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A transdisciplinary approach to the economic analysis of the European Water Framework Directive

Abstract

The Water Framework Directive prescribes economic principles to achieve its ecological targets. The aim is to establish cost-effective measures to achieve good ecological status and assess whether the costs of these measures are justifiable in view of the benefits they provide. The complex nature of water problems requires flexible decision-making embracing a diversity of 'knowledges'. Here, natural and socio-economic scientists worked together in an integrated approach 'ground-tested' through local stakeholders' knowledge and views. The aims were to: (1) develop a set of steps for implementing this transdisciplinary approach, and (2) critically reflect on the challenges of integrating different strands of knowledge to the specific context of the economics of the WFD. This was tested at a sub-catchment in Scotland. Hydro-chemical models were used to simulate effectiveness of Phosphorous pollution mitigation measures, which was then incorporated into a cost-optimization model. Costs were compared with benefits resulting from water quality improvements. This analysis was accompanied by an iterative local stakeholder consultation process. The research further analysed whether selected measures are 'future-proof' in view of climate and land-use changes. Results are used to help set the research agenda for more practical specification of economically sound and socially acceptable ways to deliver the WFD.

Keywords: cost-effectiveness, disproportionality, Phosphorous, stakeholder consultation, 'wicked' problems, water quality modelling

1. Introduction

One of the most innovative aspects of the European Water Framework Directive (WFD) is the incorporation of economic principles and tools to support delivery of ecological targets. Amongst the various economic aspects of the WFD is the use of cost-effectiveness analysis (CEA) of mitigation measures needed to achieve the ‘good ecological status’ (GES) of waters. The aim is to establish the least-costly programme of measures to be included in basin management plans (Balana et al., 2011; Perni and Martinez Paz, 2013; Skuras et al., 2014; Klauer et al., 2014a). Moreover, the WFD allows the derogation of environmental objectives if meeting them has disproportionately high costs, i.e. if the costs of the measures are higher than the resulting benefits (Martin-Ortega et al., 2014).

These principles add new challenges to the management of water resources, which is recognized to be a ‘wicked problem’ (von Korff et al., 2012; Patterson et al., 2013), that is: a problem for which it is impossible to define optimal solutions because of both uncertainty about present and future environmental conditions and intractable differences in social values (Shindler and Cramer, 1999). For example, addressing diffuse pollution requires implementation of actions involving multiple actors operating at multiple scales and influenced by a range of factors (Cash et al., 2006, Blackstock et al., 2012). Water management also commonly involves tensions and mismatches between spatial and temporal scales relating to environmental change, human behaviour and institutional processes (Cumming et al., 2006). The economic efficiency of the WFD’s programmes of measures needs to be assessed at the river basin scale by regulatory agencies, while each specific intervention requires action at the source of the problem by those responsible (e.g. field level by farmers, household level for septic tanks, local authorities for sewage plants, etc.). In addition, there are heterogeneous perceptions between different stakeholders of what constitutes proper land-management and how it affects water quality (Christen et al., submitted). Moreover, effectiveness of measures varies over small spatial scales according to soils type, slope, management, etc., whilst modelling tends to take place at a catchment scale, aggregating responses throughout the catchment to an average response. Also, it is often not possible to define simple links between chemical water quality and ecological outcome, which is the key to WFD’s pursuit of GES (Hering et al., 2010). All these elements add to the ‘wickedness’ of water management problems and help to explain the failure to deliver more substantive progress in the achievement of the WFD's objectives[1]. Finally, creating mitigation programmes for current conditions might not be 'future-proof' against climate and land-use change, potentially making GES only a temporary occurrence.

The literature covering the development of strategies to tackle 'wicked' environmental problems points clearly the need for interdisciplinarity and transdisciplinarity (Carew and Wickson, 2010; Brandt et al., 2013; Duckett et al., submitted). However, to date the economic literature on the WFD has only been able to provide partial solutions from a mono-disciplinary predominantly neoclassical perspective (Martin-Ortega, 2012). Moreover, an in-depth review of the scientific literature and policy practice on the issue of disproportionality across several countries in Europe shows that very different approaches have been taken (Martin-Ortega et al. 2014; see also Galioto et al. (2013) for an Italian case, Jacobsen (2009) for the case of Denmark; and Klauer et al. (2014b) for a German case). A transdisciplinary approach is based on the principle that the integration of other actors in the knowledge production process, in addition to specialist scientific knowledge, results in a 'final knowledge' that is anticipated to be greater than the sum of disciplinary components (Lawrence and Després, 2004; Tress et al., 2004; Mobjörk, 2010). The principle is that the complex and dynamic nature of such environmental problems requires flexible decision-making, embracing a diversity of 'knowledges' and values (Reed, 2008; Blackstock et al., 2012).

The present paper represents a practical example of how to operationalize this transdisciplinary approach to meeting WFD targets, integrating hydrological and economic modelling informed, 'ground-tested' and shaped by stakeholders' knowledge, views and perceptions. This approach was tested at the sub-catchment level in Scotland in the analysis of measures to mitigate rural diffuse pollution (phosphorus) under current and future climate conditions and land uses. The aims were to: (1) develop a set of steps for implementing this transdisciplinary approach to meeting WFD objectives, and (2) critically reflect on the opportunities and limitations of integrating different strands of knowledge to the specific context of the economic analysis of the WFD. This represents a new angle on the economic analysis of the WFD proposed so far (Martin-Ortega, 2012). Results are used to help set the research agenda for devising a more realistic economically sound and socially acceptable

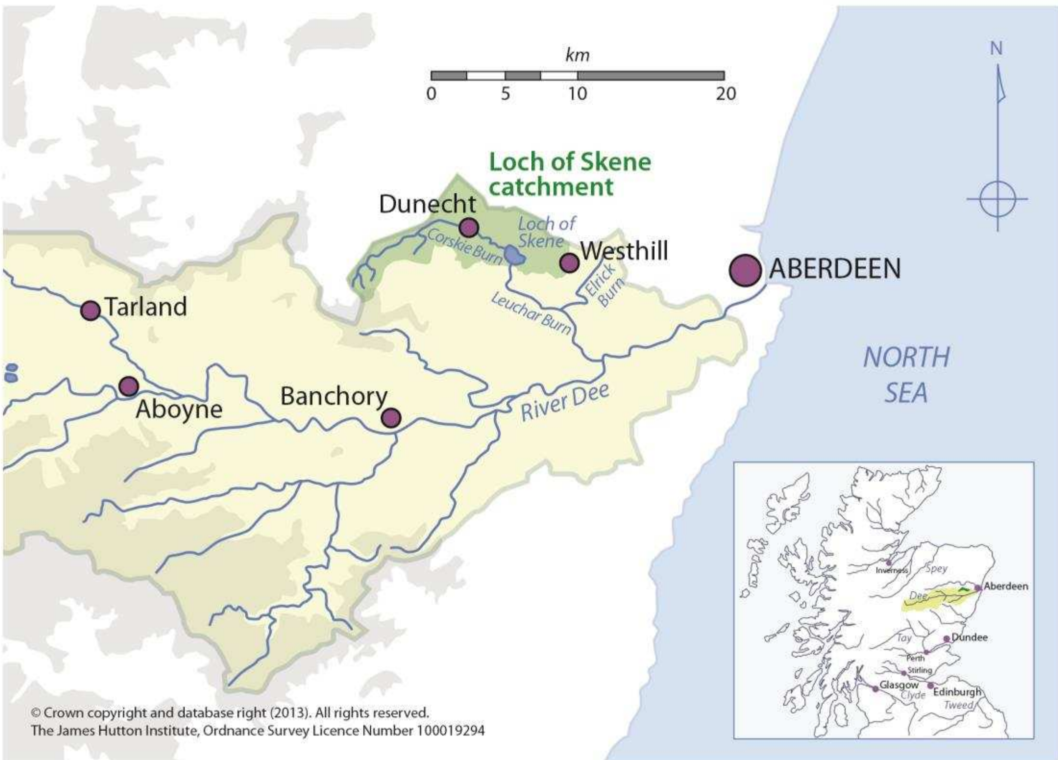
[1]The third implementation report (EC, 2012) found only a 10% predicted increase in surface water bodies likely to reach GES by 2015 -as required by the Directive- compared to 2009; leaving almost half the surface waters in Europe likely to be less than good status in 2015.

specification of management options to deliver WFD compliance under current and future conditions.

2. Case study

The transdisciplinary approach was tested in the Skene catchment, a sub-catchment of the River Dee in the north-east of Scotland. The sub-catchment lies 13 km west of the City of Aberdeen and covers an area of 48.3 km^2 (Figure 1). It is a rural, predominantly agricultural area, dominated by a single large, privately-owned estate, a characteristic land-holding and management system in Scotland (cf. McKee et al., 2013). The catchment drains into the Loch of Skene, a shallow lake (loch in Scottish dialect) with an area of 1.1 km^2. The loch is an important site for overwintering wildfowl and, as a consequence, is designated as a Site of Special Scientific Interest (SSSI), a Special Protected Area (SPA) and a Ramsar Site. The loch is used for recreational sailing between April and June thereafter poor water quality (eutrophication) prevents further use. The principal feeder stream is the Corskie Burn, which drains three quarters of the loch's catchment (34 km^2) and receives effluent from the two sewage treatment works present in the catchment. It is also the only tributary to the loch for which monitoring data (chemistry and discharge) are available.

Figure 1 The Skene sub-catchment



The Skene sub-catchment is part of the area covered by the Dee Catchment Partnership[2], a body that has been working since 2003 to protect, enhance and restore the waters of the River Dee catchment. This independent and voluntary partnership of local stakeholders and interested organisations has sought to develop a consensual and informed approach to water management. Around 20 organisations are involved, working toward the delivery of an agreed Catchment Management Plan (Cooksley, 2007).

3. Methodology

Hydro-chemical models were used to simulate sub-catchment scale effectiveness of a selection of measures for improving water quality. Results were then incorporated into a cost-optimization model, which allowed the ranking of measures according to their cost-effectiveness ratio to achieve pre-established targets of water quality improvement. These costs were then compared to the benefits resulting from the achievement of the good ecological status, elicited in an existing stated preference survey. This analysis was accompanied and sustained from the outset by an iterative consultation process with local stakeholders, whose inputs fed into the design of the analysis and also offered a way of comparing scientific results with local perceptions. The aim of the stakeholder engagement was not to substitute scientific knowledge with lay knowledge, but to gather understanding on their perceptions and practices that are otherwise unknown or inaccessible, and, further, to anticipate a reality may depart from conventional model predictions. In other words, stakeholder engagement aimed to increase the reliability of the models and make outputs more realistic. Each of the individual methodological steps (section 3.1) has its own limitations, due to different factors such as lack of data, budget restrictions and modelling capacity. However, the contribution of this research focuses on the integration process, rather than of each of the individual steps, and reflects on the challenges that need to be addressed if scientific results are to inform policy.

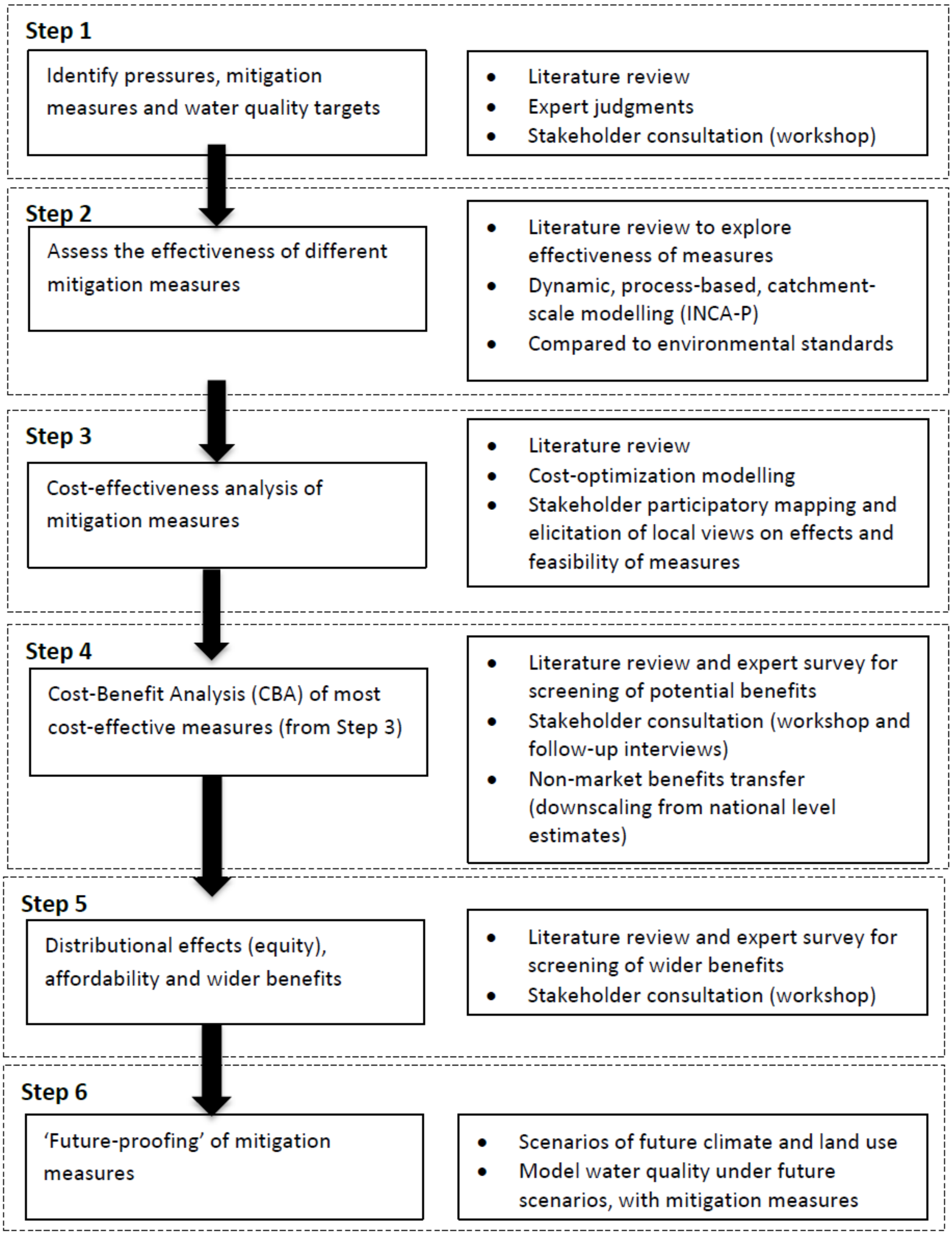
3.1. Methodological sequence

Figure 2 depicts the methodological steps followed in this research. The baseline year for the analysis was 2007 and three time horizons were used for the analysis of disproportionality,

[2] www.theriverdee.org

coinciding with the three planning cycles imposed by the WFD (2015, 2021 and 2027). The climate and land use change scenario analyses were based on projections to 2050.

Figure 2. Methodological sequence of the transdisciplinary approach to the economic analysis of the WFD



Step 1: Identify pressures, mitigation measures and water quality targets. Pressures on water quality in the study sub-catchment were identified based on previous work in the area (Balana et al., 2010). These were then presented to local stakeholders in a workshop (see Section 3.2 for details on the stakeholders involved and on the stakeholder engagement process). A

participatory discussion explored whether the pressures and sources were identified accurately according to local knowledge and whether stakeholders considered any important pressure or source to be missing from the proposed list. Workshop participants were then asked to suggest locally relevant potential measures that could be used to address those pressures.

Of the key pressures identified, phosphorus (P) is the only pressure for which the WFD sets standards for surface waters[3], and hence the one on which we focus the rest of the analysis in this study. Phosphorous targets were derived by looking at current concentrations in surface waters, and comparing these to concentrations required for GES (see Text Box 1).

Text Box 1: Phosphorous reduction targets.

Between 2007 and 2010 the Corskie Burn was classified by SEPA as having 'Moderate' chemical status with respect to mean annual soluble reactive Phosphorous (SRP) concentration. A target of around 20% reduction in mean annual SRP load (i.e. reducing from 344 to 275.5 kg/year, a reduction of 68.9kg/year), was chosen as sufficient to potentially cause a shift to a more oligotrophic macrophyte community in the Corskie Burn (Jackson-Blake et al. 2013). It should be noted that this target is based on an expert judgement, since only low frequency and relatively poor quality monitoring data are currently available (Jackson-Blake et al. 2013). However, it serves to illustrate the methodology being proposed in this paper.

Step 2: Assessing the effectiveness of different mitigation measures. The list of potential measures derived in Step 1 was narrowed down to a sub-set, selected on the basis of the existence of enough background information and data for the modelling exercise. The INtegrated CAtchment model of Phosphorus dynamics (INCA-P; Wade et al., 2002; Wade et al., 2007) was then used to simulate the current concentrations of dissolved and total P in the Corskie Burn. INCA-P is a dynamic, catchment-scale model which uses a semi-distributed approach to route water and nutrients through the terrestrial compartment and the stream. After the model had been calibrated and validated (Jackson-Blake et al., 2013), the effectiveness of each measure was estimated in terms of the associated reduction in the load of dissolved phosphorus delivered to the stream per year, and the corresponding reduction in

[3] Nitrogen is another key pollutant, but the WFD only sets standard for groundwater, which is not relevant in this case.

mean annual soluble reactive Phosphorous (SRP) concentration. These were compared to the P standard to establish compliance under the suite of measures.

Step 3: Cost-effectiveness analysis. Cost estimates were calculated to reflect income foregone and/or additional costs as a result of implementing management options, using gross margin data from a number of sources[4]. Cost estimates and cost-effectiveness model outputs were integrated in a cost optimization model, where the objective function being minimized was the sum of costs of measures to achieve target nutrient load reductions (equation 1);

$$\left.\begin{array}{l} \text{Min. } C = \sum_{m} \alpha_m C_m \\ \text{subject to:} \\ \quad \sum_{m} (EQ - R_m) \times \alpha_m \leq Q \end{array}\right\} \qquad \text{..(1)}$$

where subscript m denote the measure; α is a binary variable set equal to 1 if the measure is relevant to the SRP emission source and C is the total cost associated with the soluble reactive phosphorous (SRP) load reduction (£/year); EQ is the baseline SRP emission load; R is the SRP load reduction associated with the measure m; and Q is the SRP load above which the water body fails to achieve GES. The second part of equation 1 simply states that the sum of the load reduction from the combined measures is sufficient to achieve GES. The changes in the mitigation measures considered in this model are discrete changes. This is why we used summation instead of integral calculus for cost-aggregation. Intensification or expansion of a given defined measure (e.g. changes in the fertilizer application rate) is modelled as an additional (independent) measure. Modelling of discreet set of choices as the one proposed here is typical in empirical studies on diffuse source pollution control from agriculture (Yiridoe and Weersink, 1998; Balana et al., 2011).

Cost-effectiveness outputs calculated in this way were used to rank measures according to cost-effective ratios and were then contrasted with local stakeholders' knowledge and views on effectiveness and feasibility. A participatory mapping exercise was tested as a way of establishing spatial prioritization of the interventions. Previous studies have illustrated that

[4] Farm Management Handbook (SAC, 2008); Farm Management Pocketbook (Nix, 2011); Scottish Rural Development priorities payment rates sheet (SRDP, 2008); Defra report on cost curves of phosphorous mitigation options (Defra, 2003) and other sources available for reference in the REFRESH project report (Balana et al., 2013).

this technique – whether using computer-based models or simple annotated paper maps, as used here – promotes communication, transparency and trust between stakeholders, bringing together practitioner and scientific knowledge (Raymond et al., 2010; Irvine et al., 2009; Swetnam et al. 2011).

Step 4: Cost-Benefit Analysis of cost-effective measures. The preceding step identified the most cost-effective combination of the selected measures to achieve the SRP reduction target. To analyse disproportionality, the costs of implementing these measures were then compared with the benefits of improved water quality.

The identification of benefits was undertaken in three steps: i) screening of potential benefits based on a literature review (subsequently compiled in Martin-Ortega et al., 2014); ii) expert consultation via a survey to scientists from a range of disciplines at the James Hutton Institute; iii) consultation with local stakeholders in a workshop, where participants were asked to validate the list elaborated on the basis of i) and ii), i.e. whether the list was comprehensive and any items included in the list were considered irrelevant to the local area.

Monetary estimates of non-market benefits[5] were obtained from a published stated preferences study by Glenk et al. (2011), who estimated the benefits of improving Scottish water bodies to comply with the WFD at a national level, using a choice experiment. In that study, three categories of water quality problems in terms of ecological status ('many problems', 'few problems' and 'no problems') were defined as a simplification of the ecological status classification in the WFD. The 'no problems' category corresponded to WFD 'excellent or good status'; while 'few problems' corresponded to 'moderate status' and 'many problems' to WFD's 'poor or bad status'. To capture use and non-use values, the three categories were described to survey respondents both in terms of ecological conditions and implications for suitable recreational use, matching well the type of benefits described by the local stakeholders of our case study during the consultation process. Glenk et al. (2011) estimated, at the national level, willingness to pay (WTP) values for the improvement of ecological status in pounds per household per year per 1% of catchment area improved (£1.81 per household per year). Because they used per area of catchment as their unit of measurement, these national level values could be used to obtain WTP values for reaching

[5]Consultation with local stakeholders confirmed that significant market benefits were not to be expected in the area (see section 4.2).

GES in the study sub-catchment by multiplying marginal per hectare values by the sub-catchment area.

The use of national WTP estimates has the caveat of assuming a uniform unit value of improvement (per hectare) regardless of the specific water body, its location and use. This is, of course, a simplification. However, any benefit transfer exercise implies that the value of a certain river or lake is the same (or adjustable to) the value of another river or lake. So, in the absence of a local primary valuation, the use of national average values is considered equivalent to conventional value transfers as proposed by the valuation literature (and hence, subject to transfer errors (Wilson and Hoehn, 2006)).

Stakeholder consultation in our study confirmed theoretical expectations that beneficiaries of these non-market benefits spill beyond the boundaries of the sub-catchment (Bateman et al., 2006). To account for this spill-over effect, per year household WTP values from Glenk et al. (2011) were aggregated overall for the population of Scotland's River Basin District[6] Theoretically, benefits are expected to decline with distance from the water body and with the existence of substitutes (Bateman et al., 2006), so people living closer to the water bodies hold higher values than those living further away. As explained, we used average national marginal values of Scotland's population, and hence it is assumed that diminishing values with distance are included in that average. We are confident that the boundaries of the economic jurisdiction (i.e. area beyond which no values for water quality improvements in this sub-catchment are held) do not fall within the river basin district because previous studies in the UK have shown distance elasticities such that value terminates shortly beyond 1,000 km (Bateman et al. 2000; Hanley et al., 2003)[7], i.e. well beyond the aggregation boundaries used for this study).

For the comparison of costs and benefits, a Dual Cost-Benefit Analysis approach was judged to be the most appropriate, since it allows different discount rates to be applied to market and

[6] The population of the Scotland River Basin District adds up to 4.8 million people (SEPA, 2009). We have used a ratio of persons per household of 2.25 (ONS, 2011), totalling a population of 2,133 thousand households in the basin.
[7] For example, Hanley et al. (2003) estimate a distance decay function for use and non-use values of a river's condition as $WTP + 1 = 5.5\ (DISTANCE)^{-0.244}$, which means that WTP equates to zero (i.e. value terminates) at 1,082 km.

non-market costs and benefits (Kula and Evans, 2011)[8]. Using this approach, the Net Present Value (NPV) indicator is estimated as follows (Equation 2; Almansa et al., 2012):

$$NPV\ (r, r^e) = \sum_{t=0}^{t} \frac{F_t^m}{(1+r)^t} + \sum_{t=0}^{t} \frac{F_t^e}{(1+r^e)^t} \qquad (2)$$

Where F^m corresponds to market cash flows, F^e are the environmental cash flows, t denotes the time horizon of the evaluation, r is the usual discount rate and r^e represent the environmental discount rate ($r > r^e$). If $NPV(r, r^e) > 0$, the costs of the measures are proportionate. Discount rates of 5.5% for market costs and of 3.5% for non-market environmental benefits were applied.

Step 5: Distributional effects, affordability considerations and wider benefits. Disproportionality analysis relying only on a CBA can have undesirable social implications. Whether the cost of achieving a certain environmental target is disproportionate or not also depends on the social desirability of the distribution of benefits and costs among different socio-economic actors (Martin-Ortega et al., 2014). Although CBA approaches incorporating distributional effects exist (Pearce et al., 2006), distributional effects and equity considerations were addressed here through stakeholder consultation. How local stakeholders perceived the distribution of costs and benefits across the community was discussed qualitatively with the stakeholders during the workshops, and considered in the light of quantitative affordability indicators. We also explored the existence of benefits beyond strictly the water environment (termed here as 'wider benefits'), for example, a positive impact on carbon storage, looking at the literature and in through consultation with experts and local stakeholders.

Step 6: *'Future-proofing'* of mitigation measures. To investigate the potential impact of environmental change on water quality, the hydro-chemical model was re-run using scenarios of future climate and land use for 2050. Three climate model simulations were used, representing the average, upper and lower extreme projections from the EU FP6 ENSEMBLES project, all based on the SRES A1B emission scenario. Four storylines of

[8] The higher the discount rates are, the lower importance is attributed to costs and benefits in the future. In relation to environmental goods and services, this raises theoretical and ethical considerations about whether it is appropriate to attribute lower importance to costs and benefits of future generations in relation to current ones. To address this issue, part of the literature proposes to apply different discount rates depending on the nature of costs and benefits (Almansa and Martinez-Paz 2011). It has been argued that lower discount rates should be applied to non-market values due to sustainability and intergenerational solidarity reasons (Roumboutsos 2010; Almansa et al. 2012).

2050 land use were developed, broadly corresponding to the quadrants of the IPCC SRES scenarios representing "World Market" (A1), "National Enterprise" (A2), "Global Sustainability" (B1) and "Local Stewardship" (B2) (Brown and Castelazzi, 2014; Nakicenovic et al., 2000). Consistent with the Land Use Strategy for Scotland (2011) targets, all scenarios incorporated an increase in woodland cover and two included an increase in arable land area. INCA-P was run with each land use and climate scenario, allowing the identification of a 'worst' combined land use and climate change scenario (the SMHIRCA/BCM climate model simulation combined with the 'National Enterprise' land use scenario), and a 'best' combined scenario (the KNMI/ECHAM5r3 climate model output, combined with the 'Global Sustainability' land use scenario). INCA-P was then re-run with the cost-effective mitigation measures, together with the 'worst' and 'best' combined land use and climate change scenarios. This allows an assessment of the robustness of the measures to potential future environmental conditions.

3.2.Design of the stakeholder engagement process

This research process was designed as consultative transdisciplinarity, rather than participatory transdisciplinarity, as defined by (Mobjörk, 2010). This was so because primary objective was to gather non-scientific viewpoints and knowledge to contribute to the economic analysis of the WFD. For example, non-academic input is limited to responding to research questions already defined by the research team, rather than co-constructing a problem frame in collaboration, as is characteristic of participatory transdisciplinarity (Mobjörk, 2010). Despite this classification, participatory approaches were central to the workshop methodology of this research. Table 1 presents the transdisciplinary process and its correspondence with Lang et al.'s (2012) conceptual model.

Table 1. Project correspondence with transdisciplinary conceptual model (after Lang et al., 2012)

Phase (after Lang et al., 2012: 28)	**Project correspondence**
A: Collaborative problem framing and building of a collaborative research team	- 'Real-world' problem co-constructed by interdisciplinary team, including expert judgements. - Stakeholder recruitment including representation from scientific and non-scientific knowledge types. - Problem framing confirmed by non-academic stakeholders during first workshop.
B: Co-creation of solution-oriented and transferable knowledge through collaborative research	- Integrative and collaborative methodology adopted, including literature review, modelling, participatory mapping, expert survey and stakeholder consultation (through workshop discussions and follow-up interviews).
C: (Re)-integrating and applying the co-created knowledge	- Approval sought for research outputs from scientific and non-scientific participants (through iterative process). - Co-constructed mitigation measures.

There is no pre-defined 'recipe' for undertaking transdisciplinary research, but designing a process which focusses on the integration of stakeholders and their views as well as achieving project goals, represents good practice (cf. Brandt et al., 2013; Wiek et al., 2013). In this case, the workshops were specifically designed to address key questions relating to each of the methodological steps described in section 3.1 in the way described in Table 2. Two local workshops were carried out in half day sessions (held in February and in September 2012). Each session was followed up with a feedback questionnaire sent to participants, in addition to feedback leaflets which summarised the main workshop outcomes in non-scientific language. The questionnaire sought to gather participant views on the workshop process and facilitation, in order to improve practice in subsequent events. Questioning therefore focussed on whether the participant found the workshop professionally useful, interesting, understandable and easy to follow, as well as whether they felt they had learnt anything new (and if so, what they had learned). It also sought to establish whether the respondent would be happy to participate again in future events on similar topics, as well as providing space for further participant comments on their workshop experience. Interviews and follow-up conversations with individual participants were undertaken in order to clarify and gain further detail on specific issues, and project reports were similarly circulated and discussed with stakeholders and amended according to their feedback when deemed necessary.

Table 2. Workshop plan and participant activities – summarized

Workshop 1	**Agenda/Activities**
	• Workshop introduction: - Research team and participant introductions; - Workshop outline and purpose; introduction to the project, overview of previous work on barriers/pressures and climate change scenarios. • Plenary discussion of problems and pressures to gain participant agreement regarding key pressures. • Describing action and mitigation measures: - Identifying what measures participants currently undertake and what is possible to resolve the pressures and in order to comply with WFD (add to list generated from literature). • Evaluation of measures: - Small group discussion considering effectiveness of and costs (including implications for farm profitability). • Plenary discussion of group findings and implications for compliance/evaluation of measures with climate change. • Workshop key messages summarised, project 'next steps' outlined and participant workshop evaluation.
Interim activities	• Workshop report drafted and comments invited from participants. • Summary workshop leaflet distributed to participants. • Stakeholder analysis revised to ensure representation of interests for second workshop; invitations to second workshop sent to previous and ideal participants.
Workshop 2	• Workshop introduction: - Research team and participant introductions; - Background: project aims; - Results from previous workshop (pressures and measures); - Workshop 2 aims and outline. • Small group discussion followed by plenary gathering views on proposed list of costs and benefits of improving water quality. • Group discussion on distributional effects (cost-bearers and beneficiaries); equity and affordability considerations. • Small group discussion followed by plenary considering wider benefits of the water-improvement measures. • Small group participatory mapping of priority areas of action (i.e. participants located dots/areas of the catchment on a paper map provided, identifying priority areas for interventions). • Workshop key messages summarised, project 'next steps' outlined and participant workshop evaluation.
Post-workshop	• Workshop report drafted and comments invited from participants. • Follow up interviews with specific stakeholders • Summary workshop leaflet distributed to participants. • Final project report drafted and comments invited from participants.

Stakeholders were recruited to ensure representation of interests regarding the key research questions (i.e. in the first workshop the emphasis was on land management pressures and mitigation measures, and in the second workshop the emphasis was primarily on water quality benefits, wider benefits and distributional effects). The first workshop was attended by 18 participants (including the scientists). This group were invited to also attend the second workshop, however further stakeholder analysis was undertaken in the interim period, to ensure representation of interests. Therefore the second workshop was attended by 19 participants, with 9 joining the workshop series for the first time.

The Dee Catchment Partnership represents an ideal forum for the identification of and engagement with stakeholders. Most of the relevant stakeholders in the area are members of the Dee Catchment Partnership, including agencies who have signed up to the river basin management plan's objectives, public bodies, land managers and individual householders. The trust built locally by the Partnership also allowed access to other relevant stakeholders. Stakeholder recruitment and engagement was undertaken following the guidance of Reed and colleagues (2008; et al., 2009). No economic remuneration was provided to participants (except, in the case of farmers, covering daily expenses for the attendance to the workshops) and participation was based on the genuine interest established through the Dee Partnership. Table 3 presents the stakeholders participating in the research.

Table 3. Stakeholders involved in the research

Stakeholder type	**Number participated in first workshop**	**Number participated in second workshop**
Land owners and farm managers (including representatives of the local estate and tenants)	5	2
Other land managers (e.g. quarry, Forestry Commission Scotland)	1	
Aberdeenshire Council	2	
Local community council (representing local residents)		2
Scottish Water (public water utility)		1
Royal Society for the Protection of Birds (RSPB)	1	1
Scottish Environment Protection Agency (SEPA)	1	1
Scottish Natural Heritage (SNH)		1
Scottish Government's Rural Payments and Inspections Directorate		1
The Dee Catchment partnership	1	1
Fishery Board		1
Recreational sailing club		2
Local Biodiversity Partnership		1
Scientists (from The James Hutton Institute)	5	4
Independent experts (ornithologist/agricultural lecturer)	2	1
TOTAL:	18	19

4. Results

As mentioned, phosphorous release from sewage treatment works and agriculture was selected as the key pollutant to be targeted in this study. However, local stakeholder consultation pointed out also also other potential sources of diffuse pollution, notably that generated by urban expansion and road run-off; and other types of pressures, such as channelization, which can lead to flooding and loss of habitat diversity, and barriers to fish migration. In the loch itself, an additional source of nutrients are the faeces from winter roosting of geese and gulls, which number tens of thousands in winter (Hearn, 2004).

4.1. Cost-effective programme of selected measures

Stakeholders identified 23 measures which could potentially be relevant to improve water quality in the area. Due to limitations in the availability of complex spatially and temporally varying management data several of these measures could not be adequately assessed using the INCA-P modelling framework. Therefore only a sub-set of the identified measures, those for which there is sound scientific evidence of their effectiveness, were included in the CEA (see Table 4).

Model results providing the effectiveness of the selected measures, together with the associated costs are also shown in Table 4. A 50% and 20% reduction in fertilizer application rates to improved grassland and arable land systems respectively and investment in waste water treatment works (WWTWs) to reduce effluent SRP concentration to 1mg l^{-1} is the most cost-effective combination of the selected measures to achieve the set targets (see Text Box 1), according to model outputs. This combination falls slightly short to the 20% target (65,17 9kg/year, 19.5%). Adding a 20% stocking density reduction reaches a 21.3% load reduction, i.e. slightly beyond the target. These results would indicatively suggest that in the Corskie Burn, the 20% reduction in SRP load could be achieved at an annual cost between 36,914 and 45,934 GBP.

Table 4. Cost and effects of selected measures and cost-effectiveness ratios

Measure	Description	Effectiveness	Total costs	Cost-effectiveness	CE Ranking
		kg SRP load reduction	£ / catchment	£ / kg SRP removed	
Convert arable to grassland	20% arable to rough grazing	21.36	42,600	1,994	11
	50% arable to rough grazing	53.31	106,500	1,998	10
Reduce STW inputs	Reduce effluent concentration to 3 mg/l	20.59	29,200	1,418	7
	Reduce effluent concentration to 1 mg/l*	54.07	35,040	648	4
	Remove altogether (piped elsewhere)	70.77	46,720	660	5
Reduce manure inputs	20% stocking density** reduction	5.97	9,020	1,511	8
	50% stocking density reduction	11.66	22,550	1,935	9
Reduce fertiliser application	P fertilizer - 20% reduction to arable*	4.13	1,874	454	3
	P fertilizer - 50% reduction to arable	10.71	11,715	1,094	6
	P fertilizer - 20% reduction to grassland	3.70	-422***	-114	2
	P fertilizer - 50% reduction to grassland*	6.97	-2,638***	-378	1

* Most cost-effective combination of selected measures to achieve 19.5% SRP load reduction
** Additional measure to achieve 21.3% SRP load reduction.
*** Negative costs represent savings due to reduced costs of fertilizer application. Negative costs are usually contemplated in cost-effectiveness and marginal abatement costs models (Moran et al., 2008) – see discussion about this in section 4.3.

4.2. Benefits of improved water quality and Cost-Benefit Analysis

Consultation with local stakeholders identified that the main benefits of improved water quality in the area are non-market benefits associated with increased recreational opportunities, landscape beauty, individual and community well-being and improved habitat and wildlife (non-use values). No significant market benefits were not to be expected in the area. Improved water quality could attract more users for open access activities. However, the potential beneficial knock-on effects on the local economy (for example, in shops and pubs) were thought not to be significant, since the majority of users are local and often bring their own food and other supplies. Increased benefits from recreational fishing, which is controlled by the local private Estate, were also not expected. See results section for specification of benefits included in the analysis.

For comparison with the costs, only benefits of improving water quality in the Corskie catchment were monetized (Table 5) **Error! Reference source not found.**but it should be noted that the improvement of the stream quality has clear effects on the loch itself and beyond, notably in relation to increased sailing recreational opportunities in the form of expanded sailing season (currently constrained by algal blooms). Moreover, stakeholders also pointed out the existence of wider benefits beyond strictly the water environment, for example, a positive impact on carbon storage, enhancement of non-aquatic wildlife, the reduction of soil erosion and flooding, as well as broader positive impacts on improved sense of community and increased educational opportunities. While these wider benefits would be difficult to quantify, further work could include the modelling of effectiveness and associated costs in the loch itself to be compared with loch-related benefits.

Table 5 also shows the costs and lifetime of each of the measures comprising the most cost-effective combination of the selected measures. The NPV for each time horizon indicates, again with the necessary precautions, that the benefits of improving water quality would significantly outweigh the costs of the measures for the three time horizons in this particular area.

Table 5. Estimated costs (depending on the type of measure, these are: investment and operational cost or foregone income) and monetized benefits of measures to meet water quality targets and profitability indicators

<table>
<tr><th rowspan="2">Measure</th><th colspan="2">Annual costs</th><th rowspan="2">Measure's lifetime</th><th rowspan="2">Non-Market Benefits[**] (£ per year)</th></tr>
<tr><th>Type of cost</th><th>Value (£ per year)</th></tr>
<tr><td>50% reduction of fertilizer application to grassland system</td><td>Foregone income</td><td>0</td><td>2007-2027</td><td rowspan="4">194,639</td></tr>
<tr><td>20% reduction fertilizer application to arable land</td><td>Foregone income</td><td>1,874</td><td>2007-2027</td></tr>
<tr><td>WWTW to reduce effluent SRP concentration to meet 1mg/l.</td><td>Annualized investment and additional operational costs</td><td>35,040</td><td>2007-2027</td></tr>
<tr><td>20% stocking density reduction</td><td>Foregone income</td><td>9,020</td><td>2007-2027</td></tr>
<tr><td></td><th colspan="4">Net Present Value (GBP)</th></tr>
<tr><td></td><th colspan="2">2015</th><th>2021</th><th>2027</th></tr>
<tr><th>19.5% SRP load reduction</th><td colspan="2">1,257,081</td><td>1,917,884</td><td>2,463,981</td></tr>
<tr><th>21.3% SRP load reduction</th><td colspan="2">1,239,470</td><td>1,900,274</td><td>2,446,370</td></tr>
</table>

[*]Permanent here means sustained throughout the full period of analysis, i.e. no re-investment needed. [**] Estimates based on adjusting national average values from Glenk et al. (2011).

4.3 Distributional effects and affordability considerations

Waste water treatment works and individual farm landowners were identified in the stakeholder consultation process as the major cost bearers, which is consistent with the cost-effectiveness model results. On the other hand, it is the wider public generally who would mainly benefit from the improvement of water quality. In relation WWTW, whether their additional costs would be passed on to the general public through the increase of the water charges was an issue of debate in the stakeholder workshop. The representative of Scottish Water indicated that there has not been an increase in water charges in the area in the last three years.

In relation to farmers, Table 6 shows the ratios of annual costs and net farm income of measures under consideration, used here as a first indicator of affordability. For two of the measures identified as most cost-effective (highlighted in grey in Table 6), the reduction of 20% fertilizer in arable land, generates a cost equivalent to about 3% of farm income, while the 50% fertilizer reduction in grasslands actually produces benefits, which is coherent with previous findings from Lago (2009), who investigated the impact on profits of achieving different phosphorous loads reductions at farm level in Scotland. This is because reducing excess inputs (i.e. unnecessary fertiliser applications) can increase financial profitability. Decreased gross margins due to the application of fertilizer reduction have also been reported in the literature (e.g. Fezzi et al., 2010), but it is not infrequent to find costs savings (e.g. Panagopoulos et al. (2011), Mewes (2012)). Ultimately, the cost of reducing fertilization depends on the baseline conditions; the biophysical characteristic of the field; the input and output markets/prices, as well as the modelling approach taken, and the literature has reported both positive and negative costs for this measure (Schoumans et al. 2014, Lescot et al. 2013).

The additional measure of reducing 20% stocking density produces costs equivalent to about 11% of farm's net income, which can present a problem.

Finally, subsidies from rural development plans are available for most of the measures analysed here. This is a mechanism for costs to be transferred to the general public but it should be noted that rural development grant uptake in the area is low (Vinten et al. 2013).

Table 6. Ratio of annual costs and net farm income of measures to improve water quality (year 2007-2008)

Measures	Costs data		Ratios (farm level)	
	Annual Cost (£/ha)	**Annual Cost (£/farm)**	**Annual cost / NFI**	**Annual cost / FBI**
Convert arable to grassland				
20% arable to rough grazing	200.00	4,040	13.54%	10.30%
50% arable to rough grazing	200.00	10,100	33.86%	25.75%
Reduce manure inputs				
20% stocking density reduction	42.75	4,318	14.5%	11.0%
50% stocking density reduction	106.88	10,794	36.2%	27.5%
Reduce fertiliser application				
P fertilizer_20% reduction to arable	8.80	889	3.0%	2.3%
P fertilizer_50% reduction to arable	22.00	2,222	7.4%	5.7%
P fertilizer_20% reduction to grassland	0 (-2)	0	0.0%	0.0%
P fertilizer_50% reduction to grassland	0 (-5)	0	0.0%	0.0%

Average *Scotland's* farm size: 101 hectares (Source: http://www.scotland.gov.uk/Publications/2010/05/05134234/3).
Average farm business income: £39,219 and average Net Farm Income: £29,828 - average for all types of activities at current prices-. (Source: Farm Business Income in Scotland 2007-2008: http://www.scotland.gov.uk/Resource/Doc/282745/0085530.pdf).

4.4.'Future-proofing' of the programme of measures

The scenario analysis showed that between the present day and 2050, only small changes in rainfall and evapotranspiration are expected in this region, so little change in water quality is expected due to climate change alone. Future mean annual precipitation is projected to be equal to that during the baseline or at most 5% higher, whilst potential evapotranspiration (PET) may be 4-9% higher (Table 7). Simulated future mean annual runoff and associated discharge reflects the balance of precipitation and PET through the year, and may change by up to 7% under these climate projections. However, the likely direction of change is uncertain, with some climate models predicting an increase, others a decrease (Table 7). Agricultural phosphorus delivery to streams is dependent on runoff processes, and so the change in runoff under future climate results in a similar change in tSRP load in the stream. This loading could decrease by 3-5% or increase by up to 6%, depending on the scenario. These changes are small in themselves, and projected changes in associated in-stream concentration are smaller: decreased delivery under some scenarios (CC1 and CC3; Table 7) is offset by decreases in runoff, and so the concentration stays roughly constant compared to the baseline; in the scenario where delivery increases (CC2 in Table 7), runoff also increases and so concentration hardly changes. Climate change alone is therefore unlikely to cause any shift in environmental status of the Corskie Burn.

In some areas, future land use change has the potential to bring about far greater changes in water quality than climate change alone (e.g. Ianis et al., 2014; Dunn et al., 2012). However, in this area this does not seem to be the case. The 'best' and 'worst' land use change scenarios (see Section 3.1), when combined with the 'best' and the 'worst' climate model projections (in terms of SRP concentration), together give only small reductions in in-stream phosphorus concentration, of the order of 2 to 7% (Table 7). These equate to reductions in SRP of less than 1.4 $\mu g\ l^{-1}$, i.e. insignificant for the modelling. The small impact of land use change in this sub-catchment is because little change in arable land cover is considered plausible – any significant woodland expansion in this region is likely to take place in the middle or upper reaches of the Dee catchment, rather than on the prime arable land of the study sub-catchment.

The combined cost-effective mitigation measures bring about much larger changes in SRP (between 14 and 18%) and concentration (between 25 and 33%; Table 7) than the projected percent changes due to climate and land use change. Under future environmental conditions, the effectiveness of the combined measures causes a further 3 to 5% reduction of in-stream P concentration. This implies that any mitigation measures undertaken to improve water quality today are likely to be similarly effective by the 2050s. This conclusion is however subject to uncertainty, e.g. current climate models are poor at characterising projected changes in rainfall intensity, which is particularly important in the delivery of sediment and phosphorus to streams.

Table 7. Percentage change in climate variables, discharge and water quality variables at the catchment outflow under future environmental change scenarios. Minus symbols indicate a decrease. PET is potential evapotranspiration, SRP soluble reactive phosphorus. CC1-CC3 refer to output from the three climate model simulations used. GS and NE are the Global Sustainability and National Enterprise land use change scenarios used.

Category	Scenario	Precipitation	PET	Discharge	SRP load	SRP concentration
Climate change alone	CC1	1	6	-5	-5	-0.2
	CC2	5	4	7	6	-1
	CC3	2	9	-4	-4	1
Climate and land use change	Best (GS + CC2)	5	4	7	-0.5	-6
	Worst (NE + CC3)	2	9	-4	-8	-3
Baseline and future effectiveness of measures	Baseline + measures				-18	-33
	Best + measures	5	4	7	-19	-37
	Worst + measures	2	9	-4	-27	-37

4.5. Overall disproportionality assessment

The results detailed above indicate that derogation of WFD's objectives on the basis of disproportionality in this case study is not justifiable at the timescales analysed here, and would be even more so if the benefits to the loch and the wider benefits (e.g. carbon sequestration, reduced soil erosion) were included. It should be noted that extended time scales not only increase economic profitability due to different discount rates applied to costs and benefits (see Table 5), but would also allow sectors to prepare and adapt budgetary planning to new measures (reaching the target by 2015 might simply be technically impossible). At present rural development grants potentially limit the financial burden faced by farmers but there are likely to be farms where significant income losses can be anticipated (particularly in relation to reducing livestock density) even if the costs to the average farm are modest. Climate and land use change scenarios are unlikely compromise the effectiveness of the measures in the future in this particular case[9].

[9] For the interested reader, similar modelling studies in another catchments carried out as part of the REFRESH project have shown significant land use and climate change impacts on the effectiveness of measures (Jackson-Blake et al. 2013).

5. Discussion

This research is conceived foremost as a methodological exercise, and the specific policy implications of the case study (section 4.4) should be interpreted with care, as each of the analytical steps has its own limitations. For example, the use of national average WTP values for the estimation of non-market benefits of improved water quality is clearly subject to potentially significant transfer errors. Similarly, due to limited monitoring data and challenges the in application of complex hydro-chemical models, the simulation of P behaviour is unlikely to have fully captured the soil and in-stream processes in the sub-catchment. These limitations are well covered in the corresponding disciplinary literature (e.g. Martin-Ortega, 2012; Jackson-Blake et al., 2013). The focus and value of this paper lie in the critical reflexion of the integrated approach to the economic analysis of the WFD, whose key methodological challenges are in our view the following:

- Addressing multiple-stressors: The analysis presented here is focused exclusively on phosphorus mitigation. This is certainly a key pollutant in this catchment, and a key reason for not achieving water quality targets. However, the local stakeholder consultation highlighted other pressures on the aquatic environment that also affect the ecological status (e.g. physical modification). The current understanding of the combined effects of multi-stressors acting on water systems (and species, species interactions and species-stressor relationships) is extremely limited. Further research should consider modelling the effectiveness of measures in order to address multiple land and water usage causing multiple-stressor conditions.

- Selection of salient measures: Similarly, the cost-effectiveness analysis carried out here was limited by the number of measures that could be incorporated into the hydro-chemical modelling exercise. The primary limitation here was a lack of background data on the current status of the catchment and inadequacies in the process representation within the model for addressing a wider range of measures. As such, a number of measures identified by stakeholders as being relevant in this area could not be modelled, and it is therefore possible that potentially more cost-effective measures may not have been considered as part of the final output. More data on, for example, the number and location of septic tanks, discharges from small sewage treatment works, the number and location of livestock within the catchment, the number, location and condition of buffer strips and fences within the catchment, etc., would make a helpful contribution.

- Time lags: There is commonly a lag between the implementation of mitigation measures and the observation of improved water quality within the river system, especially for mitigation targeting P losses from agricultural areas (Kronvang et al., 2005; Meals et al., 2009). Thus it is likely that expected improvements in water quality may not be immediately apparent following management changes. This lag could have a negative influence on the attitude of farm managers to implementing such changes, as well as on the aggregation of benefits over time for the analysis of disproportionality. In our case study, we assumed GES was achieved in each of the three time horizons tested, as a sensitivity analysis, but this might not be necessarily the case. Some of the factors affecting time-lags are captured by the water quality models, but changes would still require adequate time-periods of environmental monitoring (>10 years) to demonstrate their effect in practice.

- The scale challenge: There is often a scale mismatch between the model and the solution. It seems highly likely that phosphorus emissions are highly influenced by stocking or cultivation practices close to the water body. Relatively small interventions such as the fencing of watercourses or non-cultivation of steeper slopes near streams may significantly reduce the phosphate burden. Such interventions may produce an actual change greater than that predicted in a coarser grained model. Furthermore, cost-effectiveness models may assume that the loss of income is proportionate to the areal extent of the land intervention (e.g. buffer strip size), even though the yield loss may be much less because of the lower yields in wet areas abutting watercourses. Local informants may well be aware of these spatial variations in yields as well as differential erosion vulnerabilities. Participatory mapping exercises like the one tested in this case study might help in bridging this scale issue. However, while useful in gauging the importance and spatial distribution of specific measures, these discussions, in some instances, lacked the necessary detail to enhance, at a catchment scale, confidence in hydro-chemical model parameterisation and cost optimization). Further development of this technique could be targeted to identifying pollution 'hot-spots', used to gather more monitoring data and testing of measures' effectiveness at the plot or field scale, ultimately feeding into economic and hydro-chemical modelling improvements.

- Chemical water quality and good ecological status. The analysis presented here has estimated the costs of reducing phosphorous in water, equating this to good ecological status. However, it is known that defining links between chemical water quality and ecological response is not simple (Hering et al., 2010). This is further complicated when there is a need to translate ecological status into descriptions that can be understood by the general public to enable them to express their preference (and hence the values they attribute to it) for the estimation of benefits. This implies a certain miss-match between the estimation of effectiveness in 'narrow' terms (P reduction), and broader benefits. Even more so when, as demonstrated for this case, the public perceive wider benefits (i.e. beyond those associated strictly with the water environment). It has been proposed that the application of ecosystem services-based approaches may be a useful strategy in this context (Vlachopoulou et al., 2014; Martin-Ortega, 2012), by making assessments of costs and benefits more holistic and yet systematic and linking ecosystem health to wider societal concerns (Blackstock et al., 2015). It should be noted, though, that a categorical demonstration of benefits of measures in terms of final ecosystem services still represents a significant obstacle.

- Challenges inherent to the transdisciplinary approach: Integrating different disciplinary knowledges is in itself a challenging task. When additional, non-scientific forms of knowledge are to be added into the equation, methodological and practical difficulties increase. Stakeholder engagement has proven critical to this research: it has supported the analytical process by providing sources of information and it has helped in assessing the outcomes of the economic modelling. In this regard, this project corroborates earlier evaluations of the role and benefits of transdisciplinary research (cf. Höchtl et al., 2006; Mobjörk, 2010). However, engaging with stakeholders can generate un-met expectations, for example, in relation to the issues discussed above regarding multiple stressors and the partially selective approach in relation to the measures modelled; certain bias in outcomes (depending on the type of stakeholders attending the workshops) or with regard to the legacy impact of the research (i.e. what happens once the research project ends). Engaging with stakeholders also significantly increases the length of the research, requiring several iterations, feedback processes and follow-up conversations (cf. Spangenberg, 2011). Additionally, it 'exposes' the nature of scientific knowledge; thus, it might be difficult for the non-scientific stakeholder to understand the uncertainties inherent to any scientific research and its inability to provide 'ultimate' answers. It could be argued that there is a need to shift from 'consultative' to 'participatory' transdisciplinarity, ensuring knowledge integration throughout, from problem framing to application of co-constructed knowledge. However, it should be noted that the WFD has already required complex institutional and attitudinal changes. A rapid shift to full participatory transdisciplinarity might make an already steep learning curve even more demanding and require increased resources, as locally devolved participatory system assessments need careful design and facilitation.

6. **Conclusions**

The aims of the WFD pose a technical problem: is it technically feasible to achieve (close to) natural conditions in systems which are heavily shaped by anthropogenic forces? This also represents a moral dilemma as to whether it is socially desirable to implement mitigation strategies that may critically affect farming in a world of increasing food demand. This is quintessentially a 'wicked' problem for which no 'ultimate' solution can be achieved. Transdisciplinarity is widely proposed as part of the strategy to deal with this 'wickedness', assuming that mono-disciplinary approaches are unlikely to provide adequate responses to such complex socio-ecological policy questions.

The study presented here has tested how to operationalize a consultative transdisciplinary approach to such issues within the specific context of the WFD and its economic analysis, and we are convinced that results reflect a better representation of the reality of water quality improvement (and its social and economic consequences) than any mono-disciplinary approximation. However, this is not to say that transdisciplinarity is a panacea or that it is already fully operational, at least in the context of the WFD, since critical methodological challenges, as described in this paper, remain. We believe that the research agenda should be driven by attempts to address the challenges of the integrated approach, rather than (or rather than only) on improving the sophistication[10] of the individual methods that might end up bringing them further apart. Progressing from a consultative approach towards a participatory

[10] By sophistication we do not mean robustness, which is always to be strived for.

transdisciplinary approach might help, ensuring knowledge integration throughout the process. However, this would require time to allow for institutional change and the devotion of increased resources.

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