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Exploring the meaning of disproportionate costs for the practical implementation of the Water Framework Directive

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Abstract

The Water Framework Directive (WFD) is perhaps the most ambitious piece of environmental legislation in the history of the European Union. The Directive 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 basic cost-benefit theory, this paper attempts to clarify the meaning of disproportionate cost to non-economists, and to convey a consistent interpretation that should underlie the development of a practical derogation decision making across all member states

KEYWORDS: Derogations, Cost-Effectiveness Analysis (CEA), Cost Benefit Analysis (CBA), Marginal Abatement Costs (MAC), Marginal Social Costs (MSC)

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Introduction

The European Water Framework Directive (WFD) (EC/2000/60) was adopted in 2000 with the aim of consolidating and improving European water resource management. The WFD establishes a legal framework within which to protect surface and ground waters using a common management approach and following common objectives, principles, and basic measures. It also integrates the existing European water legislation into a common framework (De Nocker et al, 2007).

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 (EC, 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 (EC, 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 (EC, 2002). 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 paper 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 paper 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 rudimentary 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.

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. Thus, costs of use, cost recovery, the recognition of the need to value benefits… 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. Economic theory does provide a useful reference point.

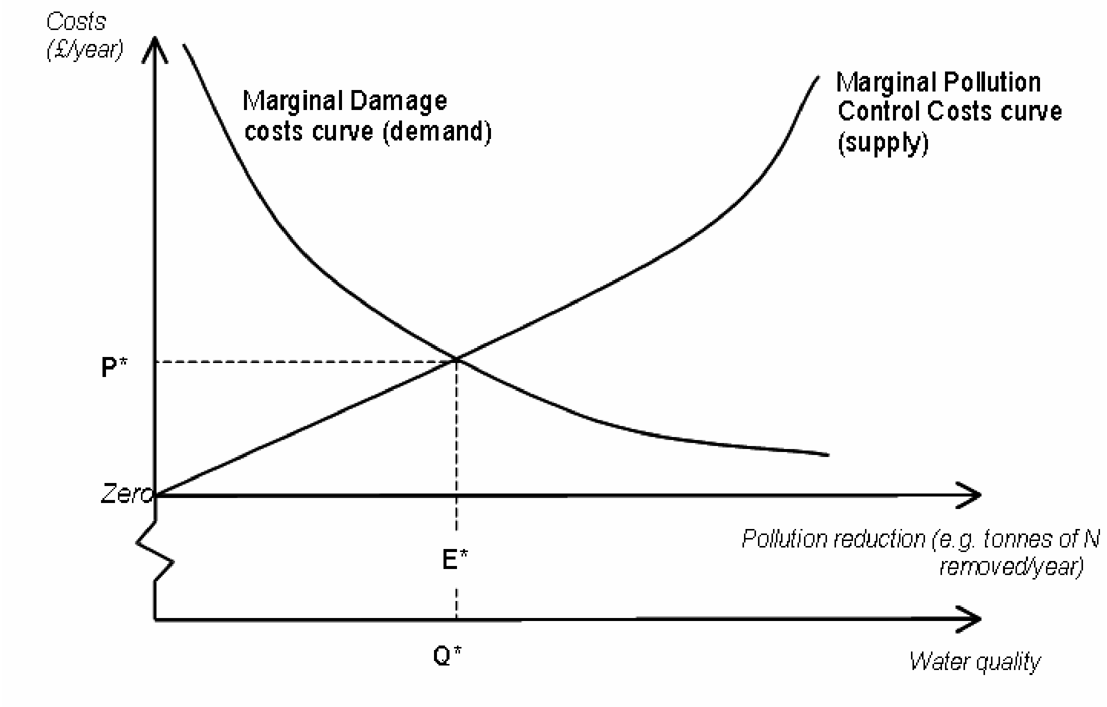
From a neo-classical welfare economics perspective, environmental degradation is depicted as one in which the activity of an economic agent (any economic agent: households, firms, governments) imposes external costs upon the rest of society in the form of pollution (Baumol and Oates 1988). This damage 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 1 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 1: 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[3]. 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 (MCA) and the environmental damage costs curve represents social costs (MSC) under perfect competition (Hanley et al, 1997).

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) 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.

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. Clearly 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

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[3] Note the important difference in economics between total and marginal costs. Marginal cost indicates the change in costs as we consider reducing one more unit of pollution.

pollution will be enough to achieve 'good status' by 2015, and if not, which 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[4] 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 1). The table also describes how different standards are being designed.

[4] 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 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 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 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.

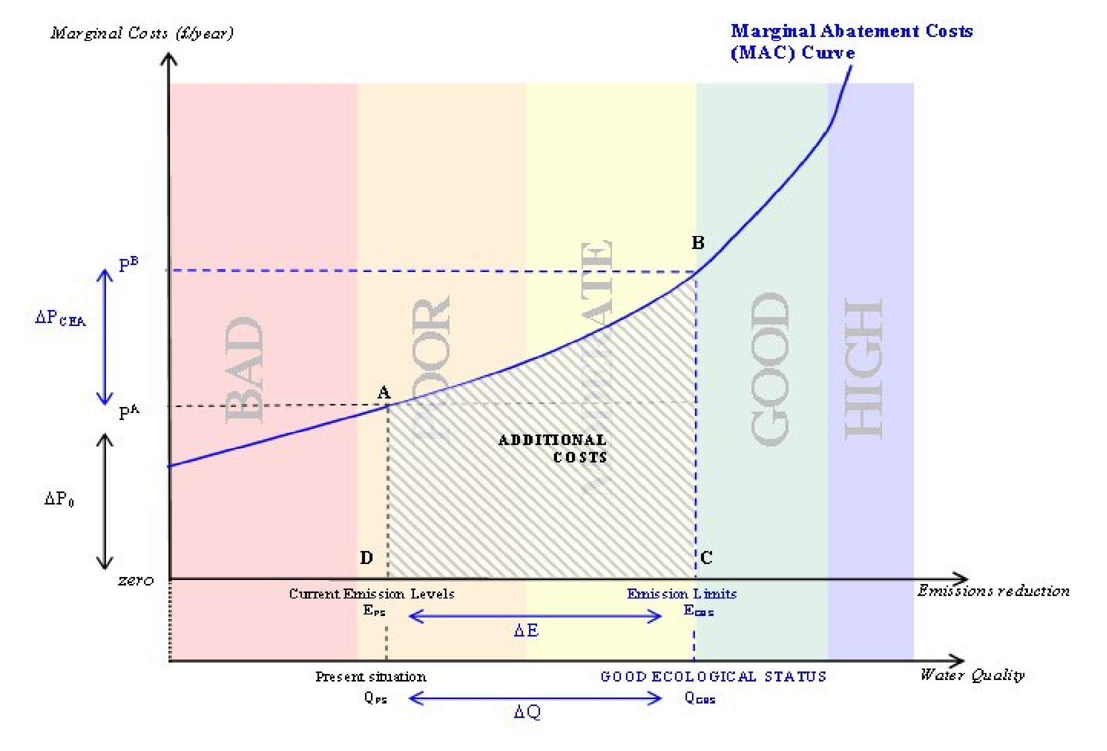
The Cost-Effectiveness Analysis (CEA)

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 (EC, 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 2 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[5] class boundaries under the WFD (bad, poor, moderate, good and high), options are ranked in increasing order of their costs per emission reduction unit.

Figure 2: The Cost-Effectiveness Analysis of water quality improvements options



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 2, 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 an 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 2***).

[5] Good status is the combination of good ecological status and good chemical status, for simplicity we now focus our analysis in the achievement of good ecological status

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[6]. Note that this analysis is described without reference to benefits other than the prescribed level of good water status.

Disproportionate Costs: a first interpretation

Consider now figure 3 that 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.

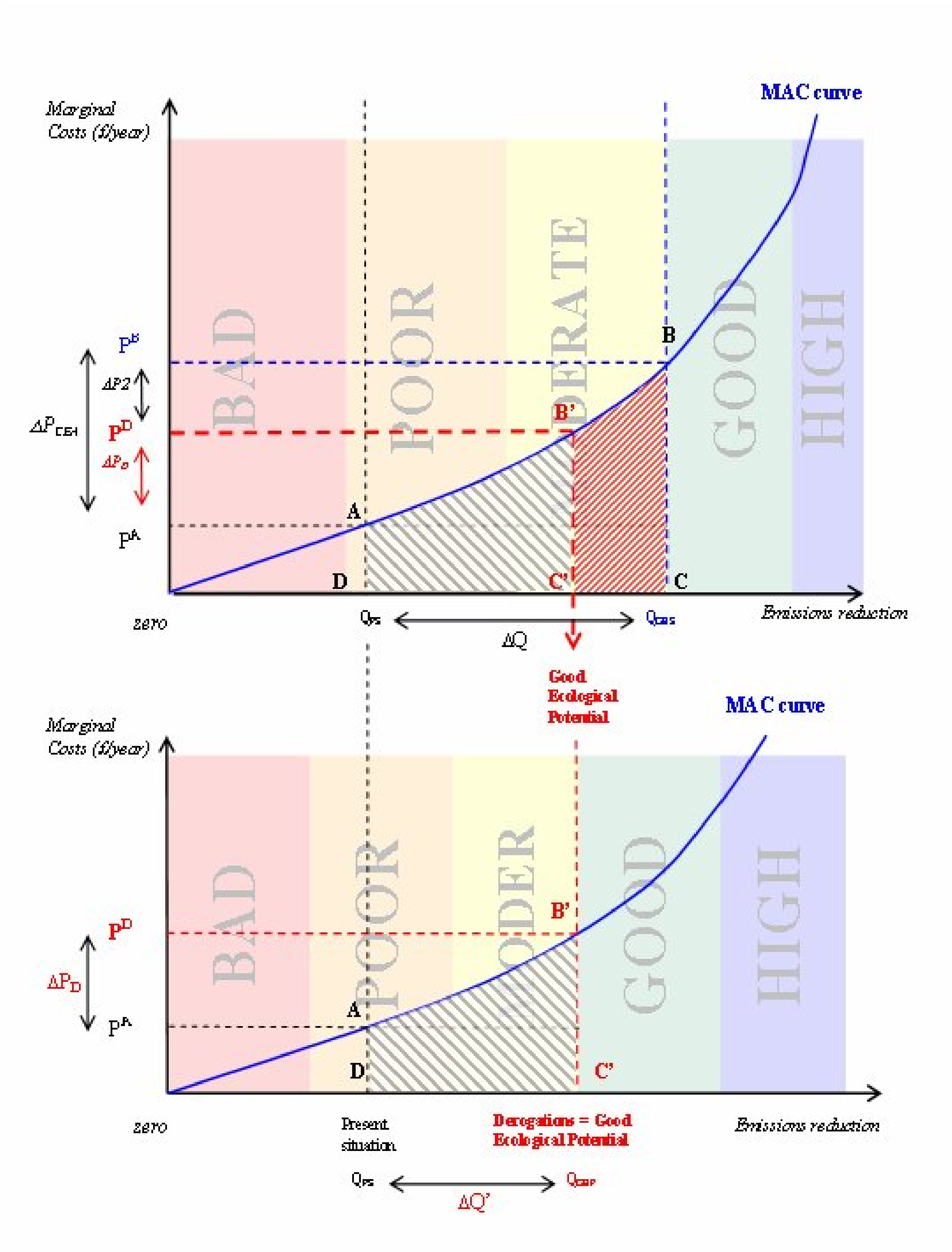
Under the WFD, this situation leaves the firm with two possible options. First, the firm may either seek to be granted time-framed derogations/exemptions. This would allow the firm to wait until new abatement technologies are available, which can reduce its overall marginal costs of compliance, and for the regulators there would be no need to lower the environmental standards. This means introducing some sort of 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***), 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 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***) 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.

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

Figure 3: Graphical representation of Disproportionate Costs



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[7]. 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 sociably 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[8]. 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 deriving from whatever uses are made of the river.

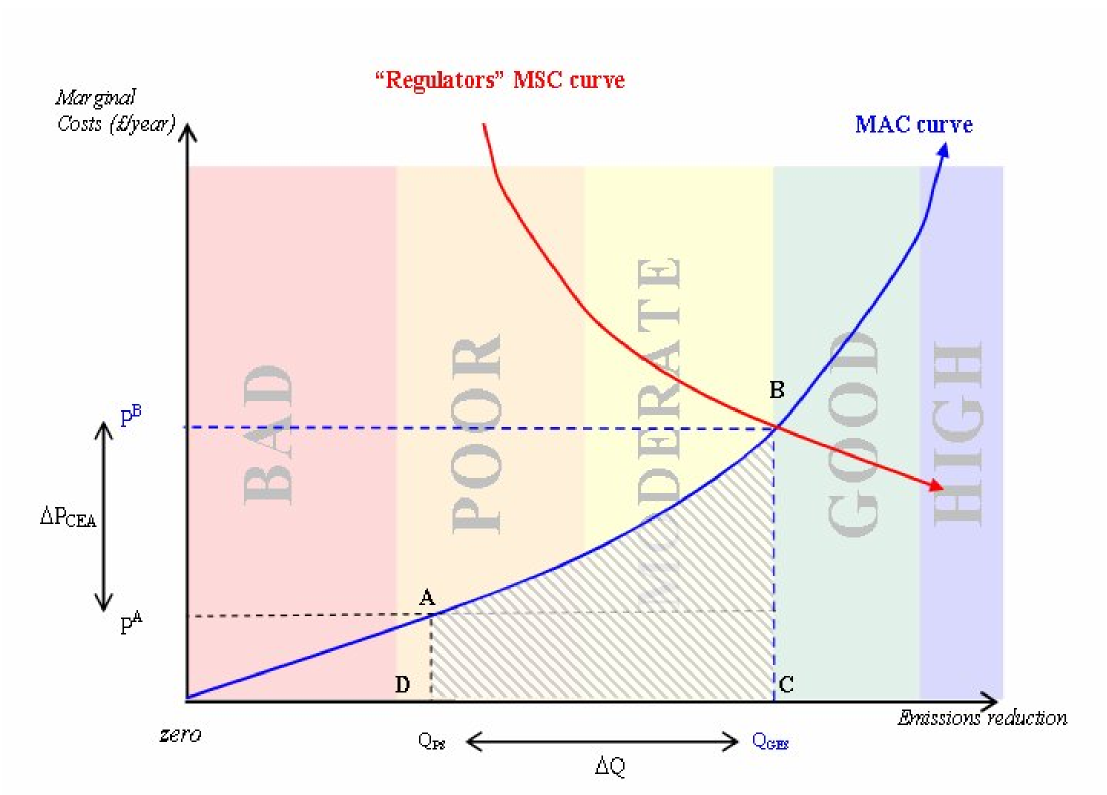
This part of the story is considered in the marginal social cost (MSC) curve (see both ***figures 4 and 5***). This curve reflects a decrease in damage costs to society (or conversely reflects the social benefit). Initial low cost abatement delivers high social costs, which progressively fall as the firm's pollution control costs increase by one extra unit. From 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 1***).

Due to the uncertainties surrounding the monetary estimation of the damage costs functions, which are mainly associated with the economic valuation of environmental improvements[9], regulators normally set the standards under other criteria. In this case, GS is defined as a function of those environmental standards necessary to support the

[7] 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.
[8] Note that this remains true even if the uses are passive or non use "existence" benefits.

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 curve (***figure 4***). 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 4)

Figure 4: The Cost-Benefit Analysis, assuming the shape of the benefits function



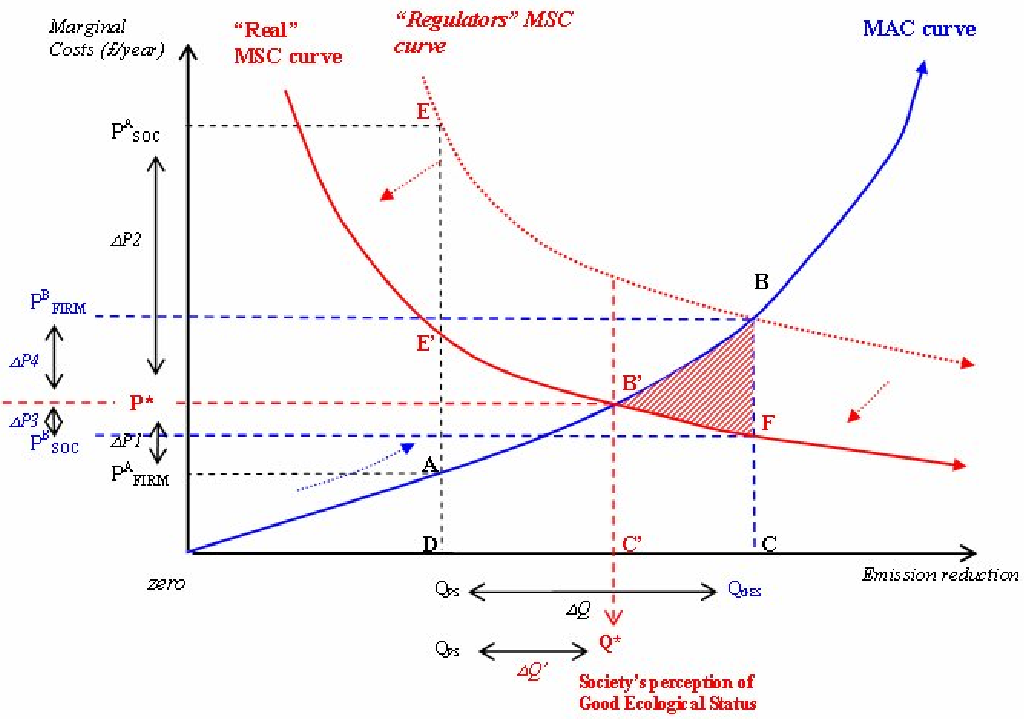
However, ***Figure 5*** shows the economic inefficiency of the standards-based system when the "real" MSC is introduced. In this hypothetical situation[10], the area BB'F (***figure 5***) represents the net loss to society as a whole, including the firm, of reducing pollution to Q_{GES} instead of Q*, 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

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

[10] 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.

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

Figure 5: Economic inefficiencies associated with assuming the shape of the damage costs curve



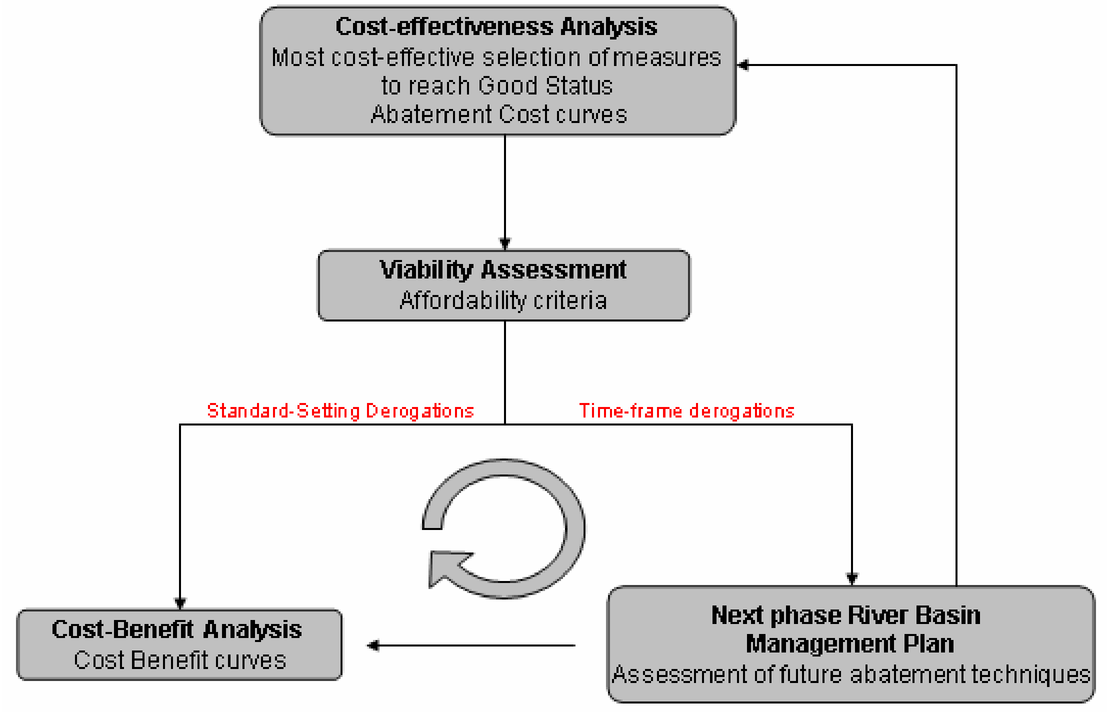
Lowering the environmental standards for a specific water body or allowing a polluter to maintain its emissions levels on the grounds of a disproportionality test, may prove one of the most controversial steps on the implementation of the WFD. Decisions may reveal issues of competitiveness between water users or uneven distribution of the financial costs associated with the Directive (Pearce 2004). 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 5) involved in complying with the Directive.

Assessment of disproportionality in practice

In this paper, 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.

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 6 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?

Figure 6 Methodological steps for the assessment of disproportionate costs



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 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 2006). 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 carry 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 marginal 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) sociably justifiable under 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. As we have introduced in this paper, the rationale for the application of CBA to justify standard-setting derogations aims to find some sort of economic efficiency on the exemptions decision making process.

There are evidently many uncertainties associated with this analysis, which are beyond the scope of this paper. 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 attribution of the responsibility to individual farmers. These uncertainties are the subject of much on-going research in Member States.

Discussion

Overall, the Water Framework Directive sets a clear course of action for many of its elements, including most of its economic elements. 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 regulators). This has been called the "public-trust" doctrine, which makes the goal of policy in face of damages, 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 the costs are found prohibitive. This differs from the normal use of CBA in policy analysis, which is widely used 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 find out 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.

Secondly, discrepancies also allow flexibility in the practical interpretation of disproportionate costs at member state level, which some countries may exploit to apply different definitions of disproportionality and different methods to reach decisions about exemptions.

Conclusion

The WFD (EC, 2000) and subsequent guidance documents on the interpretation of its economic elements (EC, 2002) provide limited guidance on the meaning of disproportionate costs for the justification for exemptions in the achievement of Good Status. This paper shows that economic theory provides a rudimentary 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 differing 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.

References:

Baumol, W., and W., Oates. (1988). The Theory of Environmental Policy, Cambridge, UK: Cambridge University Press.

Beaumont, N., J., and R., Tinch (2004) *Abatement Cost Curves: A Viable Management Tool for Enabling the Achievement of Win-Win Waste Reduction Strategies?* Journal of Environmental Management. Vol 71. pp 207-215

Chiang, A., (1984) *Fundamental Methods of Mathematical Economics*, 3rd ed. 1984, McGraw-Hill.

DEFRA (2006) *Cumulative impact of forthcoming regulatory proposals on the economics of farming – Update 1*. June 2006 http://www.defra.gov.uk/farm/policy/regulation/charge/pdf/cumulative-burdens.pdf

ECO2 (2004) *Assessment of Environmental and Resource Costs in the Water Framework Directive*. Information sheet prepared by Drafting Group ECO2. common Implementation Strategy, working group 2B. Brussels . June 2004. http://www.waterframeworkdirective.wdd.moa.gov.cy/docs/OtherCISDocuments/Economics/ECOResouceCosts.pdf

Ecologic (2005) *The Case for Valuation Studies in the Water Framework Directive.* Report for the Scotland and Northern Ireland Forum For Environmental Research (SNIFFER). Project WFD55. Edinburgh 2005.

European Commission (2000) *Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy*. Official Journal 22 December 2000. Brussels.

European Commission (2002) *Economics and the Environment. The implementation challenge of the Water Framework Directive. A guidance document*, Directorate for the Environment: Brussels.

Hanley, N., Shogren, J. F. & B. White (1997) Environmental Economics: In theory and Practice. First Edition. Macmilllan Press Limited. London. 1997

Hanley, N. and A. R. Black (2006) *Cost-Benefit Analysis and the Water Framework Directive in Scotland*. Integrated Environmental Assessment and Management. Vol 2, Issue 2, pp 156-165.

Lago, M., Mouratiadou, I., and Moran, D., (2006) *Navigating the Water Framework Directive: Implications for Agriculture and Beyond.* Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources. 1. No 061

De Nocker, L. Broekx, S., Liekens, I., Görlach, B., Jantzen, J. and P. Campling (2007). *Cost Benefit Analysis on the implementation of the Water Framework*

Directive including a specific focus on agriculture: Final Report. Report for DG Environment – Version 6.
Pearce, D.W., (1974) *Economic and Ecological Approaches to the Optimal Level of Pollution*. International Journal of Social Economics. Vol 1, Issue 2 pp 146-160.
Pearce, D.W. and R. Kerry Turner (1990) *Economics of Natural Resources and the Environment*. Harvest Wheatsheaf.
Pearce, D., (2002) *The Role of Property Rights in Determining Economic Values for Environmental Costs and Benefits*. Report to the Environment Agency. Bristol. December 2002.
Pearce, D.W., (2004) *Does European Union Environment Policy Pass a Cost-Benefit Test?* World Economics. Vol 5, No. 3 pp115-137.
Pearce, D.W., Atkinson, G. and Mourato, S. (2006) *Cost-Benefit Analysis and the Environment: Recent Developments*. OECD publications. February 2006.
RPA (2004) *CEA and Developing a Methodology to Assess Disproportionate Costs*. Final report for DEFRA, WAG, SE and DoENI. July 2004. http://www.defra.gov.uk/ENVIRONMENT/water/wfd/economics/pdf/ceafrreport.pdf#search=%22disproportionate%20costs%20rpa%20cba%20wfd%22
Tietenberg, T. (1992). *Environmental and Natural Resource Economics*. Harper Collins, New York.
UK-TAG (2006) *UK environmental standards and conditions (Phase 1)* http://www.wfduk.org/UK_Environmental_Standards/ES_Phase1_final_report/LibraryPublicDocs/UKTAG%20ReportAug%202006UKEnvironmentalStandardsandConditionsFinalReport
Varian, H., R., (2003) *Intermediate Microeconomics: A Modern Approach***.** Sixth Edition. *University of California, Berkeley*. 2002

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