean WTP £/hh/year	44.5	167.9	-32.4%	-11.5%
	Median WTP £/hh/year	25.1		-32.3%	
Scotland (Benefits Transfer)	**Mean WTP £/hh/year**	**30.1**	**148.6**		
	Median WTP £/hh/year	**17.0**			

The application of benefits function transfer offers lower mean WTP values than if the original values for England and Wales had been directly transferred without adjustment. As could be expected, due to site similarity, results for Wales under the PC model offer lower transfer errors. However, error margins for Wales are larger than the ones found for England in the DC model[29]. Overall, transfer errors are greater for the application of the PC model as compared with the DC model which may be explained by the fact that more variables were transformed in the transferred PC model and therefore, more variability introduced than in the DC model.

[29] This is due to need of large sample sizes for DC in order to decrease confidence interval width.

We now proceed to quantify the extent of the variation introduced with respect to the original model by looking at transfer errors between the adjusted variables and the originals. Table 6.8 illustrates transfer errors for each transformed variable. This table also shows the relative weight of each variable with respect to the absolute value of the dependent variable, which gives an idea of the extent of the transformations introduced to the original model[30].

Table 6.8 Transfer errors for each adjusted variable and their relative weights in the original PCCV and DCCV models

Variable	**Weight (%) absolute value dependent variable** PCCV OLS	DCCV Logit	**England & Wales** (mean) X_E	**Scotland** (mean) X_S	**Variable transfer error (% Diff)** $\frac{(X_S - X_E)}{X_E}$	References
ln_delta_hl	8.4		0.58	0.32	-43.84	SEPA (2007a)
ln_inc	35.4	45.3	6.40	6.23	-2.63	DWP (2007)
children	0.8		0.29	0.26	-10.88	Scottish Executive (2007)
edu_12	4.8		0.36	0.19	-47.22	Scottish Executive (2007)
edu_35	8.1	8.2	0.34	0.36	5.88	Scottish Executive (2007)
use	1.4		0.84	0.78	-7.14	Mori (2006)
pol_control	3.7	2.4	0.85	0.55	-35.04	Mori (2006)
$\lvert$*ln_wtp*$\rvert$	**62.6**					
$\lvert$*dc_choice*$\rvert$		**56.0**				

Overall, it was possible to introduce a higher degree of variable transformation in the PCCV OLS model than in the DCCV model. In the adjusted OLS model, we have introduced adjustments to those independent variables (or parts of the original regression) which account for almost 63% of the absolute mean value of the dependent variable (*ln_wtp*). A total of 6 variables were adjusted (table 6.8). In contrast, only 3 variables were adjusted in respect to the DCCV model, which accounts for 56% of the mean absolute value of *dc_choice* in the original model.

By looking at the percentage difference between original and adjusted mean values of each transformed variable, it is possible to further evaluate and analyse where the main

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30 In a standard regression, mean values for the dependent variable are given by the independent variables multiplied by their coefficients plus the intercept (the amount of information that cannot be explained in the model) and the error term. Weights are used to illustrate the degree of transformation which has been introduced to the original information that provide the absolute mean value of the dependent variable in these models. Weights are for absolute mean values of the dependent variable, as we need to consider also those variables with negative coefficients that would also play a part in explaining WTP values. Mean values of the dependent variable do not take into account error terms.

differences lie. This analysis will help us to shed some light upon the WTP transfer errors from the benefits function exercise reported in table 6.7.

In the original regressions, income levels hold the majority of the explanatory power of variation in WTP estimates. However contrary to expected results, income differences between England and Wales combined and Scotland only account for 2% which is not enough to explain WTP transfer errors. Higher transfer errors (beyond 35%) between original and transferred variables are found for *ln_delta_hl, edu_12, pol_control.*

Differences in *ln_delta_hl* can be easily explained by the fact that more water bodies are already classified as being of good quality in Scotland as compared to England and Wales combined. Therefore, if baseline water quality levels are of higher quality than in the original study, it would be expected that WTP values for improvements would be somewhat lower.

Scottish household statistics (Scottish Executive, 2007) were used to derive figures of education levels in Scotland for the variables *edu_12* and *edu_35.* The comparison of these statistics to the ones used for England and Wales shows large differences (c. 47%) between the number of people with low education levels (e.g. between primary and O levels). Statistical tests for equivalence of mean values for each variable have not been applied, because standard errors are often not included in reported statistics. It is also impossible to validate this figure by comparing these statistics with other data sources, as both sources are regulated by the National Statistics Office (England and Wales household data comes from the family resources survey published on an annual basis by the UK's department of Work and Pensions). Nevertheless, this raises questions about the validity of employing different data sources for the original study and the policy sites for inclusion into a benefits function transfer exercise.

This issue also applies to explaining transfer errors for the attitude to water protection and use variables (*use* and *pol_control).* It is impossible to tell if differences in mean values for these variables do reflect actual differences in peoples' preferences to water resources between the two sites or if they are just due to differences in the way information was gathered or the interviews were conducted in the studies.

It is important to remember that it is assumed in function transfer exercises that the statistical relationship between the dependent and independent variables are the same for both study as well as policy site. Accordingly, statistical significance between estimates and the notion of percentage errors presumes that the site-specific benefit estimate is a ‘‘true’’ measure of benefits for the policy site. Reported transfer errors and lower WTP estimates especially compared with Wales (one would expect comparable WTP values due to site similarities with Scotland) raises issues about transferring the benefits function from this study.

In this case, we have been unable to explain the main differences between both case study sites. Nevertheless, variability has been quantified; though it is not possible to conclude if variation is due to the fact that both sites are inherently different or because data sources employed for variable transformation are not transferrable or if transfer errors are due to some other causes. We have discovered that variable transformation has increased transfer margin errors of mean WTP and of individual variables, but have not found any evidence to validate the acceptance of the transferred values through benefits function transfer. It is obvious that the site similarity condition does not hold true in this case.

6.2.3. Conclusions

This section has assessed the suitability of applying different forms of the benefits transfer method to Scotland in order to derive benefits estimates for improvements in water quality due to the implementation of the WFD.

Following a review of available studies, we have concluded that the application of meta-analysis is not possible at this stage. We have come to this conclusion mainly because of the small number of studies available and issues regarding contextual and methodological differences between original studies, which make their aggregation impossible without inferring a high degree of manipulation. Aggregation bias would impact on any expected results. The main limitation identified was the impossibility to group original studies under the same water quality metric. Ideally, future research should focus on the identification of water quality indexes which are tailored to the WFD water status labels.

The second part of this section illustrates the results and limitations of applying a benefits function transfer exercise from a recent CVM study conducted in England and Wales to Scottish conditions. Even though we were able to introduce a high degree of adjustment to original variables in the reported benefits functions (especially in the PCCV OLS model), we were not able to deduce the main reasons for variation in the WTP estimates. Available validity tests for a benefits transfer function fail to provide any information about differences between the original valuation site and the policy site. These tests mainly relate to issues of convergence between models. We feel that it is time that the academic literature focuses on the development of simple tests that quantify/assess similarity between an original and a policy site which would allow for a quick assessment of the validity of a benefits function transfer exercise of water quality changes.

However, there is consent that benefits transfer is never preferable to an original valuation exercise. Adjusting values from elsewhere to respond to a policy need in another site should be considered as a second best option in environmental policy appraisal. We believe that our inconclusive results from the benefits transfer exercise grant the undertaking of an original valuation study to the estimation of WFD related benefits.

6.3. Part II - Using choice experiments for the estimation of the non-market benefits of the WFD

As introduced in chapter 3, for the justification of standard-setting derogations, the costs of reducing pollution at farm level need to be compared with the associated benefits of water quality improvements. The main rationale of applying benefit assessment of environmental quality improvement is that the lowering of the environmental standards needs to be: i) sociably justifiable in light of the WFD; and ii) following economic theory, the optimal point of pollution control (where costs equal benefits) is the only point when a satisfactory outcome for both society and the farmer can be found. The main rationale for the application of CBA to justify standard-setting derogations, is to achieve economic efficiency in exemptions decisions.

In this chapter, we are exploring the application of different valuation techniques for the estimation, in monetary terms, of the non-market benefits of improvements to the Scottish water environment brought about by the WFD. The previous section has offered the justification for the undertaking of an original "Scottish" valuation study. This is namely due to the impossibility to validate BFT estimates. Accordingly, this section presents the results of a practical application of a stated preference valuation technique: choice experiments (CE).

The CE survey covers a combined assessment of use and non-use type of values placed by households in Scotland on the total non-market benefits of WFD improvements. WTP estimates for water quality improvements to Scottish standing waters (lochs) and inland running waters (rivers) are estimated separately. The main aspects we wanted to be able to analyse from the survey relate to public perceptions about restoring river and loch quality to and beyond good status for the whole of Scotland, time preference for the improvements (2015 versus 2028) and whether there are regional differences in preferences for water quality changes.

The next section begins by describing the choice experiment method of environmental valuation and outlines some examples of its application to water quality improvements. This is followed by an overview of the main approach taken, which describes the survey design, and an explanation of the methodology employed. Next, an analysis of the results is presented. This section ends by offering an outline of the main conclusions of the study and highlighting areas for further research[31].

6.3.1. Choice experiments

To derive monetary estimates of overall environmental benefits (including both use and non-use values) of the WFD in Scotland, a stated preference method - Choice Experiment (CE) - has been applied. CE is used in order to derive a top down estimate of the value of the proposed changes arising from alternative WFD objectives implementation scenarios. The proposed approach is mindful of the need to derive marginal WTP values for those changes and to estimate separate values for different parts of Scotland. The main advantage of CE over other stated preference valuation techniques such as Contingent Valuation (CV) is that CE allows for the exploration of marginal changes in different attributes of the environmental good being explored, in addition to offering value estimates for the 'whole' of the policy programme. This feature of CE is in theory, particularly well suited for the development of benefits curves for the achievement of Good Status. This aspect is explored in this study.

In a choice experiment, respondents choose from a range of alternatives (or 'goods') being offered. The good is a composite bundle of attributes including a price. That alternative with the highest (expected) utility is chosen. The choice data then allow identification of choice coefficients and implicit prices. In the environmental context, the choice offered usually consists of a number of *proposed changes* and a *status quo or reference option*. The alternatives are characterised by a number of attributes that ideally comprise all relevant aspects a respondent ascribes to a certain 'good' at stake. The attributes – being quantitative or qualitative dimensions of characteristics – consist of a number of levels. Within and between the choice sets presented to the respondent, the attribute levels and thus the goods or commodities presented vary, usually according to an experimental design.

An experimental design is necessary to make sure that utility changes are identifiable for at least each of the attributes separately and is therefore a crucial part of any CE study. A main-

[31] Relevant background information for this study has been included in the annexes due to space limitations in the main body of this thesis. Annexes offer information on the overall representativeness of the survey, the analysis of responses to attitudes, opinions and uses of the water environment questions, experimental design for the choice experiment and all the questionnaire materials employed (e.g. the main survey, showcards and example choice cards).

effects only design allows for the estimation of effects in differences between levels of attributes. All second and higher level interactions between attributes are then assumed to be statistically insignificant and zero. Louviere et al. (2000) have pointed out that main effects would commonly explain between 70 – 90 percent of the variance in choice. Allowing for two-way interactions to be estimated is potentially desirable, as virtually all of the variance would be explained by main effects and two-way interactions taken together. However, an experimental design that allows for the calculation of two-way interactions is also considerably larger, implying that larger sample sizes would be required.

The experimental design of choice experiment surveys requires a careful balance between information gained from each respondent and the 'quality' of that information. Both the number of choice sets subsequently offered to respondents and the number and complexity of attributes that describe each alternative within a choice set can be expected to increase the cognitive demand of the choice exercise. Too many choice sets may lead to fatigue effects, and too many attributes may confuse some of the respondents and lead to a more frequent application of cognitive 'shortcuts' (heuristics) for choice. One such choice strategy would be to consistently choose an alternative that is more attractive in one attribute, for example price, disregarding any of the trade-offs between this attribute and the other attributes offered. Although it is not clear how, and if, both fatigue effects and choice heuristics impact on the information required to generate WTP estimates, it is common to consider these aspects in the design stage of choice experiments. Additionally, the number of choice sets per respondent and the number of attributes is further constrained by the fact that they have implications for required sample size. Sample size, in turn, is also constrained by the size of the budget available. As a rule of thumb for many applications, the number of choice sets should be somewhere between 4 – 16, and the number of attributes between 2 – 8 (Hensher et al., 2005). For example, empirical research on the appropriate number of attributes to be applied in a choice experiment has concluded that including more than 4 to 5 attributes in a choice set may lead to a severe detriment to the quality of the data collected due to the task complexity (see Mazotta and Opaluch, 1995)

As an example of applying CE to the valuation of water quality improvements, table 6.9 outlines some examples of CE attributes that have been previously used in the academic literature to value changes in water quality. Note that only two of these studies have been applied specifically to the WFD. Though neither of these is immediately fit for our purposes.

Table 6.9 CE attributes used in the literature to represent proxies for water quality improvements

Reference	Objective of the study	CE attributes used for elicitation
Adamowicz et al. (1994)	WTP for water-based recreation in Alberta (Canada).	Landscape terrain, fish size, catch rate, water quality, facilities (eg. campsite), distance from home and fish species.
Burton et al. (2000)	Public preferences for catchment management plans in the Moore Catchment (Australia) with problems of salinity, eutrophication and flooding. All problems linked to farming activities.	Area of farmland affected by salinity, area of farmland planted with trees, ecological impacts on off-farm wetlands, risk of major flood, changes in farm income and annual contribution to management plan.
Heberling et al. (2000)	Benefits of reducing pollution from acid mine drainage in Pennsylvania (US).	Overall water quality (measured according to what uses could be made of the stream with the levels: "drinkable", "fishable" and "swimmable"), miles of river restored, travel time from home to site, easy access points and household costs.
Georgiou et al. (2000)	Contingent ranking (CE related technique) to estimate the benefits of water quality improvements in the River Tame (England). Estimation WTP for marginal reductions in BOD and total ammonia.	Respondents were asked to rank 3 combinations out of 4 attributes: 1) type of fishing (trout/salmon and good game; some game fish species return; a few game fish species return; fish stocks extinct); 2) plants and wildlife (otters survive; increase in number and types of insects and greater numbers of bids; more plants and waterfowl; very limited wildlife); 3) boating and swimming (both, boating only, swimming only, neither); and 4) cost (extra council taxes): £2.50/month, £1.25/month, £0.42/month, zero).
Hanley et al. (2006c)	WTP for improvements in three components of ecological status under the WFD in the rivers Clyde (Scotland) and Wear (England)	Three attributes selected which were set at two quality levels: Fair and Good. 1) Ecology (Good: salmon, trout and course fish and a wide range of water plants, insects and birds; Fair: only course fish and a poor range of water plants, insects and birds); 2) Aesthetics/Appearance (Good: no sewage or litter; Fair: some sewage or litter); and 3) RiverBanks conditions (Good: banks with plenty of trees and plants and only natural erosion; Fair: banks with few trees and plants and evidence of accelerated erosion).
Baker et al. (2007)	WTP by households in England and Wales on improvements to the water environment brought about by the WFD	Five environmental attributes (policy objectives) which represented 3 different levels of water quality status (poor, moderate and high water quality) and household costs (extra water bills). Water quality attributes were divided as follows: 2 represented baseline water quality status at national and local levels respectively, 2 represented different levels of water quality improvements from baseline status by 2015 at national and local areas and the last one represented combined levels of improvements from baseline at local and national levels by 2027.

6.3.2. *Main approach*

Our approach follows, as a starting point, the same design used in a similar study that has recently been carried out in England and Wales (Baker et al, 2007). This study, which was undertaken by NERA for the UK CRP[32], estimated the economic value placed by English and Welsh households for water quality improvements at local and national level for non-market WFD benefits in water quality. The application of three different elicitation methods (CV Payment Card and Dichotomous Choice and CE) were tested within the same survey instrument. Although this raises issues about how WTP values may be affected by concentration and awareness levels of the respondents when faced with different elicitation formats at once and the order in which the formats are presented to respondents, we do not pursue this issue here. Nevertheless, it seemed reasonable to apply the CRP methodology to our study as there is the potential to test some of its instruments in order to obtain a consistent valuation format across the UK. From now on we will refer to this study as the "***CRP study***".

For the design of visual materials to provide background information to the respondents, the *CRP study* applied a three level status classification (i.e. high, moderate and low water quality) to match those of the WFD. In addition, national maps and pie charts were used to represent baseline water quality levels, and basic illustrations and short text to describe in simple terms the ecology associated with each quality level for generic types of water bodies (i.e. rural river, lake, urban river, coastal water body). Based on this information respondents were ask to put monetary values/make choices for/between different levels of improvements at a national and local scale.

The study presented here makes only a few changes to the CRP study design. This comprised differentiating between rivers and lochs (instead of asking respondents to value the whole of the water environment) and changing the status levels to “no”, “few” and “many” problems categories, which was considered would make it more explicit to respondents that there are interactions between pressures to the water environment and water quality. In addition, it was decided not to differentiate environmental attributes between national and local improvements of water quality as the CRP study did in order to assess

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[32] The UK Collaborative Programme for research on the economic analysis of the WFD. The CRP aims to commission research on those tools necessary for the economic analysis for the development of River Basin Management plans. For a full list of partner organisations and details of their reports, see http://www.wfdcrp.co.uk/.

differences in preferences for improvements between river basin districts in England and Wales. The main reason for this is that Scotland is essentially one river basin district (RBD). The Scottish RBD covers most of the country (central and north areas) and in the South, Scotland shares half of the Solway-Tweed RBD with England. Regional differences were evaluated by allowing through the experimental design of the CE for the estimation of regional models by location and by river basin district.

6.3.2.1. Development of baseline scenarios for rivers and lochs

The application of CE (or any stated preference technique) to any public policy change requires the definition of a counterfactual. This refers to the development of baseline policy scenarios to which the proposed improvements can be compared and scaled to by the respondents and the analyst. In this case, the counterfactual has been defined as the current or baseline water quality situation in Scotland (year 2008). We assume that no deterioration (one of the environmental objectives of the Directive) is granted at no extra costs (apart from current levels of spending in maintaining current status) and that similarly, no improvements in water quality will be achieved by 2015.

Another alternative definition of the counterfactual would have been the no policy option. However, it is impossible to assess how water quality levels would have developed without the implementation of the Directive, since its introduction in 2001, or since it was categorically transposed in Scotland in 2003 through the Water Environment Water Services (Scotland) Act. This is mainly because SEPA is using different water quality measurement units.

Baseline water quality levels for rivers and lochs have been taken from SEPA's environmental characterisation data for the Scottish River Basin District and the Solway-Tweed River Basin District (SEPA, 2005a,b). The adopted unit to measure quantity of each status level is hectares of catchment area for rivers, and hectares of surface water area for lochs. Counterfactual scenarios provide a contextual reference to respondents for the valuation exercise, and a baseline for marginal WTP estimates associated with a unit change.

The CRP study undertook extensive research on the way the general public perceives current levels and changes in the ecological status of water bodies. The study concluded that respondents found it very difficult to judge water status except at some superficial level by how it looks and smells. Accordingly, the CRP study decided to apply a very simple metric of ecological status of three levels: high, medium and low water quality[33].

The main benefits of applying a three level status classification is that respondents, when also presented with background information materials (such as baseline water quality level maps and descriptive texts and illustrations outlining the main benefits of the improvements), are able to value small differences in water quality status at national level and also the reasons why status has changed when compared with improvement scenarios. In addition, a basic three level status classification system is easy to match with the five WFD water quality status levels (poor, fair, moderate, good and high). This allows the study to capture the most policy relevant improvements brought about by the WFD, which ultimately are movements between status categories.

In our study, a simple three level water quality status classification was also applied. However, the labels of the different categories were modified, as were some of the texts to describe status and benefits of improvements. After extensive deliberation with water ecologists and experts at MLURI and testing *the CRP study* materials with the general public, it was decided that 'no, few and many problems' status labels would be more appropriate to define water quality status. There are several reasons for this: i) these labels make it more explicit to respondents that interactions exist between pressures on the water environment and water quality. Though we cannot distinguish values on the basis of types of pressures to be addressed, it was considered essential to make some reference to this issue; and ii) it offers an appropriate linkage between water quality levels and SEPA's pressures and impact data used in this study to defined baseline conditions. SEPA has identified those water bodies that most likely will reach or fail to reach Good Status by 2015. However, some water bodies still need to be classified to a high degree of confidence and therefore, were labelled as having 'few problems' in this study.

Finally, national GIS maps, pie charts to represent baseline water quality levels and the wording of status descriptions were adjusted to account for Scottish conditions. All these materials can be found in the annexes VII to XIII.

[33] This simplification is one of the main drawbacks of applying CE to complex goods and remains one of the main criticisms of stated preferences techniques (Bateman et al., 2000)

6.3.3. Choice Experiment design - Attributes and levels

The number of attributes applied was constrained by the scope of the study (i.e. the need to match Scotland's wide population), and considering the trade-offs between information gain and task complexity mentioned above. The main aspects we wanted to be able to analyse from the survey related to public perceptions about restoring river and loch quality to and beyond Good Status for the whole of Scotland, time preference of the improvements (2015 versus 2028), and whether there are significant regional differences in preferences towards water quality changes. Table 6.10 offers an illustration of the selected attributes.

Table 6.10 Choice experiment attributes

Attribute	Components	Definition
r7	*lr0, mr0, hr0*	Percent low, medium, high quality for Scottish rivers at time=0 (current conditions)
	lr7, mr7, hr7	Percent low, medium, high quality for rivers at time=7 (in 2015)
l7	*lloch0, mloch0, hloch0*	Percent low, medium, high quality for Scottish lochs at time=0 (current conditions)
	lloch7, mloch7, hloch7	Percent low, medium, high quality for lochs at time=7 (in 2015)
r20	*hriver20*	Percent high quality for Scottish rivers at time=20 (in 2028)
l20	*hloch20*	Percent high quality for Scottish lochs at time=20 (in 2028)
Price	*Bill*	Increase in water bill and other household expenses

Four environmental attributes were designed after careful consideration of the research question at stake. Their levels represent hypothetical improvement scenarios which can be thought of as different policy objectives for achieving good status in Scotland.

Two of these attributes (r7 and l7) have four levels of water quality improvements, with levels illustrating four predefined water quality improvements scenarios generated from baseline levels (no change option) for rivers and lochs in 7 years time (up to 2015). Water quality improvements were presented to respondents as the combination of percentages of the three water status categories (no, few and many problems). We simultaneously showed predefined % increases in high water quality, the subsequent increase or decrease in medium status and % decrease in low water quality. Table 6.11 illustrates how improvement levels were generated (based on *the CRP study* approach). Results for table 6.11 are in the annexes (see annex VII) [34].

[34] For a practical explanation about how these levels have been set, see Baker et al., 2007

Table 6.11 Designing water quality improvement levels - formulas

Attribute	Components	Level 1	Level 2	Level 3	Level 4
r7	**lr7**	0	1/4 lr0	1/2 lr0	3/4 lr0
	hr7	hr0+3/4(mr0-Δlr7)	hr0+1/2(mr0-Δlr7)	hr0+1/4(mr0-Δlr7)	hr0+0.1(mr0-Δlr7)
l7	**lloch7**	0	1/4 lloch0	1/2 lloch0	3/4 lloch0
	hloch7	hloch0+3/4(mloch0-Δlloch7)	hloch0+1/2(mloch0-Δlloch7)	hloch0+1/4(mloch0-Δlloch7)	hloch0+0.1(mloch0-Δlloch7)
r20	**hriver20**	95%	75%		
l20	**hloch20**	95%	75%		

r=rivers, loch=lochs, l=low, h=high, 0= current conditions; 7= 2015; 20 = 2028

Quality changes in rivers and lochs in 20 years time, r20 and l20, are represented by two predefined levels of improvements in the high quality category for rivers and lochs from baseline status by 2028. These two levels for the quality attribute in 20 years time are 75% and 95% and illustrate the total percentage of rivers and lochs classified as ‘high’ (or ‘no problems’) in 2028.

The cost attribute “*bill”* had seven levels {5,10,20,40,50,75,100} and was described as an increase in water bills and other household expenses in pounds per household per year (in perpetuity).

An important modification to *the CRP study* is that status categories are not analysed as different attributes. *The CRP study* created attributes separately for high and low water quality status (even though levels were still shown together to respondents through water quality pie-charts). In addition, their experimental design made sure that there was no correlation between these quality attributes. Arguably, this approach would sustain only if respondents' choices were influenced by the percentage levels of high and low quality separately for each of these attributes. In *the CRP study*, WTP estimates are slightly lower for avoiding an increase in the percentage of ‘low’ quality than for a one per cent increase in ‘high’ quality, but differences are far from being statistically significant. It is difficult to say to what degree respondents really considered trade-offs between high and low quality or just focused on either changes in high or low quality levels on their own.

The CRP study design also leads to combinations where an increased percentage in low quality is associated with a percentage increase in high quality. If water quality improvements are supposed to be the result of WFD implementation, such combinations seem to be quite unlikely and not very plausible. As an aside, it is questionable whether

WTP to avoid increases in low quality would be the appropriate format to assess welfare impacts if the current situation is implied as a reference basis (Knetsch, 2005). In other words, the inclusion of an independent low quality attribute may possibly not be of any use for an assessment of welfare impacts.

In the current study, it was decided that quality levels for the low and high quality categories should still be presented together (alongside the medium category), but water quality improvements should be conceived as a single attribute, where increases in 'high' water quality are always associated with decreases in 'low' water quality. We therefore estimate WTP for water quality improvements in the 'no problems' category that are assumed to include the value of reducing both 'many' and 'few' problem category percentages.

Furthermore, trials of the experimental design run before the survey, showed that respondents found it easier to base their choices in the whole graph rather than on proportions of quality, which mainly represents overall improvements scenarios with respect to the no change option. It is therefore not clear that respondents can discern between different quality categories, as it is very difficult to separate quality conditions between the scenarios.

In addition, it is easier to develop policy scenarios from a WTP estimate of attaining good status. We were interested in exploring WTP for % improvement from lower water quality status to the "no problem" category (namely the achievement of Good Status quality and beyond).

The five attributes and levels presented in tables 6.10 and 6.11 were assigned to choice sets using a fraction of the full factorial design (1/6th of 6912 questions) for an independent analysis of three different locations across Scotland: South, Central and North. In order to estimate main effects as precisely as possible the choice experiment design was constructed so that the same levels of any factor do not occur in both choices offered together in a pair[35].

In the surveys, each respondent evaluated 8 choice questions. Each choice card illustrated three options: Options A and B, which offer different improvement levels for each environmental attribute at a positive cost; and the no change option, which offer baseline

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[35] The experimental design for this study was undertaken with analytical aid from Biomathematics and Statistics Scotland (BioSS) staff. Further information on this design can be found in annex VIII.

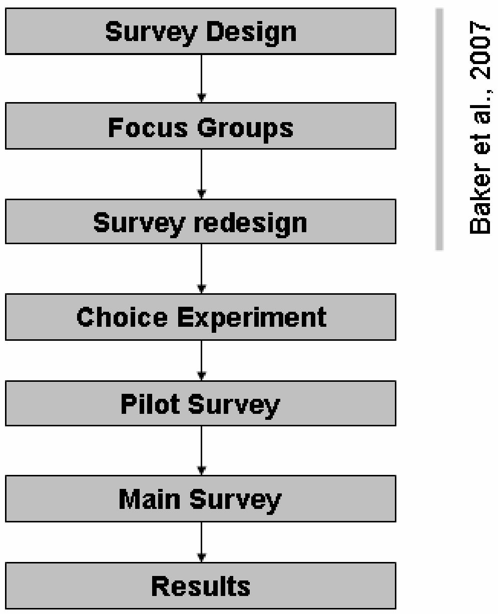
quality levels for the environmental attributes at no extra costs (zero costs). An example choice card can be found in the annexes (annex X).

Those cases where a factor had the same level in both options were excluded from the design. In addition, implausible combinations were removed; as an example, level 1 in the high quality category for r7 and l7, which was set at 83 and 81% for rivers and lochs respectively, could not be matched with level 2 for r20 and l20, which was set at 75% high status by 2028. Finally, dominant choices were intentionally left in[36].

6.3.4. Development of the survey instrument and data collection

There are several common elements to the administration of any stated preference method, specifically, the survey design and size, initial survey focus groups to explore whether the survey is comprehensible to respondents, survey pre-test and potential redesign, followed by survey administration. The final stage is data codification and analysis.

Figure 6.2 CE main survey design and administration stages



[36] Inclusion of such questions retains statistical balance in the design and hence minimises difficulties of confounding between factors, which could lead to potential difficulties in interpreting the results.

These stages are set out in Figure 6.2 and further described in turn below. Note that for our study, the first three stages (fig. 6.2) were highly influenced by the *CRP study*. The approach here follows the same design with slight modifications. Changes in the design of the choice experiment have been outlined above. Below, we describe alterations in the questionnaire design and the description of water quality improvement scenarios.

6.3.4.1. Survey design

At the survey design stage, scientific information is translated into a standard format for the CE survey. The important element to clarify is the good or environmental change to be valued. In this CE survey, the choice sets must be designed to convey the benefits related to a policy in terms of the levels of environmental attributes that we will have previously identified. This decision is in part statistically determined by the choice of the survey sample size.

Following *the CRP study* approach, this task involved the development of coloured maps, however, we introduce a distinction between rivers and lochs whereby baseline water quality levels were presented in two different maps, in order to explore if a difference in people's preferences for improvements in running and standing waters exists. SEPA's water quality characterisation data for rivers and lochs was used to develop GIS maps for the whole of Scotland (SEPA, 2005a,b and 2007a,b). These maps are presented in the annexes.

The majority of contextual instruments (i.e. explanation of attributes and levels, description of the reference baseline and scenario description) for the exercise were mostly borrowed with minor adjustments from *the CRP study*, e.g. drawings depicting water quality levels, colour schemes to reflect quality classes in the GIS maps (all these materials can be found in the annexes - Annexes IX and X). This meant that we were applying already tested materials. However, all survey instruments were retested to assess their suitability. Expert water ecologists were asked to provide their opinion on the illustrations and background texts, which have been employed to describe baseline water quality levels for rivers and lochs and the improvement scenarios. In addition, members of the public were approached in order to gauge their views and levels of understanding of all the survey instruments and background materials employed. Minor changes were required at this stage, the main one requiring a change in quality label names from “high, medium and poor” to “no, few and many problems”, as mentioned above.

In addition, a small survey pre-test consisting of 48 interviews was undertaken prior to the main survey to ensure that the survey information and the questions were being interpreted correctly. The pilot study showed that respondents were making choices consistent with theoretical expectations and that survey instruments were broadly understood. Furthermore, it proved that attributes and levels, including the price vector, had been appropriately designed[37].

6.3.4.2. Sampling frame and survey administration

Sample sizes can be based on experience in determining statistical validity using the relevant models for analysing CE data. In this case, a total number of 432 interviews were conducted as part of the main survey. This was based on the minimum number of respondents needed to cover a statistically efficient choice set sifted from a large factorial that represented all unique possible combinations of the choice set attributes and levels proposed in the study design.

Surveys were carried out using face-to-face interviews with members of the public. This part of the process was undertaken by a private market research company with previous experience in administering SP surveys. A flexible quota system approach for the survey sample was designed in order to ensure that samples were to a certain degree representative of the general population of Scotland (minimum quota limits were set up for age, social grade and location - further information in the annexes - See annex XI). This stratification was also mindful of regional variation which would allow for a geographical breakdown of the final results. Sample sizes were selected according to the CE experimental design to ensure that survey outcomes would have the required statistical relevance. This is particularly important in relation to the CE surveys to ensure that all attribute combinations are adequately covered.

Before moving on to describing in detail the methodology employed for the analysis of the choice task exercise, it is important to refer the reader to annexes XII and XIII, which report the representativeness of the sampled population as compared to Scottish Households official

[37] This involves an analysis of the pattern of choice for the proposed bid levels, i.e. create a matrix of chosen and rejected price levels to assess how often the highest price is being chosen over the lowest. As a rule of thumb, if the highest is chosen over the lowest more than 50% of the time, there is a case for increasing the price range.

statistics and outlines a brief analysis of the responses to supporting questions included in the survey. Answers to background questions provide revealed preference information about attitudes, opinions and uses of the water environment which are relevant for the modeling of choice preferences, information from these questions is used to define control variables in the logit models which are introduced below.

6.3.5. Methodology

The objective behind choice modelling is to explain an individual's choice among alternatives. An individual evaluates each alternative in a choice set of alternatives $j=1,..i,...,J$ by the 'attractiveness', or utility, of the alternative, U_j. The individual decision maker's rule is that he/she will compare $U_1, U_2,....U_i,.....U_j,.....U_J$ and choose the one which maximises his/her utility U.

An individual is assumed to have perfect knowledge on all aspects that contribute to the utility of an alternative. To the analyst, however, the set of aspects that contribute to the utility of an alternative and hence influence choice is restricted to a subset that can be observed and/or is controlled by the analyst. Although the analyst is limited with respect to the constituents of utility, the available knowledge allows making probabilistic statements on an individuals' choice.

The probability of the individual choosing alternative i over alternative j is equal to the probability that the utility of this alternative being greater than (or equal to) the utility associated with alternative j after all alternatives in the given choice set of alternatives $j=1,......i,......J$ have been evaluated.

$$Prob_i = Prob\,(U_i \geq U_j)\;\forall\; j \in j = 1,......,J; i \neq j \qquad 1)$$

Following Random Utility Theory (RUT) (e.g, Thurstone 1927; McFadden 1973; Manski 1977), the utility an individual is assumed to obtain from an alternative i is comprised of a set of observed attributes (deterministic, systematic component of utility or 'representative utility'; V_j) and a given amount of information which cannot be observed by the analyst (ε_j; also known as the (random) error term). Applying the Random Utility Model (RUM) to Equation 1) yields:

$$Prob_i = Prob\left[(V_i + \varepsilon_i) \geq (V_j + \varepsilon_j)\;\forall\; j \in j = 1,......,J; i \neq j)\right] \qquad 2)$$

Equation 2) can be re-arranged to emphasise the *random* utility maximisation rule observable by the analyst:

$$Prob_i = Prob\left[(\varepsilon_j - \varepsilon_i) \leq (V_i - V_j)\,\forall\, j \in j = 1,......,J;\, i \neq j)\right] \qquad 3)$$

Equation 3 states that the probability of an individual choosing alternative *i* is equal to the probability that the difference in the unobserved sources of utility of alternative *j* compared to alternative *i* is less than (or equal to) the difference in the observed sources of utility associated with alternative *j* after evaluating each and every alternative in the choice set of a total of $j=1,.....,i,.....J$ alternatives.

Randomness in utility maximisation is mindful of the need to find a way to deal with the unobserved individual idiosyncrasies of taste reflected by ε_j.

6.3.5.1. Multinomial Logit Model (MNL)

In the MNL model, the random individual specific terms in a set of *J* alternatives for an individual (i.e. ε_1 , ε_2 , ..., ε_i,....ε_j) are assumed to be identically and independently distributed (*IID*) across the sample population and related to the choice probability with a Type I extreme-value (Gumbel, Weibull, double-exponential) distribution. Given this assumption about ε_j we can derive a closed-form expression of the (conditional) probability to choose an alternative from a choice set of alternatives $j=1,..i,...,J$ as:

$$Prob_i = \frac{\exp(\mu' V_i)}{\sum \exp(\mu' V_J)} \qquad 4)$$

Equation 4 is basically stating that the probability of an individual choosing alternative *i* from a set of alternatives *J* is equal to the ratio between the exponential of the observed utility index for alternative *i* to the sum of the exponential of the observed utility indices for all *J* alternatives observed (also including the *ith* alternative). μ' is a scale parameter which is inversely proportional to the standard deviation of the error distribution. This parameter cannot be separately identified (typically assumed to be one) and because of the assumption stated above implies constant error variance. As $\beta \rightarrow \infty$ the model becomes deterministic (Hanley et al., 2006c).

Equation 5 illustrates an indirect utility function for alternative i (V_i) comprised of $n=1,\ldots,N$ attributes of alternative $i(X_{in})$. In this instance, it is considered linear in its arguments and additive with a constant term (θ). P_i denotes the 'price' associated with the choice of alternative i, an attribute that is of particular importance if choice modelling is applied for (economic) valuation purposes.

$$V_i = \theta_0 + \alpha P_i + \beta_n X_{in} \qquad 5)$$

Replacing V_i in Equation 4) with the indirect utility function in Equation 5) gives:

$$Prob_i = \frac{exp(\mu'(\theta_0 + \alpha P_i + \beta_n X_{in}))}{\sum exp(\mu'(\theta_0 + \alpha P_J + \beta_n X_{Jn}))} \qquad 6)$$

As the vector of attribute coefficients $\alpha,\beta' \rightarrow \infty$ the model becomes deterministic (Hanley et al., 2006c). Equation (6) can be estimated by conventional maximum likelihood procedures but bearing in mind that individual-specific characteristics (socio-economic data; such as income, age, sex...) cannot be entered into equation 5) as linear arguments, as Hanley et al., (2006c) points out; an individual's education is the same regardless of the alternative chosen, therefore these variables can only be introduced in the model by multiplication with the constant or the attributes.

6.3.5.2. Implicit prices

A common application of discrete choice models is the derivation of estimates of compensating variation or compensating surplus associated with changes in the levels of the non-price attributes being considered in the experiment. In addition to deriving estimates of welfare changes for the whole of improvement scenarios, this particular feature of choice modelling allows for the direct exploration of marginal changes over a range of (environmental) change.

Marginal WTP or implicit prices are the marginal rates of substitution (MRS) between the non-price attributes and the monetary 'price' attribute. The MRS is derived as the partial derivative of non-price attribute n with respect to price:

$$IP_n = -\beta_n / \alpha \qquad 7)$$

where α is the slope of the cost attribute and β_n is the parameter estimate of attribute n.

6.3.5.3. *Independence of irrelevant alternatives (IIA)*

In summary, there are three main assumptions that lead to the specification of the MNL model: i) the error terms are extreme value (or Gumbel) distributed; ii) the error terms are identically and independently distributed across alternatives; and, iii) the error terms are identically and independently distributed across observations/individuals.

The independence of irrelevant alternatives (IIA) property is the behavioural equivalent to the IID assumption. The IIA property states that, all else being equal, a person's choice between two alternative outcomes should be unaffected by the availability of any other choices (Cheng and Long, 2007).

Possible sources of violations of the IIA property may come from issues related to the experimental design, as choice alternatives can be mistakenly structured in such a way that they are close substitutes. In addition, the assumption of homogeneous preferences (i.e. constant error variances) among respondents is often regarded as an unrealistic behavioural assumption because each respondent may have different perception/unobserved characteristics that may influence his or her choice. Each individual places their own particular weight on their choice making, which leads to correlation across the utility of alternatives for each individual and again leads to violations of the IIA assumptions (Hensher et al., 2005 p480; Baskaran et al, 2007).

A variety of specification tests have been developed to test if the IIA assumption holds. The Hausman test (Hausman and McFadden, 1984) is the most widely applied (Hensher et al. 2005). We will provide further information about this test later in the results section.

In order to incorporate taste variation amongst respondents, some other specifications of the logit model that offer relaxation of the IIA assumption can be applied instead; such as, mixed logit (also known as Random Parameter Logit), nested logit, paired combinatorial logit and latent class logit models.

6.3.5.4. *The Mixed Logit (ML) model*

The ML model omits the three limitations of standard logit that affect the MNL model by allowing for random taste variation, unrestricted substitution patterns and correlation in unobserved factors (Campbell et al., 2006). The strong assumptions employed in MNL; *IID* (independently and identically distributed error terms) and *IIA* (independence of irrelevant alternatives) properties do not apply to this model.

Let β'_q be a vector of observed explanatory variables (including attributes of the alternatives, socio-economic characteristics of the individual...) for individual q in a choice situation with $j=1,...i,...J$ alternatives. The RUM expression of the utility associated with alternative i for individual q is:

$$U_{qi} = \beta'_{qi} x_{qi} + \varepsilon_{qi} \qquad 8)$$

, where ε_{ni} is a random error term with zero mean that is independently and identically distributed across individuals, choices and alternatives. The main difference with the MNL specification above is that β'_q is allowed to vary across individuals (and choice situations). This is achieved by introducing further stochastic elements into β'_q that are (potentially) correlated over alternatives and choices and heteroscedastic over alternatives and individuals, such that the stochastic component of the utility function of alternative i consists of two parts. Equation 9.

$$U_{qi} = \beta'_{qi} x_{qi} + \lfloor \eta_{qi} + \varepsilon_{qi} \rfloor \qquad 9)$$

, where η_{qi} is a random term with zero mean whose distribution over individuals and alternatives depends in general on underlying parameters and observed data relating to alternative i. η_{qi} can take on different distributional forms such as normal, lognormal, uniform or triangular in mixed logit models (Hensher et al., 2005). Denote the density of η_{qi} by $f(\eta_{qi} \mid \Omega)$, where Ω are the fixed parameters of the distribution. Since ε_{ni} is *IID* extreme value, the probability of choosing alternative i conditional on η_{qi} is logit:

$$L_{qi}\,(\beta_q \mid \eta_{qi}) = \frac{\exp(\beta'_q x_{qi} + \eta_{qi})}{\sum \exp(\beta'_q x_J + \eta_{qJ})} \qquad 10)$$

However, since η_{qi} is not known to the analyst, the *unconditional* choice probability (equation 11) involves integration of L_{qi} over all possible values of η_{qi} weighted by the density of η_{qi}.

$$Prob_{qi}(\beta_q \mid \Omega) = \int_{qi} Lqi\,(\beta_q \mid \eta_q i)\, f(\eta_{qi} \mid \Omega)\, \eta_{qi} \tag{11)}$$

Equation 11 illustrates the specification of the ML model, as the choice probability of $Prob_{qi}$ is a mixture of logits with $f(\cdot)$ as the mixing distribution. These probabilities do not display the *IIA* property and different substitution patterns can be attained by appropriate specifications of $f(\cdot)$. This is introduced through the random parameters by specifying each element of β'_q associated with an attribute of an alternative as, for example, having both a mean and a standard deviation; as opposed to fixed parameters which treat the standard deviation as zero such that all the behavioural information is contained in the mean (Hensher et al., 2005).

6.3.6. Results

6.3.6.1. The Hausman test

This test belongs to the category of choice partitioning tests that have been used in the literature to test for violations of the *IIA* in *MNL* models. Choice set partitioning tests compare the results from the full *MNL* model estimated with all outcomes (i.e., choices) to the results from a restricted estimation that includes only some of the outcomes. Basically, *IIA* holds when the estimated coefficients of the full model are statistically similar to those of the restricted one. If the test statistic is significant, the assumption of *IIA* is rejected and the conclusion is that the *MNL* model is inappropriate.

Equation 12 illustrates the specification of the Hausman test. This test compares the estimates $\hat{\beta}^f$, which are consistent and efficient if the null hypothesis is true, to the consistent but inefficient restricted estimates $\hat{\beta}^r$. The test is defined as:

$$H_0 : (\hat{\beta}^r - \hat{\beta}^f)'\left[Var(\hat{\beta}^r) - Var(\hat{\beta}^f)\right]^{-1}(\hat{\beta}^r - \hat{\beta}^f) \tag{12)}$$

In equation 12, $Var(\hat{\beta}^{r})$ and $Var(\hat{\beta}^{f})$ are the estimated covariance matrices. If IIA holds, equation 12, is asymptotically distributed as chi-square with degrees of freedom equal to the number of attributes considered in the restricted model. If the value of the test statistic is significantly greater than the critical value of the chi-squared distribution (at p=0.05), we conclude that the *IIA* assumption does not hold.

Table 6.12 reports the results of the Hausman test for *IIA*. This test was carried out in the general *MNL* model for the whole sample (3317 valid observations) and individually for each sub-sample divided by the three main location sites (i.e. north, south and central) and by river basin district. For each data structure, three versions of the *HM* test were computed, excluding either the reference, option A or option B.

Table 6.12 Results of the Hausman test for IIA

Sample	Number of observations	Omitted Alternative	Statistic	Significance level
All	2,858	Ref	20.31	***
	1,840	B	13.08	**
	1,936	A	8.74	
North	1,056	Ref	6.01	
	528	B	8.57	
	616	A	6.73	
Central	941	Ref	7.31	
	695	B	5.86	
	656	A	1.72	
South	861	Ref	11.30	**
	617	B	3.71	
	664	A	7.65	
SRBD	1,997	Ref	14.26	**
	1,224	B	10.51	*
	1,271	A	5.82	

Statistical significance:"*" at 90% level, "**" at 95%level and "***" at 99% level

These results of the Hausman test offer mixed conclusions. Violations of the *IIA* assumptions are reported in five out of fifteen instances. Two of the samples (i.e. north and central) pass the test. However, for the full sample, the south and the SRBD sample (which is comprised of the North and Central locations) *IIA* is rejected when some alternatives are omitted, even though these are (apart from the reference option in the full sample) rejections slightly below the 5% and 10% level.

Results indicate that the *IIA* assumption has to be rejected for the full sample. Thus model results should be treated with caution and more emphasis given to models that relax *IIA* such as mixed logit or latent class models, at least for models comprising the affected samples[38].

We now proceed to illustrate the results of the *MNL* and *ML* models for the different samples. Although results of the Hausman test point to violations of the *IIA* assumption in some of the samples, *MNL* models are still a useful starting point for the analysis of the choice data.

6.3.6.2. Choice option selection

In this section, we briefly describe our approach to identifying potential groups of respondents for exclusion from the analysis. Out of a total of 432 possible respondents to the survey (3,456 total choices), 6 respondents/entries (48 choices) had to be deleted from the sample altogether as we encounter data entry mistakes or invalid questionnaires.

Out of the remianing 3,408 observations, we had to discard 91 who answered "don't know" to one or all of the choice questions. Unfortunately, until the literature finds a method to include "don't know" responses in the analysis, this is the only possible way to deal with these responses (Hanley et al., 2006c)

Furthermore, Table 6.13 illustrates the percentage of respondents who did select the status quo over the improvement scenarios. Table 6.13 results show that the 'North' sub-sample exhibits a lower percentage of observations where the reference option has been chosen.

Selecting the "status quo" might suggest that these respondents are showing: i) "genuine zero bids", which is an indication that the respondent is not willing to pay anything because they do not value the good in a utility sense; or ii) "protest responses", which basically might reflect that the respondents do not want to make a choice/place a value for whatever reason

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38 Hausman test results for our samples are highly inconsistent; with subsamples (i.e. north and central) passing the test and pooled models (SRBD) failing to reject IIA violations. Inconsistent results have also been found in other studies and recent literature has started to question the validity of choice partitioning tests for evaluating IIA violations. Cheng and Long (2007) undertook a series of Monte Carlo simulations to evaluate different tests of IIA, including; the likelihood ratio test, the Small and Hsiao test and the Hausman test. For the Hausman test, the authors found out that the size properties of the test depend upon the data structure for the independent variables and that with some structures, this test shows substantial size distortion that is unaffected by sample size, which they offer as an explanation for inconsistencies.

apart from the one introduced before. In CE, there is no specific way of dealing with these types of responses (Hanley et al., 2006c).

Table 6.13 Choice pattern across the sample and sub-samples

Sample	Reference	Option A	Option B	Observations
All	13.8%	41.6%	44.5%	3317
North	4.0%	44.0%	52.0%	1100
Central	17.9%	42.8%	39.4%	1146
South	19.6%	38.0%	42.4%	1071
SRBD	11.1%	43.4%	45.5%	2246

6.3.6.3. MNL and ML models

Table 6.14 offers a brief description and summary statistics for the variables employed in the *MNL* and *ML* models. Water quality improvement attributes for rivers and lochs in 7 and 20 years time were estimated in the models as continuous variables using actual values. These represent percentage increases in the total area of rivers and lochs classified as having no quality problems by 2015 and 2028 from the counterfactual.

A series of control variables were seen to have an impact on the propensity to choose the reference option rather than alternatives A and B and on the attributes. These were included in the final specification of the models. These include: respondents' perceptions on current levels of rivers and lochs water quality status, attitudes towards water pollution control, levels of education, location, direct use of rivers and lochs, gender and whether the respondents gave careful consideration to the survey as judged by the interviewers or stated their households' income

Table 6.14 Variables description

Variable name	Description	Pooled		North		Central		South		SRBD	
		Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.
ASC	Alternative specific constant; =1 if alternative A and B are chosen, else 0										
River7GS	Environmental attribute - 4 levels - Percentage of RIVERS by area in Scotland in the No Problems category in 7 years										
Loch7GS	Environmental attribute - 4 levels - Percentage of LOCHS by area in Scotland in the no Problems category in 7 years										
River20GS	Environmental attribute - 2 levels - Percentage of RIVERS by area in Scotland in the No Problems category in 20 years										
Loch20GS	Environmental attribute - 2 levels -Percentage of LOCHS by area in Scotland in the no Problems category in 20 years										
Price	Attribute Cost										
Lochs_views	Perceptions of current ecological quality status LOCHS; =1 if respondents describe current quality as being "a great deal better than expected" or "somewhat better than expected"	0.07	(0.26)	0.05	(0.22)	0.10	(0.30)	0.06	(0.24)	0.07	(0.26)
Rivers_views	Perceptions of current ecological quality status RIVERS; =1 if respondents describe current quality as being "a great deal better than expected" or "somewhat better than expected"	0.18	(0.38)	0.21	(0.41)	0.23	(0.42)	0.08	(0.28)	0.22	(0.41)
Pollution control Attitudes	=1if wishing to continue improvements for pollution control	0.81	(0.39)	0.84	(0.37)	0.76	(0.43)	0.84	(0.37)	0.80	(0.40)
Income_miss	=1 if income is missing/not stated; 0 otherwise	0.47	(0.50)	0.36	(0.48)	0.25	(0.43)	0.83	(0.37)	0.30	(0.46)
Education_1	=1 if level of education between primary and O levels	0.53	(0.50)	0.54	(0.50)	0.59	(0.49)	0.45	(0.50)	0.57	(0.50)
Location_South	=1 if location is south	0.32	(0.47)	0.00	(0.00)	0.00	(0.00)	1.00	(0.00)	0.00	(0.00)
Location_Central	=1 if location is central	0.35	(0.48)	0.00	(0.00)	1.00	(0.00)	0.00	(0.00)	0.51	(0.50)
River_use	=1 if any use of rivers (often, sometimes)	0.73	(0.44)	0.81	(0.39)	0.65	(0.48)	0.75	(0.43)	0.73	(0.45)
Loch_use	=1 if any us e of lochs (often, sometimes)	0.54	(0.50)	0.59	(0.49)	0.51	(0.50)	0.52	(0.50)	0.55	(0.50)
Age<30	=1 if age below 30	0.26	(0.44)	0.28	(0.45)	0.20	(0.40)	0.31	(0.46)	0.24	(0.43)
Age_30-60	=1 if age between 30-60	0.54	(0.50)	0.62	(0.49)	0.54	(0.50)	0.45	(0.50)	0.58	(0.49)
Male	=1 if respondent is a male	0.48	(0.50)	0.52	(0.50)	0.48	(0.50)	0.43	(0.50)	0.50	(0.50)
Surv_consideration	=1if respondent gave careful consideration	0.55	(0.50)	0.91	(0.28)	0.60	(0.49)	0.13	(0.34)	0.75	(0.43)

Table 6.15 presents the estimates of the *MNL* and *ML* models for the whole sample. In the ML model a normal distribution was specified for the environmental attribute parameters while price was kept fixed. Estimates for the *ML* model were obtained using 500 random draws to simulate the sample likelihood.

Table 6.15 MNL and ML models estimates

	All Locations					
	LOGIT			MIXED		
Variable	Coeff.	SE	Sig	Coeff.	SE	Sig
ASC	1.6998	(0.3123)	***	2.3788	(0.6578)	***
River7GS	0.0189	(0.0019)	***	0.0230	(0.0030)	***
Loch7GS	0.0148	(0.0012)	***	0.0169	(0.0018)	***
River20GS	0.0069	(0.0026)	***	0.0067	(0.0031)	**
Loch20GS	0.0033	(0.0021)		0.0037	(0.0027)	
Price	-0.0166	(0.0026)	***	-0.0184	(0.0031)	***
*Lochs_views*Loch20GS*	-0.0129	(0.0031)	***	-0.0183	(0.0044)	***
*Rivers_views*River20GS*	-0.0033	(0.0027)		-0.0042	(0.0036)	
*Pollution control Attitudes*Price*	0.0090	(0.0022)	***	0.0096	(0.0025)	***
*Income_miss*Price*	-0.0056	(0.0018)	***	-0.0063	(0.0022)	***
*Education_2*Price*	0.0069	(0.0016)	***	0.0079	(0.0019)	***
*Location_South*Price*	-0.0147	(0.0023)	***	-0.0177	(0.0029)	***
*Location_Central*Price*	-0.0111	(0.0020)	***	-0.0131	(0.0023)	***
*River_use*ASC*	0.8040	(0.1337)	***	0.9830	(0.2080)	***
*Loch_use*ASC*	-0.4917	(0.1312)	***	-0.5922	(0.1730)	***
*Age<30*ASC*	1.2776	(0.1682)	***	1.5102	(0.2649)	***
*Age_30-60*ASC*	0.7002	(0.1278)	***	0.8940	(0.2032)	***
*Location_South*ASC*	-0.4009	(0.2095)	*	-0.4301	(0.2610)	*
*Location_Central*ASC*	-0.6786	(0.1942)	***	-0.8011	(0.2474)	***
*Male*ASC*	-0.2633	(0.1145)	**	-0.2971	(0.1482)	**
*Surv_consideration*ASC*	-0.9919	(0.0997)	***	-1.2226	(0.1837)	***
Standard deviation of parameters						
sdRiver7GS				0.0307	(0.0113)	**
sdLoch7GS				0.0160	(0.0383)	**
sdRiver20GS				0.0084	(0.6518)	
sdLoch20GS				0.0128	(0.4091)	
Observations	3317			3317		
Log-Likelihood	-3312.9			-3644.1		
Adj Pseudo R2	0.19			0.27		

Statistical significance:"*" at 90% level, "**" at 95%level and "***" at 99% level

Overall, both models are highly significant. Model fit as indicated by Pseudo R^2 is decent and improves for the *ML* model compared to the *MNL* specification.

In both models, the signs of the environmental and price attributes are consistent with a priori expectation and all attributes except *Loch20GS* are statistically significant at the 95 percent level or lower. This implies that respondents prefer improvements to water quality as compared to be left at baseline quality status levels. The coefficient of the Alternative Specific Constant (*ASC*) is positive and significant on average, indicating that there is a propensity to choose the water quality improvement alternatives over the status quo that cannot be explained by water quality improvement attributes. In other words, respondents on average may make use of unobserved attributes when choosing among the policy options, which could include expressing a generic preference for improvements irrespective of any combination of attribute levels. This tendency varies, however, with a number of individual specific characteristics, as we will outline below. The price attribute is negative and significant, showing that on average respondents are sensitive to increases in their water bill.

Regarding water quality improvements in 20 years (*River20GS, Loch20GS*), the attribute signs suggest that respondents prefer, on average, greater improvements in 20 years time over less (i.e. 95% over 75%). However, the coefficient for improvements in lochs in 20 years time is not significantly different from zero. Interactions with respondents' perception of river and loch water quality at present (*river_views, loch_views*) are both negative but significant only for lochs. Thus, perception of higher than expected water quality has a significant influence on preferences for improvements in lochs in 20 years time. Two possible explanations are: i) those respondents who perceived loch quality to be better than expected may accept lower levels of water quality, or ii) they may have perceived current levels of loch quality as sufficient and therefore shifted their attention to river water quality, or to remain at the baseline level.

Estimates of standard deviations for the environmental attribute parameters in the *ML* model, are also presented in table 6.15. Standard deviation parameters for quality improvements in rivers and lochs in seven years time are both statistically significant at the 95 percent level, but statistically insignificant for improvements in 20 years. This suggests the presence of unobserved heterogeneity of preferences for improvements in 7 years. This might imply that the major component of preference heterogeneity is the timing of water quality improvements which is a priori independent of the type of water body (i.e. rivers and lochs). Nevertheless, further research is needed to explain this unobserved heterogeneity of preferences by entering socio-economic variables as covariates in ML models.

Unsurprisingly, those respondents that wish to continue improvements for reducing water pollution (*Pollution_Control*) are less sensitive towards paying for them. Interaction in terms of location variables (*South, Central*) with *Price* yield negative and significant (99% level) coefficients in both models. This suggests that price sensitivity differs across regions.

Respondents below the age of 60 have a higher tendency to choose improvement alternatives over the status quo that cannot be explained by attributes compared to the reference base of respondents that are older than 60 years. The observation made from table 6.13 (choice pattern) that the status quo option was chosen more frequently in the South and Central regions is confirmed by the interactions of regional dummies with the *ASC*. Both are negative and significant, indicating that the propensity to choose improvement alternatives relative to the status quo is lower in the South and Central regions. The same effect is observed for male respondents. Interestingly, those who were rated by interviewers to have given the choice experiment careful consideration make, on average, less use of unobserved attributes for their choice. Using rivers or lochs sometimes or often has a contrary influence on choice. While river use increases the tendency to choose improvement alternatives over what would be expected from attribute information alone, loch use reduces this tendency.

6.3.6.3.1. Regional and river basin district models

In order to analyse potential regional difference in preferences, MNL and ML models were estimated for different samples of the survey by location. Model results are presented in tables 6.16 and 6.17. Table 6.16 introduces models estimates for samples of the survey undertaken in the south, central and north regions of Scotland separately. In addition, Table 6.17 shows model results by River Basin District (RBD). This is mindful of the need to develop economic analyses that are relevant for the water management units applied in the implementation of the WFD.

The coefficient of the price attribute and the fact that it is not significant in the North sample confirms our suspicions that price sensitivity differs by location. In this instance, respondents are less sensitive in the North, as compared with the other regions, to increases in their water bills in order to pay for the improvements. Nevertheless, these respondents show a strong preference towards improvements in seven years as opposed to 20 years time. Parameters reflecting river improvement in 20 years time are significant in the South and Central regions.

The variables interacting with the ASC explain the variation of the propensity to choose the water quality alternatives over the baseline best in the south region. Only age between 30 and 60 and the interviewer rating on respondents' level of consideration to the choice task are significant in the North, which leaves the ASC positive and highly significant. Whether the ASC is positive or negative overall depends on the sum of the ASC parameter and all ASC interaction parameters, where interacting variables are to be replaced by the population mean for a particular region.

Knowledge from the information present in the survey data can be extended and improved by further analysis of the supporting questions, more refined modelling of the choice data and an improved combination of both. The following aspects may be worth considering: i) an improved analysis of "uses" and "users" of rivers and lochs and how they relate to choices made; ii) entering attributes as dummy or effects coded to observe the linearity of the utility surface; iii) entering socio-economic variables as covariates in ML models to explain unobserved heterogeneity of preferences; iv) using other discrete choice models of the logit family that relax the IIA/IID assumptions such as the Nested Logit model or the Latent Class model; and v) using information on attribute considerations of respondents to account for (potential) non-compensatory decision making.

Table 6.16 Model results sampled by location

	NORTH						SOUTH						CENTRAL					
	LOGIT			MIXED			LOGIT			MIXED			LOGIT			MIXED		
Variable	Coeff.	SE	Sig	Coeff.	SE	Sig	Coeff.	SE	Sig	Coeff.	SE	Sig	Coeff.	SE	Sig	Coeff.	SE	Sig
ASC	6.5434	(1.2534)	***	13.3237	(7.6464)	*	0.4322	(0.5773)		0.5899	(0.8683)		1.9829	(0.4775)	***	2.3305	(0.6837)	***
River7GS	0.0308	(0.0033)	***	0.0614	(0.0214)	***	0.0113	(0.0036)	***	0.0143	(0.0049)	***	0.0136	(0.0032)	***	0.0203	(0.0056)	***
Loch7GS	0.0166	(0.0021)	***	0.0271	(0.0095)	***	0.0167	(0.0024)	***	0.0220	(0.0039)	***	0.0124	(0.0021)	***	0.0162	(0.0036)	***
River20GS	0.0003	(0.0043)		-0.0047	(0.0094)		0.0151	(0.0050)	***	0.0193	(0.0067)	***	0.0095	(0.0045)	**	0.0097	(0.0066)	
Loch20GS	-0.0004	(0.0034)		0.0009	(0.0076)		0.0035	(0.0042)		0.0027	(0.0060)		0.0053	(0.0037)		0.0042	(0.0055)	
Price	-0.0053	(0.0038)		-0.0132	(0.0082)		-0.0195	(0.0053)	***	-0.0252	(0.0076)	***	-0.0490	(0.0056)	***	-0.0645	(0.0075)	***
*Lochs_views*Loch20GS*	-0.0299	(0.0070)	***	-0.0560	(0.0315)	*	0.0596	(0.0115)	***	0.0781	(0.0204)	***	-0.0252	(0.0044)	***	-0.0291	(0.0069)	***
*Rivers_views*River20GS*	-0.0083	(0.0061)		-0.0136	(0.0145)		-0.0207	(0.0063)	***	-0.0254	(0.0104)	**	0.0018	(0.0039)		0.0021	(0.0054)	
*Pollution control Attitudes*Price*	-0.0048	(0.0036)		-0.0047	(0.0070)		-0.0065	(0.0041)		-0.0073	(0.0054)		0.0356	(0.0053)	***	0.0459	(0.0060)	***
*Income_miss*Price*	-0.0057	(0.0028)	**	-0.0104	(0.0057)	*	-0.0068	(0.0040)	*	-0.0081	(0.0055)		-0.0065	(0.0034)	*	-0.0069	(0.0046)	
*Education_1*Price*	0.0071	(0.0026)	***	0.0134	(0.0061)	**	0.0095	(0.0033)	***	0.0117	(0.0041)	***	0.0037	(0.0028)		0.0056	(0.0037)	
*River_use*ASC*	-0.2552	(0.5251)		-0.3309	(1.2142)		1.5484	(0.2245)	***	2.0257	(0.4449)	***	0.4703	(0.2159)	**	0.5723	(0.2708)	**
*Loch_use*ASC*	0.5819	(0.4156)		0.9360	(0.9940)		-0.7323	(0.2226)	***	-1.0059	(0.3663)	***	-0.5105	(0.2112)	**	-0.5091	(0.2503)	**
*Age<30*ASC*	-1.6719	(1.0682)		-3.4604	(2.5932)		3.1049	(0.2985)	***	3.9151	(0.6775)	***	0.2901	(0.2837)		0.1389	(0.3585)	
*Age_30-60*ASC*	-1.9483	(1.0466)	*	-3.5977	(2.8611)		2.1120	(0.2200)	***	2.7667	(0.5314)	***	-0.2599	(0.2089)		-0.3540	(0.2731)	
*Male*ASC*	0.0045	(0.3755)		-0.1609	(1.1159)		-0.2975	(0.2035)		-0.2176	(0.3029)		-0.3459	(0.1771)	*	-0.4353	(0.2274)	*
*Surv_consideration*ASC*	-2.4901	(0.3471)	***	-4.5423	(2.4071)	*	-1.3560	(0.1920)	***	-1.6846	(0.3584)	***	-0.9858	(0.1439)	***	-1.1157	(0.1959)	***
Standard deviation of parameters																		
sdRiver7GS				0.0859	(0.0440)	*				0.0272	(0.0154)	*				0.0594	(0.0169)	***
sdLoch7GS				0.0526	(0.0263)	**				0.0193	(0.0110)	*				0.0218	(0.0165)	
sdRiver20GS				0.0235	(0.0248)					0.0286	(0.6061)					0.0341	(0.0275)	
sdLoch20GS				0.0307	(0.0393)					0.0316	(0.5363)					0.0269	(0.0197)	
Observations	1100			1100			1071			1071			1146			1146		
LL	-913.03			-1208.47			-1125.57			-1176.61			-1189.71			-1259.01		
Adj Pseudo R2	0.16			0.38			0.30			0.34			0.18			0.23		

Statistical significance:"*" at 90% level, "**" at 95% level and "***" at 99% level

Table 6.17 Models results sampled by river basin

	SOUTH						SCOTTISH RIVER BASIN					
	LOGIT			MIXED			LOGIT			MIXED		
Variable	Coeff.	SE	Sig	Coeff.	SE	Sig	Coeff.	SE	Sig	Coeff.	SE	Sig
ASC	0.4322	(0.5773)		0.5899	(0.8683)		2.9890	(0.3809)	***	3.6267	(0.8761)	***
River7GS	0.0113	(0.0036)	***	0.0143	(0.0049)	***	0.0219	(0.0022)	***	0.0302	(0.0049)	***
Loch7GS	0.0167	(0.0024)	***	0.0220	(0.0039)	***	0.0142	(0.0015)	***	0.0177	(0.0029)	***
River20GS	0.0151	(0.0050)	***	0.0193	(0.0067)	***	0.0037	(0.0030)		0.0025	(0.0043)	
Loch20GS	0.0035	(0.0042)		0.0027	(0.0060)		0.0023	(0.0025)		0.0022	(0.0038)	
Price	-0.0195	(0.0053)	***	-0.0252	(0.0076)	***	-0.0254	(0.0028)	***	-0.0341	(0.0046)	***
*Lochs_views*Loch20GS*	0.0596	(0.0115)	***	0.0781	(0.0204)	***	-0.0303	(0.0035)	***	-0.0364	(0.0073)	***
*Rivers_views*River20GS*	-0.0207	(0.0063)	***	-0.0254	(0.0104)	**	-0.0001	(0.0031)		-0.0004	(0.0043)	
*Pollution control Attitudes*Price*	-0.0065	(0.0041)		-0.0073	(0.0054)		0.0139	(0.0026)	***	0.0192	(0.0036)	***
*Income_miss*Price*	-0.0068	(0.0040)	*	-0.0081	(0.0055)		-0.0030	(0.0020)		-0.0038	(0.0027)	
*Education_1*Price*	0.0095	(0.0033)	***	0.0117	(0.0041)	***	0.0048	(0.0018)	***	0.0061	(0.0024)	***
*River_use*ASC*	1.5484	(0.2245)	***	2.0257	(0.4449)	***	0.5148	(0.1951)	***	0.6232	(0.2566)	**
*Loch_use*ASC*	-0.7323	(0.2226)	***	-1.0059	(0.3663)	***	-0.4065	(0.1836)	**	-0.4571	(0.2161)	**
*Age<30*ASC*	3.1049	(0.2985)	***	3.9151	(0.6775)	***	0.2038	(0.2510)		0.0262	(0.3072)	
*Age_30-60*ASC*	2.1120	(0.2200)	***	2.7667	(0.5314)	***	-0.3623	(0.1980)	*	-0.4477	(0.2505)	*
*Male*ASC*	-0.2975	(0.2035)		-0.2176	(0.3029)		-0.1718	(0.1540)		-0.2273	(0.1877)	
*Surv_consideration*ASC*	-1.3560	(0.1920)	***	-1.6846	(0.3584)	***	-1.4321	(0.1227)	***	-1.6533	(0.2692)	***
Standard deviation of parameters												
sdRiver7GS				0.0272	(0.0154)	*				0.0453	(0.0149)	***
sdLoch7GS				0.0193	(0.0110)	*				0.0299	(0.0104)	***
sdRiver20GS				0.0286	(0.6061)					0.0180	(0.0204)	
sdLoch20GS				0.0316	(0.5363)					0.0181	(0.0170)	
Observations	1071			1071			2246			2246		
LL	-1125.57			-1176.61			-2165.93			-2467.48		
Adj Pseudo R2	0.30			0.34			0.16			0.27		

Statistical significance:"*" at 90% level, "**" at 95%level and "***" at 99% level

6.3.6.4. Implicit prices

As previously stated, the results of the *MNL* and *ML* models can be used to derive marginal measures of WTP or implicit prices; in this instance, for water quality improvements as reflected in the attributes of the models. Table 6.18 below shows estimates of implicit prices using the Delta method (Greene, 2003), along with standard errors and significance levels for the proposed attributes for the *MNL* and *ML* models obtained by applying equation 7 to the different samples. Furthermore, this table also shows 95% confidence intervals around the mean WTP[39], which were calculated using the Krinsky and Robb (1986) method using 1,000 random draws. Confidence intervals are of interest to policy for the development of water quality improvement scenarios.

Table 6.18 Willingness-to-pay (implicit prices) estimates for improvements in water quality (£/hh/year)

		MNL				ML			
Sample	**Attribute**	coeff.	(SE)	sig	(95% conf. int.)	coeff.	(SE)	sig	(95% conf. int.)
All	*River7GS*	2.04	(0.42)	***	(1.51 - 2.93)	2.18	(0.45)	***	(1.55 - 3.15)
	Loch7GS	1.60	(0.32)	***	(1.18 - 2.31)	1.60	(0.32)	***	(1.17 - 2.34)
	River20GS	0.69	(0.30)	**	(0.22 - 1.27)	0.57	(0.31)	*	(-0.14 - 0.93)
	Loch20GS	0.27	(0.23)		(-0.11 - 0.65)	0.24	(0.26)		(-0.11 - 0.82)
North	*River7GS*	3.32	(0.84)	***	(2.31 - 5.43)	3.61	(0.87)	***	(2.20 - 5.88)
	Loch7GS	1.79	(0.46)	***	(1.23 - 3.02)	1.59	(0.43)	***	(0.99 - 2.64)
	River20GS	-0.16	(0.45)		(-0.99 - 0.63)	-0.42	(0.53)		(-2.14 - 0.26)
	Loch20GS	-0.21	(0.37)		(-0.91 - 0.43)	-0.18	(0.44)		(-0.80 - 0.96)
Central	*River7GS*	0.62	(0.16)	***	(0.37 - 0.89)	0.75	(0.20)	***	(0.42 - 1.11)
	Loch7GS	0.56	(0.12)	***	(0.39 - 0.77)	0.60	(0.13)	***	(0.40 - 0.84)
	River20GS	0.45	(0.21)	**	(0.12 - 0.82)	0.37	(0.24)		(-0.24 - 0.58)
	Loch20GS	0.13	(0.17)		(-0.16 - 0.42)	0.08	(0.20)		(-0.17 - 0.53)
South	*River7GS*	0.45	(0.16)	***	(0.21 - 0.75)	0.46	(0.16)	***	(0.21 - 0.80)
	Loch7GS	0.67	(0.14)	***	(0.48 - 0.96)	0.70	(0.16)	***	(0.49 - 1.03)
	River20GS	0.54	(0.22)	**	(0.22 - 0.93)	0.47	(0.23)	**	(0.10 - 1.63)
	Loch20GS	0.29	(0.17)	*	(-0.01 - 0.59)	0.26	(0.19)		(-0.30 - 0.48)
SRBD	*River7GS*	1.53	(0.22)	***	(1.20 - 1.93)	1.63	(0.25)	***	(1.24 - 2.10)
	Loch7GS	0.99	(0.15)	***	(0.78 - 1.27)	0.95	(0.16)	***	(0.73 - 1.24)
	River20GS	0.26	(0.21)		(-0.10 - 0.61)	0.13	(0.23)		(-0.61 - 0.17)
	Loch20GS	0.00	(0.17)		(-0.27 - 0.30)	-0.02	(0.21)		(-0.22 - 0.49)

Statistical significance:"*" at 90% level, "**" at 95%level and "***" at 99% level

The values of *River7GS* and *Lochs7GS* in table 6.19 reflect the WTP or implicit prices for a 1% improvement in the total area of rivers and lochs classified as being in good status (or beyond) by 2015 per year per household (in perpetuity)[40]. *River 20GS* and *Loch20GS* WTP estimates are for the same 1% increase in river and loch quality respectively, but for improvements in 20 years time (by 2028) instead. Overall, WTP estimates for the *MNL* and *ML* are of the same magnitude for the different samples.

For the whole sample, implicit prices for the MNL and ML models show that respondents do indeed value water quality changes in rivers and lochs differently – albeit not statistically significant, with a 1% increase in the total area of rivers classified as being in Good Status by 2015 from current levels valued at around £2.00 per year per household, as compared with improvements in lochs, which are valued at around £1.60. Both estimates are statistically significant at the 99% level.

In comparison, model results indicate that respondents exhibit weaker preferences towards paying for water quality improvements in lochs and rivers in 20 years time. Implicit prices are much lower than for the 7 year attributes; with a 1 % improvement in rivers and lochs in 20 years valued at around £0.70 and £0.25 per household per year, respectively. Furthermore, *Loch 20GS* is not statistically significant in both the MNL and ML models. The main reasons for this should be further explored. An early assumption would be that this may well indicate that respondents do hold strong preferences towards lochs being improved faster.

In this respect, respondents in the north region (and to a lesser degree in the SRBD pooled model), state the strongest preferences towards faster improvements for both lochs and rivers in seven years time, as WTP values for both rivers and lochs in 20 years are not statistically significant even at the 90% margin. In contrast, respondents in the central and south regions show less pronounced differences in their hypothetical payments depending on the timing of the improvement.

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[39] We do not report here mean IP values estimated using the Krinsky and Robb method. These values nevertheless are of similar magnitude to the ones reported in table 6.20

[40] Note that, due to the experimental design employed, a 1% improvement in rivers and lochs in 7 years time classified as being in good status is confounded with a 0.27% reduction in the total area of rivers with many problems and a 0.63% reduction in the total area of lochs with many problems, respectively.

Unfortunately, it has been impossible at this stage to separate from the survey the influence of different types of uses of water (passive, option value…). Further analysis/research will attempt to cover this issue.

6.3.6.5. Testing for differences in marginal WTP between locations

A comparison across implicit prices by location is undertaken using the convolution approach proposed by Poe et al., (1994). This test was conducted to assess whether there are any significant differences between the marginal WTP derived from the MNL model by location. The hypothesis is that WTP estimates differ for attribute *i*. The following null is being tested for each attribute (13):

$$H_0: \; WTP_{LOC1} - WTP_{LOC2} = 0 \qquad 13)$$

Using a Krinsky and Robb (1986) bootstrapping procedure, a large number (e.g. 1000) of WTP estimates for the attributes are drawn from parameter estimates and the corresponding variance-covariance matrix for both regional sub-samples. WTP estimates are derived by drawing from population means and calculated using equation (7).

The procedure results in two vectors $v_i[WTP_{LOC1}]$ and $v_i[WTP_{LOC2}]$ for sub-samples LOC1 and LOC2. The difference vector between each single element of $v_i[WTP_{LOC1}]$ and each single element of $v_i[WTP_{LOC2}]$ is calculated for each attribute.[41] The one-sided significance level of difference can be derived by assessing the value of the cumulative distribution of the difference vector at zero.

The results are shown in Table 6.19. Values represent the probability of accepting the null hypothesis that the differences between Implicit Prices are equal to zero. These results are of interest to policy in case of transferring value estimates across regions. Comparisons are only useful if both implicit prices are statistically significant in the first place (results in table 6.18).

Those comparisons of implicit prices that are found to differ are highlighted in bold in table 6.19. For those cases with insignificant attribute parameters in at least one of the models, the Poe et al. test is not meaningful. If the attribute parameter is significantly different from zero in one region but not the other, we conclude that they are different, although the difference

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[41] The decreasing vector has the larger average value.

of the two (bootstrapped) WTP vectors is not necessarily significantly different. Overall, there is similarity of implicit prices between the central and south area for all the attributes. The north region shows more differences with the other regions; except for Loch20GS, which is not statistically significant in models for both regions, all other implicit prices are statistically different.

Table 6.19 Poe et al. test results for comparison between implicit prices by location

	CENTRAL	SOUTH
NORTH		
River7GS	**0.00000 ***	**0.00000 ***
Loch7GS	**0.00002 ***	**0.00142 ***
River20GS	**n.s. in North**	**n.s. in North**
Loch20GS	n.s in both	**n.s. in North**
SOUTH		
River7GS	0.23540	
Loch7GS	0.26596	
River20GS	0.37386	
Loch20GS	**n.s. in Central**	
SRBD		
River7GS		**0.00000 ***
Loch7GS		**0.06651 ***
River20GS		**n.s. in SRBD**
Loch20GS		**n.s. in SRBD**

(n.s. not statistically significant): * significant at the 90% level

Table 6.19 also illustrates comparisons of implicit prices between the Scottish River Basin (pooled north and central samples) and the south sample. As *River20GS* and *Loch20GS,* are not statistically different from zero for SRBD but are significant for the South models, all attributes differ.

This raises issues about the way water management units have been divided in Scotland. As model results indicate, respondents in the central region have more in common with respect to their preferences for water quality improvements with respondents in the south region, which are under the Solway-Tweed River Basin, than with their river basin counterparts of the north.

6.3.7. Policy analysis - Aggregation of WFD benefits

Table 6.20 offers a summary of the implicit prices for the whole sample from the ML model for rivers and lochs water quality in seven years time (up to 2015) which are applied for the aggregation of WFD benefits in Scotland. Units of WTP values/implicit prices are of pound sterling per household per year for a 1% improvement in the total area (hectares of catchment area for rivers, and hectares of surface water area for lochs) of rivers and lochs classified as being in good status (or beyond) by 2015 per year per household (in perpetuity). The following analysis and aggregation of WFD benefits only extends up to the year 2015. Our findings regarding preferences for 20 year improvements need to be given further thought, for example with respect to the treatment of the insignificant estimates for lochs.

Table 6.20 ML model WTP values for a one percent improvement in the total area of rivers and lochs classified in Good Status by 2015

	Implicit Prices mean	95% confidence intervals	
		Lower	**Upper**
		£/HH/year	
rivers7GS	2.18	1.55	3.15
Lochs7GS	1.60	1.17	2.34

The following assumptions are necessary for the development of policy scenarios and aggregation of benefits: i) constant annual linear improvements towards the objectives (up to the year 2015); ii) the survey sample is representative of the whole of Scotland (an analysis of the representativeness of the sample is offered in annex XIII). WTP estimates were adjusted to the whole of the Scottish population using data from the household projection numbers for Scotland 2006-based (GROS, 2007). This data has been used to adjust household number figures in relation with expected forecasts of household occupancy; and, iii) a 3.5% discount rate in line with UK Treasury guidance has been applied.

Only people resident in Scotland are considered. These results do not account for any possible value that visitors to the country may attribute to the overall quality of water resources. We have not separated values by types of uses of water, although, all types are covered in our survey and included in our estimates. Furthermore, estimates do not account for any possible market based benefits that Good Status may deliver in Scotland.

Policy scenarios were derived from the hypothetical water quality improvement scenarios used in the design of the environmental attributes for the choice experiment. A further policy scenario was developed and called "*SEPA scenario*". Information for this scenario comes from two recent consultation documents released by SEPA. These reports outline the main issues that the agency is expecting to address in order to deliver environmental improvements in Scotland by 2015 under compliance with the WFD. These documents also outline SEPA's own objectives as to the number of water bodies that the agency do not expect will achieve Good Status by 2015 in Scotland (SEPA 2007a,b). These objectives are presented in a suitable form for aggregation from our model based on current national conditions for rivers and lochs. Scenarios are shown in table 6.21.

Table 6.21 Levels of water quality improvements (policy scenarios) for rivers and lochs in 7 years time (2015)

Policy Scenarios	% area of RIVERS in Good Status or beyond	% area of LOCHS in Good Status or beyond
Baseline	34%	25%
Scenario 1	83%	81%
Scenario 2	65%	57%
Scenario 3	48%	38%
Scenario 4	39%	29%
SEPA Scenario	69%	67%

Table 6.22 illustrates the range of non-market benefits (in terms of their net present value) that could be accrued in Scotland from the achievement of the different scenarios by 2015. The achievement of SEPA's objectives for the improvement of river and loch water quality scenarios can deliver overall benefits beyond the billion pound barrier (for rivers and lochs combined). 95% confidence intervals are expressed in brackets.

Table 6.22 Aggregate ML estimates of WFD benefits by Policy Scenario (2015)

	Policy Scenarios	% improvement from current conditions (High status)	PV* (£/HH) to 2015		PV* (£ Million) to 2015	
Rivers	Level 1	49.0%	**373.1**	(265.3 - 538.9)	**854.8**	(607.9 - 1234.8)
	Level 2	31.0%	**236.0**	(167.8 - 340.9)	**540.8**	(384.6 - 781.2)
	Level 3	14.0%	**106.1**	(75.4 - 153.2)	**243.0**	(172.8 - 351.0)
	Level 4	5.0%	**37.9**	(26.9 - 54.7)	**86.8**	(61.7 - 125.4)
	SEPA scenarios	35.0%	**265.1**	(188.5 - 383.0)	**607.5**	(432.0 - 877.6)
Lochs	Level 1	56.0%	**313.8**	(228.3 - 457.8)	**719.1**	(523.1 - 1049.0)
	Level 2	32.0%	**179.3**	(130.4 - 261.6)	**410.9**	(298.9 - 599.4)
	Level 3	13.0%	**72.5**	(52.7 - 105.7)	**166.1**	(120.8 - 242.3)
	Level 4	4.0%	**22.3**	(16.2 - 32.5)	**51.1**	(37.2 - 74.6)
	SEPA scenarios	42.0%	**234.2**	(170.3 - 341.6)	**536.6**	(390.3 - 782.8)

PV: Present Value. 3.5% Discount Rate

Results in table 6.22 assume that each percentage improvement has the same WTP, a linear utility function has been imposed. However, this is unlikely true in reality if we consider diminishing marginal utility. As the overall quantity of surface waters regarded as being of good quality increases, we would expect individuals' WTP for improvements to decrease. This issue can be investigated by entering the quadratic expressions of rivers and lochs improvements in seven years time into our models. Accordingly, we can observe the levels of percentage improvements at which utility will begin to drop. In addition, further analysis is required for extending the aggregation of welfare changes to the year 2028 and for exploring the limitations and issues associated with the aggregation procedure. In addition, the use of standard discount rates to reflect time preferences needs to be further investigated. Arguably, as it seems from the WTP results outlined in table 6.18, respondents are applying their own discount rates. These issues have not been covered in this chapter but will be investigated in the near future.

6.3.7.1. Other policy relevant units of WTP

Benefits estimates presented in table 6.22 (above) are of use for policy in order to provide estimates of the range of possible total non-market benefits expected to arise from proposed investments in water quality or projected Programme of Measures to achieve Good Status (i.e. specially relevant for the Regulatory Impact Assessment of river management plans in Scotland). Nevertheless, other meaningful units of WTP estimates are needed in order to inform decisions on exemptions in Scotland.

Different aggregated units of benefits (i.e. WTP per hectare or per water body) can also be derived from the implicit prices estimated by the ML model shown in table 6.20. By finding out the average catchment area for rivers and surface water area for lochs that represents a 1% improvement in quality for the whole of Scotland, it is possible to derive aggregated WTP estimates per unit of area for rivers and lochs. In addition, this procedure can also be applied to derive WTP estimates for an averaged sized Scottish loch or river. These results are presented in table 6.23. Estimates are per year and for the whole of the Scottish population.

It should be noted that these expressions do not reflect that respondents were actually using information related to area or size of waterbodies when making their choices. In this respect, the figures shown in table 6.23 should be regarded as artificial ex post transformations of preference estimates rather than directly elicited preferences. Additionally, we are not considering issues of scope. In theory for any given good, WTP would increase with the scope of the change being considered (Bateman et al., 2005). In the case of the current study, the expectation would be that WTP to enhance water quality of rivers and lochs at national level would be higher than that expected at particular rivers and lochs water bodies. Furthermore, it is assumed that all stretches of lochs or rivers are equally valued. Thus, in reality we would expect some water bodies to be more valuable in relative terms than others.

Table 6.23 Annual WTP estimates and 95% confidence intervals per hectare and per average water body for rivers and lochs

	WTP £ per year	Confidence intervals (95%) - £ per year
wtp/ha of river catchment area[42]	25.43	(18.09 - 36.74)
wtp/ha of loch water surface area[43]	3,706.73	(2,696.31 - 5,407.52)
wtp/average river water body[44]	2,095.52	(1,490.14 - 3,026.99)
wtp/average loch water body[45]	11,009.22	(8,008.19 - 16,060.67)

Annual WTP per hectare estimates for rivers and lochs and net present values up to 2015, are of interest for direct comparisons with the cost curves of P mitigation strategies (BMPs) at farm level shown in chapter 4. These comparisons are illustrated in the conclusions chapter.

Furthermore, at this stage it is also possible to establish a formal comparison between original CE WTP estimates with the benefits transfer results for Scotland introduced earlier in this chapter. In Part 1 of this chapter, the BFT exercise from the Baker et al., (2007) study revealed mean WTP values for the 95% overall water quality improvement scenario in Scotland to be of the range between £30 and £149 per year per household depending on the original elicitation method (PCCV and DCCV respectively).

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[42] A 1% overall increase in the number of rivers in good condition represents an improvement in 196,431.2 hectares of river catchment area in Scotland (Anthony et al., 2005)

[43] A 1% overall increase in the number of lochs in good condition represents an improvement in 992 hectares of surface water area (SEPA, 2005a,b)

[44] The average river water body in Scotland has 82.39 ha of river catchment area (Anthony et al., 2005l).

[45] The average loch water body in Scotland has 2.97 ha of surface area (SEPA, 2005a,b).

An aggregation of CE implicit prices (table 6.20) to a 95% "Good Status" improvement scenario by 2015 for rivers and lochs, offers combined benefits of £245 per year per household (with 95% confidence intervals of £177-356). As expected, these estimates are larger because of implicit differences between the original elicitation methods employed.

Nevertheless, these aggregated CE results for the 95% overall improvement scenario are in line with the estimates reported for the CE results in the CRP study for England and Wales. Their results, which are presented in table 6.3, illustrate mean values of £300 per year per household. Again, these differences might be largely due to differences in baseline water quality levels. Furthermore, any meaningful comparisons between both CE studies should be made in terms of the implicit prices found. The CRP study results illustrate that annual WTP estimates per household for a 1% improvement in the total area of water bodies in Good Status are of the region of £0.86 for England and Wales combined (separately, £0.84 for England and £1.63 for Wales respectively). These figures are slightly lower than those found in our study (table 6.20), though our Scottish estimates bear a close resemblance to the Welsh estimates. Nonetheless, modifications introduced to the CRP CE study design disallow any direct comparisons. Further empirical research is possible to assess if these similarities hold any statistical relationship (e.g. application of Bayesian statistics[46]).

6.3.7.2. Development of simple benefits curves for rivers and lochs in Scotland

The development of benefits curves for WFD water quality improvements is fundamental for the estimation of damage cost curves associated with differing levels of water pollution. Mirroring benefits curves with the costs of abatement at a water body scale or per unit of area (ha), will be useful for the design of efficient regulation and to inform decisions about standard-setting derogations.

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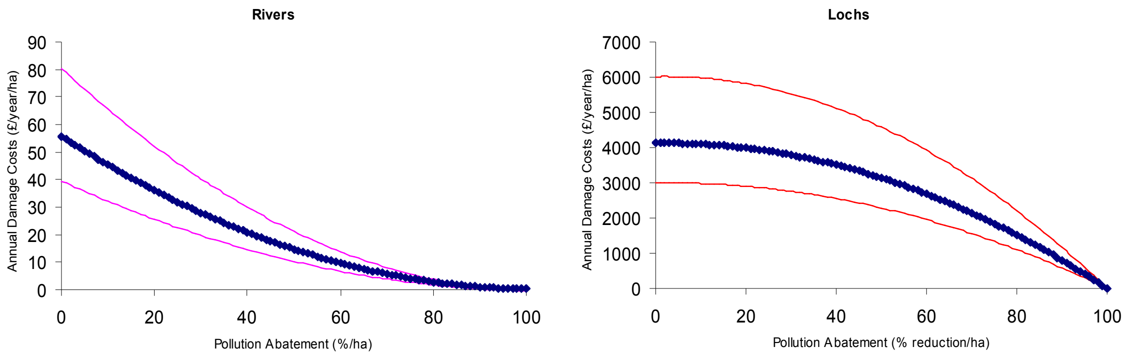
[46] The similar design of the CRP study creates opportunities to test for the performance of benefit transfer exercises using different methods and underlying assumptions. With respect to the regional differences found in Scotland, it is worth considering whether using Bayesian benefits transfer is a useful option for similar endeavours in the future. Bayesian methods allow for low sample sizes and may therefore offer a way to capture regional differences that would otherwise go unnoticed at relatively low cost. Given that model estimates are available for England and Wales, it may also be promising to test whether this information would have a significant influence on model results by comparing whether the use of preference information from England and Wales for the definition of the prior distribution has an influence on results or not (i.e. a comparison between an informative and a non-informative prior). This is a topic for further research.

The results outlined in table 6.23 (above) could be used to develop simple benefits curves for average lochs and rivers per hectare. Two main challenges: i) the shape of the benefits curve is unknown. Benefits estimates often only provide one point on the curve which forces the analyst to estimate its shape with a high margin of error. And ii) there are no definitions of status quality levels. At this stage there is no translation between definitions of Good Status (environmental standards at water body scale) and required nutrient reductions at farm scale. These therefore need to be assumed. Furthermore, for the development of damage cost functions it is necessary to assume a direct relationship between reductions in farm nutrient loads and water quality improvements. In addition, the damage cost function should account for all the social costs associated with environmental pollution. In practical terms, these include an aggregation of market and non-market benefits of all relevant types of uses of water for that specific water body (including passive or non use type of values).

The choice experiment exercise described in this chapter, illustrates WTP estimates for a one percent improvement in the total area of rivers and lochs classified at Good Status in Scotland. These estimates for a one percent improvement are confounded with a 0.27% reduction in the total area of rivers with many problems, and a 0.63% reduction of the total area of lochs with many problems. In our study the many problems category is related with the bad and poor status categories under the WFD. Relating these figures to the estimates obtained in table 6.23 for rivers and lochs uncovers WTP estimates for the few problems categories of £6.87 (£4.88-£9.92) and £2,335.24 (£1,698.68-£3,406.74) per hectare for rivers and lochs respectively.

These points can be used to explore the shape of damage costs curves for both river and loch quality improvements, which are the inverse of the benefits function (figure 6.3). No environmental damage has been assumed to deliver no welfare losses. In this respect, this assumption facilitates the development of damage costs functions as it provides a third point for estimation. For representation purposes, as there are not measurable units of GS, we have set thresholds for the different water quality status categories: the threshold between fair and GS at 66.6% and between poor and fair at 33.3% water quality levels respectively. Unfortunately, the estimated WTP values in the valuation exercise do not capture values beyond GS, which in these curves have been simulated from the identified points.

Figure 6.3 Damage costs curves for rivers and lochs (including 95% confidence intervals)



6.3.8. Conclusions

This study provides an insight into how people can value water quality improvements based on an application of national characterisation data on the state of the water environment in Scotland. The CE survey is focused on the estimation of the total economic value (i.e. use and non-use values) associated with the achievement of Good Status under compliance with the EC WFD. Furthermore, other aspects analysed, which are relevant for water policy, relate to public perceptions on restoring river and loch water quality separately, whether respondents held any preferences about the timing of the improvements (2015 versus 2028) and whether there exists regional differences in preferences towards national water quality changes.

Survey results suggest that overall, respondents prefer improvements to water quality as compared with being left at current national water quality status levels but nevertheless, they are sensitive towards large increases in their water bills/household expenses, suggesting that there is a limit to the amount of money they are willing to pay for water quality changes. Improvements in the short run, achieved in the first river basin management plan by 2015, are preferred over longer term improvements. WTP estimates show that respondents do indeed value water quality changes in rivers and lochs differently, with a 1% increase in the total area of rivers classified as being of Good Status by 2015 valued at around £2.00 per year per household, as compared with improvements in lochs, which are valued at around £1.60 per percentage improvement.

This study also provides an insight into the transferability of benefit values across different locations of the country. In this context, our survey results introduce arguments for discussion about the way river basin districts have been designed in Scotland. Respondents in the central region of the country show more closely related tendencies on their preferences towards water protection with respondents in the South than with their river basin counterparts in the North.

The survey, the methodology employed and its results, add to the growing debate on the suitability of valuation methods to estimate the non-market benefits of water quality improvements in order to inform policy analyses under the WFD (i.e. assessment of exemptions). The application of choice experiments proves very informative in this respect. Further to arguably one of the main limitations of the method, our study is proof that it is possible to borrow survey materials designed and tested elsewhere and successfully apply them to a different context with minor modifications. This decreases considerably the timing needed for carrying out the study and expenses.

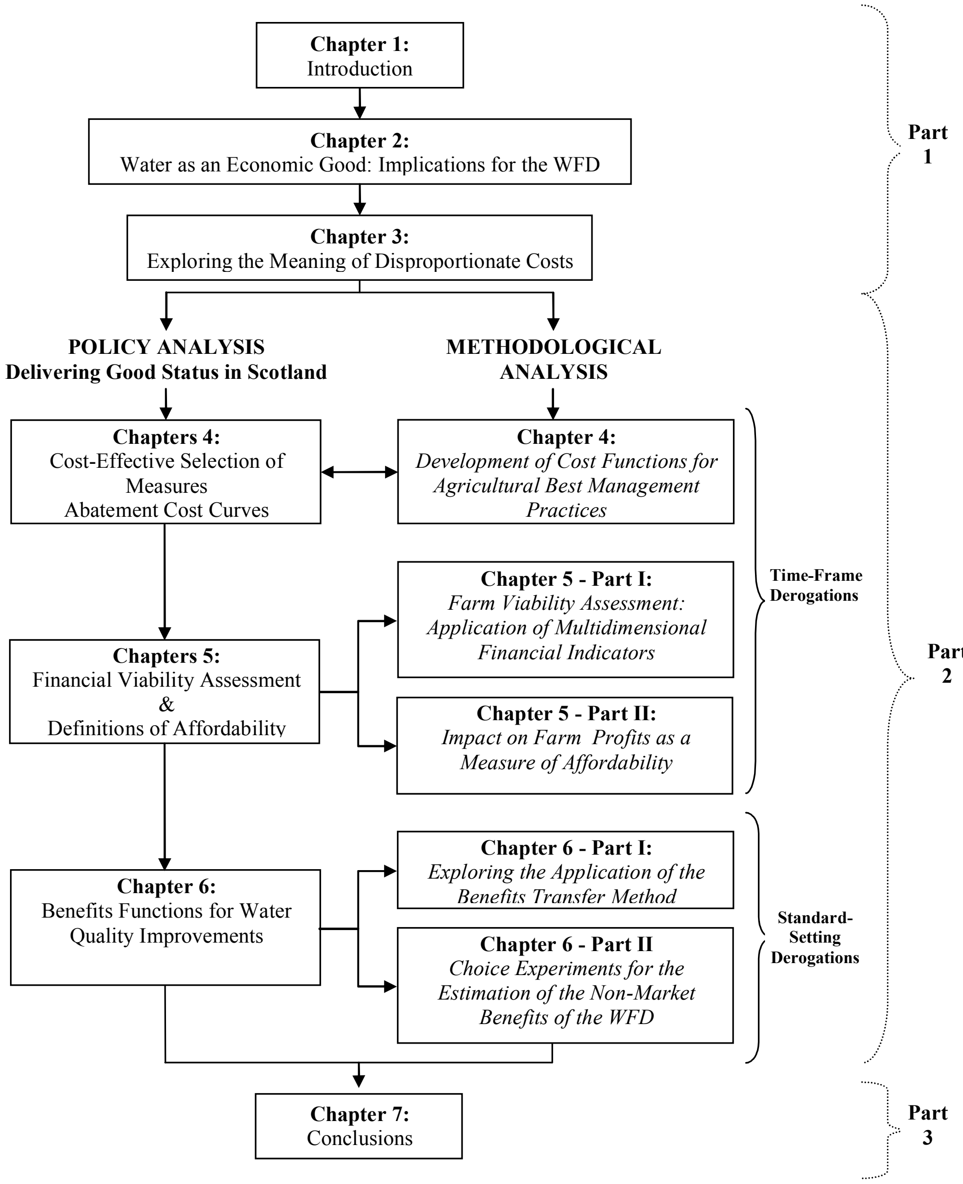
We conclude chapter 6 by reporting aggregate values for different WFD benefits policy scenarios by 2015. These estimates would also be relevant in order to assess the extent of the overall economic impacts of implementing the Directive in the country. Based on compliance targets set by the environmental regulator, the WFD can deliver non-market benefits in Scotland of over £1.1 billion in the next seven years. Finally, we illustrate an exploration into the shape of the social damage cost functions for rivers and lochs in Scotland, which are a data requisite of the proposed methodology for the assessment of disproportionate costs.

CHAPTER 7 CONCLUSIONS

7.1. Summary

This chapter presents the main conclusions of the thesis and reviews the implications both for policy and methodology. The chapter concludes with a discussion of the main strengths and weaknesses of the thesis and recommendations for future research.

Figure 7.1 Thesis Structure - Roadmap to thesis conclusions



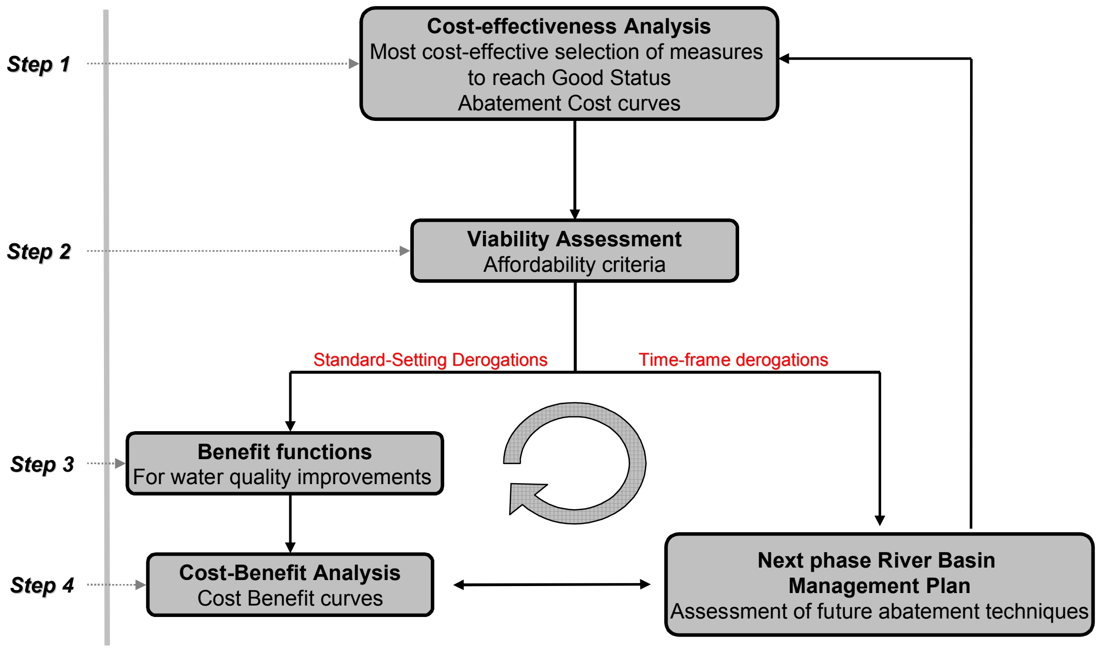
7.2. General Approach of the study

Delivering good status (GS) is cost dependent, and in some water bodies pollution abatement costs may be high or judged as disproportionate. The main aim of this research was to clarify the definition of disproportionality under the WFD and to convey a consistent method for its analysis which is mindful of several requisites. Firstly, to be fully compliant with the economic requirements of the Directive (e.g. built from the CEA for the selection of measures). Secondly, to be based on the principles of pollution control and welfare economics theory, as derogations should aim to reach socially optimal decisions. Thirdly, to take into account implicit differences between the types of derogations being sought (e.g. time-frame versus standard-setting derogations). Fourthly, coherent with current guidelines for the economic appraisal of public policy and regulations impact assessments in the UK (e.g.UK Treasury Green Book).

This thesis contends that a rational model to inform decisions on derogations under the WFD is needed, and argues that economic theory already provides a definition of disproportionate costs and the methodological tools that can inform its assessment. Building from economic theory, chapter 3 shows that, ideally, standard-setting derogations should be judged with reference to cost and benefit curves – an application of the CBA method - combined with a financial viability assessment of the firm. For the justification of time-frame derogations the assessment of benefits is not needed.

While instructive, the application of theoretical principles to water resource management is often constrained by the realities of data and administrative capacity. A major stumbling block in the theoretical approach is whether sufficient reliable costs and benefits assessment data were available. In this respect, the practical applications of the basic principles outlined in chapter 3 have presented a challenge. Figure 7.2 illustrates once again the main methodological steps followed in this thesis for the assessment of exemptions under the WFD. Throughout this thesis, we have introduced and explored the implications of using such a model for different farm systems. Overall, this research illustrates an in-depth study of the practical tests for the assessment of disproportionate costs, and seeks to answer a fundamental question for water policy/regulation: what information would be needed to judge if a hypothetical farm should be granted exemptions? In the next section we now take stock of the information gathered in the empirical chapters of this thesis and explore the advice that could be given to policy in respect to disproportionate costs and the economic tools employed.

Figure 7.2 Methodological steps for the assessment of disproportionate costs



7.3. Summary of findings - Methodological and policy implications

The WFD is expected to have extensive effects on Scottish agriculture, which is faced with the challenge of maintaining its competitiveness, while protecting water resources. Focusing the analysis on the socio-economic impacts of achieving water diffuse pollution targets for the sector, a series of independent tests for the assessment of disproportionate costs have been proposed and evaluated in this thesis. These are: i) development of abatement cost curves for agricultural Phosphorus (P) mitigation options for different farm systems (chapter 4); ii) a financial characterisation of farming in Scotland and impact on profits of achieving different P loads reductions at farm level were investigated in order to explore issues of "affordability" and "ability to pay" by the sector (chapter 5); and iii) an investigation of benefits assessment for policy analysis applying BT and discrete choice modelling to explore public preferences for pollution control and measure non-market benefits of WFD water quality improvements in Scotland (chapter 6).

In relation with the overall aim of this research, the following two sections discuss and evaluate further considerations on the assessment of disproportionate costs for the different types of derogations allowed in the Directive. Alternatively, the main conclusions of the study may be summarised in terms of the key policy and methodological contributions of each independent test, which have been highlighted in turn in each empirical chapter. Below, we briefly illustrate the links between the outcomes of each test with the original stated objectives as laid out in the introduction of this thesis.

Development of Cost Functions for Agricultural Best Management Practices

Policy Outcomes

i) The economic implications of adopting different mitigation strategies at farm level (Best Management Practices - BMPs) in order to reduce farm diffuse pollution to water have been explored.

ii) Cost-effectiveness ratios were developed as a criteria for the selection of BMPs at farm level, which is relevant in order to provide information on the most cost-effective selection of abatement techniques for PoMs to achieve GS.

iii) An assessment of the financial costs of reducing farm diffuse pollutants in order to achieve different target levels has been outlined using an application of the abatement cost curve method. This exercise would be relevant for the assessment of disproportionate costs and for the analysis of the costs and benefits of farm level mitigation option strategies for Regulatory Impact Assessment (RIA) of River Basin Management Plans (RBMPs).

Methodological Outcomes

iv) The strengths and limitations of the application of cost-effective ratios for the selection of BMPs at farm level are illustrated in chapter 4.

v) In addition, the suitability and limitations of the abatement cost curve method to estimate the extent of the financial costs associated with achieving different levels of diffuse pollutants loads reductions at farm level were explored. Chapter 4 illustrates a justification for the selection of this method over the use of dynamic optimisation to solve the cost minimisation problem for the selection of BMPs at farm level.

Financial Viability Assessment and Definitions of Affordability

Policy Outcomes

i) In chapter 5, an investigation into different definitions and measures of affordability suitable for the economic analysis of water use is offered.

ii) We have undertaken a financial characterisation of farming in Scotland, which is also relevant to understand the possible impact of other (non-water) agricultural regulations.

iii) In the second part of chapter 5, we have assessed and quantified the likely financial impacts for a typical Scottish farm of adopting different diffuse pollution mitigation strategies in order to achieve water quality improvements. This exercise would be relevant for the assessment of disproportionate costs and for the small business impact analysis of the costs and benefits of farm level mitigation option strategies for the RIA of RBMPs (Cabinet Office, 2003).

Methodological Outcomes

iv) Melichar's (1985) multidimensional financial characterisation criteria was applied to identify farms in poor financial conditions. Other issues to consider in this type of analysis, which are not covered by traditional farm financial indicators, such as technology adoption and role of off-farm income were also explored

v) An optimisation model to assess the likely changes to farm profits as a result of achieving different P loads reductions at farm level under two different scenarios: with and without government intervention (i.e. impact of regulations) was developed for the Scottish dairy and arable sectors.

Benefits Functions for Water Quality Improvements

Policy Outcomes

i) Chapter 6, and more specifically the results of the CE study, offer robust estimates of the overall benefits to society derived from the achievement of GS in Scotland, which would be relevant for the design of RBMPs.

ii) This study also explored interesting issues relevant for water policy in Scotland regarding public preferences and perceptions about restoring river and loch quality to and beyond good status for the whole of Scotland, time preference for the improvements (2015 versus 2028) and whether regional differences exist within Scotland in preferences towards changes.

iii) An evaluation into the use for water policy analysis of the benefits transfer method was also undertaken in chapter 6. The application of BT is constrained by the impossibility to validate the accuracy of the transferred values.

Methodological Outcomes

iv) Chapter 6 illustrates a practical application and evaluation of the Benefits transfer method, including a validity assessment of the transferred values.

v) The practical application and evaluation of the choice experiments method for the valuation of water quality improvements was explored. Overall, the method is highly suitable for the development of benefit functions for the WFD and for the assessment of the transferability of the estimates across different locations.

7.3.1. Time-Frame derogations

Independently of the types of derogations being sought, we have seen that a preliminary CEA of all measures available to the farmer to reduce water pollution needs to be undertaken as a requirement for the selection of PoMs to reach GS by 2015. In relation to the agricultural sector, the adoption of BMPs is especially relevant for the control of agricultural diffuse sources of water pollution to rivers. The use of cost-effectiveness indicators offers a suitable criteria for the selection of BMPs at farm level. Additionally, as a preliminary assessment of disproportionality, we investigated the economic implications of adopting different mitigation strategies in chapter 4. This chapter also presents an application of the abatement cost method to the economic analysis of mitigation options to reduce farm losses of main diffuse pollutants.

Once information about costs and effectiveness of measures has been collated, the proposed methodology (see figure 7.2) follows by suggesting that for the assessment of time-frame derogations under the WFD, information on the financial costs of measures needs to be compared with an assessment of the financial viability of the farm and the ability by the farmer to absorb the additional costs of protecting the water environment. Ultimately, this will determine farmers' efforts to achieve GS at particular water bodies (Lago et al., 2006).

In chapter 5, the question of affordability - or the evaluation of the financial impacts to individual water users - and to the degree to which costs to reach Good Status may damage their financial viability or sustainability was explored. An examination of two different practical definitions of affordability at farm level which are relevant to European water policy was undertaken: i) the use of farm financial indicators to assist in the decision-making process about derogations; and, ii) an assessment of impact on profits as a measurement unit of changes in farmers' welfare.

Table 7.1 illustrates a summary of the main results of these tests for arable and dairy farms. Results are also offered by farm size under two types of soils (sandy and clay loam soils). This table illustrates the percentage of farms, which according to Melichar's criteria (chapter 5), are classified as being in poor financial condition (vulnerable plus stressed categories). In addition, this table also summarises the results of the breakeven analysis when in our profit optimisation simulation, fixed costs are set equal to zero and profits fall to zero as a result of investing in the most cost-effective mitigation techniques (e.g. BMPs) to reduce P loads at farm level. We illustrate abatement levels and additional costs per hectare that according to the simulation would lead a typical farm to reach their break-even point.

Table 7.1 Tests results for time-frame derogations

Types of farms	Type of soil	Size	Abatement level - breakeven point (% P Reduction)	Additional costs - breakeven point (£/ha/year)	% of farms in poor financial condition
Arable	Sandy Loam Soils	All	78.7	43.35	24.59
		Small	83.4	82.38	38.07
		Medium	82.0	68.98	19.61
		Large	74.3	20.69	17.92
	Clay Loam Soils	All	79.4	106.41	24.59
		Small	84.0	171.56	38.07
		Medium	82.6	150.00	19.61
		Large	75.3	61.35	17.92
Dairy	Sandy Loam Soils	All	79.7	40.12	24.15
		Small	76.8	3.90	56.36
		Medium	79.2	33.24	20.32
		Large	80.2	47.28	20.95
	Clay Loam Soils	All	87.8	330.72	24.15
		Small	83.7	162.41	56.36
		Medium	87.2	303.41	20.32
		Large	88.5	363.80	20.95

Breakeven points on average for Scottish arable and dairy farms were found to be around 80% of P mitigation levels. Values vary (up to 10% differences) between types of farms, sizes and different soils, which is relevant if a fixed definition of disproportionate costs is applied across the sector, as we would expect different impacts for different conditions. Based on the proportion of farms classified as being in stressed and vulnerable financial health by size and type of farm, our results suggest that there is a relationship between the size of the farm and overall financial health; with a larger number of smaller farms being in an unhealthy financial condition. As an example, more than 50% of small dairy farms are classified in this category.

Overall, results presented in table 1 need to be interpreted cautiously. We would not recommend granting time-frame derogations based exclusively on these findings or the application of the proposed tests in isolation. Nevertheless, the suggested approach can be used to benchmark affordability constraints and can be applied by the regulator to identify individual cases which would need further consideration. Specifically, further considerations should include an assessment of the technical efficiency of the farm and the impact on profitability of other sources of income (i.e. off-farm income and subsidies). The exploration of these issues is especially relevant in this situation because evidence suggests that there is considerable scope for technical (cost-efficiency) improvement across the industry, and the sequential uptake of BMPs is likely to increase production efficiency whilst lowering production costs. This is a fundamental conclusion for the assessment of time-frame derogations under the WFD. Therefore, whether a farm is in poor financial condition or not, regulators also need to assess if there is scope to increase its technical efficiency before granting exceptions to achieve water quality targets. Furthermore, factors that affect farm efficiency need to be evaluated and regulations tailored accordingly. Additionally, there is potential scope to use different policy instruments (varying levels of farm support or encouragement of off-farm activities), in order to change attitudes towards the adoption of new management practices. The incorporation of these issues into a methodology to assess derogations needs to be further investigated.

7.3.2. Standard-Setting derogations

The exploration of valuation methods to estimate non-market monetary values of water quality improvements in Scotland was covered in chapter 6. The application and evaluation of BT and CE in order to derive policy relevant estimates of the overall benefits of the WFD in Scotland was undertaken.

As introduced in chapter 3, the change of environmental objectives (from GS to GP) needs to be socially justified under the WFD. We have demonstrated in chapter 3 that ideally disproportionate costs should be judged with reference to cost and benefits curves, which basically involves estimating damage functions or curves and then interpolating a demand curve from such estimates. In chapters 4 and 6, the methodological issues and data needs surrounding the development of abatement cost curves for P loads mitigation options at farm level, and the associated social damage cost curves from water pollution, were explored in turn.

In theory, efficient regulation would be achieved when environmental standards are set at the point where costs of abatement equal the associated social and environmental damage of pollution. Where costs outweigh benefits, this point offers a justification for standard-setting derogations and exemplifies inefficient regulation. Alternatively, if pollution results in overall welfare losses to society (e.g. damage larger than costs) there would be a justification for the introduction of tighter controls or other instruments to control pollution.

At this point, we undertake a formal comparison between cost and benefit curves for Scottish rivers to demonstrate a graphical interpretation of our results. These are illustrated in figure 7.3. The abatement cost curves presented in figure 7.3 were obtained in chapter 4 (figure 4.3) by plotting the respective cumulative annual total abatement costs (£/ha/year) with the associated % reduction in P loss at farm level for each BMP for four different types of Scottish farms and two different types of soils (sandy and clay loam soil). The additional % reductions in N and FIO that would arise from targeting farm P loads are also illustrated. Total abatement costs (TAC), which are the total costs of abating nutrient loads by a certain amount per year, reflect the annual equivalent costs for each measure (average Present Values - PV - to 2015 for each measure per year). In figure 7.3, cost curves are compared with the annual damage costs (£/ha) associated with different levels of pollution reduction per hectare. Damage cost curves were derived by finding the inverse of the benefits function for water quality improvements in rivers (chapter 6 – figure 6.4).

In this example, we illustrate a very simple application with only one polluter, which is represented by different types of average farm systems impacting a surface hectare of a river. This theoretical representation is used to exemplify a definition of Good Status, where costs of abatement are a function of measures to reduce farm P loads per hectare and benefits are a function of society's preferences for water quality improvements. We are not considering other competing uses of the resource and therefore, there is no need to include other types of values in our analysis.

Figure 7.3 Total abatement cost and damage cost curves for P mitigation options at farm level per hectare per year for different types of farm systems under sandy and clay loam soils.

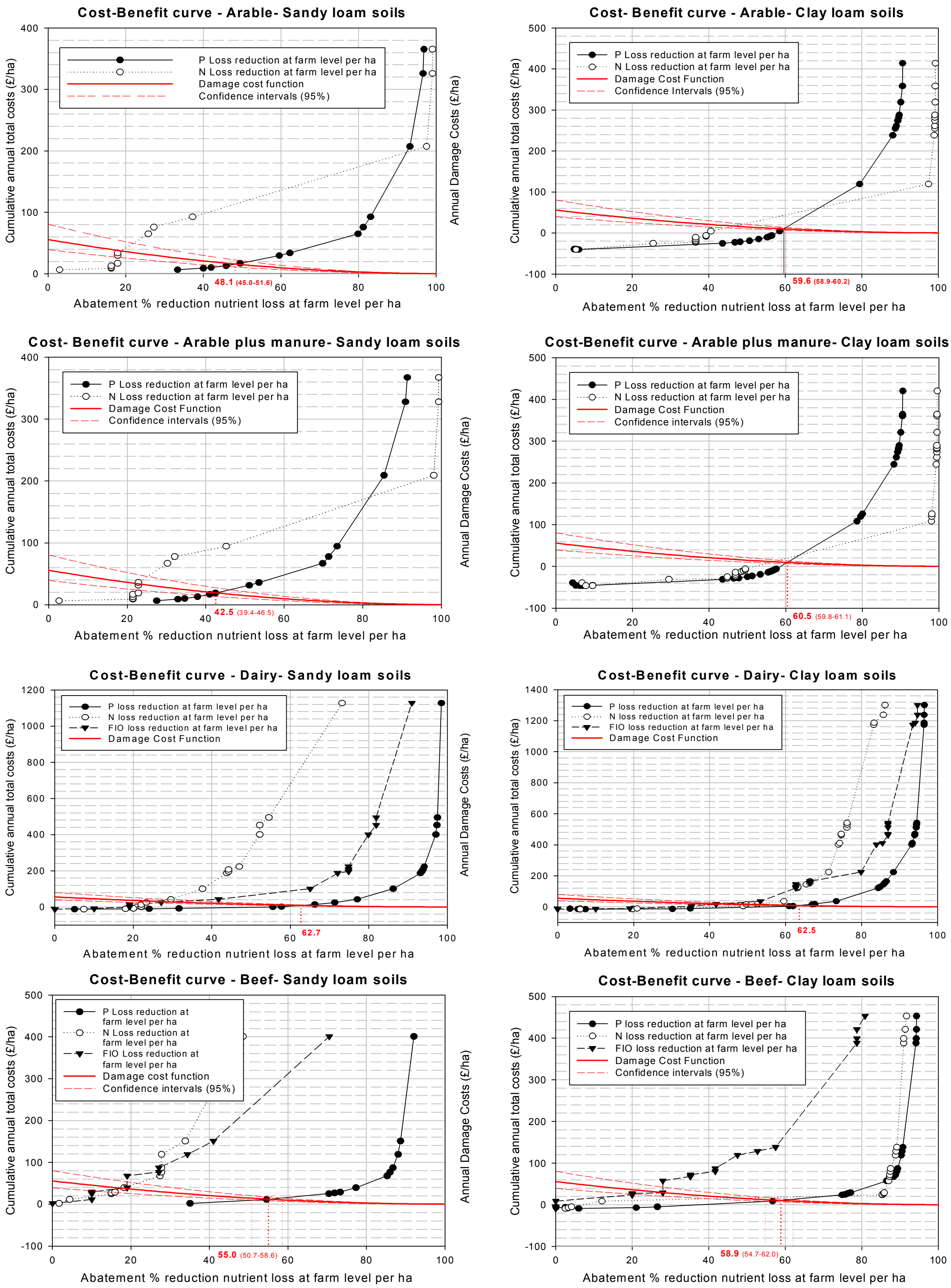


Table 7.2 below outlines the results of a graphical analysis of the curves presented in Figure 7.3 using Sigmaplot. The main benefit of this type of analysis is that there is no need to impose functional form restrictions to the abatement cost information gathered for chapter 4. Table 7.2 illustrates the levels of abatement and the extent of costs and benefits required/accrued in order to achieve P loads reductions at farm level up to the point where both curves cut for each farm system for each type of soil. 95% confidence intervals are shown in brackets.

Table 7.2 Results standard-setting derogation tests

		Efficient Abatement (%/ha)		Costs/benefits (£/ha)	
			(95% Conf. intervals)		(95% Conf. intervals)
Type of farm	**Type of soil**				
Arable	Sandy loam	48.1	(45.0 - 51.6)	15.79	(12.37 - 19.85)
	Clay loam	59.6	(58.9 - 60.2)	9.89	(7.04 - 13.68)
Arable plus manure	Sandy loam	42.5	(39.4 - 46.5)	19.18	(14.92 - 24.04)
	Clay loam	60.5	(59.8 - 61.1)	9.48	(6.74 - 13.10)
Dairy	Sandy loam	62.7		8.53	
	Clay loam	62.5		8.62	
Beef	Sandy loam	55.0	(50.7 - 58.6)	12.08	(10.01 - 14.74)
	Clay loam	58.9	(54.7 - 62.0)	10.20	(8.50 - 12.53)

Our findings uncover some other relevant conclusions for the design of farm diffuse pollution regulations. These results provide some preliminary benchmarks for the assessment of standard-setting derogations and the definition of GS. In addition, soil types are likely to play a fundamental role in the way diffuse pollution should be targeted. In clay loam soils, we find little variability in efficient levels of abatement between different farm types or management styles. Efficient reduction levels (where costs equal benefits) only vary slightly around the 60% emission reduction barrier. The lower estimate is beef farms with mean abatement levels identified at around 59%, as opposed to dairy farms which illustrate the upper limit of abatement for this type of soil at around 62.5%. In addition, the point where costs of abatement equal benefits in this type of soil are found to be quite constant between different farming systems - around £9-10 per year.

Furthermore, the impact of different farm management practices is more variable in sandy loam soils, with efficient levels of abatement varying from 42.5% on average for arable plus manure types of farms, to an upper estimate of 62.7% for dairy. These account for average differences in efficient abatement levels between farm types of around 20%. This is translated to differences in the average annual costs of abatement estimates per hectare of around £10.

7.4. Applicability of the results and main limitations of the thesis

In terms of practical applications for the successful implementation of the WFD in Scotland, these results illustrate a first exploration of the efficiency of agricultural P loads reductions at farm level and pave the way for future economic analyses of agricultural water use. Our findings highlight the need for further research into the role that the application of BMPs at farm level and different types of soils could play in finding remediation solutions for the agricultural water diffuse pollution problem in Scotland.

In terms of the impact of the Directive on the sector or the subsequent use of derogations, we have seen that this will greatly depend on SEPA's regulatory approach to the achievement of the Directive's objectives. In other words, how strictly the agency will seek the realisation of GS in Scotland and subsequently, the way in which the agency will target the reduction of agricultural diffuse pollution. Essentially, this will influence how far individual farmers will be asked to move forward in the abatement cost curves. Furthermore, we have seen that society's demand for water quality improvements and the financial conditions of the farmer, impose a limit on the practical achievement of the Directive's objectives. In this respect, our results provide preliminary benchmarks for the definition of disproportionate costs and the setting of environmental standards at farm level.

At this stage of the implementation of the WFD, there are many data limitations that have constrained our ability to assess the extent of the compliance implications for typical farms. GS standards have not yet been translated into mandated nutrient emission reductions at farm level, and farmers' current uptake of BMPs or the existing gap between baseline water quality levels in Scotland and the achievement of GS both at sectoral and typical farm level, are unknown. These are important limitations, especially relevant in this case-study where the application of BMPs offers an example of regulation that could aid farmers not only to reduce diffuse pollutant loads, which would improve the quality of the water environment, but at the same time increase their efficiency of production. Our results suggest that there is demonstrable scope for net gains to the farmer with some mitigation achieved at no extra costs. Nevertheless, we could pre-conclude that SEPA will need to proceed with caution in the setting of environmental limits and their targeting at farm level, or the agency may run the risk of enforcing the achievement of inefficient objectives and thereby, impose an unnecessary financial burden on individual farmers and the sector as a whole. This is where the use of reliable benefits estimates becomes necessary.

Overall, the applicability of our results and analysis at policy/regulatory level is mainly constrained by current approaches to the assessment of DC in Scotland (see chapter 1). In this thesis we have outlined the arguments in favour of seeking regulatory efficiency in the decision-making process of granting exemptions for the implementation of the WFD. The use of cost-benefit curves to define disproportionality is advocated, but the practical application of CBA for environmental policy/regulations appraisal is constrained in Scotland by the realities of implementation (chapter 3), and by current approaches to the use of environmental benefits assessment data for implementation of the WFD.

The main arguments against the use of economic valuation in environmental policy-making have not been covered in this thesis. This is mainly due to the fact that in practice the WFD allows for the application of different decision-making tools for the assessment of derogations and additionally, this is a topic that has already been extensively covered in the literature (for more information refer to Ecologic, 2005 or Turner, 2007). In this respect, SEPA's current approach to disproportionality conforms to the Directive's requirements for the economic analysis of water use but has many limitations. These have been outlined in Part 1 of this thesis.

Contrary to the Scottish application, CBA has become a well-established part of the policy review process for the implementation of the WFD in the rest of the UK[47] and other parts of Europe. In this respect, this study sets Scotland on an equal footing with other countries in the application of CBA for the economic analysis of water use (at least in relation to agricultural diffuse pollution concerns and public preferences for water quality improvements) and provides an alternative point of view that can easily be applied and expanded if the agency changes their current views towards economic valuation in the near future. Nevertheless, parts of our analysis may still prove useful to other agricultural policy areas outside environmental regulation, where the use of CBA is more widely recognized in Scotland, such as, for the formulation of agricultural mitigation strategies and legislation to reduce diffuse pollution at national level or for the design of supporting options (i.e. land

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[47] Relevant examples in England and Wales include an appraisal of the economic benefits brought about by measures to reduce diffuse water pollution from agriculture (IGER, 2007) or the RIA on proposals relating to tackling agricultural diffuse pollution (DEFRA, 2007b).

management contracts) to encourage implementation of BMPs at farm level. Government policies are most likely to face a cost-benefits test. In addition, new policy initiatives (including the WFD) must also be accompanied by a Regulatory Impact Assessment, which also incorporates aspects of CBA practices (Cabinet Office, 2003).

Additionally, it is important to note that the CBA process itself can be equally as valuable as the results that are generated. As Hanley and Black (2006a) acknowledge, the CBA process forces the analyst to think through, in a rigorous, consistent fashion, what the economic impacts of a project /policy/regulation will be, who will be affected and when, and the relative magnitude of any gains or losses. We believe that this study has achieved this. In this respect, this study helps to clarify the nature of agricultural water use in Scotland and how it leads to social tradeoffs with other non-agricultural users. Ultimately, this perspective contributes to the debate on how and where water is best employed to maximize its value to society.

7.4.1. Further applications of the results

The results presented in this thesis are relevant to other economic analyses such as the overall environmental impact of agriculture in Scotland, the assessment of environmental improvements benefit estimation or the assessment of the relationship between the WFD and other pieces of environmental protection legislation (e.g. the achievement of climate change targets). Some particular examples of applications are outlined below.

Our results are relevant for expanding current knowledge on the development of environmental accounts for agriculture in Scotland. These accounts are a framework for measuring and valuing the positive and negative impacts of agriculture on the environment, which alongside conventional sector accounts, would help to provide an economic measure of the sustainability of the sector; a measure of farming's external costs to other sectors; and, enhance the evidence base for priority setting within agricultural policy (Spencer et al., 2008). In this respect, our study provides further evidence on the impact of agricultural sources of diffuse pollution on freshwater quality in Scotland. In addition, benefits estimates of river and loch water quality improvements in the country can be used for the valuation of natural capital stocks and ecosystem services flows impacted by the sector. Following the recommendations of a recent review into the development of agricultural accounts for the whole of the UK (Spencer et al., 2008), our valuation exercise has been designed in line with the WFD ecological status levels, and the transferability of marginal values between different regions of the country has been evaluated. This should increase the applicability of these results for future expansions of the accounts.

Our value estimates are useful for costs and benefits transfer within Scotland and open up possibilities for other relevant economic impact analyses at different geographical scales. For example, aggregating these results with other valuation techniques, especially market-based benefits estimation, to find the total economic value of WFD related benefits for the whole of the country or for the development of social damage costs functions to water quality from other sectors (i.e. water industry, malt whisky distilleries, etc.). Moreover, abatement costs for individual farms can be up-scaled to national level, assuming homogeneity for farms within the same type and soil conditions and constant returns to scale. In this respect, cost functions could be linked with the screening tool (Anthony et al., 2005) to account for location/distance to watercourses and transfer coefficients for P. The screening tool provides estimates of source apportionment of P losses from agriculture and forestry, at water body and catchment level, for the whole of Scotland (chapter 4).

Our costs and benefits results are also relevant to other economic sectors, which (alongside agriculture) are also involved in the achievement of GS in Scotland. More precisely, this refers to the link between agriculture and the water industry and the collective achievement of water quality targets at specific catchments/water bodies. Our results increase understanding of the costs and effectiveness of removing agricultural sources of diffuse pollution. In this respect, the water industry may discover that working cooperatively with farmers in some key areas may be more cost-effective than investing in nutrient removal machinery at wastewater treatment plant level. The analysis and exploration of competing uses of the resource between these two sectors is fundamental in Scotland. This is clearly a topic for further research into the marginal costs of abatement for both sectors and into developing initiatives that could exploit the potential collaboration between farmers and Scottish Water. Evidence suggests that these initiatives involve recognition that farmers can be part of the solution and the adoption of cooperative programs between groups of farmers and between farmers and the water industry (Taylor, 2005). Some successful examples include the Drinking Water Co-operation Model in Lower Saxony (Nolte and Shepherd, 2005) and the SCAMp[48] project in England. The barriers which have to be overcome are the

[48] Developed between United Utilities and RSPB , http://www.unitedutilities.com/?OBH=3226

possible reluctance of farmers to accept any responsibility for the pollution problem and for the water supply industry to engage in negotiations/agreements with farmers.

The Abatement Cost Curve method and the results presented in chapters 4 and 6, open up the possibility for a joint assessment of water and climate change mitigation costs and benefits for the agricultural sector in Scotland. A recent DEFRA commissioned project into the development of MAC curves for Greenhouse Gases (GHGs) emissions mitigation strategies by the ALULUCF[49] sectors, concluded that there is a need to accommodate ancillary benefits and costs of reaching different GHGs emission reduction targets for the selection of suitable abatement options at farm level, as currently there are interrelated contradictions between target goals for different environmental protection policies (Moran et al., 2008). As a matter of fact, some activities designed to reduce water quality impacts may increase GHGs emissions by encouraging pollution swapping. For example, Van de Weg et al., (2008) reports on strategies that would increase N_2O emissions, while reducing water nitrate pollution, or that would lead to increased energy consumption. In this respect, it would be useful to find synergies between the identified water measures in this thesis with the GHGs mitigation measures proposed in Moran et al., (2008). Wider economic impacts would need to be investigated under two different policy scenarios; i) changes in agricultural GHGs emissions, if the achievement of "good water status" is set as a policy priority and; ii) how attaining different agricultural GHGs emission reduction targets would affect the overall reduction of water diffuse pollutants.

7.5. Main recommendations for future research

Based on the findings of the study the following research needs have been identified:

1) To expand the analysis to incorporate agricultural measures and benefits estimates for improvements in Groundwater quality in Scotland.

2) In this thesis, we have assessed the economic implications of implementing BMPs to reduce nutrient loads by hectare at the typical farm level. Whilst the application of BMPs is regarded as the most cost-effective path to reach diffuse pollution targets, the application of

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[49] Agriculture, land-use, land-use change sectors and forestry.

other measures and economic incentives at different geographical scales (farm level, catchment level or river basin level), may prove more cost-effective in practice. Future iterations of this research should aim to expand the analysis to account for these potential measures.

3) Probably the main limitation of our analysis is that we have been unable to deal with relevant uncertainty issues that are likely to affect the practical applicability of our results. Specific uncertainty issues have been highlighted throughout this thesis. In future iterations of this research, the authors would like to undertake. Firstly, a quantification of the overall uncertainty surrounding the CBA estimates presented in chapter 4, which could be explored with methods such as Bayesian Network Models in order to give a probabilistic interpretation of "at risk" estimates. This method would allow for the identification of those areas where the need for further research is most critical, to reduce the uncertainty of our estimates (see Barton et al., 2005). Secondly, to update the abatement cost curves with real farms data currently being gathered under RERAD's Research WorkPackage 3.5. And thirdly, an assessment of the potential applicability of undertaking an ecological life-cycle analysis of indicator species in order to increase accuracy in the determination of the impacts of BMPs on ecological status (IGER, 2007).

4) Finally, our exploration into the meaning of disproportionality and the application of CBA, opens the WFD to further scrutiny regarding its overall efficiency and its capacity, through the achievement of GS, to deliver net gains to society. As introduced in chapter 3 this is not an application of CBA which is encouraged by the Commission, as ultimately, the achievement of GS is independent of any justification of costs or public preferences (action only needs to be justified in the case of exemptions). This has been a common approach employed by the Commission for the application of many of its recent environmental legislation.

The overall efficiency of European environmental legislation has already been questioned. Pearce (2004) investigated if a selection of major European environmental regulations, including the WFD, would pass a cost–benefit test. He discovered that while many did pass the test, the majority did not. This conclusion led him to reflect on the willingness of Member States to sign up to inefficient regulation (Pearce, 2004). This is a fundamental question that we believe deserves further exploration, especially considering SEPA's approach to implementation in Scotland. In this respect, our results can pave the way for an overall assessment of the costs and benefits of implementing the Directive.

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ANNEXES

Annex I - Literature review of BMPs studies

Reference /year	Location	Title	Type of study	Scope	Types of farms assessed	Number/types of agricultural BMPs identified	Types of costs covered	Assessment of Effectiveness/ Diffuse pollutants covered	Baseline level of farm nutrient loads	suitability for CEA
Dickson et al (2005)	Scotland	Catchment scale appraisal of best management practices (BMPs) for the improvement of bathing water . Report for SEPA	field case study	Determine to what extent improved farm practices and BMPs could contribute to improve water quality in a small rural cathment in Scotland.	7 farms, mixed livestock/dairy practices (beef and sheep, suckler and calves, suckler and sheep and dairy)	7 measures	total investment costs	N, P and FIO (different units in within categories, difficult aggregation)	yes (comparison ex-ante and ex-post installation of BMPs)	Bad. Lack of cost data (no other types of costs than capital investment), measures of effectiveness in different units.
Frost et al, (2002)	Scotland	The impacts of agricultural environmental management: case studies from theory to practice. Report for SEPA	field case study	identification and assessment of BMPs to reduce the impact of agricultural environmental management. Not only focussed in water quality. Other environmental media covered: land and air.	hill sheep, upland stock, mixed stock and arable, dairy, general arable and intensive arable with vegetable production	BMPs to control nitrogen pollution (11 measures identified), phosphorus (10), pesticide (13), suspended solids (15), to control micro-organism pollution (10)	Capital and annual costs, separation between management payments and additional capital payments. Mixture of costs units: £/ha, £/m	N, P, pesticides, suspended solids, phaecal pathogens - qualitative assessment	Baseline loss levels defined for P and N for the different farm types	Poor, different cost categories used (difficult aggregation), no quantitative assessment of effectiveness.
McTaggart et al, (2002)	Scotland	Nitrate Vulnerable Zone Action Programme Regulations - Regulatory Impact Assessment	policy evaluation	Regulatory Impact Assessment (of proposed options) for the Action Programme regulations of Nitrate Vulnerable Zones in Scotland. Estimation of costs and benefits for the measures contained in each of the options for each farming sector within each NVZ, together with an assessment of their relative importance for reductions in leaching of nitrates	all types of farms in the Scottish NVZs (Mixed livestock, Specialist pig farms, Cropping and sheep, Cattle and sheep, Cropping and intensive livestock, Specialist poultry, Cropping and dairy, Dairy farms)	Option 1(13 measures), option 2 (10 measures)	Total annual costs (inc, maintenance and amortisation of capital) and total one-off costs for some of the measures identified. Costs for the whole of Scotland.	reduction on farm nitrate leaching. Tonnes of N reduced/year	MANNER (livestock) and NCYCLE (grassland) software used to predict baseline nitrate leaching levels for different farm types	Good but only covers BMPs for Nitrogen. Difficult to disaggregate cost information
IGER/ADAS, 2007	UK/England and Wales	An inventory of methods to control water diffuse pollution from agriculture - user manual. Report for DEFRA	Research	User manual that presents costs-effectiveness estimates of integrated diffuse mitigation measures at farm level. Latest output of an on-going DEFRA research programme to mitigate farm pollution. Summarises the results of many projects	Arable, arable plus manure, dairy, beef, broilers, breeding pigs (indoors) and breeding pigs (outdoors)	44 measures	one-off, annual cash, annualised capital (amortised over a given period of time) or annual and amortised costs (units: £/farm or ha, £/year/farm or ha)	estimated losses for N, P and FIO for sandy and clay loam soils. Units: Kg N/ha; Kg P/ha; % FIO reduction compared to baseline losses	Estimated for each type of soil for each of the farms assessed. N and P (modeling), FIO (expert judgement). Same units as effectiveness of measures	Very good. Effectivenes and baseline levels estimates in same units. Possible to calculate % reductions for each measure.
ENTEC/ADAS, 2006	UK/England and Wales	Benchmark costs database and guidance on the application of the cost-effectiveness methodology	UK guidance document	Benchmark costings under the WFD leading to a cost proforma for use by governmental departments or any other agencies involved in the implementation process of the Directive in the UK. Part of the Collaborative Research project on economics of the WFD	Different pressures/sectors assessed. BMPs for 6 farm types; grass (dairy and suckler beef), breeding pigs (indoors and outdoors), broilers, arable, arable plus manure.	44 measures (see IGER/ADAS measures)	Non recurrent (investment and design...) and recurrent costs (fixed and variable O & M). units: £, £/ha, £/ha/year, £/ha total farm area	low, medium and upper bound estimates. Cover; N, P and FIO. Units: kg N/unit, Kg P/unit, FIO % reduction	Same as IGER/ADAS. Estimated baseline loss for selected farms	Good. But excel database does not provide links between effectiveness and baseline levels.
SAC/ University of Cambridge, 2004	UK/England and Wales	The scope for regulatory incentives to encourgae increased efficiency of inputs by farmers. Report for DEFRA	Research	Increase understanding of the potential role that voluntary and regulatory instruments may have on reducing farm loads of N and P. Method: data envelopment analysis (modelling technique) to find out efficiency savings in nutrient input for different agricultural sectors	Cereal (data for a 108 farms), dairy (154 farms), sheep LFA (19 farms) and sheep lowland (118 farms) sectors were assessed	Voluntary measures: 10 for nitrogen and 7 for Phosphorus reductions	efficiency savings (£/kg nutrient/ha)	Efficiency-saving N and P use (kg/ha). Literature review of effectiveness voluntary measures.	based on estimated actual efficiency levels of nutrient use.	N/A
DEFRA, HM treasury Concultation, 2004	UK/England and Wales	Developing measures to promote catchment-sensitive farming	DEFRA/HM TREASURY consultation document	Consultation document on approaches and possible measures to improve water quality through catchment sensitive farming	Agricultural sector in England	21 general measures	Qualitative assessment of costs (no cost, low cost and high costs)	None	Qualitative assessment of diffuse sources of pollution. Review of the literature.	N/A

Reference /year	Location	Title	Type of study	Scope	Types of farms assessed	Number/types of agricultural BMPs identified	Types of costs covered	Assessment of Effectiveness/ Diffuse pollutants covered	Baseline level of farm nutrient loads	suitability for CEA
RPA, 2003	UK/England and Wales	Water Framework Directive - Indicative Costs of Agricultural Measures. Report for DEFRA	policy evaluation	new cost estimates for the agricultural sector to update Regulatory Impact Assessment of introducing the WFD in England and Wales	Dairy (south west), upland sheep (Wales), Arable farming (East Anglia)	Dairy (16 measures), Sheep (7 measures) and arable (10 measures). Identifies the % number of farms in the studied regions which would need to implement specific measures. Identifies the % of farm area for which some measures would be applicable.	One off and annual costs. Estimation of costs in terms of £/farm, aggregation by multiplying total number of farms that would require to implement each measure.	Qualitative assessment of the following farm pollutants. P, N, Sediments and soil loss, organic wastes, pesticides, veterinary medicines and FIO.	Qualitative, general statistics of farm diffuse pollution were used to assess the gap between baseline scenario and different compliance scenarios.	Poor. good indication of costs. This work was updated in ADAS, 2007
ADAS, 2002	UK/England and Wales	Methods and measures to minimise the diffuse pollution of water from agriculture - a critical appraisal. Final report to DEFRA	Review	appraise the nature and effectiveness of approaches taken in other countries to minimise diffuse pollution of water from agriculture and the policy options available to control the problem	agricultural sector in general	A wide range of pollutants were considered: nitrogen (N), phosphorus (P), sediment, pesticides, veterinary medicines, biocides, pathogens and biological oxygen demand (BOD)- 34 measures in total	No	assessment of effectiveness for some measures (%reduction)	No	Poor. Good source of information about BMPs and international experiences in their application
Dutch Ministry of Water, 2005	Holland	In pursuit of optimal measure packages: Dutch handbook on costeffectiveness analyses for the EU Water Framework Directive	Dutch guidance document	Economic frameworks for a consistent application of the cost-effectiveness analysis under the WFD across the different regions in Holland. Uses examples/case studies. Different sectors/pressures analysed.	Pig farm sector	4	Costs of reducing P loads (E/kg P discharged).	kg of P removed	Kg reduction required	Good. Only covers four measures for one agricultural sector (pig farms).
Ecologic, 2003	Germany	Handbook - Basic Principles for selecting the most cost-efficient combination of meaures for inclusion in the programme of measures as described in article 11 of the water framework directive	German guidance document	general approach for the selection of the most cost-effective combination of measures to achieve the objectives of the Directive. Uses examples/case studies. Different sectors/pressures analysed.	Agricultural sector	Annex II of the report covers some diffuse pollution measures. 4 measures, mainly covering P and N farm loads reductions.	Capital and annual costs. Different units: E/m2, E/ha, E, E/kg P discharged.	quantitative (% reduction) and qualitative	None	Poor. Different units costs estimates. Difficult to make comparisons for CEA. Effect of some measures only assessed qualitatively
Ministerio del Medio Ambiente, 2002	Spain	Analisis economico del plan de cuenca del cidacos aplicacion de la guia de analisis economico (Directive marco del agua 2000)	pilot-test study	Application of Cost-effectiveness Analysis for the selection of measures to improve water quality/quantity in the Cidacos river basin (Spain). Different sectors/pressures analysed, including agricultural pressures	Arable and livestock farms in the study area	27 measures studied. 11 in relation to water quality. however, some other measures aimed to reduce agricultural water use would improve quality as they increment flows.	total capital or annual costs for each measure (units: Euro/ha, Euro m3, Euro/ha/year)	pollutants assessed BOD, N (% reduction in pollution). method of estimation not stated.	Yes, method of estimation not stated	Good, this study is an application of CEA for the selection of measures in Spain. Specific for Spanish issues.
Hilliard et al, 2002	Canada	Agricultural Best Management Practices for the canadian prairies a review of literature	Review	literature review of existing BMPs to reduce agricultural diffuse sources of water pollution	Agricultural sector in general	14 measures	Qualitative	Qualitative	None	Poor. Good source of information, understanding BMPs. It could be used for a qualitative assessment of the cost-effectiveness of measures

Annex II - DESCRIPTION BMPs (Source: IGER/ADAS; 2007)

Category	ID	BMP Measures	Description/Rationale/Mechanism of Action	Comments/Applicability/Limitations
Land Use	1a	Convert arable land to extensive grassland: conversion to ungrazed grassland	Reduce losses of N and P by changing the land use from arable cropping to permanent grassland, either ungrazed or with a low stocking rate and zero or low fertiliser input.	The method is applicable to all forms of arable farmland but is potentially most suited to marginal arable land that was historically kept as grazing land. Benefits will be greatest on sandy and silty soils that are most prone to erosion. This is an extreme change in land use that is unlikely to be adopted by farmers without the provision of suitable incentives. It may be particularly suited to areas where the converted land would have amenity or conservation value.
	1b	Convert arable land to extensive grassland: conversion to extensive grazing		
Soil Managament	2a	Establish cover crops in the autumn: cereals	Cover crops help to reduce the mobilisation of agricultural pollutants by increasing nutrient uptake and reducing surface run-off and soil erosion. A cover crop will take up residual nitrate and other nutrients from the soil after the main crop has been harvested in the summer or early autumn, leaving less nitrate available for leaching over winter. Ensuring that the land is not left exposed helps reduce soil erosion and the mobilisation of associated pollutants. Cover crops can also help to improve soil structure compared with bare soil.	The method is relatively easy to implement and is already used in some grassland systems with the undersowing of maize and spring barley with a grass seed mixture.
	2b	Establish cover crops in the autumn: other crops		
	3	Cultivate land for crop establishment in spring rather than autumn	Autumn cultivation of land stimulates the mineralisation of N from organic matter reserves at a time when there is little N uptake by the crop, which will increase the potential for over-winter leaching losses. By cultivating in spring, there will be less opportunity for mineralised N to be leached and the N will be available for uptake by the established spring crops. This is a mobilisation method.	This is applicable to cultivations prior to the drilling of spring crops (e.g. maize, sugar beet, potatoes) or where there is a switch from winter to spring cereal cropping. It is also applicable to grassland systems where grass leys are ploughed out and re-seeded.
	4	Adopt minimal cultivation systems	The use of discs or tines to cultivate the surface as a primary cultivation in seedbed preparation or direct drill into stubbles (no-till). Minimal cultivation (rather than ploughing) may be the best way to maintain organic matter, preserve good soil structure and break up surface crusts. The resulting soil conditions should improve infiltration and retention of water, thereby reducing loss of P and sediment. Mechanism of action: This is a mobilisation method.	This method is already being adopted on a number of arable farms, with around 1.5 million hectares cultivated using discs or tines. No-till is unsuitable for light soils that are prone to capping. Minimum cultivation is less applicable in a very wet autumn and is only suitable where soil structural problems have been alleviated. Minimum tillage may increase resistant weed populations and therefore increase reliance on chemical control.
	5	Cultivate compacted tillage soils	After harvest, cultivate compacted tillage soils with discs or tines to increase surface roughness and infiltration. Carry out the cultivation in dry conditions and well ahead of the start of drainage in late autumn. Establish a vegetative cover through natural regeneration or from broadcast barley seed. Cultivation disrupts soil surface crusts and increases surface roughness. This enhances opportunities for rain to infiltrate into the soil and reduces the erosive energy of any surface flow that does occur. The method will reduce losses of P and, if manure is spread on compacted tillage soils, will also reduce	The method is applicable to the arable sector on cereal and maize land where soils are compacted, particularly in high winter rainfall areas. For the method to be effective it should be carried out in the late summer to early autumn (i.e. when soils are dry) when there are many other competing demands for the farmer's time.

		losses of FIOs. This method reduces surface run-off and soil erosion.	
6	Cultivate and drill across the slope	Cultivate and drill land along the contour to reduce the risk of developing surface flow. On fields with simple slope patterns, this measure may reduce the risk of surface run-off being initiated. The ridges created across the slope increase down-slope surface roughness and provide a barrier to surface run-off. Losses of P and FIOs are reduced as a result. This is a mobilisation method.	Applicable to all cultivated soils on sloping land. The method is more time-consuming and requires greater skill than conventional field operations. Also, this method is only likely to be effective for crops grown on gently sloping fields with simple slope patterns.
7	Leave autumn seedbeds rough	Avoid operations that create a fine seedbed that will 'slump' and run together. Leaving the autumn seedbed rough encourages infiltration and reduces the development of surface flow, thereby reducing the loss of P and FIOs.This is a mobilisation and transport method. A more open seedbed is achieved by using a reduced number of cultivations, particularly from powered cultivation equipment, and by avoiding the use of a heavy roller. This helps to reduce the risk of surface flow by preventing soil capping and enhancing infiltration of surface water into the soil. A rough seedbed also helps to break up any surface flow that is generated, reducing the risk of sheet wash and rill/gully development.	Applicable to the establishment of crops in the arable sector (particularly on sandy and silty soils). It is most applicable to winter cereal crops that can establish well in coarse seedbeds.The method is not well suited to crops such as oilseed rape, sugar beet and reseeded grasslands that require fine, clod-free seedbeds.
8	Avoid tramlines over winter	Delay the establishment of tramlines until the spring. Tramlines are generally established for combinable crops at the time of drilling. Compacted tramlines can result in the channelling of surface water and the development of rills and gullies on erosion susceptible soils. Avoiding tramlines over winter therefore helps prevent soil erosion, accelerated run-off and the loss of P. This is a mobilisation method. Avoiding the compaction produced by tramlines over winter helps prevent soil mobilisation and surface run-off.	applicable to winter cereals in most arable farming systems, particularly on light soils in areas with higher winter rainfall. It is not applicable to most oilseed rape crops, due to the need to apply pesticides during the autumn period. It is not a straightforward method to implement as farmers generally need to access winter cereal fields in the autumn to apply pesticides. To do this while avoiding the compaction associated with tramlines may only be possible by using low ground-pressure vehicles. Such a management practice would probably necessitate a change of herbicide policy on the farm and increase costs. The approach is compatible with the Environmental Stewardship scheme and there is no conflict with other methods.
9	Establish in-field grass buffer strips	establish grass buffer strips on sloping fields along the land contour, in valley bottoms or on upper slopes to reduce and slow down surface flow. Cut regularly in the first 12 months to control annual weeds and encourage grasses to tiller. In-field buffer strips can reduce P and, where manures are applied to tillage land, FIO losses by slowing run-off and intercepting the delivery of sediment. The Entry Level Environmental Stewardship scheme offers options for strips between 2 and 6 m in width. Also, under the Higher Level Stewardship Scheme, there is the option to establish in-field grass areas to prevent erosion and run-off (with a maximum permissible area of 30% of each field). The strip acts as a natural buffer to reduce the transfer of diffuse pollutants in surface run-off from agricultural land to water. Buffer strips can act as a sediment-trap, as well as helping to	applicable to all arable farming systems on sloping land. They are particularly suited to fields with long slopes, where high volumes of surface run-off can be generated. The buffer strips will reduce the length of fields, but increase the time taken for field operations by around 10%. They may be more effective when combined with additional riparian buffer strips (Method 43).

		reduce nutrient and pesticide losses in run-off. The strip has no effect on nitrate other than pro rata for the area taken out of production (i.e. the buffer strip is similar to unfertilised grass).	
10	Loosen compacted soil layers in grassland fields	Reduce surface run-off from grass fields by shallow spiking or subsoiling to disrupt compacted soil layers, reducing the infiltration of rainwater and slurry into the soil and thus increase the frequency of surface run-off.. These operations should be carried out in dry conditions. method reduces the risk of pollutants being transported to watercourses in surface run-off. Trampling by livestock, particularly cattle, and the passage of heavy farm traffic can compact the upper layers of grassland soils in both grazing and silage fields. Because the soil is cultivated only infrequently, the compaction persists and may build up over a number of years. The reduced porosity impedes the percolation of rainwater and slurry and increases the risk of surface run-off. Shallow spiking, slitting or subsoiling breaks up this compacted layer and allows more rapid infiltration of water, thus reducing run-off from the soil surface. In addition, soil aeration is improved and roots are able to penetrate deeper into the soil, which will increase nutrient uptake from deeper soil layers.	method applicable to grassland farms but most particularly those with high stocking rates of cattle and a high proportion of older swards. Compaction is most likely to occur on medium and fine textured soils. There are few limitations to the adoption of this method although the field operations may be more difficult on stony soils.
11	Maintain and enhance soil organic matter levels	Low soil organic matter levels are a concern in some arable systems. They can give rise to soil structural problems and increased risk of soil erosion. Maintaining or enhancing the content of soil organic matter helps to reduce the risks of surface run-off and erosion, and enables the efficient use of soil nutrients and added mineral fertiliser. These benefits should be effective in reducing P losses. This is a mobilisation and transport method. I.	applicable to all arable farming systems, particularly on low organic matter soils that are structurally unstable. Grasslands tend to be characterised by higher organic matter contents and a more stable structure.Practicability depends on the local availability of organic manures. There is usually ample opportunity for the spreading of organic manure at some point in an arable rotation.
12	Allow field drainage systems to deteriorate	Allow existing (old) drainage systems to naturally deteriorate, i.e. cease to maintain them. Some drainage systems will survive for decades with little management, therefore this is a long-term option. Other drainage systems may take only a few years to deteriorate. Drainage systems can accelerate the delivery of agricultural pollutants from land to a watercourse, by acting as a preferential (by-pass) flow route. Allowing drainage systems to deteriorate therefore reduces hydrological connectivity and the transfer of pollutants to the watercourse. When drains have deteriorated, water is forced to percolate through the soil at a slower rate.	applicable to the intensive grassland sector on heavy soils. It is a relatively easy option to implement but is unlikely to be acceptable to farmers, particularly in areas of heavy soils where waterlogging is a problem. Measure compatible with the Higher Level Environmental Stewardship Scheme. without an effective drainage system, economically sustainable arable cropping would not be possible on many heavy soils. If the drainage status deteriorated greatly, it is likely that a farmer would revert the arable land to grassland or other alternative land use. Similarly, the method is not applicable for farmers growing potatoes and sugar beet on unstable, silty soils. Method is easy to implement as no action is necessary. However, there will be considerable resistance from farmers to adopting the method as a deliberately managed activity without financial incentive. It is probable that with increasing soil wetness, it would also be necessary to reduce stocking rates on livestock farms (see Method 15).

Livestock Management	13	Reduce overall stocking rates on livestock farms	Reducing the stocking rate reduces the amounts of N, P and FIOs deposited in fields in excreta and handled in manures. It also allows mineral fertiliser inputs to be reduced and reduces poaching of the soil. Livestock excreta deposited in the field and applied in manures are important sources of N, P and FIOs. Reducing the number of stock will reduce the amounts of excreta and manure produced per unit area.	applicable to all livestock farms but will have the greatest impact on intensively stocked units that produce large quantities of excreta and where the risk of soil structural damage is greatest. method would be relatively simple to implement but would have a serious impact on profitability. The main factor limiting its adoption would be the major reduction in farm income resulting from reduced stock numbers.
	14	Reduce the length of the grazing day or grazing season	Reduce the length of time livestock are allowed to graze in the fields, either by keeping stock inside during the night or by shortening the length of the grazing season, particularly in autumn. Reducing the time animals spend grazing reduces the amount of urine deposited in fields.	applicable to intensive livestock farms where animals graze outside between spring and autumn and where there is already suitable housing. It will be most effective on free-draining or shallow soils, which are most susceptible to nitrate leaching. Reducing the length of the grazing day is most suited to dairy farms, where cows can be kept indoors between the afternoon milking and morning milking. Shortening the grazing day or grazing season will both increase the time for which animals are housed and increase the amount of manure produced. The method will only be effective if suitable precautions are taken to minimise losses from this manure when it spread on the fields. Increasing the amount of time when animals are housed creates additional work on mixed farms. Reduced grazing is likely to increase the proportion of grass utilised by cutting. The increased labour costs would reduce profitability significantly, particularly on farms with a high dependency on grass forage. Reducing the length of the grazing season goes against the current trend of maximising the use of grazed grass by extending the grazing season.
	15	Reduce field stocking rates when soils are wet	When soils are wet, the numbers of livestock per unit area or the time stock spend on the field should be reduced sufficiently to avoid severe poaching and compaction of the soil. Decreases the risk of surface run-off and transport of pollutants to watercourses.Lower stocking rates will also reduce the amount of excreta deposited in these areas and the amounts of pollutants available for loss. Potential benefits are greater for P and FIOs than for nitrate because a greater proportion of the loss of these pollutants occurs via surface flow.	applicable to all livestock farms where animals are kept outside but most particularly to those with high stocking rates, where extended grazing is practised or where stock are wintered outdoors. On some farms, it may only be necessary to install temporary fences to exclude stock from temporarily wet areas of particular fields. Poaching is likely to be more severe with cattle grazing than with sheep. Although outdoor pigs are particularly damaging to the soil, the method is of limited applicability to these units as they are usually set-stocked and do not have the option of moving stock to other fields or indoors. Fine-textured, less-permeable soils are most susceptible to poaching and the risk is increased in high-rainfall areas.This method will only be fully effective if methods are adopted to reduce losses from this additional manure when it is spread onto land. Profitability would be seriously reduced on farms that are highly dependant on grass forage and are dominated by fine-textured soils.
	16	Move feed and water troughs at regular intervals	Feed troughs, feeding racks and water troughs for outdoor stock should be re-positioned at intervals to reduce damage to the soil and improve the distribution of excreta. Troughs and racks should be moved more frequently when the soil is wet and most easily poached. They should not be sited close to water courses. Animal movements in fields are concentrated around feeding points and water troughs. This results in large inputs of	applicable to beef/sheep systems than dairy, where feed is usually provided close to or on the farmstead (except for buffer feeds). It is especially relevant to farms where livestock are wintered outside. Feed troughs and feeding points are already routinely moved on some farms.This is a simple method with few limitations to its implementation. It is more difficult to vary the position of water troughs. This would probably require use of a bowser or installation of a number of permanent drinking points within the field, as used on

			excreta to these areas, which become a source of high levels of N, P and FIOs.	dairy farms that employ a strip-grazing system.
	17	Reduce dietary N and P intakes	Adjust the composition of livestock diets to reduce the total intake of N and P per unit of production: by avoiding diets that contain N and P in excess of the dietary requirement of the animal and by formulating diets that increase the efficiency of N and P utilisation by the animal. this will reduce the amount of N and P excreted, either directly to fields or via manure, and thereby minimise additions to the pools of N and P that are sources of diffuse pollution.	Potential for applying the method: Benefits will be greatest on intensive dairy, pig and poultry units and least on those feeding a largely forage diet. Short-term benefits of reducing N and P in run-off will be greatest on less-permeable soils, and for nitrate leaching on sandy and shallow soils. The longer-term benefits of reducing soil nutrient loadings will be effective on all soil types. The extent to which these methods can be applied depends on the proportion of farms currently feeding excess N and P or not already using feed supplements.
	18	Adopt phase feeding of livestock	Manage livestock in smaller groups, divided on the basis of their individual feed requirements. Feed the groups separately with rations matched to the optimum N and P requirements of the animals within each group (phase feeding). Phase feeding allows more precise matching of the ration to the individual animal's nutritional requirements. Nutrients are utilised more efficiently and less of the dietary N and P is excreted, thereby reducing the N and P content of manures. This reduces the amount of N and P available for loss when these manures are applied to fields and the potential accumulation of N and P in the soil. Mechanism of action: Livestock at different growth stages or stages of the reproductive or lactation cycle have different optimum feed requirements. However, because of limited labour and housing facilities, livestock with different feed requirements are often grouped together and receive the same ration. As a result, some stock will receive higher levels of N and P than they can utilise efficiently and will excrete the surplus (see Method 17).	applicable to all livestock systems except those based primarily on grazing. It would be effective at reducing losses of P and N in run-off from fine textured soils and in reducing nitrate leaching from free-draining soils. method is more suited to larger units where there would be greater numbers of animals in the individual feeding groups. It would be most effective if adopted in combination with the actions described in Method 17 to reduce dietary N and P intakes. As with Method 17, it is important that improvements in N and P utilisation are used to reduce total N and P inputs rather than as an opportunity to increase agricultural output from the unit, which would lessen the impact on losses.In the ruminant sector, this method reflects current practice where cows are grouped according to yield. However, practical application may be difficult on dairy units where cows are fed a single diet across a range of yields.
Fertiliser Management	19	Use a fertiliser recommendation system	This involves the use a recognised fertiliser recommendation system to plan fertiliser applications to all crops, Do not exceed optimum recommended rates, to time fertiliser applications to minimise the risk of loss of nutrients (e.g. avoid autumn N applications and early spring timings to drained clay soils), to take full account of manure inputs when planning mineral fertiliser applications, ensure accurate use of mineral fertilisers by proper maintenance, setting and calibration of spreading machinery and the use of good quality fertilisers. Fertiliser recommendation systems take account of the following factors: soil nutrient supply based on soil analysis or climate, previous cropping and soil type, crop nutrient requirements for a	A good fertiliser recommendation system ensures that the necessary quantities of the essential crop nutrients are only available when required for uptake by the crop. As a result, the amount of excess nutrients in the soil is reduced to a minimum. The system also ensures that the soil is in a sufficiently fertile state to maximise the efficient use of nutrients already in the soil, or supplied from other sources such as organic manures. Maintaining an appropriate balance between nutrients is also important to maximise the efficient uptake of all nutrients and reduce losses to a minimum. Measure can be used in all farming systems, but are particularly effective in intensive grassland, arable and horticultural systems. The method would have less impact in extensive grassland systems, as according to fertiliser practice surveys, most grassland soils receive less N than is

		given soil and climate, crop requirement for nutrients at the various growth stages, the amount of nutrients supplied to the crop by added manures and by previous manure applications and soil pH and the need for lime. this Measure will reduce the risk of applying more fertiliser nutrients than the crop needs and will minimise the risk of causing diffuse water pollution by nitrate and P.	recommended by RB209.The method would require investment in education and guidance.
20	Integrate fertiliser and manure nutrient supply	involves: use a recognised fertiliser recommendation system (e.g. RB209, PLANET and other supplementary guidance) to make full allowance of the nutrients applied in manures and reduce mineral fertiliser inputs accordingly, use manure analysis to gain a better understanding of nutrient applications and supply, to keep records of mineral fertiliser and organic manure inputs to individual fields Mechanism of action: The amount of nutrient is reduced at source. Mineral fertiliser applications are reduced to no more than is required for optimum economic production levels and to maintain adequate levels in the soil.	applicable to intensive grassland and arable systems, but also relevant to extensive grassland systems where breeding ewes are brought onto more fertile low-lying ground in late autumn to early winter. The method is effective wherever mineral fertilisers are used to top-up the nutrients supplied in organic manures. This method could be easily implemented via advice, education and guidance. Particular guidance is required with soiil and manure sampling, on-farm analysis of manure, and interpretation of results.
21a	Reduce fertiliser application rates: 20% reduction in N	Reduce the amount of N and P fertiliser applied to crops by a certain percentage below the economic optimum. Rationale: On most fields, limiting the amount of N fertiliser applied to crops will reduce the quantity of residual nitrate in the soil after harvest. In the short term, limiting P fertiliser rates can reduce the amount of soluble P lost from the system. In the long term, reducing P fertiliser rates can reduce the amount lost as particulate P. The amount of fertiliser applied is reduced at source. There will be a slight reduction in the amount of residual soil nitrate available for leaching in the autumn. However, there will be no effect on the amount of nitrate mineralised from soil organic matter. This mineralised nitrate forms the larger part of the nitrate pool that is available for leaching over the autumn and winter. In the longer term, where soil P reserves are allowed to run down, there will be a reduction in soluble P loss. Limiting P fertiliser applications in any one year will reduce the amount of P that can be lost in surface run-off or in drain-flow. However, where organic manures are applied to the soil, there will be little net effect from reducing mineral fertiliser rates.	applicable to all farming systems where fertiliser is used. The method would have a significant impact on crop yields. The impact of reducing fertiliser P would be greatest and immediate for crops that are particularly responsive to the nutrient (e.g. potatoes and some vegetable crops). Reductions in N fertiliser would have an immediate impact on all crops other than legumes. For most crops, any reduction in fertiliser N would cause a small but economically significant reduction in yield.
21b	Reduce fertiliser application rates: 20% reduction in P		

22	Do not apply P fertilisers to high P index soils	Do not apply mineral P fertiliser to soils that have an ADAS Soil P Index of 4 or above. The amount of P lost by erosion or leaching depends on the soil P content. Losses in solution increase rapidly once soil P reserves reach elevated levels, e.g. ADAS Soil P index 4 or above. Losses can be minimised by maintaining soil P levels at Index 2 or by allowing the P content of high P index soils to run down. If mineral P fertiliser is not applied and the P content of high P index soils is allowed to decline, the amount of P lost with eroded soil particles and in solution will be reduced. Soil P is adsorbed on soil particles and is lost when sediment is eroded from fields in surface flow and in drain flow. The higher the soil P reserves, the greater the amount of P lost with the transported soil. The amount of P lost in soil solution is also greater on high P index soils, particularly on P-saturated soils. Balancing P inputs to crop offtakes and not applying P to soil with high P reserves must also take account of the P supplied in manure applications (see Methods 20 and 33). However, the run-down of high soil P reserves is a gradual process and full benefits will only be achieved in the longer term (>10 years).	applicable to all farming systems, but would have greatest effect in intensive grassland and arable systems. method could be easily implemented via advice, education and guidance. Particular guidance is required with soil sampling, analysis and interpretation of Soil P Index levels. There would be resistance to adopting the method for those crops (e.g. potatoes) that can respond to P mineral fertiliser on high P Index soils.
23	Do not apply fertiliser to high risk areas	Do not apply mineral fertiliser at any time into hedges or ditches or to field areas where there are direct flow paths to watercourses. For example, areas with a dense network of open drains, wet depressions (flushes) draining to a nearby watercourse, or areas close to road culverts. Fields with high P index soils should also be considered as area with a high risk of P loss (see Method 22). Avoiding fertiliser spreading to hydrologically connected areas helps prevent the mobilisation and transfer of agricultural pollutants from land to water.	potentially applicable to all grassland farming systems, but may be most applicable to the extensive grassland sector, where open drains and waterlogged areas are common. It is also applicable to arable fields with hedges, ditches and areas close to road culverts.It is an easy option to implement, although some farmers may be resistant to not applying fertiliser to grassland that contains areas prone to waterlogging or to grassland areas with a dense network of open drains. Avoiding fertiliser spreading in high-risk areas is compatible with the Environmental Stewardship Scheme and there is no conflict with other methods.
24	Avoid spreading fertiliser to fields at high-risk times	involves: do not spread mineral fertiliser at times when there is a high risk of surface flow or rapid movement to field drains from wet soils, do not spread N fertiliser between September and February when there is a high risk of nitrate leaching loss, unless there is a specific crop requirement during this time and do not apply N fertiliser when there is little or no crop uptake. Fertiliser timing affects the mobilisation of nutrients being released from land to water. Avoiding spreading fertiliser to fields at high-risk times reduces the availability of nitrate for loss through leaching and of P for loss in surface run-off or rapid preferential flow. Surface run-off is most likely to occur when rain falls onto sloping ground with soils that are saturated, frozen or snow covered. Rapid preferential flow of fertiliser nutrients through the soil is most likely to occur from drained soils when they are wet and rainfall follows soon after fertiliser has been applied. The method aims to prevent nutrients being added at times when there is	potentially applicable to most farming systems, i.e. all which use mineral fertiliser. Fertiliser timing to avoid high-risk periods is compatible with the Environmental Stewardship Scheme and there is no conflict with other methods. It would be relatively acceptable to the farmer, although the prediction of rainfall and restriction on the timing of mineral N applications may cause practical difficulties for some farmers. The adoption of this method will require a degree of education and advisory activity to persuade farmers that the spreading of fertiliser at high-risk times (e.g. when soils are ‘wet’ and surface run-off or drain flow losses may occur) should not be undertaken. Farmers may be particularly reluctant to avoid applying fertiliser to drained clay soils in early spring to promote early season crop growth.

			rapid transfer of water from the soil surface to water bodies or rapid leaching to ground water.	
Manure Management	25	Increase the capacity of farm manure (slurry) stores	expand facilities for collection and storage of slurry and dirty water to allow them to be spread at times when there is a low risk of run-off and when there is an actively growing crop to utilise the nutrients supplied in the manure. Collection and storage of slurry and dirty water provides flexibility about when to apply these materials to fields. There will be fewer occasions when a lack of storage capacity forces farmers to apply manures at times when there is a high risk of polluting ground or surface waters. If a farm has little or no storage for slurry and dirty water, the farmer will be obliged to spread these materials as they are produced. This will inevitably result in applications at times when there is a risk of nitrate leaching and of N, P and FIOs from the manure being transported to watercourses in surface run-off or in drainflow.	applicable to livestock farms that have limited manure storage facilities. The provision of adequate storage facilities is most important on farms that handle their manure as slurry and those that produce dirty water. In contrast, solid manure can be stored in field heaps, or sometimes in the animal house, prior to land-spreading at a time of year that presents less risk of pollution. The method would be effective on all types of soil. method will only be effective if implemented in conjunction with Methods 31 - 36 (where relevant) and particularly where the actions in Methods 26 - 30 have also been adopted.
	26	Minimise the volume of dirty water produced	By minimising unnecessary dirty yard areas, avoiding excessive use of water in washing down yards, buildings, etc., preventing unnecessary mixing with clean water from uncovered clean yard areas and from roofs, etc., roofing over yard areas and covering dirty water and slurry stores. Minimising the volume of dirty water produced reduces the volume to be stored and spread. Farms will be less likely to run out of storage space during winter and be forced to spread dirty water or slurry at times when there is a high risk of pollution occurring.	method is mainly applicable to farms with cattle, particularly dairy farms, though most livestock farms will produce some dirty water. The method will be effective in reducing losses from fine textured and capping soils where there is the greatest risk of run-off and on free-draining soils where there is a high risk of nitrate leaching. There are few limitations to the adoption of this method though there may be practical limitations to the roofing of yards and covering of dirty water or slurry stores. The extent to which yard areas can be reduced is limited by the need to avoid overcrowding that might adversely effect herd health and milk quality. Preventing unnecessary inputs of rainwater will be most effective in high rainfall areas. Using a pressure washer to wash down yards uses more water than a non-pressurised supply.
	27	Adopt batch storage of slurry	Description: Store batches of slurry for at least 90 days before spreading on fields and do not add fresh slurry to the store during this storage period. FIOs die off during storage. There are fewer microorganisms in the material that is spread and therefore less risk of FIOs entering water bodies via surface run-off or percolation to field drains. Numbers of FIOs decline during storage and this can be an effective means of reducing bacterial numbers in the slurry. It is less effective for controlling the protozoan parasite, Cryptosporidium. If there is run-off or percolation into field drains following slurry application, the transported material will contain many fewer FIOs compared with ‘fresh’ slurry. The method is primarily directed at reducing pathogen loads and will have little effect on nitrate or P losses.	applicable to livestock farms that produce slurry. Potential benefits would be greatest on sloping ground where the risk of surface run-off is greatest and on soils where drainflow is likely to occur following slurry spreading. The method requires that slurry is stored without any additions of fresh material during the 90-day storage period, otherwise the added slurry would contaminate the stored material with fresh, viable microorganisms. In most cases, this will require more than one store.

28	Adopt batch storage of solid manure	Description: Store solid manure for at least 90 days before spreading on fields and no fresh manure should be added to the heap during this storage period. FIOs die off during storage. There are fewer organisms in the material that is spread and therefore less risk of microorganisms from the manure entering water bodies via surface run-off or percolation through the soil to field drains. Also, the readily available N and total N content of stored farmyard manure will be lower than in the fresh manure, which will lessen the risk of nitrate leaching losses.	applicable to livestock farms that produce solid manure and have only a single store where fresh manure is continuously added to that already present. Potential benefits will be greatest on impermeable soils where the risk of surface run-off is greatest, on drained clay soils with rapid by-pass flow routes to drains and on freely drained soils that are susceptible to nitrate leaching. Storage facilities for solid manures can be constructed relatively simply and cheaply (see Method 32) and there are therefore few limitations to adopting this method. If manure from loose-housed cattle is only removed from the animal house at the end of the winter housing period, a 90-day storage period would restrict its use on some spring-sown crops, e.g. maize.
29	Compost solid manure	Description: encourage the breakdown of solid manures by actively composting the manure heap and turn the solid manure heap twice in the first seven days of composting to facilitate aeration and the development of high temperatures within the heap. This is a source method that uses aerobic microbial metabolism to increase temperatures sufficiently to inactivate pathogens and to reduce the readily available N content of manures.	Applicable to farms with solid manures, particularly in areas where there is a high risk of pathogen transfer to water systems. It can be easily incorporated into normal farm operations using standard farmyard machinery. A degree of education and guidance is necessary in the first few months of operation.
30	Change from slurry to a solid manure handling system	Change from a system where the manure from housed animals is collected as a liquid slurry to one where animals are kept on a bed of straw to produce a solid manure. Solid manures are more easily stored than slurries and present less risk of pollutant loss when they are spread.	applicable to those farms with housed stock that currently handle all or part of their manure as a liquid slurry. It is not applicable to sheep or poultry units as these do not produce slurries. It will be most effective on sloping and less permeable soils where the risk of surface run-off is greatest, on free draining sandy or shallow soils that are prone to nitrate leaching and on drained clay soils where rapid losses can occur in drainflow from wet soils. There will be additional labour requirements associated with spreading straw in the animal house. Solid manure is less easily handled than liquid slurries. It cannot be pumped and cannot be used with umbilical spreading systems.
31	Site solid manure heaps away from watercourses and field drains	Where solid manure is stacked in the field or outside of buildings, the heap should not be sited over field drains or close to a watercourse (i.e. at least 10 m separation). Siting manure heaps away from drains reduces the risk that preferential flow of effluent though the soil might transport N, P and FIOs to field drains or via surface run-off into a watercourse.	applicable to all farms that produce or import solid manure and store it in the field. Benefits are likely to be greatest on heavier soils, where there is a greater risk of surface run-off and where drains are more likely to be present. method is simple to implement with few limitations to its use. However, it will be difficult to find suitable positions for manure heaps on those farms where most fields have a system of closely-spaced drains. The method will provide little additional benefit where Method 32, to site manure heaps on concrete and collect effluent, has already been adopted and properly implemented.
32	Site solid manure heaps on concrete and collect the effluent	When stored outside, manure heaps should be sited on an impermeable concrete base with facilities for collecting the effluent that drains from the heap and the effluent should be spread on the land when there is little risk of it causing pollution. This method prevents seepage and accumulation of high concentrations of soluble N and P in the soil below the heap, which may subsequently be leached to surface and ground waters or flow directly through cracks to field drains. The concrete surface also reduces the area of soil compaction caused by farm machinery during loading and unloading of	applicable to all livestock farms that produce solid manure (and to arable farms that import manure) and currently do not take these precautions. The action will be most effective on heavier soils, where there is a greater risk of surface run-off and where field drains are more likely to be present, and on sandy soils or shallow soils over permeable rocks where the risk of leaching is greatest. The method would be simple to adopt and there are few limitations on where it could be implemented. If the precaution to site manure heaps away from watercourses and drains (Method 31) was already being observed, the additional benefits of this method would be largely confined to reductions in nitrate leaching, as

		manure. Collection of the effluent prevents overland flow from the heap, which could otherwise transport N, P and FIOs to watercourses. The effluent can be spread at a later date when soil conditions are suitable and the nutrient content can be utilised by the crop.	the impact of P and FIO losses in surface run-off would already be minimised.
33	Do not apply manure to high-risk areas	For example, directly adjacent to a watercourse, borehole or road culvert, to shallow soils over fissured rock or cracked soils over field drains, to areas with a dense network of open (surface) drains, or to wet depressions (flushes) draining to a nearby watercourse. High risk areas also include fields with high P index soils (P Index 4 and above) and manure should not be applied to these areas at any time. These are areas where there is a particularly high risk of rapid transport of solutes or suspended material to watercourses and inputs of potential pollutants to these areas should be avoided wherever possible. Losses of P on eroded soil particles and by leaching are greatest on high P index soils. Applying manures to these areas will further increase the excessive P content of the soil and increase the amounts lost.	applicable to all farms applying manures and where these ground conditions occur. These will mainly be livestock farms. Although most hydrologically well-connected areas are likely to be easily identified, some old, but still functioning, drainage networks may not be known to the farmer. Wet areas affected by spring lines are difficult to work and may already be excluded from the agricultural area. On some farms, particularly intensive dairy farms, with a history of high P use and of spreading manures on the same fields, a large proportion of the farm may be classified as having high P index soils and be excluded from receiving further applications. In these circumstances, it may be necessary to export surplus manure to other farms (Method 37).
34	Do not spread farmyard manure to fields at high-risk times	Avoid spreading straw-based FYM to fields at times when there is a high risk of surface run-off or of rainfall causing losses by leaching. There is a risk of pollution if solid manures are spread under conditions where heavy rain could transport N, P and FIOs in surface run-off. As solid manures have a low moisture content compared with slurries, they do not themselves add sufficient water to the soil to initiate surface run-off or preferential flow to field drains. Pollutants will only be transported to watercourses when there is heavy rainfall following the application. Avoiding spreading solid manures at times when these conditions are likely to occur minimises this risk. Fresh FYM has a higher content of readily-available N and FIOs and generally presents a greater risk of pollution than FYM that has been stored for several months.	applicable to livestock farms producing solid manure and to other farms that import fresh solid manure and spread it directly to fields. The risk of run-off is greatest on impermeable soils and on slopes. High-risk times will be most frequent in winter when soils are wet, particularly in high rainfall areas. Provided the farm has some storage for solid manure or can leave it in the animal house until conditions improve, there are few limitations to adopting this method. However, the method may limit opportunities for applying manure before some spring-sown crops.

35	Do not spread slurry or poultry manure to fields at high-risk times	Description: Do not apply slurry or poultry manure to fields at times when there is a high risk of surface run-off; e.g. in winter when soils are saturated or frozen hard or when heavy rain is expected in the next few days. Do not apply slurry or poultry manure to fields at times when there is a high risk of rapid percolation to field drains; e.g. in winter and spring when soils are wet or in summer when soils are dry and cracked over drains. And, do not apply slurry or poultry manure to fields late in the growing season (i.e. autumn/early winter) or when there is no crop to utilise the added N. Slurry and poultry manure have high contents of readily-available N. Avoiding applications of these materials at times when surface run-off or rapid preferential flow to drains is likely to occur reduces the risk of these flows transporting pollutants to watercourses. Avoiding applications in autumn or early winter helps to avoid a build-up of soil nitrate that may be leached over winter.	method is limited to those farms producing animal slurry or importing slurry (including liquid sewage sludge) and those using poultry manure. High-risk times will be most frequent in high rainfall areas and on sloping sites with impermeable soils, on shallow or sandy soils and on artificially drained soils where there are preferential loss pathways. For slurry, this method will only be applicable to those farms that have sufficient storage capacity to allow a choice of when to apply slurry. Over 15% of the farms in a recent survey had little or no storage. Even where storage is adequate for normal conditions, exceptional weather or poor planning can create a situation where stores are full during a high-risk period so that land spreading is the only option. It would generally be acceptable to apply slurry to grass later in the season than for other crops, as long as the sward continued to take up N.
36	Incorporate manure into the soil	using a plough, discs or tines. The rapid soil incorporation of manures can reduce the loss of P and FIOs in surface runoff. This is a mobilisation and delivery method. Incorporation of manure can reduce the detachment and entrainment of manure particles by increasing surface roughness,promoting infiltration and largely preventing the exposure of manure to the hydrological forces of raindrop impact, surface run-off and drain flow losses. Rapid soil incorporation of manure (i.e. within 6 hours of spreading for slurry and 24 hours for solid manures) also reduces the volatilisation of ammonia by reducing the exposure of manure to the air.	Applicable to the arable sector on all soil types and to maize growing in the dairy sector. In most circumstances this method can be carried out as part of normal field preparations, although not commonly within 24 hours of spreading. Where contractors are carrying out the spreading it would require either additional investment in machinery for the agricultural contractor or a degree of co-ordination between farmers and contractors.
37a	Transport manure to neighbouring farms 5km	For farms within the 2002 designated Nitrate Vulnerable Zones (NVZs) where organic manure N loadings averaged over the tillage area exceed 210 kg/ha of total N each year or where they exceed 250 kg/ha over the grassland area, organic manures in excess of this loading must be transported to other farms (or stocking rates must be reduced – see Method 13). In England and Wales from 19 December 2006, the organic manure N limit for tillage land within these NVZs will be reduced to 170 kg total N/ha. Where there is a surplus of nutrients, farm manures can be exported to neighbouring farmland. As a result, farms are able to balance the input of nutrients with the capacity of land to absorb those nutrients. Current regulations concentrate on N, but it is possible to introduce limits on P loading as well.	applicable to the intensive grassland, indoor pig and poultry sectors. In 1996, an estimated 40% of poultry manures, 15% of pig manures and 2% of cattle manures were exported from the unit of production. The method is reasonably easy to implement and enforce since it is based on livestock numbers and recordable vehicle movements. The method is most easily applied where receiving farm holdings are in close proximity (e.g. within 5-20 km) The receiving farms must have the land capacity to absorb the transported organic N (and P) load, and if transport takes place during NVZ Action Programme 'closed periods' they must have sufficient storage
37b	Transport manure to neighbouring farms 20km		
38	Incinerate poultry litter	Transport poultry litter to an incinerator where it is burnt. The manure and the N, P and FIOs it contains are removed from the farm and eliminated as a source of diffuse pollution. Removing the manure from the farm removes the source of pollution.	method is only applicable to poultry litter and some dry layer manures. The moisture content of straw-based farmyard manures is too high for incineration. Applicability will be limited by the availability of suitable incineration facilities within an acceptable distance of broiler and turkey farms.

Farm infrastructure	39	Fence off rivers and streams from livestock	Erect stock-proof fences in grazing fields and on trackways adjoining stream and rivers. Livestock, particularly cattle, can cause severe damage to stream and river banks when attempting to gain access to drinking water. The vegetative cover is destroyed and the soil badly poached, leading to erosion of the bank and increased transport of soil particles and associated P into the watercourse. Livestock also add N, P and FIOs by defecating and urinating directly into the water. Fencing to prevent access to the banks eliminates this source of pollution. Because of the importance of surface flows in transporting P and FIOs, this method has a greater impact on losses of these pollutants than for nitrate.	applicable to farms with grazing livestock and to all soil types. Benefits will be greatest on intensively stocked farms, particularly those with cattle. The method may be less feasible on some upland beef/sheep farms with extensive areas of rough grazing and considerable lengths of unfenced stream banks. Fortunately, pollutant inputs to these streams are likely to be smaller than on more intensively stocked farms. There may also be a need to provide an alternative source of drinking water.
	40	Construct bridges for livestock crossing rivers and streams	allows livestock to cross rivers and streams without damaging the banks and to prevent animals urinating and defecating directly into the water. Where livestock ford rivers and streams, they can erode banks, disturb the stream bed and increase inputs of sediment to the watercourse. Stock can also add pollutants directly by urinating and defecating into the water. Provision of bridges removes the need for fording watercourses and eliminates this source of pollution.	applicable to livestock farms where there are stream crossings without bridges. It is particularly applicable to dairy farms where cows are typically moved between the fields and dairy twice a day. This method will only be effective when combined with Method 39, to fence off other areas of river and stream bank from livestock. There are few circumstances that would limit the adoption of this method. It may be impractical on some upland farms with extensive areas of rough grazing and many streams and crossing points.
	41	Re-site gateways away from high-risk areas	Move gateways located in high-risk areas, such as at the bottom of a slope and near to a watercourse, to lower risk areas on upper slopes.Many fields have gateways located at the bottom of a slope and near to a watercourse. Increased activity occurs around gateways, including trampling by livestock (particularly on dairy farms) and compaction by machinery. Repositioning the gateway would decrease the loss of P associated with sediment losses and of FIOs from grass fields, by reducing hydrological connectivity. There would be minimal effect on nitrate losses.	potentially applicable to all farming systems in sloping areas and is relatively easy to implement. Re-locating gates from high risk to lower risk areas should be practicable on most fields in sloping areas. Farmers may be reluctant to re-locate gateways but if it could improveopportunities for access then it may be seen as being advantageous, particularly in wet years. Practicability will be reduced where new tracks have to be constructed in addition to the new gateways. Re-siting gateways is compatible with the Environmental Stewardship Scheme and thereare no conflicts with other methods.
	42	Establish new hedges	Lay new hedges along fence lines and use them to re-create old field patterns that serve to break the hydrological connectivity of the landscape into its constituent parts. Increasing the number of hedgerows can help to reduce P and FIO losses by trapping sediments and lowering run-off volumes (breaking up slopes). Hedges can also help to protect soils from wind erosion. Hedges also act as ‘natural’ buffer strips and sediment traps.	most applicable to the arable sectors where hedgerows have been removed but is potentially applicable to all farming systems. There is great potential for this method in areas with complex soil or landscape patterns, particularly on erosion susceptible sandy and silty soils. Planting hedges and making fields smaller will increase the time required for field operations and would be resisted by many larger arable farms. On grassland farms it may help with stock management and provide useful shelter in summer but considerable investment and time is involved in establishing the hedgerows. On most farms the laying of hedges would have to be carried out over a number of years to fit in with current farming operations. It is compatible with the Environmental Stewardship Scheme and there would be no conflicts with other methods.

43	Establish riparian buffer strips	Establish a vegetated and unfertilised grass/woodland buffer strip alongside watercourses. The grass/woodland strip will act as a 'natural' buffer feature to reduce the transfer of P and FIOs from agricultural land to water. Riparian buffer strips can reduce pollution delivery in two ways. They distance agricultural activity from the riparian area and therefore reduce direct pollution from inorganic fertiliser and organic manure additions. They are also used to intercept overland flow from agricultural land just before it reaches the watercourse. Buffer strips can therefore act as a sediment trap, as well as helping to reduce nutrient transfers.	most effective at retaining sediment when overland flow is shallow and slow, therefore they are particularly suited to low-lying and gently undulating landscapes where the topography does not concentrate the flow into channels. They are potentially applicable to all farming systems. This measure requires a certain amount of investment to establish but once established require little maintenance. Farmers may have issues related to controlling weeds from the strips but the impact is less than from in-field grass buffer strips as it is usually the less productive land that is lost. They are compatible with the Environmental Stewardship Scheme and there is no conflict with other methods.
44	Establish and maintain artificial (constructed) wetlands	Construct or establish wetlands with fences and channels that will be sufficient to capture run-off and sediment from a field or group of fields or farm hard-standings that regularly discharge run-off water and sediment. Constructed wetlands are used for the 'treatment' of wastewater generated from farm hard-standing areas and to intercept run-off water from a field or group of fields. They can trap sediment and through the retention of run-off, reduce N, P (soluble and particulate) and FIO loads in water exiting the wetland. Wetlands act by intercepting pollutant delivery, providing a 'buffer zone' and can potentially clean up polluted water. They can be natural or artificial, permanent or temporary, with water that is static or flowing, fresh or brackish. Constructed flow wetlands can be either surface (overland) flow or subsurface (percolation) flow systems.	Wetlands can potentially be applied to all farming systems on soils with moderate to poor drainage, but are particularly applicable in the intensive livestock and arable sectors. They are not effective on free-draining soils, where drainage water moves to the groundwater. Their construction is compatible with the Environmental Stewardship Scheme and there is no conflict with other methods. Artificial wetlands are difficult and expensive to implement as a pollution control method. Their construction will inevitably involve the loss of some agricultural land. However, where they can be used to address a potential pollution problem they are reasonably acceptable to farmers. The outflow of water from artificial wetlands into a watercourse may require a discharge consent from the Environment Agency. There will also be a need to obtain Environment Agency approval if the wetland is being used to treat farm hard-standings run-off. Constructed, subsurface flow systems require maintenance due to deposition of sediment and organic matter that can result in some sections becoming impermeable. Current experience from wastewater treatment suggests that action is required every 5-7 years.

Annex III- BMPs cost effectiveness indicators per farm type

Table III.1 Cost-effectiveness indicators for BMPs to reduce agricultural P loads (£/Kg P removed per ha or the equivalent £/% reduction) for Arable farms

ARABLE

Sandy loam soil

ID	Category	BMP Measures	targeted pollutant N	P	FIO	Abatement Costs* £/ha	Reduction in pollutant loss Total P (KG P/ha) Baseline loss: 0.3	Cost-Effectiveness (£/% Reduction in P loss per ha)	Cumulative % P Loss at farm level per ha Inc. Compounding	ALTERNATIVE REDUCTIONS (BENEFITS) Reduction in pollutant loss Nitrate (KG N/ha) Baseline loss: 51	Cumulative % N Loss at farm level per ha Inc. Compounding
43	Farm infrastructure	Establish riparian buffer strips	x	x		51.73	0.1	1.55	33.33	1.5	2.94
2a	Soil Managament	Establish cover crops in the autumn: cereals	x	x		19.68	0.03	1.97	40.00	7	16.26
41	Farm infrastructure	Re-site gateways away from high-risk areas		x		9.50	0.01	2.85	42.00	0	16.26
6	Soil Managament	Cultivate and drill across the slope		x		23.62	0.02	3.54	45.87	0	16.26
5	Soil Managament	Cultivate compacted tillage soils		x		31.50	0.02	4.72	49.48	1	17.90
42	Farm infrastructure	Establish new hedges		x		100.00	0.06	5.00	59.58	0	17.90
8	Soil Managament	Avoid tramlines over winter		x		35.43	0.02	5.31	62.28	0	17.90
9	Soil Managament	Establish in-field grass buffer strips	x	x	x	248.82	0.14	5.33	79.88	4.9	25.79
3	Soil Managament	Cultivate land for crop establishment in spring rather than autumn	x	x		86.61	0.02	12.99	81.22	1	27.25
2b	Soil Managament	Establish cover crops in the autumn: other crops	x	x		133.86	0.03	13.39	83.10	7	37.23
1a	Land Use	Convert arable land to extensive grassland: conversion to ungrazed grassland	x	x		917.53	0.18	15.29	93.24	49	97.54
1b	Land Use	Convert arable land to extensive grassland: conversion to extensive grazing	x	x	x	951.00	0.15	19.02	96.62	31	99.03
7	Soil Managament	Leave autumn seedbeds rough		x		314.96	0.02	47.24	96.85	0	99.03

*NPV/ha over an 8 year period (up to 2015). Discount rate 3.5%

Clay loam soil

ID	Category	BMP Measures	targeted pollutant N	P	FIO	Abatement Costs* £/ha	Reduction in pollutant loss Total P (KG P/ha) Baseline loss: 2.3	Cost-Effectiveness (£/% Reduction in P loss per ha)	Cumulative % P Loss at farm level per ha Inc. Compounding	ALTERNATIVE REDUCTIONS (BENEFITS) Reduction in pollutant loss Nitrate (KG N/ha) Baseline loss: 47	Cumulative % N Loss at farm level per ha Inc. Compounding
4	Soil Managament	Adopt minimal cultivation systems	x	x		-314.96	0.11	-65.85	4.78	2.5	5.32
22	Fertiliser Management	Do not apply P fertilisers to high P index soils		x		-7.87	0.03	-6.04	6.02	0	5.32
44	Farm infrastructure	Establish and maintain artificial (constructed) wetlands	x	x		120.30	0.92	3.01	43.61	10	25.46
2a	Soil Managament	Establish cover crops in the autumn: cereals	x	x		19.68	0.13	3.48	46.80	7	36.57
41	Farm infrastructure	Re-site gateways away from high-risk areas		x		9.50	0.06	3.64	48.19	0	36.57
6	Soil Managament	Cultivate and drill across the slope		x		23.62	0.11	4.94	50.67	0	36.57
5	Soil Managament	Cultivate compacted tillage soils		x		31.50	0.11	6.59	53.03	0	36.57
8	Soil Managament	Avoid tramlines over winter		x		35.43	0.11	7.41	55.27	0	36.57
19	Fertiliser Management	Use a fertiliser recommendation system	x	x		15.75	0.03	12.07	55.86	2	39.26
21b	Fertiliser Management	Reduce fertiliser application rates: 20% reduction in P		x		18.11	0.03	13.88	56.43	0	39.26
3	Soil Managament	Cultivate land for crop establishment in spring rather than autumn	x	x		86.61	0.11	18.11	58.52	1	40.56
1a	Land Use	Convert arable land to extensive grassland: conversion to ungrazed grassland	x	x		917.53	1.16	18.19	79.44	45	97.47
1b	Land Use	Convert arable land to extensive grassland: conversion to extensive grazing	x	x	x	951.00	0.97	22.55	88.11	27	98.92
2b	Soil Managament	Establish cover crops in the autumn: other crops	x	x		133.86	0.13	23.68	88.78	7	99.08
43	Farm infrastructure	Establish riparian buffer strips	x	x		51.73	0.05	23.80	89.03	1	99.10
42	Farm infrastructure	Establish new hedges		x		100.00	0.08	28.75	89.41	0	99.10
23	Fertiliser Management	Do not apply fertiliser to high risk areas	x	x		62.99	0.05	28.98	89.64	0.1	99.11
24	Fertiliser Management	Avoid spreading fertiliser to fields at high-risk times	x	x		49.00	0.03	37.57	89.77	0.1	99.11
9	Soil Managament	Establish in-field grass buffer strips	x	x	x	248.82	0.09	63.59	90.17	4.5	99.19
7	Soil Managament	Leave autumn seedbeds rough		x		314.96	0.11	65.85	90.64	0	99.19
12	Soil Managament	Allow field drainage systems to deteriorate	x	x		445.17	0.01	1023.88	90.68	1.5	99.22

*NPV/ha over an 8 year period (up to 2015). Discount rate 3.5%

Table III.2 Cost-effectiveness indicators for BMPs to reduce agricultural P loads (£/Kg P removed per ha or the equivalent £/% reduction) for Arable plus Manure farms

ARABLE PLUS MANURE

Sandy loam soil

ID	Category	BMP Measures	targeted pollutant			Abatement Costs*	Reduction in pollutant loss Total P (KG P/ha)	Cost-Effectiveness (£/% Reduction in P loss per ha)	Cumulative % P Loss at farm level per ha	ALTERNATIVE REDUCTIONS (BENEFITS) Reduction in pollutant loss Nitrate (KG N/ha)	ALTERNATIVE REDUCTIONS (BENEFITS) Cumulative % N Loss at farm level per ha
			N	P	FIO	£/ha	Baseline loss: 0.4		Inc. Compounding	Baseline loss: 57	Inc. Compounding
43	Farm infrastructure	Establish riparian buffer strips	x	x		51.73	0.11	1.88	27.50	1.5	2.63
2a	Soil Managament	Establish cover crops in the autumn: cereals	x	x		19.68	0.03	2.62	32.94	11	21.42
41	Farm infrastructure	Re-site gateways away from high-risk areas		x		9.50	0.01	3.80	34.61	0	21.42
6	Soil Managament	Cultivate and drill across the slope		x		23.62	0.02	4.72	37.88	0	21.42
5	Soil Managament	Cultivate compacted tillage soils		x		31.50	0.02	6.30	40.99	0	21.42
34	Manure Management	Do not spread farmyard manure to fields at high-risk times	x	x		15.75	0.01	6.30	42.46	1	22.80
42	Farm infrastructure	Establish new hedges		x		100.00	0.06	6.67	51.09	0	22.80
8	Soil Managament	Avoid tramlines over winter		x		35.43	0.02	7.09	53.54	0	22.80
9	Soil Managament	Establish in-field grass buffer strips	x	x		248.82	0.14	7.11	69.80	5.5	30.25
3	Soil Managament	Cultivate land for crop establishment in spring rather than autumn	x	x		86.61	0.02	17.32	71.31	1.5	32.09
2b	Soil Managament	Establish cover crops in the autumn: other crops	x	x		133.86	0.03	17.85	73.46	11	45.19
1a	Land Use	Convert arable land to extensive grassland: conversion to ungrazed grassland	x	x		917.53	0.18	20.39	85.40	55	98.08
1b	Land Use	Convert arable land to extensive grassland: conversion to extensive grazing	x	x	x	951.00	0.15	25.36	90.88	37	99.33
7	Soil Managament	Leave autumn seedbeds rough		x		314.96	0.02	62.99	91.33	0	99.33

*NPV/ha over an 8 year period (up to 2015). Discount rate 3.5%

Clay loam soil

ID	Category	BMP Measures	targeted pollutant			Abatement Costs*	Reduction in pollutant loss Total P (KG P/ha)	Cost-Effectiveness (£/% Reduction in P loss per ha)	Cumulative % P Loss at farm level per ha	ALTERNATIVE REDUCTIONS (BENEFITS) Reduction in pollutant loss Nitrate (KG N/ha)	ALTERNATIVE REDUCTIONS (BENEFITS) Cumulative % N Loss at farm level per ha
			N	P	FIO	£/ha	Baseline loss: 2.5		Inc. Compounding	Baseline loss: 51	Inc. Compounding
4	Soil Managament	Adopt minimal cultivation systems	x	x		-314.96	0.11	-71.58	4.40	3.5	6.86
20	Fertiliser Management	Integrate fertiliser and manure nutrient supply	x	x		-47.24	0.02	-59.05	5.16	1.5	9.60
22	Fertiliser Management	Do not apply P fertilisers to high P index soils		x		-7.87	0.03	-6.56	6.30	0	9.60
36	Manure Management	Incorporate manure into the soil	x	x		0.00	0.02	0.00	7.05	-1	7.83
44	Farm infrastructure	Establish and maintain artificial (constructed) wetlands	x	x		120.30	0.98	3.07	43.49	12	29.52
2a	Soil Managament	Establish cover crops in the autumn: cereals	x	x		19.68	0.13	3.79	46.43	11	44.72
41	Farm infrastructure	Re-site gateways away from high-risk areas		x		9.50	0.06	3.96	47.71	0	44.72
6	Soil Managament	Cultivate and drill across the slope		x		23.62	0.11	5.37	50.01	0	44.72
35	Manure Management	Do not spread slurry or poultry manure to fields at high-risk times	x	x		15.75	0.06	6.56	51.21	2	46.89
5	Soil Managament	Cultivate compacted tillage soils		x		31.50	0.11	7.16	53.36	0	46.89
8	Soil Managament	Avoid tramlines over winter		x		35.43	0.11	8.05	55.41	0	46.89
19	Fertiliser Management	Use a fertiliser recommendation system	x	x		15.75	0.03	13.12	55.95	1.5	48.45
33	Manure Management	Do not apply manure to high-risk areas	x	x		15.75	0.03	13.12	56.48	0.5	48.95
34	Manure Management	Do not spread farmyard manure to fields at high-risk times	x	x		15.75	0.03	13.12	57.00	0.5	49.45
21b	Fertiliser Management	Reduce fertiliser application rates: 20% reduction in P		x		18.11	0.03	15.09	57.51	0	49.45
1a	Land Use	Convert arable land to extensive grassland: conversion to ungrazed grassland	x	x		917.53	1.24	18.50	78.59	49	98.02
3	Soil Managament	Cultivate land for crop establishment in spring rather than autumn	x	x		86.61	0.11	19.68	79.53	1.5	98.08
43	Farm infrastructure	Establish riparian buffer strips	x	x		51.73	0.06	21.55	80.02	1	98.11
1b	Land Use	Convert arable land to extensive grassland: conversion to extensive grazing	x	x	x	951.00	1.03	23.08	88.25	31	99.26
2b	Soil Managament	Establish cover crops in the autumn: other crops	x	x		133.86	0.13	25.74	88.86	11	99.42
42	Farm infrastructure	Establish new hedges		x		100.00	0.08	31.25	89.22	0	99.42
23	Fertiliser Management	Do not apply fertiliser to high risk areas	x	x		62.99	0.05	31.50	89.43	0.1	99.42
31	Manure Management	Site solid manure heaps away from watercourses and field drains	x	x	x	15.75	0.01	39.37	89.48	0.2	99.42
24	Fertiliser Management	Avoid spreading fertiliser to fields at high-risk times	x	x		49.00	0.03	40.83	89.60	0.1	99.42
9	Soil Managament	Establish in-field grass buffer strips	x	x		248.82	0.1	62.20	90.02	4.9	99.48
7	Soil Managament	Leave autumn seedbeds rough		x		314.96	0.11	71.58	90.46	0	99.48
32	Manure Management	Site solid manure heaps on concrete and collect the effluent	x	x		32.60	0.01	81.50	90.50	0.2	99.48
12	Soil Managament	Allow field drainage systems to deteriorate	x	x		445.17	0.01	1112.92	90.53	2	99.50

*NPV/ha over an 8 year period (up to 2015). Discount rate 3.5%

Table III.3 Cost-effectiveness indicators for BMPs to reduce agricultural P loads (£/Kg P removed per ha or the equivalent £/% reduction) for Dairy farms

DAIRY

Sandy loam soil

ID	Category	BMP Measures	targeted pollutant N	P	FIO	Abatement Costs*	Reduction in pollutant loss Total P (KG P/ha)	Cost-Effectiveness (£/% Reduction in P loss per ha)	Cumulative % P Loss at farm level per ha	ALTERNATIVE REDUCTIONS (BENEFITS) Reduction in pollutant loss Nitrate (KG N/ha)	Cumulative % N Loss at farm level per ha	% Reduction in pollutant loss FIO per ha	Cumulative % FIO Loss at farm level per ha
						£/ha	Baseline loss: 0.2		Inc. Compounding	Baseline loss: 61	Inc. Compounding		Inc. Compounding
20	Fertiliser Management	Integrate fertiliser and manure nutrient supply	x	x		-94.49	0.01	-18.90	5.00	4.5	7.38	0	0.00
35	Manure Management	Do not spread slurry or poultry manure to fields at high-risk times	x	x	x	15.75	0.04	0.79	24.00	7	18.01	10	10.00
33	Manure Management	Do not apply manure to high-risk areas	x	x	x	15.75	0.02	1.57	31.60	1.5	20.02	0	10.00
43	Farm infrastructure	Establish riparian buffer strips	x	x	x	64.00	0.07	1.83	55.54	1.5	21.99	10	19.00
41	Farm infrastructure	Re-site gateways away from high-risk areas		x		9.50	0.01	1.90	57.76	0	21.99	0	19.00
42	Farm infrastructure	Establish new hedges		x		100.00	0.04	5.00	66.21	0	21.99	0	19.00
16	Livestock Management	Move feed and water troughs at regular intervals	x	x	x	88.98	0.03	5.93	71.28	1	23.27	10	27.10
25	Manure Management	Increase the capacity of farm manure (slurry) stores	x	x	x	142.00	0.04	7.10	77.02	5	29.56	20	41.68
30	Manure Management	Change from slurry to a solid manure handling system	x	x	x	470.00	0.08	11.75	86.21	7	37.64	40	65.01
37a	Manure Management	Transport manure to neighbouring farms 5km	x	x	x	685.03	0.1	13.70	93.11	6	43.77	20	72.01
39	Farm infrastructure	Fence off rivers and streams from livestock	x	x	x	76.80	0.01	15.36	93.45	0.5	44.24	10	74.81
10	Soil Managament	Loosen compacted soil layers in grassland fields		x		85.04	0.01	17.01	93.78	0	44.24	0	74.81
18	Livestock Management	Adopt phase feeding of livestock	x	x		128.35	0.01	25.67	94.09	3	46.98	0	74.81
37b	Manure Management	Transport manure to neighbouring farms 20km	x	x	x	1421.25	0.1	28.42	97.05	6	52.19	20	79.84
15	Livestock Management	Reduce field stocking rates when soils are wet	x	x	x	413.38	0.02	41.34	97.34	0	52.19	10	81.86
17	Livestock Management	Reduce dietary N and P intakes	x	x		334.64	0.01	66.93	97.47	3	54.54	0	81.86
13	Livestock Management	Reduce overall stocking rates on livestock farms	x	x	x	5060.18	0.08	126.50	98.48	25	73.17	50	90.93

*NPV/ha over an 8 year period (up to 2015). Discount rate 3.5%

Clay loam soil

ID	Category	BMP Measures	targeted pollutant N	P	FIO	Abatement Costs*	Reduction in pollutant loss Total P (KG P/ha)	Cost-Effectiveness (£/% Reduction in P loss per ha)	Cumulative % P Loss at farm level per ha	ALTERNATIVE REDUCTIONS (BENEFITS) Reduction in pollutant loss Nitrate (KG N/ha)	Cumulative % N Loss at farm level per ha	% Reduction in pollutant loss FIO per ha	Cumulative % FIO Loss at farm level per ha
						£/ha	Baseline loss: 2.8		Inc. Compounding	Baseline loss: 34	Inc. Compounding		Inc. Compounding
20	Fertiliser Management	Integrate fertiliser and manure nutrient supply	x	x		-94.49	0.09	-29.40	3.21	2	5.88	0	0.00
21b	Fertiliser Management	Reduce fertiliser application rates (reduction in P by a 20%)		x		-15.75	0.06	-7.35	5.29	0	5.88	0	0.00
22	Fertiliser Management	Do not apply P fertilisers to high P index soils		x		-7.87	0.06	-3.67	7.32	0	5.88	0	0.00
35	Manure Management	Do not spread slurry or poultry manure to fields at high-risk times	x	x	x	15.75	0.69	0.64	30.16	5	19.72	10	10.00
33	Manure Management	Do not apply manure to high-risk areas	x	x	x	15.75	0.19	2.32	34.90	0.5	20.90	10	19.00
44	Farm infrastructure	Establish and maintain artificial (constructed) wetlands	x	x	x	120.30	1.12	3.01	60.94	12	48.82	20	35.20
41	Farm infrastructure	Re-site gateways away from high-risk areas		x		9.50	0.07	3.80	61.91	0	48.82	0	35.20
16	Livestock Management	Move feed and water troughs at regular intervals	x	x	x	88.98	0.37	6.73	66.95	0.5	49.57	10	41.68
19	Fertiliser Management	Use a fertiliser recommendation system	x	x		15.75	0.06	7.35	67.66	2	52.54	0	41.68
25	Manure Management	Increase the capacity of farm manure (slurry) stores	x	x	x	142.00	0.49	8.11	73.32	5	59.52	20	53.34
37a	Manure Management	Transport manure to neighbouring farms 5km	x	x	x	685.03	1.16	16.54	84.37	3	63.09	20	62.68
23	Fertiliser Management	Do not apply fertiliser to high risk areas	x	x		55.12	0.08	19.29	84.82	0.1	63.20	0	62.68
18	Livestock Management	Adopt phase feeding of livestock	x	x		128.35	0.17	21.14	85.74	2	65.36	0	62.68
43	Farm infrastructure	Establish riparian buffer strips	x	x	x	64.00	0.06	29.87	86.04	1	66.38	10	66.41
42	Farm infrastructure	Establish new hedges		x		100.00	0.09	31.11	86.49	0	66.38	0	66.41
30	Manure Management	Change from slurry to a solid manure handling system	x	x	x	470.00	0.39	33.74	88.37	5	71.33	40	79.84
37b	Manure Management	Transport manure to neighbouring farms 20km	x	x	x	1421.25	1.16	34.31	93.19	3	73.86	20	83.88
39	Farm infrastructure	Fence off rivers and streams from livestock	x	x	x	76.80	0.06	35.84	93.34	0.5	74.24	10	85.49
15	Livestock Management	Reduce field stocking rates when soils are wet	x	x	x	413.38	0.25	46.30	93.93	0.5	74.62	10	86.94
36	Manure Management	Incorporate manure into the soil		x		52.76	0.03	49.24	94.00	0	74.62	0	86.94
17	Livestock Management	Reduce dietary N and P intakes	x	x		334.64	0.17	55.12	94.36	2	76.11	0	86.94
24	Fertiliser Management	Avoid spreading fertiliser to fields at high-risk times	x	x		153.62	0.04	107.54	94.44	0.1	76.18	0	86.94
10	Soil Managament	Loosen compacted soil layers in grassland fields		x		85.04	0.02	119.05	94.48	0	76.18	0	86.94
13	Livestock Management	Reduce overall stocking rates on livestock farms	x	x	x	5060.18	0.98	144.58	96.41	10	83.19	50	93.47
26	Manure Management	Minimise the volume of dirty water produced	x	x	x	102.00	0.01	285.60	96.43	0.1	83.24	10	94.12
14	Livestock Management	Reduce the length of the grazing day or grazing season	x	x	x	413.38	0.03	385.82	96.46	5	85.70	10	94.71
12	Soil Managament	Allow field drainage systems to deteriorate	x	x		499.27	0.01	1397.96	96.48	1	86.12	0	94.71

*NPV/ha over an 8 year period (up to 2015). Discount rate 3.5%

Table III.4 Cost-effectiveness indicators for BMPs to reduce agricultural P loads (£/Kg P removed per ha or the equivalent £/% reduction) for Beef farms

BEEF

Sandy loam soil

ID	Category	BMP Measures	targeted pollutant N	P	FIO	Abatement Costs* £/ha	Reduction in pollutant loss Total P (KG P/ha) Baseline loss: 0.2	Cost-Effectiveness (£/% Reduction in P loss per ha)	Cumulative % P Loss at farm level per ha Inc. Compounding	ALTERNATIVE REDUCTIONS (BENEFITS) Reduction in pollutant loss Nitrate (KG N/ha) Baseline loss: 18	Cumulative % N Loss at farm level per ha Inc. Compounding	% Reduction in pollutant loss FIO per ha	Cumulative % FIO Loss at farm level per ha Inc. Compounding
34	Manure Management	Do not spread farmyard manure to fields at high-risk times	x	x		15.75	0.07	0.45	35.00	0.3	1.67	0	0.00
43	Farm infrastructure	Establish riparian buffer strips	x	x	x	76.80	0.06	2.56	54.50	0.5	4.40	10	10.00
37a	Manure Management	Transport manure to neighbouring farms 5km	x	x		110.24	0.07	3.15	70.43	2	15.02	0	10.00
31	Manure Management	Site solid manure heaps away from watercourses and field drains	x	x	x	15.75	0.01	3.15	71.90	0	15.02	0	10.00
33	Manure Management	Do not apply manure to high-risk areas	x	x		15.75	0.01	3.15	73.31	0.2	15.96	0	10.00
16	Livestock Management	Move feed and water troughs at regular intervals	x	x	x	81.89	0.03	5.46	77.31	0.5	18.30	10	19.00
37b	Manure Management	Transport manure to neighbouring farms 20km	x	x		228.34	0.07	6.52	85.25	2	27.38	0	19.00
32	Manure Management	Site solid manure heaps on concrete and collect the effluent	x	x	x	68.60	0.01	13.72	85.99	0.1	27.78	10	27.10
10	Soil Managament	Loosen compacted soil layers in grassland fields		x		85.04	0.01	17.01	86.69	0	27.78	0	27.10
15	Livestock Management	Reduce field stocking rates when soils are wet	x	x	x	255.90	0.02	25.59	88.02	0	27.78	10	34.39
14	Livestock Management	Reduce the length of the grazing day or grazing season	x	x	x	255.90	0.01	51.18	88.62	1.5	33.80	10	40.95
13	Livestock Management	Reduce overall stocking rates on livestock farms	x	x	x	1999.95	0.06	66.67	92.03	4	48.51	50	70.48

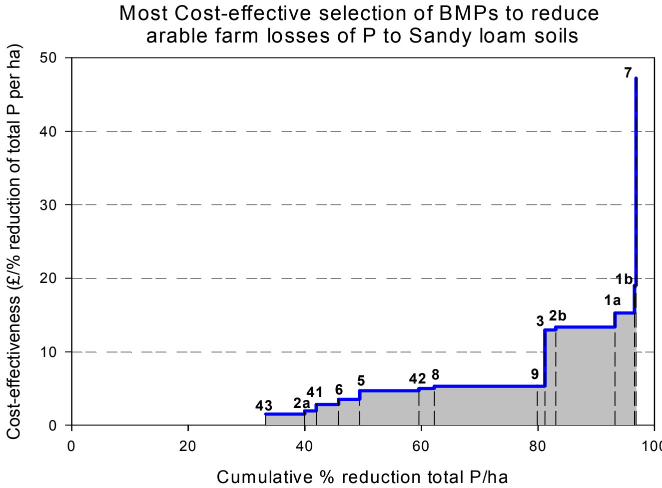
*NPV/ha over an 8 year period (up to 2015). Discount rate 3.5%

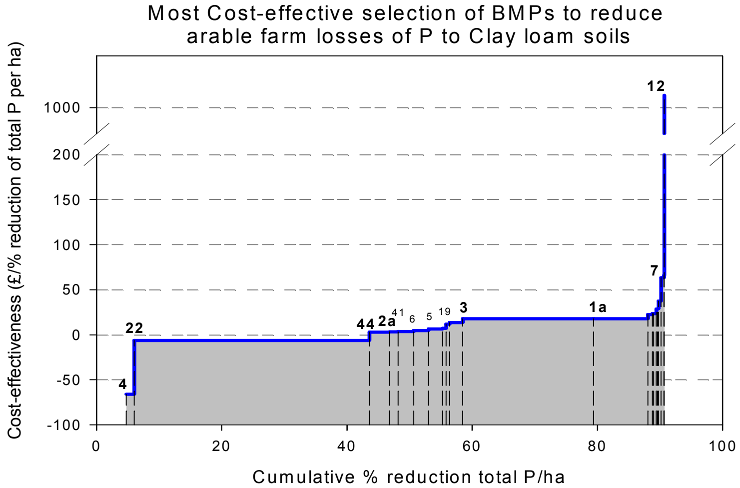
Clay loam soil

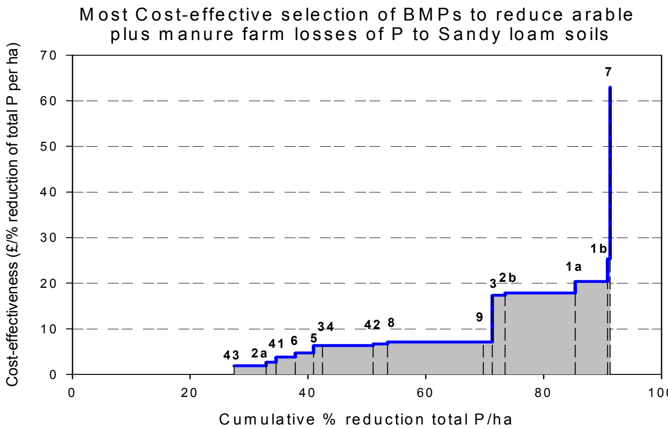
ID	Category	BMP Measures	targeted pollutant N	P	FIO	Abatement Costs* £/ha	Reduction in pollutant loss Total P (KG P/ha) Baseline loss: 1	Cost-Effectiveness (£/% Reduction in P loss per ha)	Cumulative % P Loss at farm level per ha Inc. Compounding	ALTERNATIVE REDUCTIONS (BENEFITS) Reduction in pollutant loss Nitrate (KG N/ha) Baseline loss: 12	Cumulative % N Loss at farm level per ha Inc. Compounding	% Reduction in pollutant loss FIO per ha	Cumulative % FIO Loss at farm level per ha Inc. Compounding
20	Fertiliser Management	Integrate fertiliser and manure nutrient supply	x	x		-47.24	0.001	-472.44	0.10	0.3	2.50	0	0.00
21b	Fertiliser Management	Reduce fertiliser application rates; 20% Reduction P		x		-15.75	0.03	-5.25	3.10	0	2.50	0	0.00
22	Fertiliser Management	Do not apply P fertilisers to high P index soils		x		-7.87	0.03	-2.62	6.00	0	2.50	0	0.00
34	Manure Management	Do not spread farmyard manure to fields at high-risk times	x	x		15.75	0.16	0.98	21.04	0.1	3.31	0	0.00
33	Manure Management	Do not apply manure to high-risk areas	x	x		15.75	0.07	2.25	26.57	0.1	4.12	0	0.00
37a	Manure Management	Transport manure to neighbouring farms 5km	x	x		110.24	0.41	2.69	56.68	1	12.11	0	0.00
44	Farm infrastructure	Establish and maintain artificial (constructed) wetlands	x	x	x	120.30	0.42	2.86	74.87	10	85.35	20	20.00
41	Farm infrastructure	Re-site gateways away from high-risk areas		x		9.50	0.03	3.17	75.63	0	85.35	0	20.00
19	Fertiliser Management	Use a fertiliser recommendation system	x	x		15.75	0.03	5.25	76.36	0.3	85.72	0	20.00
31	Manure Management	Site solid manure heaps away from watercourses and field drains	x	x	x	15.75	0.03	5.25	77.07	0.1	85.84	10	28.00
37b	Manure Management	Transport manure to neighbouring farms 20km	x	x		228.34	0.41	5.57	86.47	1	87.02	0	28.00
16	Livestock Management	Move feed and water troughs at regular intervals	x	x	x	81.89	0.14	5.85	88.36	0.2	87.23	10	35.20
23	Fertiliser Management	Do not apply fertiliser to high risk areas	x	x		31.50	0.04	7.87	88.83	0.1	87.34	0	35.20
32	Manure Management	Site solid manure heaps on concrete and collect the effluent	x	x	x	68.60	0.03	22.87	89.16	0.1	87.45	10	41.68
24	Fertiliser Management	Avoid spreading fertiliser to fields at high-risk times	x	x		54.72	0.02	27.36	89.38	0.1	87.55	0	41.68
15	Livestock Management	Reduce field stocking rates when soils are wet	x	x	x	255.90	0.09	28.43	90.34	1.2	88.79	10	47.51
39	Farm infrastructure	Fence off rivers and streams from livestock	x	x	x	76.80	0.02	38.40	90.53	0.2	88.98	10	52.76
43	Farm infrastructure	Establish riparian buffer strips	x	x	x	76.80	0.02	38.40	90.72	0.2	89.17	10	57.48
13	Livestock Management	Reduce overall stocking rates on livestock farms	x	x	x	1999.95	0.37	54.05	94.15	2	90.97	50	78.74
10	Soil Managament	Loosen compacted soil layers in grassland fields		x		85.04	0.01	85.04	94.21	0	90.97	0	78.74
12	Soil Managament	Allow field drainage systems to deteriorate	x	x		177.85	0.01	177.85	94.27	0.5	91.35	0	78.74
14	Livestock Management	Reduce the length of the grazing day or grazing season	x	x	x	255.90	0.01	255.90	94.33	0.5	91.71	10	80.87

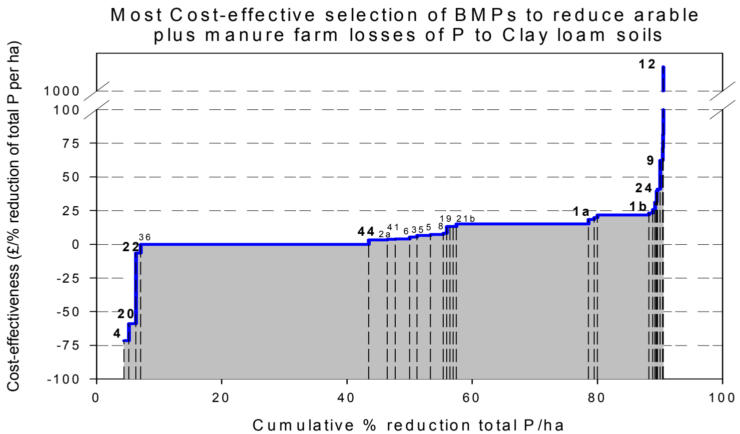
*NPV/ha over an 8 year period (up to 2015). Discount rate 3.5%

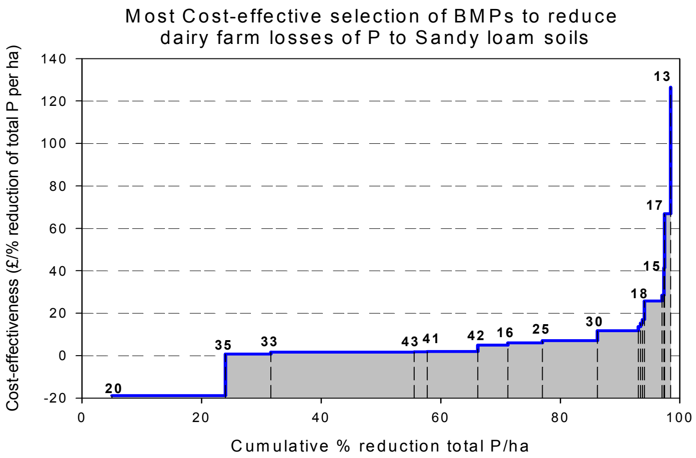
Annex IV- CEA graphs for the selection of measures

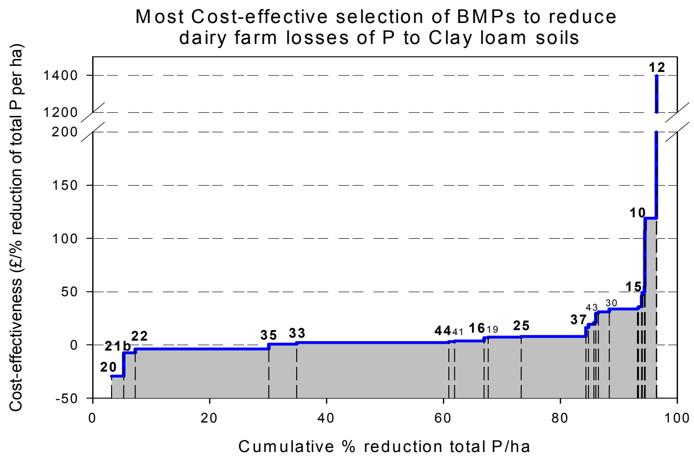


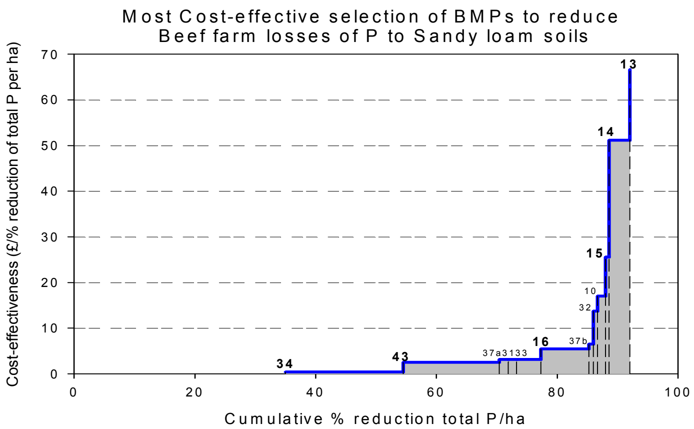


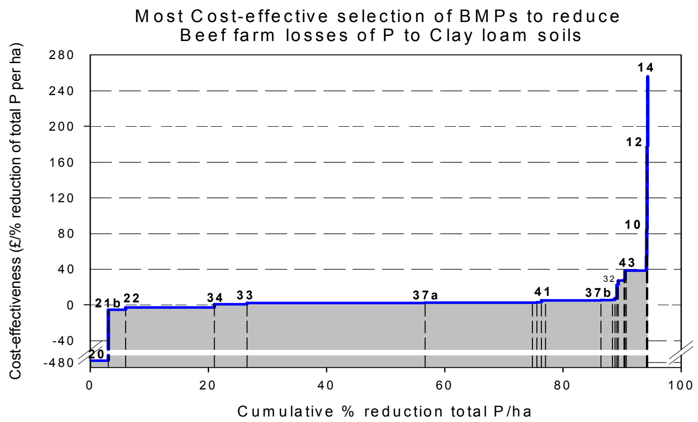





Most Cost-effective selection of BMPs to reduce
dairy farm losses of P to Sandy loam soils
Cost-effectiveness (£/% reduction of total P per ha)
Cumulative % reduction total P/ha
20
35
33
43
41
42
16
25
30
18
15
17
13


Most Cost-effective selection of BMPs to reduce
dairy farm losses of P to Clay loam soils
Cost-effectiveness (£/% reduction of total P per ha)
Cumulative % reduction total P/ha
20
21b
22
35
33
44
41
16
19
25
37
43
30
15
10
12


Most Cost-effective selection of BMPs to reduce
Beef farm losses of P to Sandy loam soils
Cost-effectiveness (£/% reduction of total P per ha)
Cumulative % reduction total P/ha
34
43
37a3133
16
37b
32
10
15
14
13


Most Cost-effective selection of BMPs to reduce
Beef farm losses of P to Clay loam soils
Cost-effectiveness (£/% reduction of total P per ha)
Cumulative % reduction total P/ha
20
21b
22
34
33
37a
41
37b
32
43
10
12
14

Annex V Criteria developed by Melichar (1985) for the classification of farms by financial position[1]

If Debt/Asset ratio is:	And if Return on Assets is:	And If Return on Equity Is:	Then Financial Position is:
Operators with Equity under $50,000			
Under 40	Above 0	N/A	Good
40 to 70	Above 5	N/A	Good
Over 70	Above 15	N/A	Good
Under 40	-5 to 0	N/A	Fair
40 to 70	0 to 5	N/A	Fair
Over 70	5 to 15	N/A	Fair
Under 40	-15 to -5	N/A	Stressed
40 to 70	-5 to 0	N/A	Stressed
Over 70	0 to 5	N/A	Stressed
Under 40	Under -15	N/A	Vulnerable
40 to 70	Under -5	N/A	Vulnerable
Over 70	Under 0	N/A	Vulnerable
Operators with Equity above $50,000			
Under 40	Above 0	Above 0	Good
40 to 70	Above 5	Above 5	Good
Over 70	Above 15	Above 15	Good
If not already classified as "good", Then			
Under 10	Above -15	Above -15	Fair
10 to 40	Above -5	Above -5	Fair
40 to 70	Above 0	Above 0	Fair
Over 70	Above 5	Above 5	Fair
If Not already classified as "good" or "fair", Then:			
Under 10	N/A	N/A	Stressed
10 to 40	Above -15	Above -15	Stressed
40 to 70	Above -5	Above -5	Stressed
Over 70	Above 0	Above 0	Stressed
If Not already classified as "good", "fair" or "stressed", Then:			Vulnerable

N/A Not Applicable

Source: Wadsworth and Bravo-Ureta, (1992)

[1] Equity in dollars 1985 prices. For our analysis, this figure has been inflated to reflect 2007 prices and converted into pounds

Annex VI Summary of BMPs to reduce farm diffuse pollutants in most cost-effective order

Arable farms		Dairy farms	
Sandy Loam Soils	**Clay Loam Soils**	**Sandy Loam Soils**	**Clay Loam Soils**
Establish riparian buffer strips	Adopt minimal cultivation systems	Integrate fertiliser and manure nutrient supply	Integrate fertiliser and manure nutrient supply
Establish cover crops in the autumn: cereals	Do not apply P fertilisers to high P index soils	Do not spread slurry or poultry manure to fields at high-risk times	Reduce fertiliser application rates (reduction in P by a 20%)
Re-site gateways away from high-risk areas	Establish and maintain artificial (constructed) wetlands	Do not apply manure to high-risk areas	Do not apply P fertilisers to high P index soils
Cultivate and drill across the slope	Establish cover crops in the autumn: cereals	Establish riparian buffer strips	Do not spread slurry or poultry manure to fields at high-risk times
Cultivate compacted tillage soils	Re-site gateways away from high-risk areas	Re-site gateways away from high-risk areas	Do not apply manure to high-risk areas
Establish new hedges	Cultivate and drill across the slope	Establish new hedges	Establish and maintain artificial (constructed) wetlands
Avoid tramlines over winter	Cultivate compacted tillage soils	Move feed and water troughs at regular intervals	Re-site gateways away from high-risk areas
Establish in-field grass buffer strips	Avoid tramlines over winter	Increase the capacity of farm manure (slurry) stores	Move feed and water troughs at regular intervals
Cultivate land for crop establishment in spring rather than autumn	Use a fertiliser recommendation system	Change from slurry to a solid manure handling system	Use a fertiliser recommendation system
Establish cover crops in the autumn: other crops	Reduce fertiliser application rates: 20% reduction in P	Transport manure to neighbouring farms 5km	Increase the capacity of farm manure (slurry) stores
Convert arable land to extensive grassland: conversion to ungrazed grassland	Cultivate land for crop establishment in spring rather than autumn	Fence off rivers and streams from livestock	Transport manure to neighbouring farms 5km
Convert arable land to extensive grassland: conversion to extensive grazing	Convert arable land to extensive grassland: conversion to ungrazed grassland	Loosen compacted soil layers in grassland fields	Do not apply fertiliser to high risk areas
Leave autumn seedbeds rough	Convert arable land to extensive grassland: conversion to extensive grazing	Adopt phase feeding of livestock	Adopt phase feeding of livestock
	Establish cover crops in the autumn: other crops	Transport manure to neighbouring farms 20km	Establish riparian buffer strips
	Establish riparian buffer strips	Reduce field stocking rates when soils are wet	Establish new hedges
	Establish new hedges	Reduce dietary N and P intakes	Change from slurry to a solid manure handling system
	Do not apply fertiliser to high risk areas	Reduce overall stocking rates on livestock farms	Transport manure to neighbouring farms 20km
	Avoid spreading fertiliser to fields at high-risk times		Fence off rivers and streams from livestock
	Establish in-field grass buffer strips		Reduce field stocking rates when soils are wet
	Leave autumn seedbeds rough		Incorporate manure into the soil
	Allow field drainage systems to deteriorate		Reduce dietary N and P intakes
			Avoid spreading fertiliser to fields at high-risk times
			Loosen compacted soil layers in grassland fields
			Reduce overall stocking rates on livestock farms
			Minimise the volume of dirty water produced
			Reduce the length of the grazing day or grazing season
			Allow field drainage systems to deteriorate

Annex VII - CE Water Quality Levels

Levels of water quality improvements (policy scenarios) for rivers in 7 years time (2015)

Water Quality Descriptors	Legend	No change	Level 1	Level 2	Level 3	Level 4
R7LOW	**Many Problems**	16%	0%	4 %	8%	12%
R7HIGH	**No Problems**	34%	83 %	65%	48%	39%
R7MED	**Few Problems**	50%	17%	31%	44%	49%
∆R7LOW		0	-16%	-12%	-8%	-4%

Levels of water quality improvements (policy scenarios) for rivers in 7 years time (2015)

Water Quality Descriptors	Legend	No change	Level 1	Level 2	Level 3	Level 4
L7LOW	**Many Problems**	44%	0.00%	11%	22%	33%
L7HIGH	**No Problems**	25%	81%	57%	38%	29%
L&MED	**Few Problems**	31%	19%	32%	40%	38%
∆L7LOW		0	-44%	-33%	-22%	-11%

Annex VIII - CE Experimental Design

The choice experiment involved a formal comparison of preferences for four levels of each of two attributes (rivers at 7 years and lochs at 7 years) and two levels of each of two attributes (rivers at 20 years and lochs at 20 years). Respondents were asked to choose between policy scenarios, presented in pairs, each scenario being defined by varying levels of each attribute and an associated price for each option, which took the form of an implied increase in water bills and other household expenses. Pairs of scenarios were selected according to a formal statistical design that allowed efficient estimation of preferences between attributes and price levels.

In order to estimate main effects as precisely as possible the choice experiment design was constructed so that the same levels of any factor do not occur in both choices offered together in a pair. Consequently the full design for a choice experiment with this number of factors and levels would be $12^3 * 2^2 = 6912$ questions. If each respondent is asked a set of eight questions unique to that individual then a total of 862 respondents would be required. Such a design would be fully efficient for the estimation of main effects. However, if this was repeated in each of the three Scottish regions (South, Central and North) then a total of 2586 respondents would be required. In order to reduce this to a more realistic size, fractional factorial designs were used in which only a fraction of the possible questions were asked.

A pilot study was undertaken in which a total of 48 respondents were asked eight questions each. This constitutes a 1/18th fraction of a complete factorial design as a total of 384 questions (1/18th of 6912) were asked. In the main study 144 respondents were questioned in each of the three regions and this constitutes a 1/6th fraction of the full factorial design. The fractions were determined using the theory of fractional factorial designs. The 12 permitted pairings for rivers and lochs at eight years and for price differences were each defined as 12-level factors (table 5 below) which can then be expressed as combinations of two 2-level pseudo-factors and a single 3-level pseudo factor. A 1/6th fraction design can be generated by applying two sifts (one sift based on a combination of 2-level factors and 2-level pseudo-factors, and a further sift based on a combination of 3-level pseudo-factors). A 1/18th fraction design can be generated by applying three sifts (one sift based on a combination of 2-level factors and 2-level pseudo-factors, and a further two sifts each based on a combination of 3-level pseudo-factors). During the main study the same 862 questions were asked in each of the three regions but a separate randomisation of questions to respondents was undertaken for each region.

Table 5 Price levels

Level	First price Option A	Second price Option B	Difference	Absolute difference
1	£5	£20	-£15	£15
2	£20	£5	£15	£15
3	£5	£50	-£45	£45
4	£50	£5	£45	£45
5	£10	£40	-£30	£30
6	£40	£10	£30	£30
7	£10	£75	-£65	£65
8	£75	£10	£65	£65
9	£20	£75	-£55	£55
10	£75	£20	£55	£55
11	£20	£100	-£80	£80
12	£100	£20	£80	£80

Survey
WFD CE Main

Good morning/afternoon/evening. My name is.............from Feedback Market Research. In conjunction with the Macaulay Institute and the Scottish Agricultural College we are currently conducting research on behalf of the Scottish Government, investigating important issues about environmental policy.

Could you please spare some time to take part in this study? (**IF NECESSARY SAY**: The interview will take around 25- 30 mins to complete). As a thank you for your time you will be provided with **£5**?.

This is a bona fide market research exercise. It is being conducted under the Market Research Society Code of Conduct which means that any answer you give will be treated in confidence.

Many of the following questions relate to people's use of Scottish rivers and lochs. If you think that a particular question or part of a question does not apply to you please answer "don't know".

INTERVIEWER

INTERVIEWER ID. #

DATE

CHOICE SET ID #
(ENVELOP)

TIME STARTED

TIME ENDED

INTERVIEW LENGTH (MINS)

LOCATION

SECTION A: INTRODUCTORY SECTION

Q1 First, I'm going to read a list of national concerns. For each issue, please tell me whether you feel the amount of money we are spending as a nation in Scotland is too much, just about the right amount, or too little.
[HAND RESPONDENT SHOWCARD 1A] RECORD RESPONSES IN THE TABLE BELOW

		Too much	About the Right Amount	Too little	Don't Know ***[DO NOT READ]***	Refused ***[DO NOT READ]***
1a	Pensions	1	2	3	8	9
1b	Social Security	1	2	3	8	9
1c	Education	1	2	3	8	9
1d	Reducing Water Pollution	1	2	3	8	9
1e	Transport	1	2	3	8	9
1f	Policy and Criminal Justice	1	2	3	8	9
1g	Reducing Air Pollution	1	2	3	8	9
1h	Health Care	1	2	3	8	9

Q2 I now want you to consider this list of areas where water and sewerage services could be improved if we spent more.
[HAND RESPONDENT SHOWCARD 1B]
Which of these issues do you think should be the first and second highest priority for investment?
2a First priority=____________ 2b Second priority=____________
[RECORD RESPONSES TO Q2a AND Q2b USING THE FOLLOWING CODE:]

Issues	First Priority	Second Priority
Reducing the risk of drinking water discoloration	1.1	2.1
Reducing the risks of sewer flooding	1.2	2.2
Fixing leaks and pipes	1.3	2.3
Improving river water quality	1.4	2.4
Reducing the risks of interruptions to supply	1.5	2.5
Improving bathing water quality	1.6	2.6
Protecting animal and plant life around waterways	1.7	2.7
Don't know ***[DO NOT READ]***	1.9	2.9

Q3 Please look at this card **[HAND RESPONDENT SHOWCARD 2].** It contains five statements about pollution control and the costs of pollution control. Please read these statements, and then tell me which you agree with most.
RECORD A, B, C, D, E, F, OR G ________________

A Protecting the environment is so important that pollution control requirements and standards cannot be too strict, and continuing improvement must be made regardless of cost

B Protecting the environment is important and continuing improvements should be funded, provided that they are not excessively costly

C We are spending about the right amount on cleaning up the environment and don't need to increase or decrease this spending

D We have made enough progress on cleaning up the environment and should now concentrate on holding down costs rather than requiring stricter pollution controls

E Pollution control requirements and environmental quality standards have gone too far and they already cost most than they are worth

F Don't know ***[DO NOT READ]***

G Refused ***[DO NOT READ]***

Q4 How many people including yourself, partner and children live in your household? __________ **[RECORD NUMBER]**

(IF MORE THAN ONE PERSON, ASK 4A. OTHERWISE SKIP TO Q5).

4a. How many of these people are under 16? __________ **[RECORD NUMBER]**

In this survey we will ask you questions about how you value lochs and rivers. The questions we ask will only deal with freshwater bodies. Oceans or other salt water, including sea lochs, will not be included. When we say loch in this survey, we mean standing body of fresh water, including natural lochs, ponds and reservoirs created by damming rivers. When we say river in this survey, we mean any flowing body of water fed by run-off from rain or snow. This involves rivers, creeks and any other streams.

Now we would like to ask you some questions about how you use lochs and rivers in Scotland

Q5 Please look at this card **[HAND RESPONDENT SHOWCARD 3].**
This card lists a number of possible **river based activities**. I would like to know if you, or any members of your household, have done any of these things often – that is more than six times in the last twelve months, or sometimes – that is between three and six times in the last twelve months, or rarely – that is only once or twice in the last twelve months, or not at all. Remember only in Scotland.

RECORD RESPONSES IN THE TABLE BELOW

Activity	**Often** (More than 6 times in the last 12 months)	**Sometimes** (Between 3 and 6 times in the last 12 months)	**Rarely** (Once or twice in the last 12 months)	**Never** (Zero times in the past 12 months)	**Don't know** ***[DO NOT READ]***	**Refused** ***[DO NOT READ]***
Q5a Fishing/Hunting	*1 - Often -*	*2 - Sometimes -*	*3 - Rarely -*	*4 - Never -*	*8 - Don't know-*	*9 - Refused-*
Q5b Non-motor water based activity (e.g. swimming, canoeing....)	*1 - Often -*	*2 - Sometimes -*	*3 - Rarely -*	*4 - Never -*	*8 - Don't know-*	*9 - Refused-*
Q5c Motorised water based activity (e.g. motor boating, Jet Skiing...)	*1 - Often -*	*2 - Sometimes -*	*3 - Rarely -*	*4 - Never -*	*8 - Don't know-*	*9 - Refused-*
Q5d Natural beauty/Peace and quiet in the proximity of a river (e.g. hiking, camping, walking, running, cycling, sitting nearby, wildlife observation....)	*1 - Often -*	*2 - Sometimes -*	*3 - Rarely -*	*4 - Never -*	*8 - Don't know-*	*9 - Refused-*

Q6 Please look again at this card **[HAND RESPONDENT SHOWCARD 3].**
This card lists a number of possible **loch based activities**. I would like to know if you, or any members of your household, have done any of these things often – that is more than six times in the last twelve months, or sometimes – that is between three and six times in the last twelve months, or rarely – that is only once or twice in the last twelve months, or not at all. Remember only in Scotland.

RECORD RESPONSES IN THE TABLE BELOW

Activity	**Often** (More than 6 times in the last 12 months)	**Sometimes** (Between 3 and 6 times in the last 12 months)	**Rarely** (Once or twice in the last 12 months)	**Never** (Zero times in the past 12 months)	**Don't know** ***[DO NOT READ]***	**Refused** ***[DO NOT READ]***
Q6a Fishing/Hunting	*1 - Often -*	*2 - Sometimes -*	*3 - Rarely -*	*4 - Never -*	*8 - Don't know-*	*9 - Refused-*
Q6b Non-motor water based activity (e.g. swimming, canoeing....)	*1 - Often -*	*2 - Sometimes -*	*3 - Rarely -*	*4 - Never -*	*8 - Don't know-*	*9 - Refused-*
Q6c Motorised water based activity (e.g. motor boating, Jet Skiing...)	*1 - Often -*	*2 - Sometimes -*	*3 - Rarely -*	*4 - Never -*	*8 - Don't know-*	*9 - Refused-*
Q6d Natural beauty/Peace and quiet in the proximity of a loch (e.g. hiking, camping, walking, running, cycling, sitting nearby, wildlife observation....)	*1 - Often -*	*2 - Sometimes -*	*3 - Rarely -*	*4 - Never -*	*8 - Don't know-*	*9 - Refused-*

SECTION B: WATER QUALITY LEVELS
[READ OUT SLOWLY AND CLEARLY]

Please look at this card. [HAND RESPONDENT SHOWCARD 4] This card describes what the environment in and around a loch or river site might be like. In this card, and for the remainder of the survey, we will use the three-colour system on this card to describe the three possible water quality levels at any water site.

Dark Blue – No Problems. Water quality is "High". There will be a diverse and natural range of plants, insects, fish, birds and other animals. Water will generally have the right degree of clarity, no noticeable pollution, and generally be suitable for contact activities.

Mid Blue – Few Problems. Water quality is "Medium". There will be plants, insects, fish, birds and other animals, but there will be some fish and other wildlife missing. Water will be slightly murky or discoloured in parts, and there will sometimes be visible pollution in some places, and some algal blooms. Water will be suitable for contact activities in some areas but not others.

Light blue – Many Problems. Water quality is "Low". There may be limited or no plants or wildlife, or the water may be dominated by a single plant species. Water will generally be murky or discoloured, and may sometimes be bad-smelling in some places. There may also regularly be visible pollution in some places, and frequent algal blooms. Water will be unsuitable for contact activities.

[Please keep this card in front of you for the rest of this survey].

I am now going to show you an example water site for both rivers and lochs. The illustrations are text-book drawings, which match the descriptions, but the colours, the number and type of species present, and their sizes depend on the location, time of day and season.

There are a few important ideas in these illustrations. First, "No Problems" means "high water quality" and that having the right amount of naturally occurring plants and animals. Second, some water sites are naturally more murky than others so clear water does not necessarily mean high quality. Lastly, so we can focus on the number of plants and animals present in the river or loch, the illustrations will not show any rubbish like plastic bottles or plastics bags.

[HAND RESPONDENT NEXT SHOWCARD, FIRST 4A, FOLLOWED BY 4B. READ OUT CLEAR AND SLOWLY, THE TEXT BELOW THAT CORRESPONDS TO THE SHOW CARD GIVEN]

[CARD 4A RIVER]
This card gives an example of rural rivers.
A Rural River that has Dark Blue quality will have a varied fish population, possibly including trout and salmon as well as coarse fish. It will support diverse and native plants and animals. It will have the right degree of clarity most of the time with no noticeable pollution. There will be natural and seasonal variations in water levels and flow. It will generally be suitable for contact activities.
A Rural River that has Mid Blue quality will have coarse fish, possibly trout but few salmon. It will support some plants and animals. Water will have slightly less than the right degree of clarity, becoming murkier after rain. In some cases river bed, banks, water levels and/or flow may be affected by human pressures. It will be suitable for contact activities in some areas but not others.
A Rural River that has Light Blue quality will have very few plants, fish or other animals. It will have cloudy, discoloured and possibly bad-smelling water. River bed, banks, water levels and flow will be noticeably affected by human or animal pressures. It will be unsuitable for contact activities Please note that in some places, there may be some pollution-tolerant fish in low quality rural rivers.

[CARD 4B LOCH]
This card gives an example of how Lochs are classified.

A loch that has Dark Blue quality will have a diversity of underwater plants, floating lilies, and tall flowering plants. It will have a varied fish population, including trout and coarse fish. Insects such as dragonflies are present. The water will have the right degree of clarity and no noticeable pollution. There will be natural and seasonal variations in water levels. It will generally be suitable for contact activities.
A Loch that has Mid Blue quality will have some underwater and floating plants in shallow areas and around the loch. There will be some coarse fish and other animals present but limited. Insects are rare. Water will have slightly less than the right degree of clarity and occasionally discoloured water. It will be suitable for contact activities in some areas but not others.
A Loch that has Light Blue quality will have very few plants, except blanket weed, and very few fish or other animals, except worms and leeches. It will have cloudy, discoloured and possibly bad-smelling water. It will be unsuitable for contact activities. Please note that in some places, there may be some pollution-tolerant fish in low quality lochs.

[READ OUT SLOWLY AND CLEARLY]
Please remember that this survey focuses specifically in the way we care about rivers and lochs separately. Different water site types and areas may have different characteristics even when they are both classified as having the same quality.

Now I would like you to look at a map of Scotland showing how rivers have been assessed nationally **[HAND RESPONDENT MAP 1 –NATIONAL RIVER WATER QUALITY.]** This map shows only major rivers, but does not show all of the smaller rivers and canals. Instead, the map has been divided into catchment zones and each has been coloured for ecological quality. All land drains its water, including rainfall, to rivers eventually. The land areas which contribute to a particular stretch of river represent a catchment zone.
The quality of one catchment zone may influence one or more of its neighbouring catchments, but it also may have a very different quality level depending on the direction of water flow and on the ability of the particular river environment to absorb pollutants. The pie chart at the top shows the overall proportions of the area on the map that have No Problems, Few Problems and Many Problems.

Q7 Looking at this map, what do you think about the ecological quality of Scottish rivers? Please look at this card **[HAND RESPONDENT SHOWCARD 5].** Is it A, B, C, D, or E?
.
[RECORD A, B, C, D, E, F, OR G] ____________

A	A great deal better than expected	**F**	Don't know ***[DO NOT READ]***
B	Some what better than expected	**G**	Refused ***[DO NOT READ]***
C	About as expected		
D	Somewhat worse than expected		
E	A great deal worse than expected		

Q7b Why is it?
[RECORD VERBATIM BELOW]

____

____

____

____

Q8 Now I would like you to look at a map of Scotland which illustrates only lochs water quality. **[HAND RESPONDENT MAP 2 – "NATIONAL LOCH WATER QUALITY"]** This map shows the ecological quality of all lochs in Scotland.

Looking at this map, what do you think about the ecological quality of lochs in Scotland? Please look at this card **[HAND AGAIN RESPONDENT SHOWCARD 5].** Is it A, B, C, D, or E?

.

[RECORD A, B, C, D, E, F, OR G] ____________

A A great deal better than expected

B Some what better than expected

C About as expected

D Somewhat worse than expected

E A great deal worse than expected

F Don't know ***[DO NOT READ]***

G Refused ***[DO NOT READ]***

Q8b Why is it?

[RECORD VERBATIM BELOW]

____

____

____

____

____

SECTION C: IMPROVEMENTS AND BENEFITS

[READ OUT SLOWLY AND CLEARLY]

Over the past 10 years, significant improvements have been made to water sites. Now a new law is in place, which has as its first goal to make sure that the water environment does not get any worse from now on at any site. It then aims to make substantial improvements within the next 7 to 20 years by reducing the amount of Low Quality and increasing the amount of High Quality sites in the water environment.

Please look at this next card. **[HAND RESPONDENT SHOWCARD 6].**

The potential benefits of the new law include the following:
Improvements from Many Problems to Few Problems will make conditions better for plants, fish and animals in and around the water , decrease visible pollution, and allow for boating, angling and other activities around the water.
Improvements from Few Problems to No Problems will end visible signs of pollution, improve conditions for all water-based activities, and provide good conditions for a diversity of plants, animals, fish and birds.

Please note that the new law will not affect your drinking water quality, it will not directly create more access for recreational water users; and it will not be responsible for cleaning up general rubbish like plastic bags and bottles.

Q9 **[HAND RESPONDENT SHOW CARD 7.]** Please look at this card and choose which of these possible improvements is most important to you, **[RECORD 1ST RANKED BENEFIT, A-G]**
.
[RECORD A, B, C, D, E, F, OR G] ____________

A Improved conditions for fishing

B Improved conditions for water contact activities. For example; canoeing, rowing, rafting, surfing, windsurfing, diving, wading, paddling, or swimming in the sea or rivers or lakes (not in swimming pools)

C Improved conditions for other activities on, or around the water. for example; narrowboating, walking, running, cycling, or sitting nearby)

D Knowledge of improved habitats for plants, fish and other animals

E Other [Please specify] ____
____
____

F All of equal importance ***[DO NOT READ]***

G Don't know ***[DO NOT READ]***

SECTION D1: WATER QUALITY VALUATION
[READ OUT SLOWLY AND CLEARLY]

Water quality is affected by pollution from households, farms and businesses, and climate change. Some works are needed just to prevent water sites from getting worse. The government's policy is that the polluter will have to pay for these works. This will make some every day products more expensive and will increase household water and sewerage bills too.

The government has estimated that these extra costs to each household, including yours, will be of a certain amount (£X) per year, in terms of higher water and sewerage bills (through council taxes) and higher prices on everyday products.

Improving the environment requires more cutting of pollution, which will make products more expensive and will further increase your household expenditure. I am now going to show you cards, which have two or three options for water environment improvements. For all the options, steps will be taken so there will be no worsening of the water environment at any site, the most cost-effective works will be used, the money will be
ring-fenced to make the improvements, and information will be made available to the public on progress towards the improvements.

On each card, I would like you to tell me which option you most prefer. The choices you make will be used to help decide how far to go with making improvements so will influence everyone's payment for improvements.

CHOICE EXAMPLE
[READ OUT SLOWLY AND CLEARLY]

Please look at this card. **[HAND RESPONDENT EXAMPLE CHOICE CARD 1]**. In the top half of the table are some pie-charts. For each option (Option A – No Change and Option B-water quality improvements option) the first row of pie charts shows how much of Scottish rivers - the area on the first map - will be classified Light Blue, Mid Blue and Dark Blue in 7years time, in 2015. The second row of pie charts, shows the same information but for Scottish lochs.

The left-hand Option A is the No Change option. There is no deterioration but also no improvements, so the No Problem, Few Problems and Many Problems areas stay the same as shown in the maps earlier. For example, currently **16** percent of Scottish rivers are classified as having many problems and low water quality, 50 percent do have a few problems which translates to Medium Water Quality and 34 percent do not have problems and their quality can be regarded as high or very good. These will stay the same if you choose this option and will result in the addition to household costs and water bills of £ X per year due to the steps to stop any worsening.

In Option B, some improvements are made and so the amount of Light Blue comes down, and the amount of Dark Blue goes up over the next 7 years until 2015. The second row relates to Scottish lochs water quality in 2015.

The third row shows the percentage of rivers in Scotland that will be Dark Blue quality in 20 years time. By this time, all Light Blue areas will have been tackled so that the remainder will all be Mid Blue or Dark Blue. Those areas remaining Mid Blue will be evenly spread around the country. The fourth row shows the same information for lochs.

For rivers in 20 years time, In Option A, there will be no improvements, and so the amount of Dark Blue will remain as it is now in the year 2028, meaning that 34% of Scottish rivers, as shown in the maps, will be then classified as having good quality or no problems (the rest will have few and many problems in the proportions shown in the pie chart for rivers); However with improvements in Option B, 75 percent of rivers in Scotland will be Dark Blue, with the remainder being Mid Blue.

The forth row shows the same information for Scottish lochs. In Option A, there will be no improvements, and so the amount of Dark Blue will remain as it is now in the year 2028, meaning that 25% of Scottish lochs will be classified then as having good quality or no problems; in Option B, 75 percent of Scottish lochs will be Dark Blue, with the remainder being Mid Blue.

The fifth row shows how much your household would pay every year for the improvements in the Option above. For Option A – No Change, there would small additional charges (£X) to stop the water environment getting worse. For Option B, the increase in your household bills would be £Y every year. These payments are in addition to the payment to ensure no water site gets worse, and would continue indefinitely.

Please review the options and think about how you would select between Option A and B taking into account the amount of the water environment improved, the amount of time it would take, and the increase in your household bills. You should choose Option A unless you think Option B is better.

It is important for us to get realistic choices from you regarding the values of these programmes, so before you make some real choices, please consider your household budget and all of the things that you and your household need or would prefer to spend your money on before you decide. Please also bear in mind that your water bill and other household expenses may change in future for other reasons not related to the water environment, and your income may also change in future. Your choices will influence how far to go with improvements, so will influence everyone's payment for improvements.

Q.10 I will now show you a card which is exactly the same as the two option examples except the payment amounts are filled in and you have a choice of three Options A, B, or C. Please select your preferred option, A, B, or C.

[RECORD OPTION CHOSEN FOR EXAMPLE CHOICE CARD] 10a________________

10b Why did you select this Option? **[RECORD VERBATIM]**

____

____

____

____

____

Q11 [IF RESPONDENT CHOOSES "Option A - No Change" AND PROVIDES AN EXPLANATION LIKE: "THE ENVIRONMENT HAS IMPROVED FOR ZERO COST", INTERVIEWER SHOULD READ: "There will be no improvements if Option A is chosen, would you like to re-consider your choice?"

[RECORD REVISED RESPONSE AS Q10 AND REPEAT Q10a]

Q11 revised answer __________________

Q11a revised answer __________________

I'd now like you to repeat what you have just done for each of the following cards.
[HAND RESPONDENT REMAINING CHOICE CARDS]
All cards will have three options to choose between,
Option A - No Change will be the same for every card.

VERY IMPORTANT!
INTERVIEWER: SELECT ONE SET (sealed envelop) OF CHOICE OPTION CARDS FOR THIS RESPONDENT AND RECORD THE REFERENCE NUMBER BELOW:

M / P ☐ ☐ ☐ ☐

NOW GO THROUGH EACH OF THE OPTIONS 1-8 IN TURN WITH THE RESPONDENT. PLEASE FOLLOW THE ORDER IN WHICH THE PAGES HAVE BEEN STAPLED. RESPONDENTS ARE NOT ASKED TO JUSTIFY THEIR DECISIONS WHEN LOOKING AT THESE CARDS. QUESTION 12 AIMS TO DO THIS.

POINT TO CHOICECARD **OPTION 1** ON THE SET CHOSEN, AND ASK:

1a) Which option do you **most** prefer?

A	1
B	2
C	3
Don't know	Y

1b) Which option do you **least** prefer?

A	1
B	2
C	3
Don't know	Y

POINT TO CHOICECARD **OPTION 2** ON THE SET CHOSEN, AND ASK:

2a) Which option do you **most** prefer?

A	1
B	2
C	3
Don't know	Y

2b) Which option did you **least** prefer?

A	1
B	2
C	3
Don't know	Y

POINT TO CHOICECARD **OPTION 3** ON THE SET CHOSEN, AND ASK:

3a) Which option do you **most** prefer?

A	1
B	2
C	3
Don't know	Y

3b) Which option did you **least** prefer?

A	1
B	2
C	3
Don't know	Y

POINT TO CHOICECARD **OPTION 4** ON THE SET CHOSEN, AND ASK:

4a) Which option do you **most** prefer?

A	1
B	2
C	3
Don't know	Y

4b) Which option did you **least** prefer?

A	1
B	2
C	3
Don't know	Y

POINT TO CHOICECARD **OPTION 5** ON THE SET CHOSEN, AND ASK:

5a) Which option do you **most** prefer?

A	1
B	2
C	3
Don't know	Y

5b) Which option did you **least** prefer?

A	1
B	2
C	3
Don't know	Y

POINT TO CHOICECARD **OPTION 6** ON THE SET CHOSEN, AND ASK:

6a) Which option do you **most** prefer?

A	1
B	2
C	3
Don't know	Y

6b) Which option did you **least** prefer?

A	1
B	2
C	3
Don't know	Y

POINT TO CHOICECARD **OPTION 6** ON THE SET CHOSEN, AND ASK:

7a) Which option do you **most** prefer?

A	1
B	2
C	3
Don't know	Y

7b) Which option did you **least** prefer?

A	1
B	2
C	3
Don't know	Y

POINT TO CHOICECARD **OPTION 6** ON THE SET CHOSEN, AND ASK:

8a) Which option do you **most** prefer?

A	1
B	2
C	3
Don't know	Y

8b) Which option did you **least** prefer?

A	1
B	2
C	3
Don't know	Y

Q12 In making your choices between options on the cards, what factors did you consider, and which were the most important?
[RECORD VERBATIM BELOW]

____

____

____

____

INTERVIEWER: NOW SCORE THROUGH THE SET OF CHOICE OPTIONS USED SO THAT YOU DO NOT USE IT AGAIN.

SECTION E: BACKGROUND INFORMATION

In order to ensure that we survey people from all walks of life, I would now like to ask you some questions about you and your household. I would like to reassure you that all responses will be kept strictly confidential.

Q13 Gender

Male	1
Female	2

Q14 Please could you tell me your age?

A	18-24	1
B	25-29	2
C	30-44	3
D	45-59	4
E	60-64	5
F	65-74	6
G	75+	7

Q15 [HAND RESPONDENT EDUCATION CARD] What is the highest level of education you completed on this card? Do not include specialized schools like secretarial, art, or trade schools.

[RECORD a, b, c, d, e, f or g] ____________________

A	Primary
B	O levels, GCSE or CSE (1 or more passes), NVQ Level 1or foundation Level GNVQ
C	5 or more O levels, CSE grade 1's or GCSE grades A-C; School certificate; 1 or more A levels or AS levels; NVQ level 2 or intermediate GNVQ.
D	2 or more A levels; 4 or more AS levels; Higher School Certificate; NVQ Level 3; or advanced GNVQ.
E	First Degree, Higher Degree, NVQ levels 4 and 5; HNC; HND; Qualified Teacher Status; Qualified Medical Doctor; Qualified Dentist; Qualified Nurse; Midwife; or Health Visitor.
F	DON'T KNOW ***[DO NOT READ]***
G	REFUSED ***[DO NOT READ]***

Q15 **[HAND RESPONDENT EMPLOYMENT STATUS CARD]** Which of the categories on this card describes your current employment status?

[RECORD A-L] ______________________________

A Working full-time employee (31+ hours)
B Working part-time employee (1-30 hours)
C Working self-employed
D Working and full time student
E Not working - seeking work
F Not working - full time student
G Not working retired
H Not working - looking after home/children
I Not working - permanently sick / disabled
J Not working - other
K Don't know ***[DO NOT READ]***
L Refused ***[DO NOT READ]***

Q16 What is your partner's employment status?

[RECORD, AS AVOBE, A-L] ____________________
M IF NO PARTNER

Q17 **[HAND RESPONDENT INCOME CARD]** For classification purposes only, please tell me which category best describes the total income that you (and all other members of this household) earned during 2007 before taxes. Please be sure to include each member's wages and salaries, as well as net income from any business, pensions, benefits dividends, interest, tips, or other income. Just tell me the letter that best describes your household's income.

VERY IMPORTANT - <u>IF THE RESPONDENT FAILS TO PROVIDE AN ANSWER TO THIS QUESTION</u> (meaning he/she ANSWERS "DON'T KNOW" OR REFUSE"). PLEASE READ THE FOLLOWING:

"WE WOULD LIKE TO REMIND YOU THAT ALL YOUR ANSWERS ARE IMPORTNAT TO US, IN SPECIAL TO THIS QUESTION. THIS IS A BONA FIDE MARKET RESEARCH EXERCISE AND IT IS BEING CONDUCTED UNDER THE MARKET RESEARCH SOCIETY CODE OF CONDUCT WHICH MEANS THAT ANY ANSWERS YOU GIVE WILL BE TREATED IN CONFIDENDENCE".

INTERVIEWER PLEASE ENCOURAGE THE RESPONDENT TO ANSWER THIS QUESTION.

[RECORD A-Q FROM TABLE BELOW] ____

	Per Week	Per Year
A	Up to £86	Under £4,500
B	£87 to £125	£4,500 to £6,499
C	£126 to £144	£6,500 to £7,499
D	£145 to £182	£7,500 to £9,499
E	£183 to £221	£9,500 to £11,499
F	£222 to £259	£11,500 to £13,499
G	£260 to £298	£13,500 to £15,499
H	£299 to £336	£15,500 to £17,499
I	£337 to £480	£17,500 to £24,999
J	£481 to £576	£25,000 to £29,999

K	£577 to £769	£30,000 to £39,999
L	£770 to £961	£40,000 to £49,999
M	£962 to £1,441	£50,000 to £74,999
N	£1,442 to £1,922	£75,000 to £99,999
O	£1,923 or over	£100,000 or over
P	Don't know ***[DO NOT READ]***	
Q	Refused ***[DO NOT READ]***	

Q18 **[HAND RESPONDENT BENEFITS CARD]** Are you, or is your partner, in receipt of any of the benefits on this card?

[RECORD A-L FROM TABLE BELOW, CODE ALL THAT APPLY] ______________________

A Unemployment related benefits, or National Insurance Credits

B Income support (not as an unemployed person)

C Sickness or disability benefits (not including tax credits)

D State pension

E Family related benefits (excluding child benefit and tax credits)

F Child benefit

G Cold weather payment

H Housing, or council tax benefits

I Tax Credits

J Other (Please specify)____

K Don't know ***[DO NOT READ]***

L Refused ***[DO NOT READ]***

Q19 **[HAND RESPONDENT ACCOMMODATION STATUS CARD]** Which of these ways describes how you occupy your accommodation?

[RECORD A-I FROM TABLE BELOW] ____

A Own outright

B Own with a mortgage loan

C Shared ownership

D Rented from council / local authority

E Rented from Housing Association / registered social landlord

F Rented from private landlord or letting agency

G Rented from other

H Don't know ***[DO NOT READ]***

I Refused ***[DO NOT READ]***

Q20 Are you or any persons in your household members of organisations which are involved with lakes, streams, rivers, estuaries or coastal waters? These organisations might include fishing clubs, surfing groups, environmental groups, local council groups focusing on water.

[RECORD A-L FROM TABLE BELOW, CODE ALL THAT APPLY] ______________________

A	Royal Society for the Protection of Birds (RSPB)
B	Surfers Against Sewage / Marine protection society
C	Canoeing/Boating/Rowing/Windsurfing club
D	Angling club
E	Ramblers Association
F	Friends of the Earth / Greenpeace /WWF
G	National Trust
H	Local Wildlife Trust or environmental organisation
I	Other similar organisation
J	Don't know ***[DO NOT READ]***
K	Refused ***[DO NOT READ]***
L	NO MEMBER OF ANY GROUP

Q21 Last question! What is the first part of your postcode? (For example B22)

[VERBATIM] ______________________

MANY THANKS AND CLOSE

FOR INTEVIEWER USE ONLY:

INTERVIEWER'S NAME/CODE:		
DATE:	**TIME:**	**LOCATION:**
FIRST PART OF POSTCODE WHERE THE INTERVIEW TOOK PLACE:		
INTERVIEWER: PLEASE COMPLETE THE FOLLOWING QUESTIONS AS SOON AS POSSIBLE AFTER THE INTERVIEW. **The first two questions are only concerned with how the respondent answered the choice questions in these survey, which asked them to make choices, and estimate their overall willingness to pay for water quality improvements. the last question is about respondents fatigue**		

Qi1 Irrespective of whether or not the respondent answered all of the questions in this questionnaire, in your judgement, how did the respondent understand what he or she was being asked to do in the questions?

[RECORD 1-7 FROM LIST BELOW________________________

1) Understood completely
2) Understood a great deal
3) Understood somewhat
4) Understood a little
5) Did not understand very much
6) Did not understand at all
7) Other (SPECIFY):

Qi2 Which of the following descriptions best describe the degree of effort the respondent made to arrive at a value for the survey's choice cards?

[RECORD 1-4 FROM LIST BELOW________________________

1) Gave the questions careful consideration
2) Gave the questions some consideration
3) Gave the questions very little consideration
4) Other (SPECIFY):

Qi3 Which of the following best describes the degree of fatigue shown by the respondent?

[RECORD 1-3 FROM LIST BELOW________________________

1) Maintained concentration throughout the survey
2) Lessened concentration in the later stages
3) Other (SPECIFY):

CARD 1A

	Too Much	About the Right Amount	Too little
Pensions			
Social Security			
Education			
Reducing Water Pollution			
Transport			
Policy and Criminal Justice			
Reducing Air Pollution			
Health Care			

CARD 1B

	First Priority	Second Priority
Reducing the risk of drinking water discoloration		
Reducing the risks of sewer flooding		
Fixing leaks and pipes		
Improving river water quality		
Reducing the risks of interruptions to supply		
Improving bathing water quality		
Protecting animal and plant life around waterways		

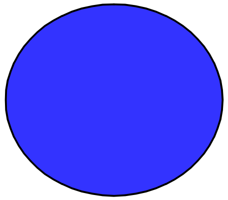
CARD 2

A	Protecting the environment is so important that pollution control requirements and standards cannot be too strict, and continuing improvement must be made regardless of cost
B	Protecting the environment is important and continuing improvements should be funded, provided that they are not excessively costly
C	We are spending about the right amount on cleaning up the environment and don't need to increase or decrease this spending
D	We have made enough progress on cleaning up the environment and should now concentrate on holding down costs rather than requiring stricter pollution controls
E	Pollution control requirements and environmental quality standards have gone too far and they already cost most than they are worth

CARD 3

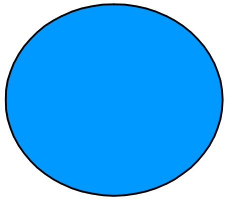
Activity	Often (More than 6 times in the last 12 months)	Sometimes (Between 3 and 6 times in the last 12 months)	Rarely (Once or twice in the last 12 months)	Never (Zero times in the past 12 months)
Q6a Fishing/Hunting	***- Often -***	***- Sometimes -***	***- Rarely -***	***- Never -***
Q6b Non-motor water based activity (e.g. swimming, canoeing....)	***- Often -***	***- Sometimes -***	***- Rarely -***	***- Never -***
Q6c Motorised water based activity (e.g. motor boating, Jet Skiing...)	***- Often -***	***- Sometimes -***	***- Rarely -***	***- Never -***
Q6d Natural beauty/Peace and quiet (e.g. hiking, camping, walking, running, cycling, sitting nearby, wildlife observation....)	***- Often -***	***- Sometimes -***	***- Rarely -***	***- Never -***

CARD 4 - Water Quality Levels



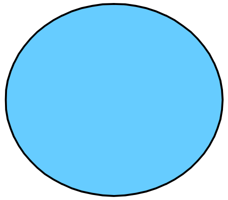
Dark Blue - quality is "HIGH" - NO PROBLEMS

- There will be diverse and natural range of plants, insects, fish, birds and other animals
- Water will generally have the right degree of clarity and there will be no noticeable pollution
- Water will generally be suitable for contact activities, such as rowing or wind surfing



Mid Blue - quality is "MEDIUM" - FEW PROBLEMS

- There will be plants, insects, fish, birds and other animals, but there will be some fish and other wildlife missing
-Water will be slightly murky or discoloured in parts, and there will sometimes be visible pollution in some places, and some algal blooms
- Water will be suitable for contact activities in some areas but no others



Light Blue - quality is "LOW" - MANY PROBLEMS

- There may be limited or no plants or wildlife, or the water may be dominated by a single plant species
- Water will generally be murky or discoloured, and may sometimes be bad-smelling in some places. there may also regularly be visible pollution in some places, and frequent algal blooms
- Water will be unsuitable for contact activities

CARD 4A - RIVER

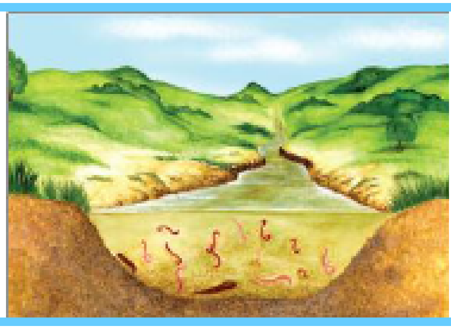
High - NO
Quality- PROBLEMS

Varied fish population, including trout and salmon as well as coarse fish. Supports diverse and native plants and animals. Water with right degree of clarity most of the time and no noticeable pollution. Natural and seasonal variations in water levels and flow. Generally suitable for contact activities.

Medium - FEW
Quality - PROBLEMS

Coarse fish, possibly trout but few salmon. Supports some plants and animals. Slightly less than the right degree of clarity, becoming murkier after rain. In some cases river bed, banks, water levels and/or flow may be affected by human pressures. Suitable for contact activities in some areas but not others.

Low - MANY
Quality - PROBLEMS



Few plants, fish or other animals. Physical barriers to migratory fish present. Often cloudy, discoloured and possibly bad-smelling water. River bed, banks, water levels and flow will be noticeably affected by human or animal pressures. Unsuitable for contact activities.

CARD 4B -LAKE

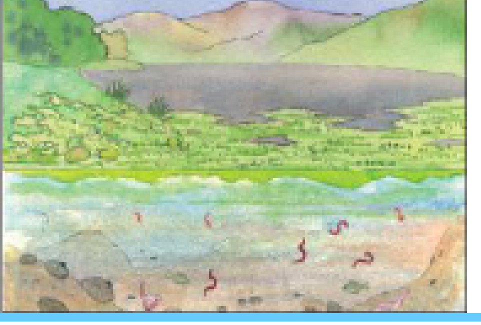
High - NO
Quality- PROBLEMS

A diversity of underwater plants, floating lilies and tall flowering plants. Varied fish population, including trout and coarse fish. Insects such as dragonflies are present. Water with right degree of clarity and no noticeable pollution. Natural and seasonal variations in water levels. Suitable for contact activities.

Medium - FEW
Quality - PROBLEMS

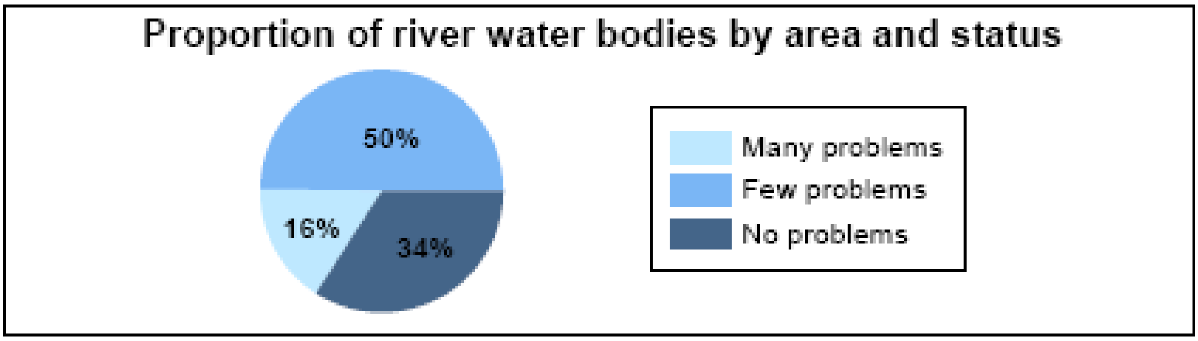
Some underwater and floating plants in shallow areas and around the lake. Some coarse fish and other animals present but limited. Insects are rare. Slightly unclear and occasionally discoloured water. suitable for contact activities in some areas but no others.

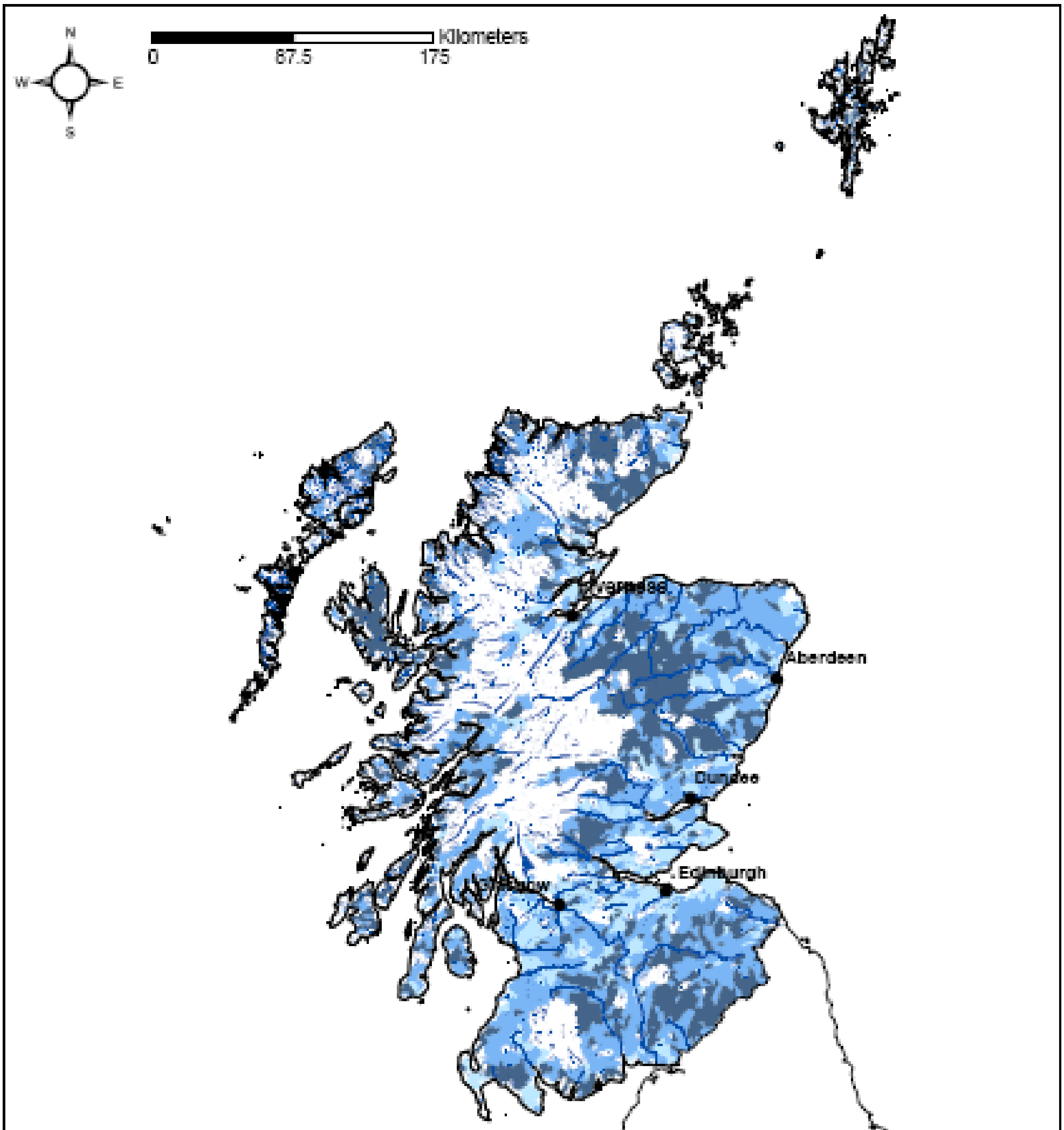
Low - MANY
Quality - PROBLEMS



Very few plants, except blanket weed, and very few fish or other animals, except worms and leeches. Cloudy, discoloured and possibly bad-smelling water. Unsuitable for contact activities.

MAP 1 - River Water Quality - Scotland





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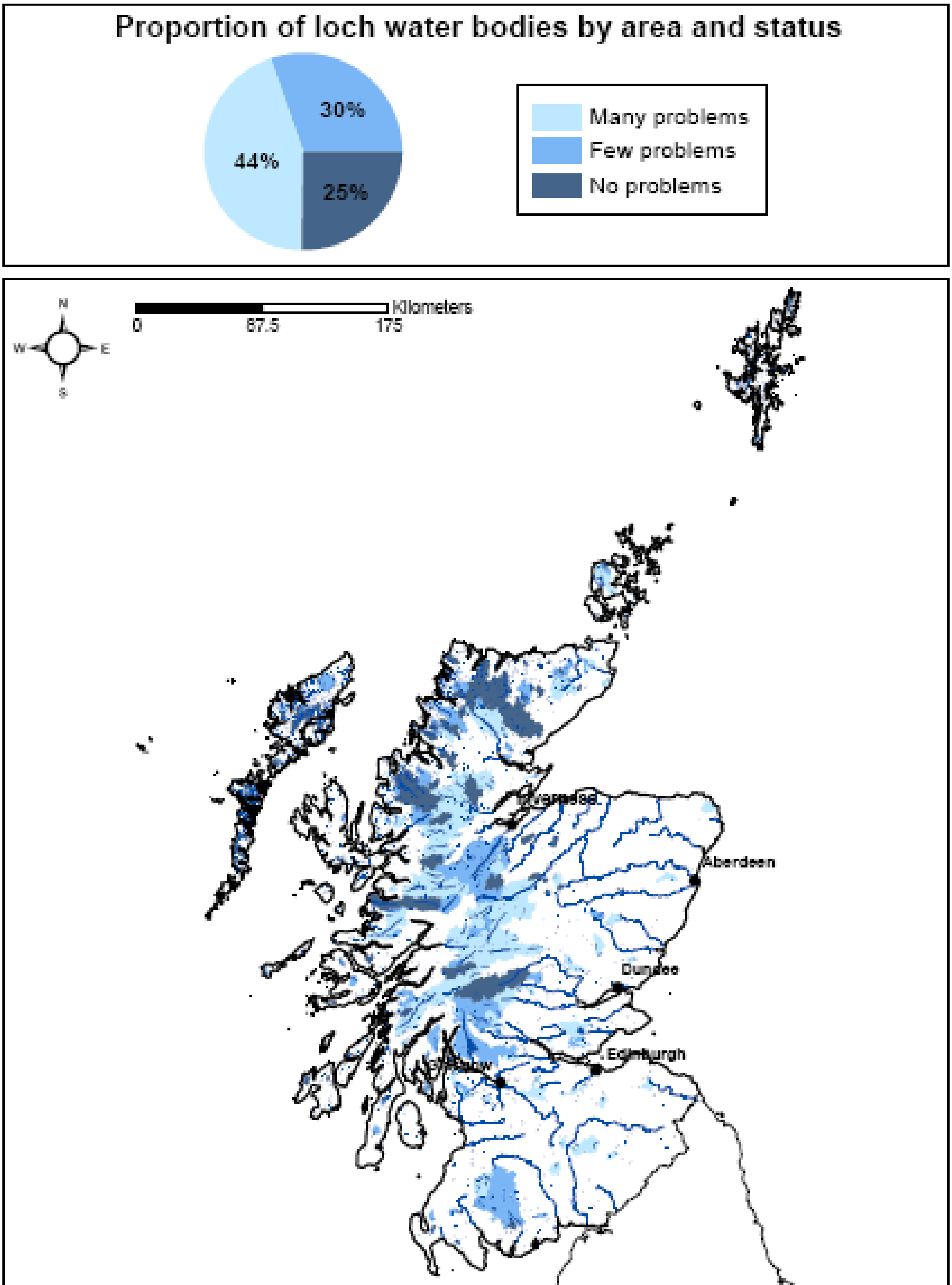
SNIFFER (2006) Diffuse Pollution Screening Tool: Stage 3
SNIFFER Project WFD77

MAP 2 - Loch Water Quality - Scotland



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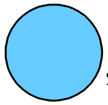
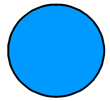
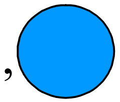
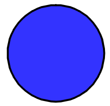
CARD 5

A	A great deal better than expected
B	Some what better than expected
C	About as expected
D	Somewhat worse than expected
E	A great deal worse than expected

CARD 6 - Water Environment Benefits

<u>The potential benefits of the new law include the following</u>:

Moving from "Many Problems", , to "Few Problems", , will make conditions better for plants, fish and animals in around the water, decrease visible pollution, and allow for boating, angling and other activities around the water.

Moving from "Few Problems", , to "No Problems",  , will end visible signs of pollution, improve conditions for all water-based activities, and provide good conditions for a diversity of plants, animals, fish and birds.

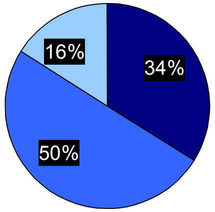
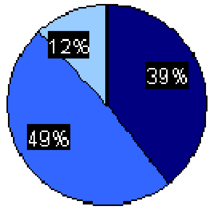
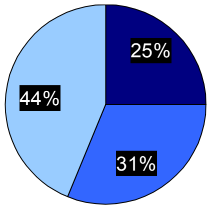
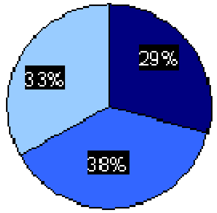
Please note that the new law will not affect your drinking water quality, it will ***<u>not</u>*** directly create more access for recreational water users; and it will ***<u>not</u>*** be responsible for cleaning up general rubbish like plastic bags and bottles.

CARD 7 - BENEFITS

A	Improved conditions for fishing
B	Improved conditions for water contact activities. For example; canoeing, rowing, rafting, surfing, windsurfing, diving, wading, paddling, or swimming in the sea or rivers or lakes (*not in swimming pools*)
C	Improved conditions for other activities on, or around the water. for example; narrowboating, walking, running, cycling, or sitting nearby)
D	Knowledge of improved habitats for plants, fish and other animals
E	Other *[Please specify]*

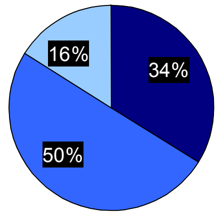
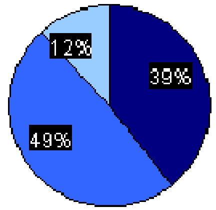
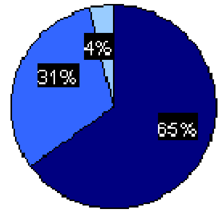
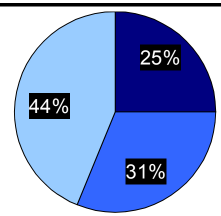
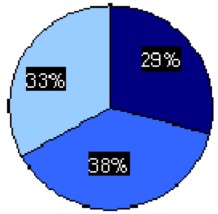
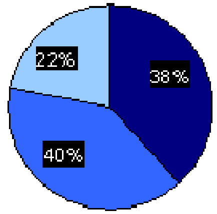
EXAMPLE CHOICE CARD 1

No Problems	Few Problems	Many Problems

CHOICE CARD SET CODE: Example 1	Option A - No Change	Option B
Status of **RIVERS** in 7 years time	 NOW AND 2015	 IN 2015
Status of **LOCHS** in 7 years time	 NOW AND 2015	 IN 2015
Status of **RIVERS in 20 years time**	Same as now IN 2028 34 %	IN 2028 75 %
Status of **LOCHS in 20 years time**	Same as now IN 2028 25 %	IN 2028 75 %
Annual increase in your water bill and other household payments. Note this payment will be added to the cost of avoiding any worsening of the water environment	£X	£ Y Per year
	Option A - No Change	**Option B**
Remembering all the things you could do with your money, **which option would you choose**, A or B?		

EXAMPLE CHOICE CARD 2

No Problems	Few Problems	Many Problems

CHOICE CARD SET CODE: Example 2	**Option A - No Change**	**Option B**	**Option C**
Status of **RIVERS** in 7 years time	 NOW AND 2015	 IN 2015	 IN 2015
Status of **LOCHS** in 7 years time	 NOW AND 2015	 IN 2015	 IN 2015
Status of **RIVERS in 20 years time**	Same as now IN 2028 34 %	IN 2028 95 %	IN 2028 75 %
Status of **LOCHS in 20 years time**	Same as now IN 2028 25 %	IN 2028 95 %	IN 2028 75 %
Annual increase in your water bill and other household payments. Note this payment will be added to the cost of avoiding any worsening of the water environment	NONE	£ 50 Per year	£ 5 Per year
	Option A - No Change	**Option B**	**Option C**
Remembering all the things you could do with your money, **which option would you choose**, A , B or C?			

AGE

A	18-24
B	25-29
C	30-44
D	45-59
E	60-64
F	65-74
G	75+

EDUCATION

A	Primary
B	O levels, GCSE or CSE (1 or more passes), NVQ Level 1or foundation Level GNVQ
C	5 or more O levels, CSE grade 1's or GCSE grades A-C; School certificate; 1 or more A levels or AS levels; NVQ level 2 or intermediate GNVQ.
D	2 or more A levels; 4 or more AS levels; Higher School Certificate; NVQ Level 3; or advanced GNVQ.
E	First Degree, Higher Degree, NVQ levels 4 and 5; HNC; HND; Qualified Teacher Status; Qualified Medical Doctor; Qualified Dentist; Qualified Nurse; Midwife; or Health Visitor.

EMPLOYMENT STATUS

A	Working full-time employee (31+ hours)
B	Working part-time employee (1-30 hours)
C	Working self-employed
D	Working and full time student
E	Not working - seeking work
F	Not working - full time student
G	Not working retired
H	Not working - looking after home/children
I	Not working - permanently sick / disabled
J	Not working - other

INCOME

	Per Week	Per Year
A	Up to £86	Under £4,500
B	£87 to £125	£4,500 to £6,499
C	£126 to £144	£6,500 to £7,499
D	£145 to £182	£7,500 to £9,499
E	£183 to £221	£9,500 to £11,499
F	£222 to £259	£11,500 to £13,499
G	£260 to £298	£13,500 to £15,499
H	£299 to £336	£15,500 to £17,499
I	£337 to £480	£17,500 to £24,999
J	£481 to £576	£25,000 to £29,999
K	£577 to £769	£30,000 to £39,999
L	£770 to £961	£40,000 to £49,999
M	£962 to £1,441	£50,000 to £74,999
N	£1,442 to £1,922	£75,000 to £99,999
O	£1,923 or over	£100,000 or over

BENEFITS

A	Unemployment related benefits, or National Insurance Credits
B	Income support (not as an unemployed person)
C	Sickness or disability benefits (not including tax credits)
D	State pension
E	Family related benefits (excluding child benefit and tax credits)
F	Child benefit
G	Cold weather payment
H	Housing, or council tax benefits
I	Tax Credits
J	Other (*Please specify*)

ACCOMMODATION STATUS

A	Own outright
B	Own with a mortgage loan
C	Shared ownership
D	Rented from council / local authority
E	Rented from Housing Association / registered social landlord
F	Rented from private landlord or letting agency
G	Rented from other

MEMBERSHIPS

A	Royal Society for the Protection of Birds (RSPB)
B	Surfers Against Sewage / Marine protection society
C	Canoeing/Boating/Rowing/Windsurfing club
D	Angling club
E	Ramblers Association
F	Friends of the Earth / Greenpeace /WWF
G	National Trust
H	Local Wildlife Trust or environmental organisation
I	Other similar organisation

Annex XI - Survey Quota

	Survey - Water quality CE
Total sample size	490
Pilot sample size	48
Main sample size	442
Sampling procedure	Target population: Representative sample of Households in Scotland. HH main bill payer. Locations: Division of Scotland into 3 regions: Choose locations in South (Dumfries and Galloway); Central (Central belt including Edinburgh, Glasgow and Stirling); North (Highlands and Aberdeenshire). Respondents have to live within 15 miles of each of the chosen locations (record home postcode). Use of Census (or Scottish Household survey) data and definitions of urban/rural to select sample points from each area Sampling quotas: sex (45%+male/45%+female), age (3 cat; 18-34,35-54,55-75), Socioeconomic grade (SEG), reasonable spread from across categories: - AB Higher and intermediate managerial/administrative/professional - C1 Supervisory, Clerical, junior managerial/ administrative/professional - C2 Skilled Manual Workers - D Semi-skilled and un-skilled manual workers - E On state benefits, unemployed, lowest grade workers, Students - X Not applicable (people aged 15 or under or aged 75 or over
Type of interview	Face-to-face in home

Annex XII - Characteristics of the sampled population

Table 1 Comparison of survey sample to official Scottish Household statistics

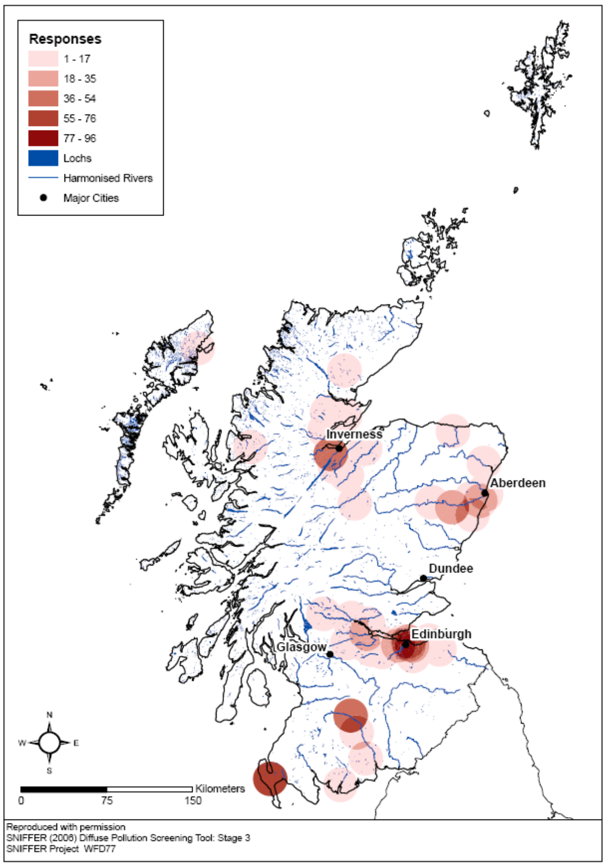
Demographic variable	Survey Participants n=426 Percent	Scottish Household statistics* n=19,233 Percent
Employment Status (16 years or older)	inc. partners (n=652)	
Self employed	1	7
Full time employment	44	49
Part time employment	10	14
Looking after home/family	10	8
Permanently retired from work	17	3
Unemployed and seeking work	7	4
At school/Higher/further education	1	5
Permanently sick or disabled	6	8
Age		
18-24	16.0	13.6
25-44	41.5	32.1
45-59	22.3	25.9
60-74	15.3	19.8
75 plus	4.9	8.6
Education		
O Grade, Standard Grade or equivalent	30	19
Higher, A level, HNC/ HND or equivalent	36	32
Degree, Professional qualification	22	24
Gender		
Male	47.4	48
Female	52.6	52
Number of people in Household		
One	23	32
Two	34	35
Three	21	15
Four	14	12
Five or more	9	5
Whether household has children		
Yes	35.6	26
Mean number of children	1.67763	1.68
Household Income		
Less than £300 per week	45.0	39.1**
£300-£699	36.0	38.1**
£700-999	8.6	12.4**
More than a £1000 per week	10.4	10.4**
Tenure of household		
Owned outright	16	29
Own with a mortgage loan	27	36
Rent - Local Authority/Scottish Homes	39	17
Rent - Housing Assoc/Co-op	10	8
Rent - Private landlord	7	7

*2005/2006 Scottish Household Survey
** Family Resources Survey 2005-06 (n=1892)

Responses by location

Figure 1 below maps the location of the place of residence of the 432 respondents to the main survey and indicates the proportion of responses in a 10 mile radius area by postcode. The sample was evenly divided between three regions in Scotland (south, central and north areas).

Figure 1 Place of residence of the participants in the survey



Annex XIII - Results on supporting questions (attitudes, opinions and uses of the water environment)

Table 1 Opinions on Levels of National Spending on Public Services - Percentages (comparison with the CRP study results in brackets)

	Too much	About right amount	Too little	Don't know	Refused	Total
Pensions	1.4	15.5	59.6	23.2	0.2	426
	(2.7)	*(19.1)*	*(74.3)*	*(4.0)*		*(1389)*
Social Security	17.6	30.3	33.6	18.3	0.2	426
	(29.5)	*(38.5)*	*(22.8)*	*(9.1)*		*(1389)*
Education	2.8	29.1	60.6	7.3	0.2	426
	(3.5)	*(35.5)*	*(57.4)*	*(3.5)*	*(0.1)*	*(1389)*
Reducing Water Pollution	2.8	26.3	35.9	34.7	0.2	426
	(1.8)	*(32.8)*	*(52.7)*	*(12.7)*		*(1389)*
Transport	12.7	32.4	40.1	14.6	0.2	426
	(9.7)	*(35)*	*(52.6)*	*(2.7)*		*(1389)*
Policy and Criminal Justice	7.7	24.9	47.4	19.5	0.5	426
	(7.9)	*(40)*	*(49.5)*	*(2.6)*	*(0.1)*	*(1389)*
Reducing Air Pollution	3.8	22.8	45.1	27.7	0.7	426
	(3.5)	*(29.6)*	*(58.8)*	*(8.2)*		*(1389)*
Health Care	2.6	27.0	65.7	4.5	0.2	426
	(3.0)	*(28.2)*	*(67.7)*	*(1.1)*	*(1.0)*	*(1389)*

Compared to *the CRP study* survey, the "don't know" response option has a much higher response rate in all categories (table 1). Across all public services, "*too little*" is the most frequently used response category regarding national spending. In relation to water pollution, public spending is considered "too little" by 35.9% of the respondents. In contrast, only 2.8% of the respondents consider that "too much" money is spent in reducing water pollution. It is worth to note that 34.7% of the respondents answered "don't know" to this question, which is the highest "don't know' response amongst all categories.

Table 2 Opinions on Scottish Water expenditure priorities - percentages (England and Wales in brackets)

Expenditure categories	**First Priority**		**Second Priority**	
Reducing the risk of drinking water discoloration	28.6	*(20.1)*	14.8	*(12.5)*
Reducing the risks of sewer flooding	22.8	*(30.7)*	23.7	*(28.9)*
Fixing leaks and pipes	22.3	*(34.0)*	13.1	*(22.8)*
Improving river water quality	8.5	*(5.7)*	12.0	*(10.7)*
Reducing the risks of interruptions to supply	4.7	*(3.7)*	9.2	*(8.0)*
Improving bathing water quality	0.7	*(0.8)*	4.5	*(2.5)*
Protecting animal and plant life around waterways	10.6	*(5.0)*	20.4	*(14.3)*
Don't know	1.9	*(0.1)*	2.3	*(0.3)*
total	426	*(1389.0)*	426	*(1389)*

As in the *CRP study* results for England and Wales, expenditures in the provision of water services is regarded as the most important spending priority to be undertaken by water companies (table 2). Reducing the risk of drinking water discoloration and reducing the risks of sewer flooding are being seen as the top first priority (28.6% of the Scottish respondents) and top second priority (23.7%).

Compared with England and Wales, water companies' spending on partially related WFD outcomes (i.e improving river water quality, reducing the risks of interruptions to supply, improving bathing water quality, protecting animal and plant life around waterways) score higher for both as a first priority (24.4% in Scotland as opposed to15.2% in E&W) and as a second priority (46.0% to 35.5% respectively). In relation to the specific main benefits of the WFD, "*protecting animal and plant life*" is the second most preferred option as a second expenditure priority with 20.4% choosing this option.

Table 3 Attitudes towards paying for environmental Protection and improvements

Statements	Scottish Survey	*E&W*
Protecting the environment is so important that pollution control requirements and standards cannot be too strict, and continuing improvement must be made regardless of cost	38.0	*(46.2)*
Protecting the environment is important and continuing improvements should be funded, provided that they are not excessively costly	43.2	*(39.4)*
We are spending about the right amount on cleaning up the environment and don't need to increase or decrease this spending	8.5	*(6.8)*
We have made enough progress on cleaning up the environment and should now concentrate on holding down costs rather than requiring stricter pollution controls	5.4	*(4.3)*
Pollution control requirements and environmental quality standards have gone too far and they already cost most than they are worth	2.6	*(3.0)*
Don't know	2.1	*(0.4)*
Refused	0.2	
Total	426	*(1389)*

The vast majority of Scottish respondents, as their English and Wales counterparts, are inclined to pay for improving and protecting the environment in general. 38% of the people interviewed in Scotland would agree that environmental improvements should take place regardless of costs (table 3). Almost half of the sample (43.2%) showed a strong preference for controls that are not excessively costly.

Table 4 Opinions on most important WFD benefits - Proportion of Sample

Most Important WFD Benefit	Proportion of Sample (%)	
Direct Use Benefits	Scotland	*E&W*
Improved conditions for fishing	5.9	*(2.7)*
Improved conditions for water contact activities. For example; canoeing, rowing, rafting, surfing, windsurfing, diving, wading, paddling, or swimming in the sea or rivers or lakes (not in swimming pools)	10.1	*(23)*
Improved conditions for other activities on, or around the water. for example; narrowboating, walking, running, cycling, or sitting nearby)	17.1	*(20.7)*
All use benefits	33.1	*(46.4)*
Other types of use/non-use benefits		
Knowledge of improved habitats for plants, fish and other animals	60.1	*(49.2)*
Other	0.0	*(1.9)*
All of equal importance	5.2	*(1.9)*
Don't know	1.6	*(0.6)*
Total	426	*(1389)*

Around two thirds of the people surveyed in Scotland stated that knowledge of improved habitats are, in their opinion, the most important benefits to be accrued from the WFD (table

4). Around a third of the sample (33.1%) stated that improved conditions for direct use benefits were most important.

Rivers and Lochs: Uses and perception of current status

Table 5 Uses of rivers and lochs in the last 12 months - Percentages

DIRECT USES		**Often** (More than 6 times)	**Sometimes** (Between 3 and 6 times)	**Rarely** (Once or twice)	**Never**	**Don't know**	**Total**
Fishing/Hunting	River	8.5	7.5	6.3	77.7	0.0	426
	Loch	6.3	8.7	4.5	80.3	0.2	426
Non-motor water based activity (e.g. swimming, canoeing....)	River	4.2	12.9	9.2	73.7	0.0	426
	Loch	2.8	8.5	5.9	82.6	0.2	426
Motorised water based activity (e.g. motor boating, Jet Skiing...)	River	0.5	2.6	5.2	91.8	0.0	426
	Loch	0.5	3.3	4.0	92.0	0.2	426
Enjoyment of natural beauty/peace and quiet (e.g. hiking, camping, walking, running, cycling, sitting nearby, wildlife observation....)	River	41.1	29.8	9.9	19.2	0.0	426
	Loch	20.9	31.2	14.3	33.3	0.2	426

Table 5 above illustrates the results from the question on uses of rivers and lochs by the respondents.

Table 6 Reactions to River and Loch water environment quality maps

	Rivers	**Lochs**
A great deal better than expected	2.3	0.2
Some what better than expected	15.3	6.8
About as expected	35.7	27.5
Somewhat worse than expected	39.4	50.7
A great deal worse than expected	4.5	12.4
Don't know	2.8	2.3
Total	426	426

Public perceptions of current water quality levels are somewhat different to scientific knowledge of current status at national level. For lochs and rivers, 50% and almost 40% (respectively) of the respondents stated that baseline water quality maps depict in overall a lower water quality picture than they had expected be the case (table 6). 12.4% of the respondents believe that loch water quality is much worse than they had expected. A little over a third of the respondents believe that national river water quality is about as they had expected and around a quarter believe that the same is true for national baseline loch water quality levels.