Journal of Environmental Management 161 (2015) 163–172

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Research article

Integrated cost-effectiveness analysis of agri-environmental measures for water quality

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ARTICLE INFO

Article history:
Received 27 November 2014
Received in revised form
16 May 2015
Accepted 19 June 2015
Available online xxx

Keywords:
Cost-effectiveness
Integrated approach
Mitigation measures
Water quality
Dee catchment
Nitrate
Phosphorus

ABSTRACT

This paper presents an application of integrated methodological approach for identifying cost-effective combinations of agri-environmental measures to achieve water quality targets. The methodological approach involves linking hydro-chemical modelling with economic costs of mitigation measures. The utility of the approach was explored for the River Dee catchment in North East Scotland, examining the cost-effectiveness of mitigation measures for nitrogen (N) and phosphorus (P) pollutants. In-stream nitrate concentration was modelled using the STREAM-N and phosphorus using INCA-P model. Both models were first run for baseline conditions and then their effectiveness for changes in land management was simulated. Costs were based on farm income foregone, capital and operational expenditures. The costs and effects data were integrated using 'Risk Solver Platform' optimization in excel to produce the most cost-effective combination of measures by which target nutrient reductions could be attained at a minimum economic cost. The analysis identified different combination of measures as most cost-effective for the two pollutants. An important aspect of this paper is integration of model-based effectiveness estimates with economic cost of measures for cost-effectiveness analysis of land and water management options. The methodological approach developed is not limited to the two pollutants and the selected agri-environmental measures considered in the paper; the approach can be adapted to the cost-effectiveness analysis of any catchment-scale environmental management options.

1. Introduction

The need for economic analysis for supporting water management and policy decisions in Europe is explicitly recognised in the EU Water Framework Directive (WFD). EU member States were required to adopt cost-effective water resource management measures to achieve 'good ecological status' for all waters by 2015 (EC, 2003; WATECO, 2003). In relation to the WFD, cost-effectiveness analysis (CEA) involves an integrated appraisal technique that provides a ranking of a set of management measures on the basis of their costs and effectiveness for achieving the objectives set out in the Directive. This entails the assessment of implementation costs of measures and their impacts on the water bodies to meet the pre-specified water quality and/or quantity objective at a minimum economic cost.

Balana et al. (2011) highlighted that costs and effectiveness figures vary not only among EU Member States but also within a country across the landscape and farming systems. Estimates of the costs and effectiveness of measures depend on: (a) how specific measures are implemented, (b) the environmental conditions, (c) scale (both spatial and temporal), (d) baseline situation, and (e) land use types and management practices. Uncertainty and time-scales of effectiveness and obtaining accurate cost estimates of measures over a period are another key challenges in assessing the cost-effectiveness of measures to reduce diffuse pollution from agriculture (Collins et al., 2014). Challenges in assessing the effectiveness of measures are partly due to spatial scale issues. Measures are implemented at farm level, whilst ecological targets are set at the sub-catchment or catchment scale (Bouraoui and Grizzetti, 2014). Temporal issues also need to be taken into account, as achievement of good ecological status over short time-scales appears problematic due to lags in water quality responses (Kronvang et al., 2005; Meals et al., 2010). Different combinations of these

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http://dx.doi.org/10.1016/j.jenvman.2015.06.035
0301-4797/

factors result in different effectiveness and cost estimates. Thus, identification of localized, targeted and context specific mitigation measures and the assessment of their respective costs and effects could help achieve the Directive's objectives in a more cost-effective manner than standardized prescriptions of mitigation measures (Balana et al., 2012).

In assessing the cost-effectiveness of agricultural related measures, Balana et al. (2011) highlighted the need to capture the heterogeneity of real world farms instead of considering 'representative' farms, due to the variation in site-specific cost and effectiveness of measures in mitigating pollutant losses. Balana et al. (2011) advocated a need for better integration of biophysical modelling with economic analyses in order to assess multiple effects of mitigation strategies. Such an approach requires the assimilation of detailed data at an appropriate resolution within an interdisciplinary framework in order to establish the environmental and socio-economic criteria used in the cost-effectiveness modelling. One reason why this is important is to ensure internal consistency in terms of the physical definitions of the measures that are being implemented, as a mismatch in this may have significant implications both for the costing of the actions and their effectiveness in practice. A good example of this is a constructed wetland where it is necessary to define not only the location and proposed management of the wetland in order to determine its effectiveness (Berninger et al., 2012), but also its size, method of construction and continued management requirements in order to evaluate its cost.

Few studies have adopted such an integrated approach and the range of mitigation measures explored has been limited. One exception is the study undertaken by Ghebremichael et al. (2013), where an integrated tool was developed to aid in the identification and mitigation of critical source areas of pollution at a catchment scale, while maintaining economic viability at the farm scale. However, Panagopoulos et al. (2011) recognised that there are major limitations of such a combined methodology, linked to the deficiencies of the process-based modelling tools in representing natural processes and mitigation measures and the true costs of their implementation.

A key objective of the EU-FP7 REFRESH project was to develop a system to enable water managers to design cost-effective restoration programmes for freshwater ecosystems at the local and catchment scale, in the context of the Water Framework Directive, accounting for expected future impacts of climate and land use change. To demonstrate this approach, the EU-FP7 REFRESH project (http://www.refresh.ucl.ac.uk/) identified six demonstration catchments across Europe (England, Scotland, Greece, Finland, Norway, and Czech Republic), broadly representing the major climatic regions, land use and water body pressures present across Europe. This paper presents an empirical application of the integrated hydro-chemical and economic modelling approach in the River Dee catchment (one of the six demonstration catchments in the project), in North East Scotland. The methodological approach has been adapted and applied in the other five demonstration catchments of the REFRESH project. The generic step-by-step assessment approach adopted in this paper can be adapted and applied in any catchment or sub-catchment scale cost-effectiveness study.

2. Integrated modelling approach

A common framework for integrating hydro-chemical and economic analysis was developed within the REFRESH project for application across the six demonstration catchments. CEA methodologies are themselves comprised of various sequential steps (Defra, 2005; WATECO, 2003); these were combined with hydro-chemical modelling steps to create an integrated modelling methodology. Briefly, the steps involved were:

Step 1: Assessment of current water quality and ecological status to identify water quality issues and set targets. This includes identification of the major pressures (e.g. excess nutrient inputs) and their sources (e.g. agricultural activities, sewage effluent).

Step 2: Identification of a set of mitigation measures. The types of measures chosen depend on the nature of the key pressures and their sources identified in step 1 above.

Step 3: Baseline application of hydro-chemical models to reproduce recent hydrological and chemical responses.

Step 4: Collaborative refinement of mitigation measures, between hydro-chemical and socio-economic scientists, such that the measures can be realistically represented within the models at the relevant scale.

Step 5: Hydro-chemical simulation of the effectiveness of each individual mitigation measure compared to the baseline.

Step 6: Cost estimates of mitigation measures. This involves the assessment of the economic costs (such as material/resource costs, labour costs, capital costs, and operational costs) of the selected mitigation measures.

Step 7: Assessment of the cost effectiveness of individual mitigation measures and identification of the most cost-effective combination of measures to achieve water quality targets. This involves application of an optimization approach to integrate economic cost and effectiveness data in order to determine the least cost way of achieving environmental objectives.

Additional steps were also included in the REFRESH integrated modelling approach to explore the future robustness of the mitigation measures under scenarios of climate and land use change. However the results of these steps are not reported here.

3. Study area and identification of key pressures

The River Dee catchment (North East Scotland; Fig. 1) is a large (*ca.* 2100 km^2), relatively unspoilt area, famed for its salmon fishing, shooting and hill walking. It has been designated at European level for the species it supports, in particular Atlantic salmon (*Salmo salar*), freshwater pearl mussel (*Margaritifera margaritifera*) and otter (*Lutra lutra*). The catchment is subject to significant pressures, including morphological alterations and nutrient inputs from sewage and agriculture, and the area remains a top conservation priority. Within the Dee catchment there is significant heterogeneity in climate, topography, soils and land use, and consequently the key determinants and targets for water quality are variable across the catchment. Two sub-catchments of the Dee were selected for more in-depth study to explore the cost-effectiveness of mitigation measures under different conditions – the Tarland Burn catchment and the Corskie Burn catchment (see Fig. 1).

Tarland Burn catchment, in the middle reaches of the Dee, is characterized by a mixed land use. The catchment supports a wide range of land uses. On the upper slopes heather moorland used for sport shooting gives way to plantation forestry which in turn meets the upper fields of the farms in which beef cattle and sheep are grazed, interspersed between fields of improved grass. Agricultural activity is typically mixed cereal and livestock, dominated by the fattening of cattle and malting barley production and some sheep on improved pasture on the uplands. There are 4369 cattle and 8566 sheep in total (Agricultural Census, 2008) in the farms located in the catchment. An estimated total of 4472 people are living in and around this sub-catchment. Water quality examinations at the lower end of the Tarland sub-catchment (SEPA, 2009) indicated that faecal indicator organisms (FIO), N and P loadings were significant issues compromising water quality in Tarland. These issues are related to diffuse pollution whose main sources are agricultural

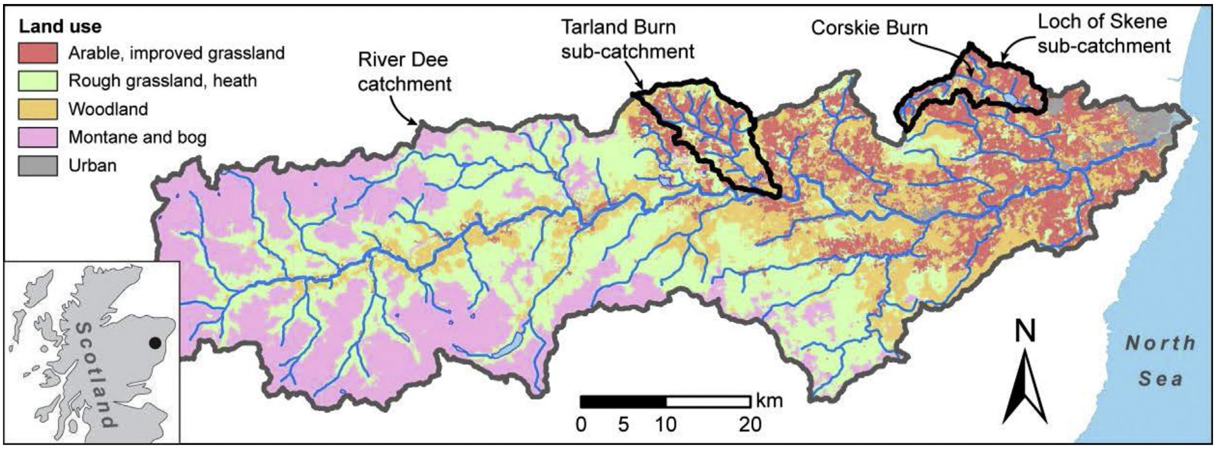

Fig. 1. Map of land use in the River Dee catchment in north east Scotland simplified from LCM2007 land use. ©Crown copyright and database right (2012) Ordnance Survey Licence Number 100019294.

activities. Water quality problem due to diffuse inputs of nutrients primarily nitrogen pollution and sediments from agriculture are the key issues in the catchment.

The Corskie Burn catchment is in the lower parts of the Dee catchment and is a predominantly agricultural catchment which feeds a shallow lake (the Loch of Skene) with an area of *ca.* 1 km^2 (Fig. 1). Both the stream and the lake were classified as having 'Poor' ecological status in 2008, due to phosphorus loading from sewage treatment works and diffuse agricultural inputs, and barriers to fish migration. High nutrient inputs lead to annual cyanobacterial blooms within the lake, which may be toxic to humans, mammals and fish. The catchment is mainly rural and predominantly under agricultural use. The major uses of the land are for arable farming (40%) and grassland (37%) and forestry (14%). The farming is typically mixed cereal and livestock, dominated by the breeding and fattening of cattle and mixed arable production and some sheep. There are about 4377 cattle and 5000 sheep (Agricultural Census, 2008) in the farms located in the catchment. The study of the Corskie Burn focussed on mitigation of P pollution, including two point sources (WWTWs) and diffuse agricultural sources.

4. Identification of mitigation measures

Identification of mitigation measures was based on previous work in the catchments (Defra, 2004; Cuttle et al., 2007; Newell Price et al., 2011), expert consultations and stakeholder input (Martin-Ortega et al., 2012; Martin Ortega et al., 2015). The Scottish Rural Development Programme (SRDP 2007–2013) document was also consulted. Catchment level specific factors and attributes such as dominant land use type and farming systems, environmental pressures and sources of pressures were taken into account for selecting the set of measures. A total of 10 broad N mitigation measures in the Tarland were considered for cost-effectiveness analysis, further broken down into 23 sub-measures (Table 1).

Note that the level at which the measures are implemented across the catchment (e.g. 20% or 50% reduction in fertilizer application rate) was selected to reflect the practicalities of achieving uptake of different measures. Even where cost-effectiveness is identified as being good, there is unlikely to be full uptake, due to a range of other barriers (e.g. social, attitudinal and differences in perception) that commonly inhibit diffuse pollution mitigation efforts (Christen et al., 2015; Wynn et al., 2001). A similar but simplified set of measures was examined in the Corskie Burn catchment for mitigation of diffuse P pollution (Table 2).

In addition to measures linked to the land use sectors, due to the wastewater treatment works (WWTWs) being an important source of P in the Corskie Burn sub-catchment, three specific measures targeting WWTWs were also identified.

5. Methods

5.1. Modelling N in the Tarland catchment

The STREAM-N model (Dunn et al., 2013) was used to simulate hydrology and stream nitrate concentrations within the Tarland Burn catchment. The N model is based on a daily balance of inputs and outputs of total mineral N (NO_3^- and NH_4^+) according to land use and crop types which are assigned to each 100 m grid cell within the model. The grid structure of the STREAM-N model simplifies integration of field-scale land use and characterisation of spatial properties such as the proximity of different land parcels to the stream network. The hydrology is driven by inputs of precipitation and evapotranspiration, with a simple degree-day snow module to account for snow accumulation and melt in the winter. The daily balance calculates the amount of N available for leaching at any time, using the terms given in Equation (1), where $availN_t$ is the N available for leaching at time-step t, *fertN* is the input of N as inorganic fertiliser, *cropN* is the uptake of N by crops, *depN* is the input of N from atmospheric deposition, *minerN* is the release of N by mineralisation, *denitN* is the loss of N by de-nitrification, *OrgN* is the available fraction of mineral N in added organic material. Different values for *fertN* and *cropN* are estimated according to crop type and take seasonally varying values.

$$availN_t = availN_{t-1} + fertN_t - cropN_t + depN_t + minerN_t - denitN_t + orgN_t \ldots \tag{1}$$

A baseline application of the STREAM-N model to the Tarland Burn catchment is presented in Dunn et al. (2013). This study demonstrated that the model was capable of simulating stream flows and nitrate concentrations at the catchment outlet with an acceptable degree of accuracy, and that the model was also capable of simulating the variability in responses as a function of land use at scales of 6 km^2 or greater. At smaller spatial resolutions the accuracy of nitrate simulations was poorer and this was believed to be linked to a lack of detailed knowledge of the management practices that are applied at a field scale.

Table 1
N mitigation measures considered in the Tarland sub-catchment.

No.	Main measure	Sub-measures	Coded as
1	Convert arable to extensive grassland	• 20% change to extensive improved grassland	1 a(i)
		• 50% change to extensive improved grassland	1 a(ii)
		• 20% change to rough grazing	1 b(i)
		• 50% change to rough grazing	1 b(ii)
2	Winter cover crops	• Uptake of 20 kg/ha through winter	2
3	Reduce stocking rates	• 20% reduction applied across all fields	3 a(i)
		• 50% reduction applied across all fields	3 a(ii)
		• Complete livestock removal from 20% of land	3 b(i)
		• Complete livestock removal from 50% of land	3 b(ii)
4	Nutrient management planning	• For 20% of arable crops	4 a
		• For 20% livestock	4 b
5	Reduce fertiliser application rates	• 20% reduction applied across all crop land	5 a(i)
		• 50% reduction applied across all crop land	5 a(ii)
6	Increase capacity of farm manure stores	• 2 kg/ha reduction in grassland N excess	6 a
		• 5 kg/ha reduction in grassland N excess	6 b
7	Fence off streams from livestock	• Reduction of 10 kg/ha for stream reachesbounded by grassland	7
8	Establish riparian buffer strips	• 6 m buffer strips adjacent to arable land, no management	8 a(i)
		• 6 m buffer strips adjacent to arable land, with management	8 a(ii)
		• 10 m buffer strips adjacent to arable land, no management	8 b(ii)
		• 10 m buffer strips adjacent to arable land, with management	8 b(ii)
9	Establish and maintain constructed farm wetlands (CFW)	• CFW type 1: farmyard and field interception	9 a
		• CFW type 1: field interception	9 b
10	Septic tank management measures	• 12.5% efficiency of N removal from septic tanks	10

The suite of measures presented in Table 1 was implemented in the STREAM-N model in a variety of different ways according to the nature of the measure. Some of the measures (e.g. reductions in fertiliser application rates) involved a simple modification to the model input across the whole catchment, whilst others (e.g. managed buffer strips) involved spatially explicit modifications applied at specific points in space. For each of the measures, Table 3 summarises how it was implemented in STREAM-N and the related assumptions.

Outputs from the effectiveness simulations included the percentage reduction in stream nitrate concentrations at the outlet of the catchment, total reduction in nitrate load at the catchment scale and the reduction in nitrate loss (kg N/ha) at the mitigation site. Mitigation measures were initially tested individually, then, following assessment of cost-effectiveness, the effectiveness of a bundle of measures was also evaluated.

5.2. Modelling P in the Corskie Burn

A similar approach to that presented above for N was used to simulate P dynamics and mitigation in the Corskie Burn catchment. In this case the INCA-P (INtegrated CAtchment Model of Phosphorus dynamics) model was used to simulate daily in-stream mean phosphorus concentrations for baseline conditions and to evaluate the effects of changing land management. INCA-P is a dynamic, process-based model that uses a semi-distributed approach to simulate the flow of water and nutrients through the terrestrial system to river reaches (Wade et al., 2002). INCA-P requires input time series of precipitation, air temperature, hydrologically effective rainfall and soil moisture deficit. Phosphorus inputs to the system include daily application rates from fertilizer, manure spreading, grazing animals, and atmospheric deposition and point inputs to the stream (septic tanks and sewage treatment works). For each reach in the catchment the model outputs include daily time series of flow, and concentrations and loads of soluble reactive phosphorus (SRP), total phosphorus, particulate P and suspended sediment. Routine stream water monitoring data from the Scottish Environmental Protection Agency (SEPA) were used to calibrate and test INCA-P.

5.3. Estimation of costs of mitigation measures

Cost estimates of measures were based on income foregone and additional costs as a result of implementing the measure. Costs were estimated using farm level data from the Farm Management Handbook (SAC, 2008), the Farm Management Pocketbook (Nix, 2011), Scottish Rural Development Programme –rural priorities payment rates sheet (freely accessible here: http://scotland.gov.uk/Topics/farmingrural/SRDP/Background/RDCsmanagementpaymentrates), ADAS mitigation measures –

Table 2
P mitigation measures considered in Corskie Burn sub-catchment.

No.	Main measures	Sub-measures	Coded as
1	Convert arable to grassland	• 20% arable to rough grazing	1 a
		• 50% arable to rough grazing	1 a
2	Reduce WWTWs inputs	• Reduce effluent concentration to 3 mg/l	2 a
		• Reduce effluent concentration to 1 mg/l	2 b
		• Remove altogether (piped elsewhere)	2 c
3	Reduce manure inputs	• 20% reduction across improved grassland 3	3 a
		• 50% reduction across improved grassland	3 b
4	Reduce fertiliser application	• 20% reduction to arable	4 a
		• 50% reduction to arable	4 b
		• 20% reduction to grassland	4 c
		• 50% reduction to grassland	4 d

Table 3
Methods and assumptions used to implement the selected mitigation measures in the model and associated assumptions.

Code[a]	Method of implementation	Assumptions
1 a	Fields defined as arable in baseline model are converted to improved grassland with a reduced N excess to reflect extensive grazing.	Reduction in animal numbers is proportional to N excess on improved grassland.
1 b	Fields defined as arable in baseline model are converted to rough grazing.	
2	Fields which will grow a spring cereal are defined with a preceding winter catch crop.	No fertiliser is added to the catch crop and it uptakes a total 20 kg ha^{-1} N between Sep–Dec and Feb–Mar.
3 a	A uniform reduction in the N excess is applied to all improved grassland fields.	Reduction in animal numbers is proportional to N excess on improved grassland.
3 b	A proportion of improved grassland fields are converted to rough grazing.	
4 a	Fertiliser inputs are increased by a factor of 1.3 for all arable cropping.	Based on unpublished farm nutrient budget data (A Sinclair, *pers comm*). Model presently assumed good nutrient management therefore effect is assessed as an *increase* in N loss under poor management.
4 b	The N excess for improved grassland is increased from 22 kg ha^{-1} to 47 kg ha^{-1}.	Based on unpublished farm nutrient budget data (A Sinclair, *pers comm*). Assessed as an *increase* in N loss under poor management.
5	Fertiliser application rates are reduced from baseline level with crop uptake also modified to reflect sub-optimal yield.	20% reduction in fertiliser leads to 5% reduction in yield, 50% leads to 16% reduction in yield (Von Blottnitz et al., 2006). Reduction in crop uptake is assumed proportional to reduction in yield.
6	N excess for improved grassland areas is reduced to account for improved handling of manure.	Reduction in N loss is typically between 2 and 5 kg ha^{-1} averaged over the farm area (Cuttle et al., 2007).
7	All model stream cells with adjacent grazed land have their N input decreased by 10 kg ha^{-1}.	Based on estimated time cattle stand in water (Gary et al., 1983), total cattle numbers and % of improved grassland bounded by a stream.
8 a	All model stream cells with adjacent crop land have their fertiliser inputs reduced pro-rata according to buffer width.	Fertiliser inputs to the fields are reduced according to reduction in area. All streams are assumed already to have a 2 m buffer and only the additional width is considered.
8 b	In addition to fertiliser reductions as per 8a, 1% of N in sub-surface flow is removed per m of buffer width.	Buffer is managed to enhance wet anaerobic conditions, leading to increased de-nitrification.
9 a	Point source reduction in stream N load.	Each farm holding >50 ha installs a constructed wetland to treat farmyard runoff which leads to a 5.9 kg ha^{-1} decrease in N loss (based on Gouriveau et al., unpublished data).
9 b	One large wetland draining 47 ha of land is implemented in the model. N is removed from all flow (surface and sub-surface) routed through the wetland.	Efficiency of N removal by wetland is 47%.
10	Point source reduction in stream N load, based on estimate population served by septic tanks.	N content of domestic wastewater estimated at 45 mg l^{-1} Amador et al. (2007). Improved treatment of septic waste can reduce N loss by 12.6%.

[a] See Table 1 for a description of the measure associated with each code.

user guide (Newell Price et al., 2011) and a Defra report (Defra, 2004).

Cost estimates for the WWTWs related measures were obtained as follows: two WWTWs in Corskie Burn, at Dunecht and Lyne of Skene settlements were considered. The population in both settlements is less than 1000. According to information from Scottish Water (Bell D., Pers. comm.), Dunecht WWTW is designed to process up to 3 times the Dry Weather Flow, which for this site is 12 m^3/day. Annual wastewater processing capacities for plants were computed on the basis of this figure. It was indicated that two biological treatment types are in use –Rotating Biological Contractor (RBC) and Activated Sludge (AS) for Lyne of Skene and Dunecht plants, respectively. A recent work on treatment potential and costs of various WWTW technologies in UK (EA, 2012) highlighted the costs of various wastewaters P treatment technologies (Fig. 2).

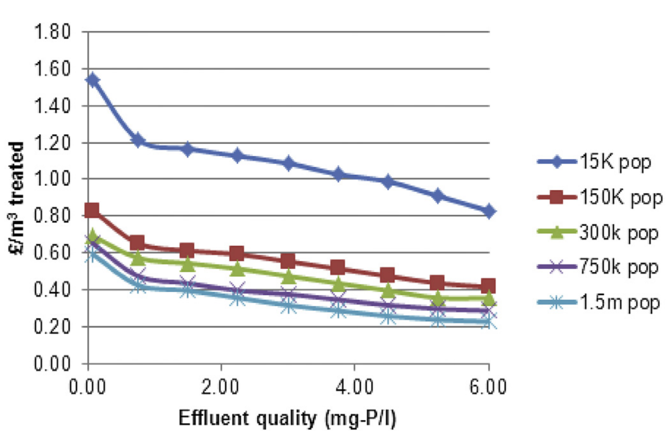

Fig. 2. Cost curve for total treatment costs (Net Present Value (NPV)) over 20 years) versus target effluent quality for a number of different wastewater treatment works size categories, based on population served (source: EA, 2012).

Based on the population size served by the two WWTWs, the relevant cost curve in Fig. 2 is the upper dark–blue curve, because the population in both settlements is lower than the 15,000 population. On the basis of Fig. 2, we used the following costs for the three WWTW-related measures considered in our study: (1) Remove all effluent (reduce effluent concentration to zero): £1.6/m^3 treated; (2) Reduce effluent concentration to 1 mg/l: £1.2/m^3 treated; and (3) Reduce effluent concentration to 3 mg/l: £1/m^3 treated. As the population sizes in both settlements are lower than those represented in the upper dark–blue curve in Fig. 2, these cost figures we derived can be considered as a conservative (lower) cost estimates Further details on cost data sources and the estimation methods can be seen in Balana et al., 2013.

5.4. Integrated cost-effectiveness modelling

The costs and effects data were integrated using the 'Risk Solver Platform' (v. 12.5.1.0) optimization tool in Excel. This method indicates implementation of a simple cost-minimization technique where the objective function was the aggregate cost of measures to achieve N or P pollutant reduction in a given water body. This can be presented as an optimization problem (Equation 2).

$$\begin{aligned} &Min.C = \sum_m \sum_s \alpha_{m,s} C_{m,s} \\ &subject\ \ to: \\ &\sum_m \sum_s (E_s - R_{m,s}) \times \alpha_{m,s} \le Q \end{aligned} \tag{2}$$

where $\alpha_{m,s}$ is a binary variable assuming the value of 1 if mitigation measure m is selected at source s and 0 if the measure is not selected; $C_{m,s}$ is the total cost (£/year) of putting measure m at source s; E_s is the amount of nutrient load entering into the water body without measure m (i.e. the reference state); $R_{m,s}$ is the efficiency of measure 'm' in reducing the nutrient load; and Q denotes the nutrient load beyond which the water body fails to achieve the pre-set environmental objective in the WFD. Equation (3) gives the constraints that must hold.

$$\sum_m \alpha_{m,s} = 1, \quad \forall s \text{ and } \alpha_{\mathrm{m,s}} \in [0,1], \forall \; s, m \tag{3}$$

6. Results

6.1. Effectiveness of N measures

The baseline model simulations indicate that the total catchment nitrate loss is around 114 tonnes/year, which equates to 16.1 kg NO_3—N $ha^{-1}y^{-1}$. Source apportionment of the total loading indicates that around 52, 50 and 22 tonnes of nitrate come from arable crops, improved grasslands, and other land uses respectively. In terms of nitrate-N leaching per unit area, arable land has the greatest losses, averaging 50.3 kg $ha^{-1}y^{-1}$, followed by 24.7 kg $ha^{-1}y^{-1}$ from improved grassland and only 5.4 kg $ha^{-1}y^{-1}$ from other land uses.

Table 4 summarises the effectiveness of the different measures in terms of the % reduction in stream nitrate concentrations and % reduction in nitrate-N load at the location where the mitigation measure is implemented. In terms of the overall effectiveness of individual measures, adoption of certain single stringent measures can reduce mean stream nitrate concentrations by more than 10% of the baseline. These measures include a 50% cut in fertilizer application (corresponding to a 21% load reduction), 50% arable to rough grazing conversion (with a 19% load reduction), 50% removal of livestock (with a 14% load reduction) and conversion of 50% of arable to extensive improved grassland (with a 13% load reduction); see Table 4 for more details. It is worth reiterating at this stage the scale dependency of these results in the sense that we have (somewhat arbitrarily) assumed potential feasible uptake of measures at scales of up to 50% of the total available area. Clearly, the likelihood of this being achieved in practice is linked to the economic viability of the measures and their attractiveness as practical options for land managers. Consequently, the hydro-chemical assessment alone is of limited value in assessing mitigation potential at the catchment scale. The specific load reductions at a point also highlight that some mitigation measures could be of greater value if implemented at a larger scale. For example, the constructed wetland measure (9b) was assessed as generating a 17 kg ha^{-1} reduction in nitrate-N load, but our evaluation only

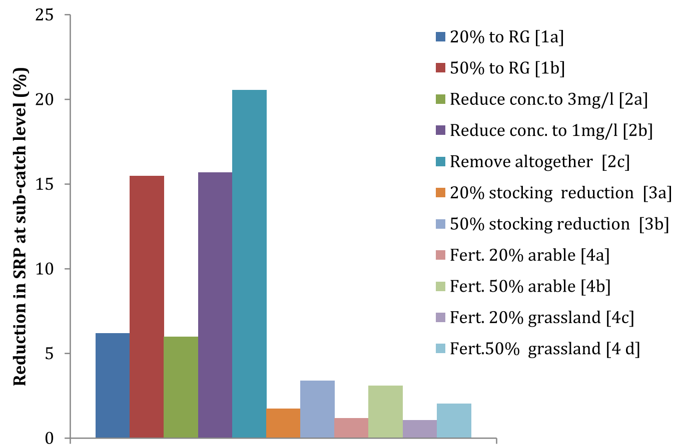

Fig. 3. Effectiveness of P mitigation measures in the Corskie Burn catchment.

Table 4
Summary of STREAM-N simulations of the effectiveness of individual measures.

Mitigation measure code	% Reduction in stream nitrate-N concentration	% catchment changed	Nitrate-N load reduction at site of change (kg N/ha)
1			
a(i)	4.99	2.90	27.09
a(ii)	13.28	7.32	27.81
b(i)	6.72	2.90	37.00
b(ii)	18.58	7.32	39.17
2	4.78	10.88	7.07
3			
a(i)	3.39	27.42	1.91
a(ii)	8.48	27.42	4.77
b(i)	5.61	5.51	15.84
b(ii)	14.10	13.71	15.98
4			
a	2.54	2.91	13.33
b	3.83	5.50	10.75
5			
a(i)	8.34	14.62	8.74
a(ii)	20.62	14.62	21.64
6			
a	1.54	27.42	0.87
b	3.86	27.42	2.17
7	2.48	3.15	10.01
8			
a(i)	0.40	1.84	3.80
a(ii)	2.01	1.84	15.89
b(i)	0.81	1.84	7.59
b(ii)	3.46	1.84	27.45
9			
a	5.10	11.04	5.88
b	0.78	0.66	16.92
10	0.19	4.00	0.61

looked at implementation of one such wetland within the catchment.

6.2. Effectiveness of P measures

Fig. 3 presents the effectiveness of individual measures (in kg SRP-P reduced per catchment per year) for the Corskie Burn catchment. Of these, sewage-related measures and the conversion of arable land to grassland appear to be the most effective, although large uncertainties in the baseline effluent loading to the stream, as well as in hydro-chemical model output, mean that further work is required. Current results suggest, for instance, that measures targeting the two WWTWs could by themselves reduce the SRP-P load up to the target of 20% (assuming the measure removing the effluent altogether is achievable, e.g. by piping it elsewhere). This seems highly stringent to achieve; however less stringent methods such as P removal at WWTWs to meet effluent concentrations of 1 mg/l or 3 mg/l may still be more effective than agricultural measures in the Corskie Burn catchment, at least in terms of SRP-P reductions.

6.3. Cost-effectiveness of measures

Fig. 4 presents cost-effectiveness results for N mitigation measures in the Tarland Burn catchment. In interpreting Fig. 4 it is important to remember that the most cost-effective measures have a low cost associated with them per kg reduction in nitrate. Of the measures considered, the reduction in fertiliser applications is the most cost-effective, followed by a range of the ‘soft-engineering’ type approaches, including the establishment of riparian buffer strips and constructed wetlands. In general the measures related to livestock activities were found to be less cost-effective than those related to arable cropping management. In particular, reducing livestock numbers or stocking density was the least cost-effective measure (ca. £21/kg NO_3–N/ha).

Our analyses show that reducing fertilizer application in the arable sector, even up to 50%, is more cost-effective than the other arable measures. Previous Defra-commissioned projects in England and Wales (e.g. Defra, 2004) found similar results. Two explanations are commonly provided for this result. First, applying less fertilizer means saving money that would have been spent on fertilizer; secondly, fertilizer reduction up to a certain threshold level has little impact on crop yield. The combined effect resulted in ‘fertilizer reduction’ being a cost-effective measure.

Table 5 presents the combination of low-cost measures to achieve the water quality targets set for mean annual nitrate concentrations in the Tarland Burn. The most cost-effective means of achieving a 10% reduction in nitrate concentration is through a combination of reducing fertiliser applications by 20%, establishing 10 m well-managed buffer strips along riparian crop land and establishing one large-scale constructed wetland. These three measures can achieve the 10% N reduction at a total cost of over £15,000. However, if we set a more stringent concentration standard (mean annual concentration of 2.5 mg NO_3^-–N/l), the cost-effective combination of measures required to achieve this target cost about 7 times as much as the cost of a 10% N reduction target (£110,803). The 2.5 mg NO_3–N/l standard requires the adoption of three additional measures (Table 3) on top of those identified for the 3 mg/l target.

As far as the SRP mitigation measures are concerned, reductions in fertilizer applications in grassland systems resulted in savings of money (Fig. 5). Conversion of arable land to rough grazing systems is the most expensive measure. Measures related WWTWS are modest in terms of cost-effectiveness.

The Corskie Burn was classified by SEPA as having ‘Moderate’ phosphate chemical status, which should have enabled a clear target reduction to be set to bring it up to ‘Good’ status. However, on examination of the monitoring data the stream was found to have been misclassified, with several outlying points exerting undue influence on the annual mean. In fact, when four outliers were excluded, the stream shifts to ‘Good’ chemical status for the entire 2000–2010 period. More, better quality monitoring data is clearly

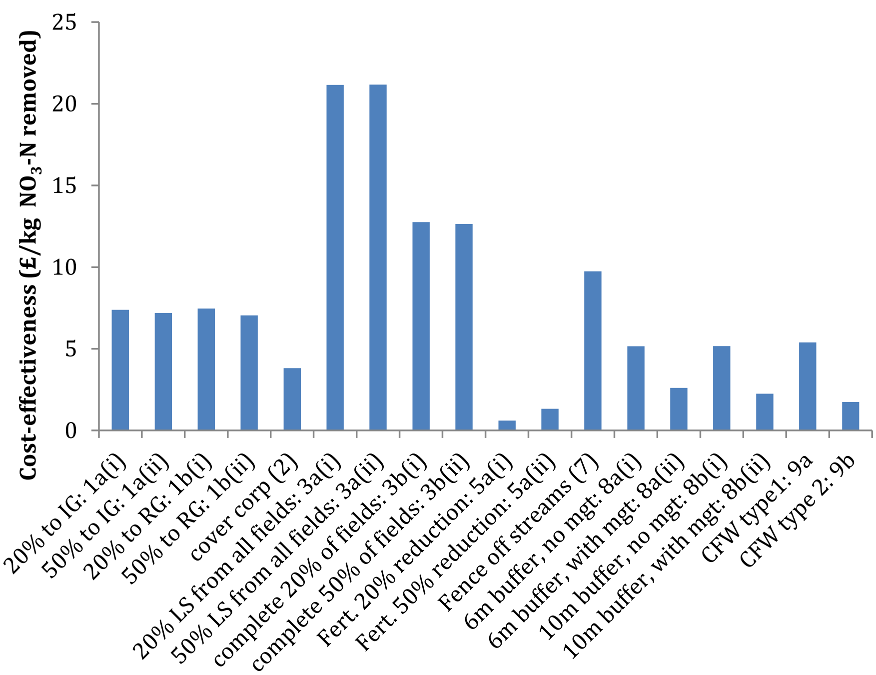

Fig. 4. Cost-effectiveness of N mitigation measures (Tarland catchment)*. [*Refer Table 1 for the full descriptions of measures. IG = improved grassland; RG = rough grassland; LS = Livestock; Fert. = fertilizer; mgt = management; CFW = constructed farm wetland].

Table 5
Cost-effective combination of measures for attaining target annual mean NO_3–N concentrations of 3 mg/l and 2.5 mg/l.

10% N reduction target (3 mg NO_3–N/l)	TC(£/catch)	CE (£/kg NO_3–N)
Measures		
• 20% reduction in applied N fertilizer – 5a(i)	5480	0.61
• Constructed field wetland (CFW) area for field interception – (9a)	1384	1.74
• 10 m buffer adjacent to arable land (with management) – 8b(ii)	8580	2.25
15% N reduction target (2.5 mg NO_3–N/l)		
Measures		
• 20% reduction in applied N fertilizer-Arable land – 5a(i)	5480	0.61
• Constructed field wetland (CFW) area for field interception – (9a)	1384	1.74
• 10 m buffer adjacent to arable land (with management) – 8b(ii)	8580	2.25
• Winter cover crops (uptake of 20 kg through winter) –(2)	20,790	3.82
• 20% reduction in applied N fertilizer-grass land – 5b(i)	33,569	3.93
• Arable grassland conversion (20% to improved grassland) – 1a(i)	41,000	7.38

TC = total cost; CE = cost-effectiveness ratio.

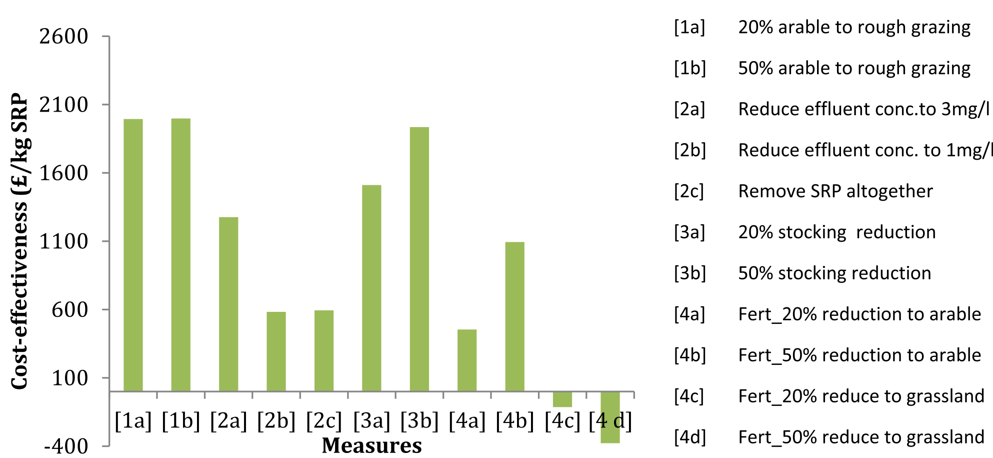

Fig. 5. Cost-effectiveness of P measures in the Corskie Burn catchment.

needed. Instead, a target 30% reduction in mean annual phosphate concentration was chosen, enough to cause a potential shift to a more oligotrophic macrophyte community (Jackson-Blake et al., 2013). As can be seen in Table 6, this target could be best achieved using a combination of: (i) a 50% reduction of fertilizer application to grassland system; (ii) a 20% reduction of fertilizer application to arable land; and (iii) investment in WWTWs to reduce effluent SRP concentration to 1 mg/l. The 30% reduction in mean annual SRP concentration in the Corskie Burn could therefore be achieved with an annual total cost of just under £40,000/year.

7. Discussion

The results reported in this paper were based on the cost-effectiveness work conducted at one of the selected demonstration catchments for the EU FP7 REFRESH project. Unlike most previous WFD-related cost-effectiveness studies which have been based on expert opinion effectiveness data (Balana et al., 2011), the REFRESH project pursued an ambitious approach to undertake integrated hydro-chemical and cost-effectiveness modelling. This has been a challenging exercise due to a range of factors such as the uncertainties inherent in environmental changes, the reliability or robustness of model predictions, and defining the appropriate spatial and temporal scales. Despite these challenges, we have developed a methodology for integrating catchment modelling outputs and economic analyses, and demonstrated its successful application in the case studies presented in this paper. The effectiveness data were derived from hydro-chemical modelling, and were integrated with the economic data to generate the cost-effectiveness results. Moreover, stakeholders (farmers, land managers and Scottish public agencies) were engaged in the selection of modelled measures, their potential application and impacts and costs.

One of the novelties of the research reported in this paper is acknowledging the variability within a catchment and disaggregated analysis of cost-effectiveness within a larger catchment. Most water management related cost-effectiveness studies are at river basin scale (Paulsen and Wernstedt, 1995; Cools et al., 2011) or at national scale (Schou et al., 2000; Brady, 2003; Hanley and Black, 2006) or even regional scale (Gren et al., 1997; Froschl et al., 2008). Such large scale analyses tend to fail to reflect the internal variability inherent in environmental systems. The results from the Dee

Table 6
Cost effective combination of P measures in the Corskie Burn catchment.

Measures for a 30% reduction in mean annual SRP concentration	Total cost (£/catch)	Cumulative (%) effectiveness (annual mean concentration)
• 50% reduction of fertilizer application to grassland system	Cost saving (zero cost assumed)	1.60
• 20% reduction fertilizer application to arable land	1874	2.60
• Reduction in effluent SRP concentration to 1 mg/l.	35040	30.00

catchment clearly demonstrate this fact. The studies on the two smaller catchments within the larger Dee catchment explored different aspects of water quality and consequently identified different measures as being key to success. Although results are preliminary, for the Corskie Burn catchment there could, for example, be an 'easy' win to be made in the context of P by tackling the major point sources of pollution, whereas in the Tarland Burn cost-effective mitigation of N is most likely to involve 'soft engineering' approaches alongside fertiliser efficiency. These findings are consistent with a move towards the identification of pollution hotspots as being the most efficient means of tackling water quality problems. Kovacs et al. (2012), for example, found that intervention with better management practices on a properly selected small proportion of the total catchment area (1–3%) is sufficient to reach a remarkable improvement in water quality compared to more broad brush approaches. Ideally, a fully integrated CEA would also take on board the consideration of multiple pollutants to evaluate multiple benefits, but either way it is clear that sub-catchment scale analysis provides more robust and realistic evidence for achieving environmental targets cost-effectively, compared with more regional analyses.

Representation of systems at a sub-catchment scale also permits a level of local knowledge to be integrated within the assessment whilst enabling a broader overview to be obtained than would be possible using a farm-scale approach. However, there are some limitations in relation to the hydro-chemical modelling in terms of the type of management interventions that can be represented by the models, the manner in which they are implemented and the uncertainties associated with the farm practices at a sub-catchment scale. Dunn et al. (2013) highlighted some of the difficulties in representing agricultural management including limitations in input data and the spatial aggregation implicit in most catchment models. For example, farmyard runoff from animal housing and slurry stores is dependent not just on how many livestock that are kept, but the duration of their indoor housing, the handling of the waste products and capacity of storage. Therefore improvements that can be gained from better management will vary on a case by case basis. Similarly, whilst implementation of buffer strips along riparian corridors should lead to a decrease in overall fertiliser inputs, other benefits such as uptake of N or trapping of sediments are highly dependent on the condition and management of the buffer strip. Thus, in evaluating the hydro-chemical effectiveness of mitigation measures every model will make some key assumptions about the implementation of the measure. Whilst data collation and process representation linked to these measures is more practical to achieve at a farm-scale, delivery and assessment of the overall benefits at a larger scale is required from the regulatory perspective. However, costs and effectiveness data generated at a small-sale, for instance, from farm accounts data, can be used to predict catchment level impacts. For example, Fezzi et al. (2010) assessed the costs of Water Framework Directive-related measures on farm accounts data and predicted catchment scale impacts for the case study of the agriculturally diverse Yorkshire Derwent catchment, in the North of England. Similarly, the methodological approach presented in this paper (the 7 steps in Section 2) can be adapted for cost-effectiveness analysis at a larger catchment or river basin scale.

From an economic perspective, CEA is not an economic efficiency analysis due to the fact that physical environmental targets are exogenously determined and they are not based on social-optimum criterion. Furthermore, the predetermined value of environmental targets or standards used in CEA greatly affects the results of the analysis, and yet setting appropriate nutrient reduction targets may be problematic. For example, the WFD currently does not specify target nitrate concentrations, so in this study several targets were selected, based on expert judgement of likely reductions required for low nutrient invertebrate communities.

There are variations in the interpretation of WFD implementation between member countries. For example, the Flemish (Belgium) interpretation of WFD standards for total nitrogen and nitrate (Cools et al., 2011) indicates that for total N, good status is below 4 mg N/L; very good status is below 3 mg N/L, whilst for nitrate, good status is between 2 and 10 mg nitrate-N per litre and very good status is less than 2 mg nitrate-N per litre. Meanwhile, in the UK no specific targets are given for surface water nitrate concentrations, despite their importance in causing surface water eutrophication, and in the context of the EU Habitats Directive higher water quality standards are required for some key ecological species. For example, for freshwater pearl mussels (*M. margaritifera*) to survive and reproduce, stream nutrient levels must be close to reference conditions for ultra-oligotrophic rivers. Thus, for the purpose of establishing nitrate concentration levels in the Tarland Burn, we reviewed a range of literature (both grey and published) and expert consultation. Nonetheless, targets were still challenging to set in this study, due to the miss-classification of the water chemistry status of the Corskie Burn (Section 6.3). It is therefore possible that different cost-effective measures would have been identified had different targets been set. Thus, CEA studies should focus not only on the cost and effects data, but also examine baseline classifications and accuracy of the environmental targets.

8. Conclusion and implications for future research

We have presented a methodology for identifying cost effective combinations of measures to reach water quality targets using a case study catchment in North East Scotland. This methodology involves linking process-based, dynamic, catchment-scale hydro-chemical modelling results with an economic analysis of measures. The generic integrated methodological approach developed in this paper can be adapted and applied to other river catchment in Europe or elsewhere. The adaptability of the methodology was tested in five different countries/catchments selected for the REFRESH project in Europe: The Thames catchment (England), Louros catchment (Greece), Pyhajarvi/Ylaneenjoki catchment (Finland), Vansjø-Høbol catchment (Norway), and Vltava catchment (Czech Republic) which broadly represent the major climatic regions and mix of land uses in Europe.

From the findings in this research the following major conclusions are derived: (1) identification of key pressure sources and targeted measures at sub-catchment scale are the best way to achieve cost-effective pollution mitigation; (2) livestock measures appear to be more costly to implement than arable sector measures when attempting to reduce nitrate inputs in a mixed agricultural catchment; (3) investment in WWTW may be a cost-effective strategy to reduce in-stream phosphate concentrations, even for a largely rural, agricultural, catchment; and (4) in the Dee catchment as a whole the WFD objectives could be achieved in a cost-effective way if targeted mitigation measures were implemented in arable agriculture sector and wastewater treatment works. Generally, we found that disaggregated cost-effectiveness assessment within a catchment is an appropriate approach for data generation, analysis, and stakeholder engagement.

The analysis presented here is focused exclusively on phosphorus and nitrogen mitigation. Though these were identified as key pollutants for not achieving water quality targets, other significant pressures on the aquatic environment affect the ecological status in the study catchment, and elsewhere (e.g. physical modification). Thus, future research should focus on addressing multiple-stressors that compromise water quality in the catchment.

Acknowledgements

This work was conducted under the EU FP7 REFRESH Project (Adaptive Strategies to Mitigate the Impacts of Climate change of European Freshwater Ecosystems: Grant Number 244121) and the Scottish Government's Rural and Environment Science and Analytical Services division (RESAS) Strategic research programme (2011–2016) Theme 2. We gratefully acknowledge the support from the EU and RESAS. The views expressed in this paper are those of the authors and do not necessarily reflect those of the James Hutton Institute, RESAS or EU. We would like to extend our sincere thanks to our colleagues Bill Slee, Rachel Helliwell, and Andy Vinten (James Hutton Institute, UK); Demetris Psaltopoulos and Dimitris Skuras (University of Patras, Greece); Prof. Andrew Wade (Reading University, UK); Martin Kernan (University College London, UK); Angel Perni and all colleagues contributed to the WP5 and WP6 of the REFRESH Project in the six Demonstration catchments and the stakeholders participated in the workshops.

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