Journal of Agricultural Economics, Vol. 57, No. 2, 2006, 221–237

Analysing the Agricultural Costs and Non-market Benefits of Implementing the Water Framework Directive

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Abstract

Implementation of the Water Framework Directive (WFD) represents a fundamental change in the management of water in Europe with a requirement that member states ensure 'good ecological status' for all water bodies by 2015. Agriculture is expected to bear a major share of WFD implementation costs as it is compelled to reduce the emission of diffuse water pollutants. The research outlined here comprises interdisciplinary modelling of agricultural land use, hydrology and consequent water quality effects to consider both agricultural costs and the non-market recreational use (and potentially non-use) values that implementation of the Directive may generate. A theme throughout the research is the spatial distribution of the costs and benefits of WFD implementation, which is addressed through the use of GIS techniques in the modelling of agricultural land use, the integration of land use and hydrological models, and the estimation, aggregation and transfer of the economic value of the benefits.

Keywords: *Benefits transfer; GIS; non-point pollution; Water Framework Directive.*

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1. Introduction: The Water Framework Directive

The Water Framework Directive (WFD) (European Parliament, 2000) represents a fundamental change in the management of water quality in Europe. The Directive imposes outcome-based targets, requiring a shift away from standards framed in terms of the chemical composition of water in favour of an approach which assesses the ecological quality of water bodies. These standards will be water body specific and hence require differentiated action. However, there is a general requirement to improve all European waters to 'good ecological status' by 2015. Although the definition of such status depends upon reference conditions, it is generally agreed that implementation of the WFD will require substantial reductions in pollutant inputs to rivers both from point and diffuse sources. While good ecological status is required, there is reference within the Directive to the costs of such improvements and the scope for time derogations and the setting of less restrictive targets because of disproportionate costs. The assessment of disproportionality is to be by Member States and is intended to be based on an economic assessment.[2]

The WFD specifically requires the control of diffuse emissions into water bodies, the primary source of which is agriculture. The impacts of the WFD in the UK are uncertain but may well be substantial. A recent report suggested that to meet the requirements of the WFD by simply reducing inputs and production might require large-scale changes in agricultural land use including reductions in fertiliser application rates to crops and grass by 50%, halving sheep stocking rates and cutting cattle stocking rates by 25% (Haygarth *et al.*, 2003). Such major changes are serious for an already economically fragile rural economy and appear to clash with other official priorities for supporting the farming sector (DEFRA and HM Treasury, 2004). Another area of uncertainty concerns the benefits which implementation of the WFD might generate. Many benefits from WFD implementation are anticipated including enhancing the ecological and chemical status of waterbodies and wetlands and reducing pollution of them, lessening the frequency and extent of floods and droughts, encouraging participation in water-based recreation, etc. The economic benefits are likely to be many although only a minority are likely to be easily amenable to quantification, for example, reduced water treatment costs. One important motivation for the WFD appears to be the creation of non-market environmental benefits, such as open-access recreation (see Articles 4, 9 and 11 of the WFD and Environment Directorate-General, 2005).

In this context there is an urgent policy requirement for better understanding of the aggregate agricultural costs and non-market benefits of WFD implementation, as well their distribution and the ability of those bearing the costs to do so. However, it seems self-evident that the impacts of the policy will be geographically uneven. The extent and spatial variability of the changes required is illustrated in Figure 1, which shows the current assessment of the risks of rivers in England and Wales failing WFD objectives in terms of phosphate, nitrogen and pesticide levels. As can be seen, the changes required for WFD compliance are likely to be substantial and geographically variable.

[2] Disproportionality is not interpreted simply as where costs exceed benefits. The ability to pay of those affected by the measures is also to be part of the assessment (RPA, 2003).

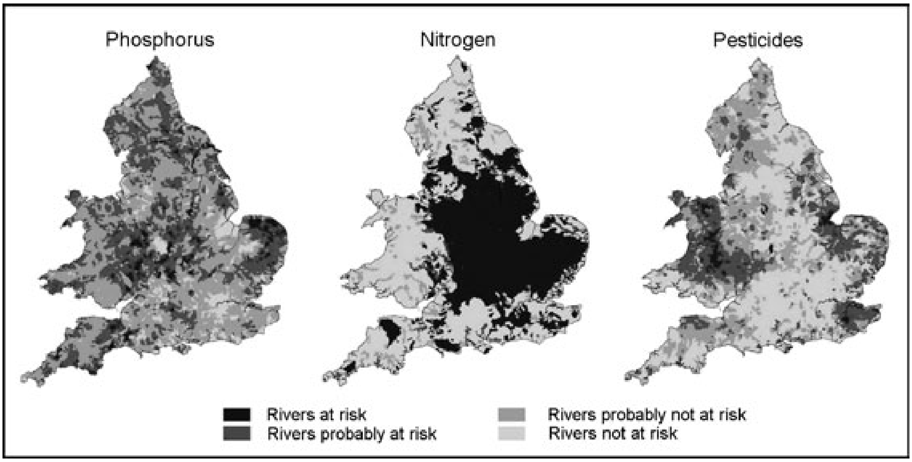

Figure 1. The risk of rivers in England and Wales failing WFD objectives.
***Source*:** Adapted from Environment Agency (2004).

Variation in the physical environment and agri-economic context implies that WFD implementation will involve spatially differentiated responses from farmers and land managers. For example, contrasting soil types, topography, rainfall, etc. could mean that the impacts of a given land use upon water quality will differ radically by location. Similarly the benefits of WFD-induced water quality changes will vary across space in a complex manner, depending not just on the distribution and physical response of catchments, rivers and estuaries, but also upon the distribution of present and potential future beneficiaries. A primary challenge for research is, therefore, to capture the complex interplay and spatial distribution of key costs and benefits of possible WFD implementation strategies.

The Catchment Hydrology, Resources, Economics And Management (ChREAM) project,[3] recently funded under the RELU programme, sets out to assess the agricultural costs and key non-market benefits associated with the introduction of the WFD and to consider the impacts of alternative implementations of that policy. These impacts will involve geographically varied changes in land use patterns and water quality. Hence the spatial dimension is central to the project's aims and methodology. The project comprises integrated hydrological, economic, agronomic and geographical elements designed to allow analysis of the effects of WFD implementation from required changes in water quality back through the hydrological system to consequent constraints upon land use and resulting impacts upon farm revenues

[3] The ChREAM project (RES-227-25-0024) is funded by the UK Research Councils' Rural Economy and Land Use (RELU) Programme. RELU is funded jointly by the Economic and Social Research Council, the Biotechnology and Biological Sciences Research Council and the Natural Environment Research Council, with additional funding from the Department for Environment, Food and Rural Affairs and the Scottish Executive Environment and Rural Affairs Department. For project details see http://www.uea.ac.uk/env/cserge/research/54.htm. A paper detailing the research is provided at http://www.uea.ac.uk/env/cserge/pub/wp/edm/edm_2006_05.htm.

and profits. To do this an integrated model of land use and hydrology at the catchment scale is being developed.

There have been important precedents combining land use and hydrological models, perhaps most notably in the UK, the NERC/ESRC Land Use Modelling Programme, NELUP (O'Callaghan, 1995, 1996; Moxey and White, 1998). In light of this, the core of this paper focuses upon the contemporary intellectual and practical challenges facing the ChREAM project as it begins its research, and the methodological and empirical contributions it seeks to make. By way of introduction, the study area and existing hydrological models of that area are now briefly described and the land use modelling approach outlined. The paper then assesses the challenges for and contribution to be made by the project in the assessment of the scale and distribution of WFD non-market benefits. The paper concludes by discussing the project's approach to interdisciplinarity.

2. Integrated Hydrological–Economic Modelling of the Agricultural Costs of WFD Implementation

2.1. The study area

The Humber basin (Figure 2) was chosen as the study area. The basin covers an area of some 25,000 km^2, most of which is put to diverse agricultural uses, and has a population of over 10 million people. The basin drains 28% of England, mainly via its two principal river catchments, the Ouse and the Trent.

A comparison of Figures 1 and 2 indicates that the Humber is an ideal case study area as it captures a full range of emission levels of key pollutants such as phosphates, nitrogen and pesticides. In addition to its overall size and significance, the contrasting characteristics of sub-catchments across the Humber basin provide useful variation for subsequent extrapolation.

2.2. Hydrological modelling

The Humber catchment has been the focus of considerable prior modelling including the EU EUROCAT[4] and Land Ocean Interaction Study (LOIS[5]) programmes (Cave *et al*., 2003; Neal and Davies, 2003; Andrews *et al*., 2005); both involved collaborative work between various of the ChREAM researchers. The hydrological model within the ChREAM project integrates this prior work with an existing 'catchment supply' model (CASCADE[6]) which combines flows of water, nutrients, pesticides, etc. across hydrologically representative units (HRUs[7]). The ChREAM

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[4] 'European catchments: catchment changes and their impact on the coast' (EUROCAT) was an EU FP5 project modelling point and diffuse sources of nutrients in seven catchments including the Humber.

[5] The LOIS programme (1992–98) was funded by NERC and included simulating fluxes and transformations of sediments, nutrients and contaminants from the catchment to the edge of the continental shelf.

[6] The CASCADE (Catchment Scale Delivery) model simulates the catchment scale transfer of nutrients, sediment and other material from the land surface and the soil, through a river network to a catchment outlet.

[7] HRUs are distinct sub-catchment units, typically 2–8 km^2 in extent.

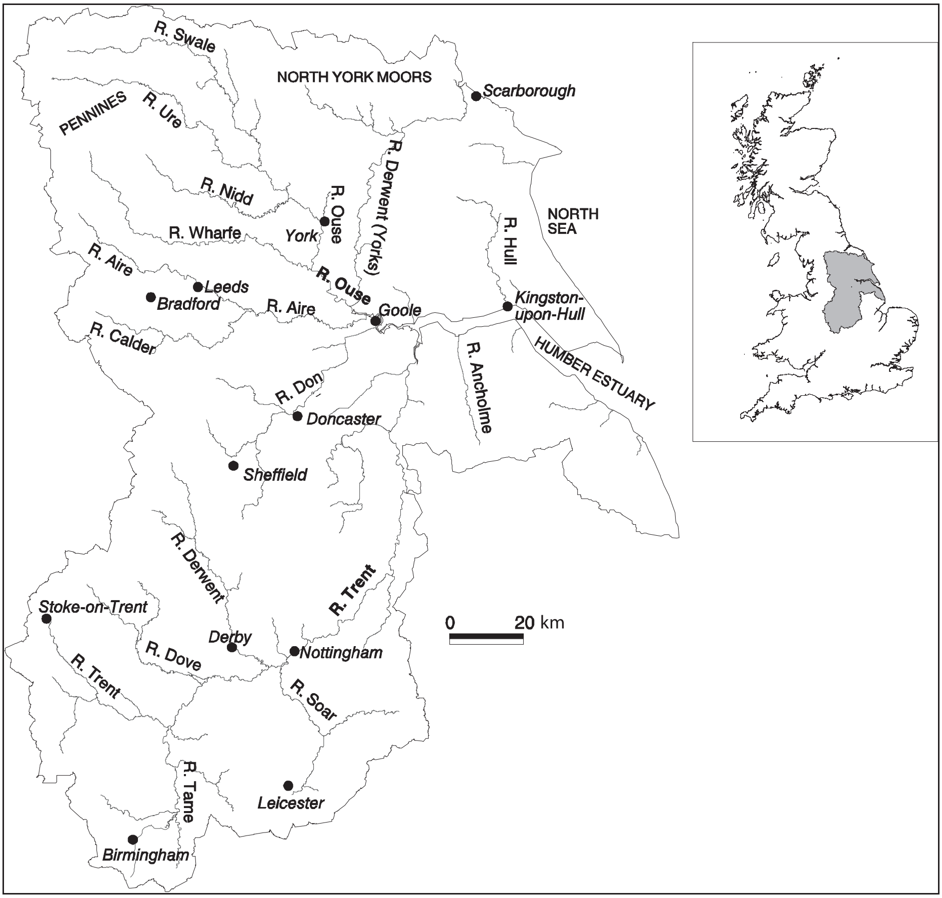

Figure 2. The case study area: the Humber Basin's rivers and major cities.
Maps are based on Ordnance Survey material with the permission of Ordnance Survey on behalf of the Controller of Her Majesty's Stationary Office © Crown copyright. All rights reserved. NERC, 100017897, 2005.

model also incorporates a further module to assess point source discharges, e.g., from sewage treatment works (Kay *et al*., 2005). Synthesis with an 'in-stream' component (QUESTOR[8]) is intended to capture pollutant transport and impacts (Hutchins *et al*., 2006).

An initial research objective is to capture the processes by which potential pollutants such as nitrogen, phosphorus, pesticide and faecal material are attenuated before entering the river channel. This entails modelling how the application of, say, a specified fertiliser at a given location will, after accounting for the environmental and land use characteristics of that location (soil type and depth, slope, underlying geology, cropping pattern, etc.), leach into water bodies (Davies and Neal, 2004; Neal *et al*., 2005). The modelling must capture the modification of diffuse pollutants in-river, supplemented by the addition of point source discharges

[8] QUESTOR (Quality Evaluation and Simulation Tool for River-systems) is software used to support in-stream water quality modelling.

and analyse how together these pollutants affect aquatic biology. In combination, such analysis takes us from a given land use and nutrient application regime at a specified location, to, for example, the prediction of the rate of algal bloom within a receiving waterway.

The spatial dimension is crucial to all aspects of the project, from water process modelling, through land use interactions, to the analysis of the non-market benefits of water quality improvement. Consequently, the project will make extensive use of spatially sensitive modelling routines and geographical information systems (GIS).

2.3. Land use modelling and integration with the hydrological model

A number of studies have already examined the impact of water pollution policies upon agriculture (Rigby and Young, 1996; Rigby, 1997; Gömann *et al*., 2004). Within the ChREAM project, farm activities will be modelled by an econometric process analysis (Antle and Capalbo, 2001) estimated using Farm Business Survey (FBS) panel data, Agricultural Census information, and other geographical context variables. Production decisions will be specified as a function of farm characteristics, inputs, bio-physical characteristics of the production site, and market conditions. Other types of production models, such as farm programming models, derive optimal resource allocations for one or more farms which are 'representative' of the area or region. These models are extremely amenable to the imposition of (environmental) restrictions, for example, through restrictions in fertiliser application or abstraction rates. However, they are more limited in application where there is highly significant spatial variation and heterogeneity both in environmental characteristics and processes and in farm-economic behaviour. For example, the relative homogeneity of production systems on the two Scottish catchments modelled in Hanley *et al.* (2006) allows production to be modelled as a single representative farm producing potatoes, arable crops and grassland. This is in stark contrast to the size of, and heterogeneity within, the Humber catchment with farm types ranging from intensive arable or livestock production in lowland areas to more extensive mixed enterprises in upland regions. In the proposed approach, the farm or site-specific spatial information are included in the model to control for such effects. This will remove possible biophysical sources of bias in the estimates.

The integration of farm economic and hydrological models will be achieved at the scale of HRUs. The 'catchment supply' model, CASCADE, is based on monthly profiles of nutrient inputs (e.g., nitrates) from a range of land use (e.g., different crop type) and livestock sources. Ward scale data from the DEFRA June Agricultural Census[9] on land cover and livestock numbers will be interpolated to provide baseline profiles for HRUs[10] and also provide a framework for scaling up production decisions modelled at the farm scale to generate alternative sets of such profiles. This will allow different economic or policy scenarios to be modelled in terms of their implications for such variables as fertiliser use. These estimates will, in turn, become inputs for the hydrological models. By exogenously setting water quality to some policy determined level (such as that commensurate with 'good ecological status' under the WFD) it should also be possible to drive these linkages 'backwards'

[9] See http://www.defra.gov.uk/esg/work_htm/publications/cs/farmstats_web/default.htm

[10] For a discussion of relevant techniques see, for example, Moxey and Allanson (1994).

from the hydrological to farm models, determining the maximum level of farm inputs commensurate with that particular outcome.

The economic welfare implications of any state (pre- or post-WFD) of the production process can be captured via the associated profit function given the expected output price. This procedure simultaneously estimates both supply and cost functions and is designed to incorporate alterations to parameters such as the output price changes associated with CAP reform or revisions to agri-environmental policy regimes.

The integrated nature of this modelling approach and its ability to represent different levels of farming activity also provides a framework to analyse farmer reactions to shocks such as a limit on input use resulting from WFD implementation measures. Limits upon fertiliser inputs are likely to generate an incentive for farmers to substitute away from intensive nutrient input production towards alternative activities. However, sole reliance upon a model estimated within one state to accurately predict for the potentially substantially different farm business environment post-WFD implementation is risky. Therefore, the response modelling exercise will be supplemented in the research by a two-stage farm survey. In the first stage, information will be gathered regarding farmers' own estimates of their likely response to the input and activity constraints imposed by WFD implementation. This will feed into the integrated modelling information through the probabilities of simulated response strategies. The second survey will take resulting model predictions back out to farms and elicit their response to these estimates, these responses being incorporated within a further revision of the model. This allows refinement of the models and their simulated responses to WFD implementation to better capture the range of adjustment measures producers undertake. The surveys will also be used to test various models of farm behaviour (a recognised research priority for WFD implementation; Haygarth *et al*., 2003), especially deviations from a standard profit maximising assumption. These will again be used to refine the agricultural simulation process.

3. Assessing the Non-market Benefits of the WFD

Thus far the focus has been upon the means of assessing the scale and distribution of the agricultural costs of WFD implementation. It is claimed that the Directive will also deliver substantial benefits. However, perhaps surprisingly for such a major policy change, there does not appear to have been any formal economic assessment of these benefits. While the policy may generate a number of market benefits, such as reductions in water treatment costs for water companies and manufacturers, our research focuses upon the less tangible, but potentially major, non-market values generated by the WFD. These benefits include use values such as improved opportunities for, and qualities of, informal recreation and non-use benefits, such as the values individuals may hold for improvements in wildlife habitat which are not incorporated within recreation and amenity values. These benefits form an important part of the assessment of the proportionality of costs which should inform WFD implementation.

3.1. Use and non-use values and preference robustness

It is an early research objective of the ChREAM project to decide the non-market values upon which to focus and the methods most appropriate for estimating those values. In part, this focus will be informed by interactions with stakeholder groups including representatives from DEFRA, the Environment Agency, Ofwat, Yorkshire Water and the CLA. An overriding concern will be the likelihood of providing significant contributions to methodology in this field. There are a number of possible methods to be considered. Where preferences for environmental goods are well formed (as described in basic economic theory[11]) then it is a reasonable test of valuation techniques as to whether robust and accurate estimates of corresponding values can be obtained. However, if preferences are only poorly formed[12] (and perhaps internally inconsistent) then conventional economic valuation techniques of this kind cannot provide robust and accurate measures of such constructs. The benefits generated by the WFD are likely to provide examples of both well-formed and poorly formed preferences. For example, compared to inexperienced visitors, high-intensity recreational users of waterways, such as anglers, are more likely to have well-formed preferences for those resources and have robust values for the changes generated by the WFD. Following Plott (1996), it is the high degree of consumption (use) experience which leads to the development of robust, theoretically consistent, economic preferences. Following this logic, it seems likely that the absence of such experience is likely to mean that preferences for the non-use benefits of the WFD will be less well formed and, therefore, derived values are more liable to exhibit anomalies relative to standard theoretical expectations.

Arguably, these problems can be exacerbated by the fact that such non-use values can only be elicited through ‘stated preference’ (SP) methods (Bateman *et al*., 2002). These methods apply surveys designed to directly ask a representative sample of respondents about their values for some change in the provision of a good, such as a WFD-inspired increase in the quality of river water and the consequent effects upon flora and fauna. SP studies typically obtain these values *via* one of two methods:

- By asking respondents in a choice experiment (CE) to choose their preferred option among several, with each option defined by a series of attributes of the river and some cost attribute. Implicit trade-offs can be observed and values inferred from the (repeated) choices made.
- By asking respondents direct contingent valuation (CV) questions regarding willingness-to-pay (WTP) or willingness-to-accept (WTA) compensation for changes in the provision of some good.

Each approach has strengths and weaknesses. However, both are liable to the charge that, armed only with poorly formed prior preferences, respondents may react to SP surveys by ‘constructing preferences’ based upon what they perceive to

[11] For example, they are transitive; if A is preferred to B and B preferred to C then A should also be preferred to C. Such theoretically consistent preferences are the basis of economic decision-making techniques such as cost–benefit analysis.

[12] For example, if the value of a good varied according to factors such as the presence or absence of an inferior substitute (Bateman *et al*., 2005). Notice however, that such effects can occur for market priced goods (Doyle *et al*., 1999) leading to the likelihood that double standards are applied to valuations depending upon whether they are elicited within or without markets. Indeed Shogren (2006) argues that all values, market or otherwise, are contextual and, therefore, to some degree malleable according to the frame in which they are obtained.

be cues given by the valuation survey itself or the context in which it is undertaken (Tversky and Kahneman, 1974; Bateman and Mawby, 2004).[13] Such preferences are malleable, changing with the frame of the question rather than just the nature of the good and as such are frequently judged to be unsuitable for inclusion within a strict cost–benefit analysis. Even when prior preferences are well formed, unfamiliarity with such hypothetical markets adds a further layer of complexity to the uncertainty associated with non-use values. In effect, respondents need not only to 'discover' their preferences for such goods, but also learn how these valuation methods work (Braga and Starmer, 2005).

Addressing such research challenges is an important part of the justification for projects such as ChREAM and successful SP analysis of benefits has the potential to provide a rich diversity of information about such values. Extending the Plott argument, it may be possible for individuals to develop robust, consistent preferences and an associated understanding of contingent and choice markets as they experience valuation experiments. Techniques to promote such learning in valuation studies offer some promise here, including those pioneered by Bjornstad *et al.* (1997) and our own work on the transparency of valuation tasks (Bateman *et al.*, 2004). The ChREAM project will seek to extend previous work in this area, for example, applying and extending research in the psychological sciences showing that the presentation of information in visual, as opposed to numeric and textual, form can substantially reduce choice errors (Hibbard and Peters, 2003). In our own work, we have used virtual reality visualisations as a medium for enhancing understanding and eliciting values for land use change (Bateman *et al.*, 2006b; Dockerty *et al.*, 2006). Throughout such exercises, the key test is to assess the comparative stability and theoretical consistency of preferences both before and after the application of such preference discovery and visual information tools.

It seems clear that non-use, rather than use, preferences are likely to be less well formed and, therefore, more difficult to assess. Whether or not our own research will include non-use values remains, at present, an open question. However, the greater robustness of users' preferences does suggest that these provide a superior target for applying these techniques. Furthermore, unlike the non-use case, in estimating use values, we are not constrained solely to SP methods. We can also examine users' preferences via their behaviour as analysed through revealed preference (RP) techniques.

3.2. The spatial aspect of revealed preferences (RP)

The most commonly applied RP technique is the travel cost random utility model.[14] This examines data on recreationalists' choices of visits to a variety of sites offering

[13] Indeed it may well be that, even when preferences are well formed, respondents may 'truthfully overstate' their values because of the 'focusing illusion' (Kahneman and Sugden, 2005) that '*nothing in life is as important as you think it is when you're thinking about it*'. For this reason the Revealed Preference analysis discussed subsequently might be particularly important.

[14] Such methods actually rely upon the same underlying random utility model of behaviour (McFadden, 1974) as SP approaches. However, the RP use of observed behaviour is often viewed as an advantage. Whether this aura of respectability is completely deserved is a point of debate as RP methods are dependent upon a number of assumptions holding (see, for example, Randall, 1994; Champ *et al.*, 2003).

different recreational experiences. These experiences are characterised by the attributes of each site including its water quality, facilities and accessibility. Attributes vary from site to site (e.g., different activities and water qualities) as do economic costs incurred visiting each site (in terms of varying direct travel expenditures, different travel times, etc.). Accordingly, visit choices reveal how recreationalists are prepared to make trade-offs between site attributes and costs. In turn, this provides a basis for our own research in estimating in monetary terms the benefits of WFD-inspired improvements in water quality.[15]

Travel cost analysis is inherently spatial and is, therefore, also highly amenable to the analytic opportunities afforded by application of GIS. The technique explicitly recognises that recreationalists' choices are made in the context of numerous substitute and complementary sites. Inadequate incorporation of all pertinent alternative sites is liable to result in inaccurate estimates of benefit values, these being either too high or too low depending upon the balance of substitutes or complements which are ignored. The ChREAM research project aims to substantially improve the travel cost technique through application of GIS both to: (i) improve the way in which site accessibility is measured (Bateman *et al.*, 1996), allowing for the distribution and varying road speed and congestion quality of the road network and; (ii) ensure the comprehensive inclusion of all alternative sites (Jones *et al.*, 2002). This latter extension will include not only other waterways, but also other types of recognised outdoor recreation site (woodlands, wetlands, beaches, National Parks, etc.) and man-made recreation facilities (built heritage, zoos, parks, cinemas and other urban attractions which might conceivably substitute for waterway visits). Such qualitatively different sites are frequently omitted from travel cost analyses of open-access recreational sites, an omission which may have resulted in an overestimation of the value of a given site. Furthermore, the modelling of this data will seek to assess the cross-substitution effects of improvements in the water quality not just of one waterway but multiple and perhaps all waterways, as envisaged under the WFD. As each site is improved to 'good ecological status' so the marginal value of improving the next site declines. Incorporating this shift is vital to the accurate estimation of total values.

The implementation of RP techniques clearly entails a number of spatial analytic issues including the aggregation of benefits across the population, benefits transfer and the distributional analysis of those benefits. Throughout these assessments of benefits a GIS-based approach will be used, together with census demographic data, to provide a spatial element to the analysis. These issues of benefit aggregation, transfer and distribution are now discussed in more detail.

3.3. Benefits aggregation

As noted above, the value an individual holds for the improvement of a certain site will vary according to numerous spatially variable factors such as the availability of substitute sites. Furthermore, when aggregating across individuals, their characteris-

[15] This technique has been applied in numerous previous studies to estimate the recreational value of water quality improvements, for example, through changes in fish species densities (Kling and Thomson, 1996) or through changes in measured water quality (e.g., Feenberg and Mills, 1980).

tics also vary spatially (for example, those with higher incomes may live closer to a given set of sites). A consideration of space is, therefore, vital to the accurate aggregation of benefit values for improvements to amenity waterways (and indeed most spatially confined environmental resources). The spatial analytic capabilities of geographical information systems (GIS) provide an ideal medium for harmonising the diverse data necessary to undertaking such an aggregation exercise (Bateman *et al.*, 2000). In particular, GIS readily allows the researcher to specify a valuation function which varies across space according to a variety of factors including: (i) the distribution of rivers, lakes, estuaries, etc.; (ii) the change in quality to those resources, with improvements tending to convert former non-users into users in a spatially non-random manner; (iii) the accessibility of complementary and substitute assets; and (iv) the distribution and socio-economic/demographic characteristics of the relevant population. The inclusion of such factors allows the analyst to observe any 'distance decay' in values as we consider households which are progressively more remote from a given improvement. Furthermore, once such a valuation function is estimated, by applying it within a GIS to data detailing explanatory variables for all locations we can define the appropriate 'economic jurisdiction' (that area within which values are non-zero) for calculating total benefit values (Bateman *et al.*, 2006a). This avoids common aggregation problems associated with artificial 'political jurisdictions' typically defined by convenient administrative areas rather than the benefits generated by a scheme.[16]

3.4. Benefits transfer

The estimation of spatially sensitive valuation functions also allows us to investigate the potential for 'benefit transfers' (Brouwer, 2000; Ready *et al.*, 2004). Here value functions, estimated as outlined above, are applied to generate values for sites which may not of themselves have been part of a valuation exercise. In theory, once a robust valuation function has been estimated the researcher need only know the attribute levels and improvement scheme which characterises an unsurveyed site to estimate (via the valuation function) the benefit generated by improving that site. However, in practice the track record of benefits transfer work is strewn with failure with many analyses failing recognised tests of transfer validity (see, for example, Vandenberg *et al.*, 2001; Ready *et al.*, 2004).

Despite the empirical problems of benefit transfer, its potential to obviate the need for conducting individual site surveys each time an environmental improvement is to be assessed makes it an attractive method. Indeed this approach underpins contemporary Environment Agency assessments in England. However, at present these are conducted not via a value function transfer but rather by transferring mean values from a relatively small pool of existing studies, of which few, if any, were conducted for such a purpose. Transferring well-specified functions

[16] The typical problem here arises when survey sampling is predefined to occur in a set area which is not representative of the economic jurisdiction. In such cases, aggregation approaches which do not rely upon valuation functions but instead use survey sample means may result in substantial bias, although this is not inevitable. For example, if sampling is confined to an area close to a site but the sample mean is applied to a larger aggregation area then total value assessments may be upwardly biased.

(if they can be estimated) rather than mean values allows the analyst to obtain a value which is adjusted for the characteristics and environs of the site in question. The key issue, therefore, is to determine the robustness or otherwise of the valuation function to be transferred. This is typically undertaken by taking a function estimated from one subset of observations and using it to estimate values for an alternative set of sites for which independent value estimates are already held. The ChREAM project will investigate such robustness analysis. In doing so, we will be guided in part by our recent findings which suggest that the transfer of statistical 'best-fit' functions may inflate value estimation errors (Brouwer and Bateman, 2005). This is because such functions may include site-specific contextual factors which are not relevant to those sites to which the function is transferred. Conversely, the specification of functions on the basis of those general factors identified in core economic theory (e.g., cost, income level and usage parameters) could produce valid transfers which outperform simple mean value approaches in terms of the errors generated.

3.5. Distributional analysis

A final advantage offered by our spatially sensitive GIS-based methodology is an ability to examine the distributional implications of the water quality improvements offered by the WFD. While valuation studies often note an association between WTP and household income, the implications of this association are rarely explored. Furthermore, given that the distribution of benefits is likely to be both spatially and socially uneven, the potential exists for some groups to fare better than others in capturing the non-market benefits of WFD implementation. The estimation of a value function, which varies across space and socio-economic dimensions, allows us to use the GIS to link census measures of deprivation and corresponding WFD benefit values. It seems likely that many environmental benefits, such as those generated by National Parks, are disproportionately captured by the wealthier sections of the society. This may also be the case with water quality benefits, although the case is less clear-cut. Informal waterway recreation is open to all and, unlike National Parks, is much more widespread and accessible. Furthermore, if the WFD is indeed implemented as it stands then it seems likely that the benefits and beneficiaries may be clustered within urban and surrounding areas. However, while this Directive seems potentially redistributive in terms of its benefits, their urban concentration contrasts markedly with the predominantly rural and agricultural incidence of WFD costs. The extent of the tensions between efficiency, equity, certainty of achieving targets and other policy-relevant criteria can be explored through the simulation of alternative WFD implementations in which these criteria are given different weights.

4. Further Issues: Outputs, Users and Interdisciplinarity

As noted earlier in this paper, previous hydrological–economic research was undertaken under the NELUP programme. In commenting on this programme following its conclusion, Moxey and White (1998) highlighted three key problems which had limited its effectiveness: (i) an overly ambitious attempt to produce a complete software 'Decision Support System' (DSS) capable of dealing with too great a diversity of scenarios and which research users saw as usurping the decision process; (ii) an attempt to cater to an overly diverse stakeholder group and one which was not identified from the outset; and (iii) discipline territoriality between researchers. In formulating the ChREAM project, the researchers have attempted to set the project on a course designed to avoid these problems, although only time will tell how successfully.

To be more specific, the project builds on a history of collaboration between a number of the partners and substantial previous experience of interdisciplinary working (see Turner, 2000; Cave *et al.*, 2003). Combining GIS methods with valuation techniques to produce spatially distributed benefits assessments has been pioneered by project members (Bateman *et al.*, 2003). As a consequence, issues such as the integration of data and units of analyses are being addressed at the outset because of the potential difficulties resulting from fundamental differences in the levels of spatial and temporal aggregation across economics, ecology and hydrology (Moxey and White, 1998). Practical measures (such as the use of Access Grid Node videoconferencing and budget provision for researchers to spend time at other institutions) are being used to address the problems that can arise from physical distance between partners (Tress *et al.*, 2005) because, as commented in past work among the project team, 'there is no substitute for regular contact between researchers from different disciplines in the context of a common research problem and a joint learning curve' (Turner, 2000: 459). Such visits and discussions have already clarified that the resolution of available databases make it impractical to achieve integration of hydrological and economic models at the level of the individual farm and instead a framework of geographical areas is being sought where FBS, Agricultural Census and Hydrological Response Units can be linked with both sufficient detail and accuracy.

Providing a clear focus and objectives is also important in such interdisciplinary research. To this end, ChREAM is not attempting to provide a DSS addressing a wide range of scenarios and instead adopts a narrower remit concerning the estimation of select agricultural costs and non-market benefits within a single (if complex) policy environment: the implementation of the WFD. Allied to this, the project is seeking the early and ongoing assistance of a relatively focussed group of users, principally policy makers and those affected by the WFD. Through such an approach, we hope to achieve a situation where discipline experts can combine across subject or paradigm boundaries to generate new insights and work with non-academic partners in a way that is characteristic of transdisciplinary studies as defined by Tress *et al.* (2005). Such an approach seems to us central to the RELU programme and necessary to address the challenges posed by WFD implementation.

5. Concluding Remarks

The spatial variation in the distribution of the agricultural costs and non-market benefits of alternative WFD implementations is a central theme of this study. On the cost side, the combination of spatially explicit physical and farm economic data underpins a new hydrological–economic model of agricultural land use. It is the spatial dimension which also underpins the linkage of the economic analysis with models of nutrient transport, diffuse pollution and consequent biological effects within the water environment. Similarly, it is the spatial dimension which underpins the variegated impacts upon farm welfare predicted when the linked models are driven 'backwards', from water quality to farm production.

Analysis of the farm level costs likely to be imposed by the WFD is balanced by investigation of some of the key non-market potential benefits which the Directive may generate. Here we investigate an integrated mix of RP and SP methods to assess the non-market values which are a major focus of the policy. As in our farm modelling exercise, the use of GIS-based methodology is critical in extending previous research to address the aggregation, transfer and distribution issues of such benefits.

The programme of research is highly interdisciplinary and policy orientated but, as the discussion above indicates, it also addresses current frontier issues in the literature on land use modelling, non-market benefits analysis and the spatial aspects of both. It is not the first, nor is it likely to be the last, large-scale land use–hydrological modelling study. Nevertheless, we contend that its clear focus upon a defined and contemporary policy-relevant issue, combined with new approaches to established research problems, should constitute a useful contribution to knowledge in this area.

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