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Cost-Effective Mitigation of Diffuse Pollution: Setting Criteria for River Basin Management at Multiple Locations

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Abstract A case study of the Yorkshire Derwent (UK) catchment is used to illustrate an integrated approach for assessing the viability of policy options for reducing diffuse nitrate losses to waterbodies. For a range of options, modeling methods for simulating river nitrate levels are combined with techniques for estimating the economic costs to agriculture of modifying those levels. By incorporating spatially explicit data and information on catchment residence times (which may span many decades particularly in areas of groundwater discharge) a method is developed for efficient spatial targeting of measures, for example, to the most at-risk freshwater environments. Combining hydrological and economic findings, the analysis reveals that, in terms of cost-effectiveness, the ranking of options is highly sensitive to both (i) whether or not specific stretches of river within a catchment are regarded as a priority for protection, and (ii) the criterion of nitrate concentration deemed most appropriate as an indicator of the health of the environment. Therefore, given the focus under European legislation upon ecological status of freshwaters, these conclusions highlight the need to improve understanding of mechanistic linkages between the chemical and biological dynamics of aquatic systems.

Keywords Water framework directive · Diffuse pollution · Nitrate · Cost-effectiveness

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Introduction

Widespread areas of England and Wales are at risk of failing to meet the 2015 requirement of "good ecological status" of the EU Water Framework Directive (WFD) (2000/60/EC) (Environment Agency 2008; Bateman and others 2006a). There is consensus that this is in the main attributable to excess nutrients (Kronvang and others 2005). The legislation calls for catchment-wide management of water resources and water quality. Modeling tools, driven by national input data coverages that represent the catchment landscape, the river drainage network and land management economics are needed by policymakers to identify the scenarios that best achieve compliance. From a research perspective there has been recent and ongoing momentum to link assessments of economics and water quality to guide and prioritize future decision making (Bateman and others 2006a; Fezzi and others 2008a).

Previous studies that have sought to predict changes in nitrate-N leaching (as a consequence of prescribed management change; e.g., Defra 2007) have made quantifications in terms of field scale loads (kg N ha^{-1}). Here the focus is different, being at catchment-scale where the relative effectiveness of scenarios can only be assessed comprehensively when flow dilution has been considered. In other words, nutrient concentrations must be considered and only then when the influence of point sources and in-river processes has been taken into account. It is believed that nutrient concentrations and their seasonal variability (rather than absolute annual nutrient loads) impinge upon the biological response in rivers, which has a direct bearing on ecological status (Maidstone and Parr 2002; Wade and others 2002).

Here we use a case-study of the Yorkshire Derwent to indicate how process-orientated mathematical models of diffuse pollution, in-river water quality and farm economics can be linked together to provide a flexible tool for policy support. This builds on a previous investigation (Hutchins and others 2008) that illustrated the potential heterogeneity of catchments (in this instance the Yorkshire Derwent) in terms of agricultural systems, hydrochemical dynamics, the relationships between land-use and water quality and the impact of land-use change on water quality. We incorporate spatially explicit data to develop a method for efficient spatial targeting of measures to, for example, the most at-risk freshwater environments. Our efficiency measure combines both hydrological and financial assessments within a single, policy compatible, cost-effectiveness assessment. The results are examined in the context of expert knowledge on groundwater hydrochemistry and residence times.

Given a focus upon generating policy relevant findings the case study is executed with reference to three recently proposed measures for WFD implementation taken from Defra (2007):

- Scenario A. 20% cut in fertiliser application;
- Scenario B. 20% cut in stocking density;
- Scenario C. 20% conversion of arable land to extensive permanent grassland.

Trade-offs between water quality improvements and economic cost have been evaluated in terms of mean nitrate-N concentration at a single downstream location (Fezzi and others 2008a). The present article extends analysis of these policy measures to:

(i) a range of locations upstream given the need to protect all freshwaters under the WFD;
(ii) pinpoint different measures of nitrate status which may be especially pertinent indices of ecological sensitivity, for example high concentrations and season-specific criteria, and how these differ in terms of cost-effectiveness.

In this way the approach embodies flexibility to address specific improvements in water quality that may be especially beneficial ecologically. By using a river model splitting the network up into small reaches it also allows focus to be made on sub-catchments that may be of particular interest in policy management initiatives such as the Defra Catchment Sensitive Farming (CSF) programme (Defra 2004).

Methodology

Farm Economic Modeling

The economic impact on farms is estimated here following the approach introduced in Fezzi and others (2008a, b). It is assessed in terms of changes in farm gross margin (FGM), defined as the difference between revenues arising from the different activities carried on within the farm and variable costs. Due to the difficulty of allocating total farm fixed costs (such as labor, machinery, etc.) to individual activities within the farm, FGM is typically the variable of interest in agricultural economics. This approach draws upon farm level account data supplied by the Farm Business Survey (Defra and National Assembly for Wales 2005) to estimate changes in FGM arising from various WFD related policy options. From this analysis it is clear that even within specific farm types (e.g., cereal farms, dairy farms) there is substantial variability in FGM. This variability is due to the heterogeneity in farming practices and in particular to the different activities carried on within a farm. Incorporating this variation is important for policy to identify the financial impacts of options such as for those (typically cropping) farms for which a 20% reduction in fertilizer use would also mean a (small) improvement in FGM. Fezzi and others (2008a) estimate regression models that relate the FGM changes arising from some WFD policy option with the different activities carried on within a farm (e.g., hectares of cereals, number of cows). Equipped with these statistical relationships between farm characteristics and potential WFD impacts, we can predict the likely FGM changes in any specific area within the UK for which the relevant pattern of agricultural land use are known. The present article applies these relationships to predict the economic impact of different policy options within the Yorkshire Derwent catchment. Cost estimates are calculated using 2005 farm account data and therefore are valid if the prices of the agricultural commodities do not vary substantially from the ones in this baseline year.

Case Study Catchment

Modeling was undertaken for the 1586 km^2 Derwent catchment draining to Buttercrambe (NGR SE 731587) in North Yorkshire, UK (Fig. 1). Annual average rainfall is 779 mm, although it exceeds 1000 mm in the North York Moors at the northern edge of the catchment. Of this rainfall, approximately 59% is accounted for as evapotranspiration (Centre for Ecology and Hydrology 2003). In 2000, land-cover proportions were 42%, 27%, 15% and 13% for arable, grass, woodland and upland cover respectively (Fuller and others 2002). The remainder was urban and suburban. Soil permeability varies greatly across the catchment, with consequent spatial differences in stream baseflow index (BFI) (Institute of Hydrology 1980) at gauged sites (Table 1). For the baseline simulation of water quality against which scenario analysis was to be compared, land-use data sets were estimated for the 2000–2003 period. These were derived from a combination of LCM2000 (Fuller and others 2002)

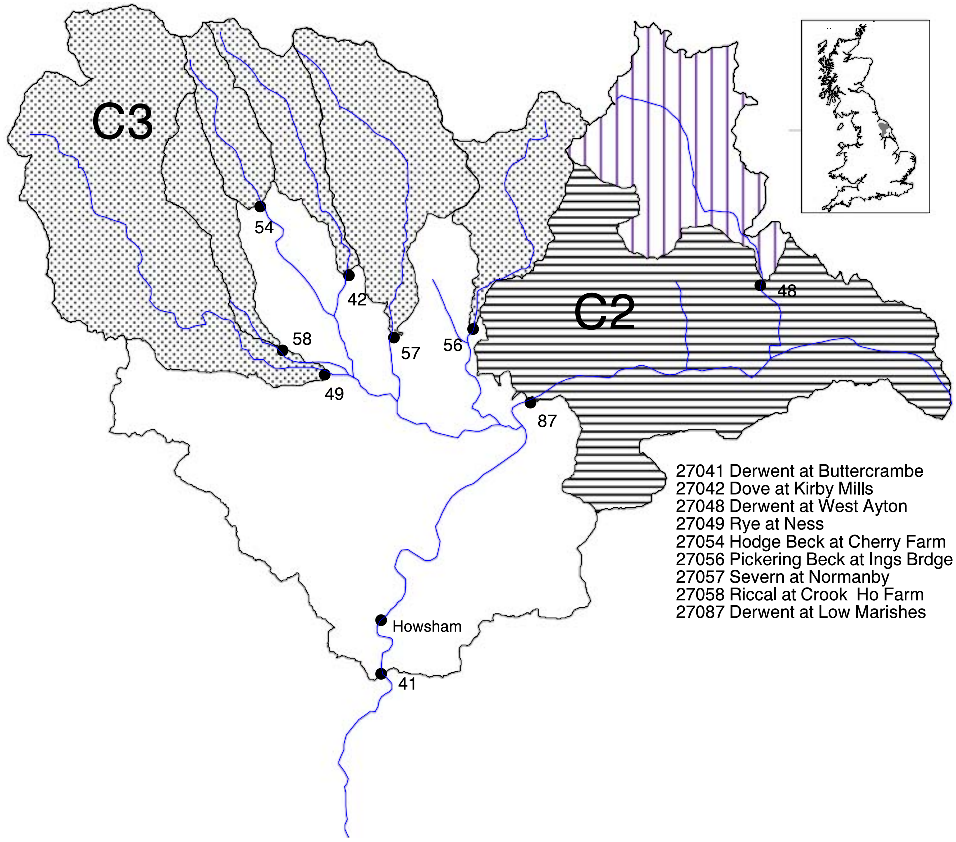

Fig. 1 Map of the Derwent catchment. Model output was assessed at five locations: Site 1: 27049 (labeled 49 on the map); Site 2: below the confluence downstream of 27054 and 27042; Site 3: 27056; Site 4: 27087; Site 5: 27041. Shaded areas represent sub-catchments in hydrometric area 27. Scenario C1 was applied uniformly across the entire Derwent catchment. Scenario C2 (*horizontal line shading*) was focused in the sub-catchment at Site 4 and Scenario C3 (*stippled shading*) in the sub-catchments at Sites 1, 2 and 3. The area upstream of 27048 (*vertical shading*) was excluded from diffuse pollution modeling and scenario analysis as high flows from this area are diverted out of the catchment via the Seacut to the North Sea at Scarborough

Table 1 Assumed current ("baseline") levels of N applied and leachable nitrate-N [example for most abundant soil HOST class (24)]; and the changes in crop residue nitrate-N and leachable nitrate-N resulting from reductions in N input

Land-use	N applied (kg ha^{-1})	Leachable NO_3-N (HOST 24) (kg ha^{-1})	kg ha^{-1} change under 20% reduction in fertilizer	
			crop residue NO_3-N	leachable NO_3-N
Dairy, permanent grassland	120	15	n/a	−4.7 (−10.4)
Dairy, temporary grassland	165	26	n/a	−7.3 (−17.6)
Beef, cattle, and sheep, permanent grassland	45	8	n/a	−1.4 (−2.0)
Beef, cattle, and sheep, temporary grassland	90	25	n/a	−4.5 (−7.5)
Winter wheat	197	40	−2.7	−1.1
Winter barley	139	39	−2.0	−0.8
Spring barley	105	40	−1.5	−0.8
Potatoes	160	76	−6.7	−3.6
Sugar beet	95	92	−1.0	−0.5
Winter OSR	209	78	−5.3	−2.2
Spring OSR	131	86	−3.3	−1.8

Leachable nitrate-N is used as an input to the CASCADE model. For grasslands, leachable nitrate-N is calculated using NCYCLE; bracketed values refer to a 50% fertilizer cut

and EDINA 2 km grid data aggregated from the Defra 2004 Agricultural Census cropping and livestock statistics.

Water Quality Modeling

A combination of the CASCADE and QUESTOR models was used (Hutchins and others 2006, 2007) to represent the daily dynamics of catchment systems in terms of diffuse and point source pollution and in-river processes. CASCADE (Cooper and Naden 1998) represents the sources (inputs) and hydrological mobilization of diffuse pollutants through the soil and their delivery to river channels. A catchment is divided geographically into hydrological response units. These units are hydrologically independent

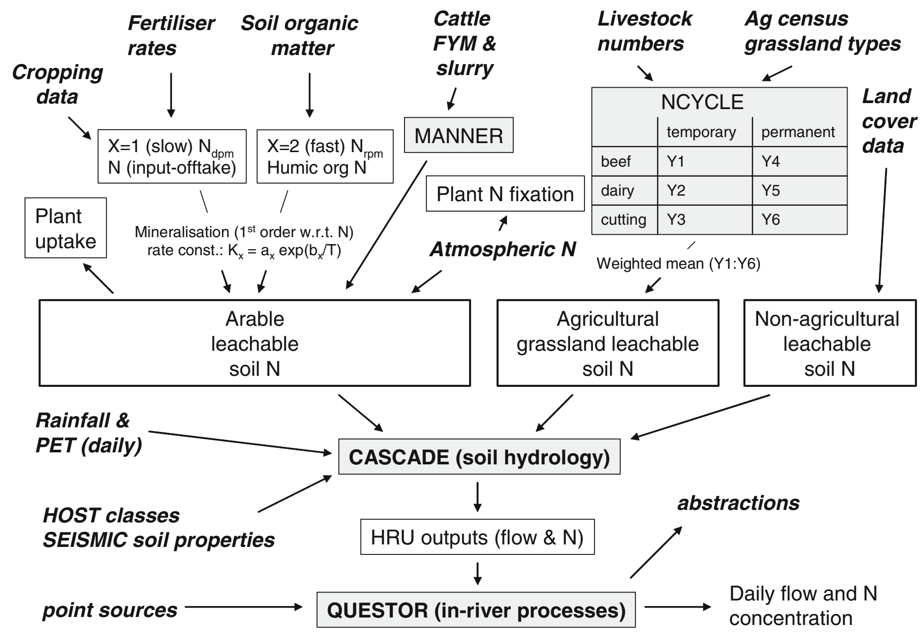

Fig. 2 Diagram illustrating the CASCADE–QUESTOR N model. Input data are indicated in bold italic type. Discrete model codes are represented by shaded boxes. Further details are given by Hutchins and others (2008)

of each other and are of approximate size 5 km^2. The mixing of the diffuse inputs with point sources (e.g., discharges from sewage treatment works) and their modification caused by in-stream processes and abstractions is represented by QUESTOR (Eatherall and others 1998; Boorman 2003, 2007), which splits the Derwent catchment into a branched network of 65 reaches. For nitrate-N, the CASCADE model includes land-use specific and soil-type specific estimates of nitrate-N available for leaching from the soil. These are provided as input to the model on a monthly basis and are sensitive to a wide range of arable crop types (including cereals, potatoes, OSR, sugar beet and other major crops, discriminating between winter and spring sowing), grassland management systems (beef, dairy and cutting systems of different production intensities and different grassland age) and fertilizer regimes (quantity of inorganic fertilizer; type, quantity and timing of applied livestock waste). The approach makes use of model representations of these systems, validated and extensively applied in numerous other studies. Concepts adopted include those of Lord (1992) and Sylvester-Bradley (1993) to represent arable crop residue N content, and the MANNER model (Chambers and others 1999) which covers the fate of manure N applied to arable land. Grassland systems are considered using the NCYCLE model (Scholefield and others 1991). The use of these concepts for derivation of the nitrate-N inputs to the model are described more fully elsewhere (Hutchins and others 2008). A simplified diagrammatic representation of how the component parts of the model fit together (Fig. 2) illustrates the linkages between model codes and input data sources.

Scenario Specification

Current fertilizer application rates were taken from the British Survey of Fertiliser Practice (2005) as detailed in Table 1.

Values for leachable N under grasslands embody sensitivity to grassland system, sward age, previous land cover and atmospheric input. Livestock numbers were used to estimate the proportion of grassland under different cutting systems. For past landcover, the relative likelihood of it being grass or arable was assumed uniform across the Derwent, a value estimated from 1990 landcover statistics, and the information used to help pinpoint an appropriate quantity of leachable N by grassland system. The model requires annual mean N fertilizer rate as an input parameter. The effect of reducing this by 20% was made (Scenario A). Based on Defra's *RB209* document (Defra 2000) and other evidence reported in the literature (Nevens and Rehuel 2003; Whitehead 1995; Jarvis and others 1995) it is concluded that a 50% reduction in N will lead to an approximate 20% reduction in grass dry matter yield (see Table 1). The assumption that this leads to a 20% reduction in stocking rates is made (Scenario B). Literature suggests that the relationship is linear at the levels of fertilization of relevance. This is borne out by modelling. A 40% cut in fertilizer on grasslands leads to leachable N values roughly 15–20% higher than for a 50% cut. To summarize, mechanistically for grasslands, Scenarios A and B both involve a reduction in fertilization rate.

For arable crops, the crop residue, defined as plant N uptake minus plant N offtake is calculated in the model.

The yield [N offtake divided by (weight per weight) N content] is not calculated, as it is not an essential intermediate step in the determination of leachable nitrate-N. For a range of crops, research findings (e.g., Lord 1992) permit confidence in what will happen to crop residues if fertilizer levels are reduced by a specified amount below or at the economic optimum (Scenario A). Less comprehensively understood is what happens to the component parts (uptake and offtake); and additionally the optimum levels themselves have not been built into the model. The assumption is made that they are not being exceeded. If this were to be the case, and there is no clear evidence of this happening to a significant extent, then the drop in leachable nitrate-N following reduction in input would be more dramatic. In addition to the crop residues, there are other sources of leachable nitrate-N under arable land use (soil organic matter, managed additions of livestock waste and atmospheric deposition). These are considered separately by the model (see Fig. 2).

When converting 20% of the arable land to extensive grassland the option (as specified in Defra 2007) of zero N application ungrazed grassland was considered (Scenarios C). All arable land accounted for as set-aside, fruit or vegetables was excluded from the conversion. The impacts of these proposed changes on leachable nitrate-N were explored using NCYCLE. The initial impact following the change and the impact 20 years into the future was evaluated. The leachable nitrate-N was substantially higher in grassland greater than 20 years old than grassland 11–20 years old. It appears that steady state takes at least 20 years to be attained and that the benefits of the change, though substantial and dramatic initially, are reduced over time.

Scenario C, representing conversion of arable land to zero N application grassland, was carried out in three alternative ways, in order to explore impacts of spatially targeting a measure. An area of 117 km^2 represents the 20% of the arable land in the Derwent to be converted. Firstly a uniform conversion across the entire catchment was made (C1). Secondly (C2) the conversion was targeted wholly within the 230 km^2 Low Marishes (NGR SE 833774) sub-catchment (excluding the area upstream of West Ayton (NGR SE 990853)) which is highlighted under CSF documentation as being intensively arable with high nutrient status. Thirdly, the conversion was targeted wholly within the sub-catchments of the Rye, Riccal, Hodge Beck, Dove, Seven and Pickering Beck (C3). The Rye is also prioritized under CSF, with a prevalence of arable crops on steeply sloping land susceptible to sediment loss. The locations of these sub-catchments and of 5 sites where detailed evaluation of policy options was undertaken are shown in Fig. 1.

Explanation of the Presentation of Results

Meteorological data from 2000–2003 were used to drive both the scenarios and the reference nitrate-N modeling. Results are presented in terms of the predicted change occurring when the system has attained a steady state. It is known that it may take many years for the effects of a change in N fertilizer regime to be seen in water quality even at the field drainage scale (Burt and Haycock 1993; Shaffer 2002). At the scale of a large catchment (e.g., Yorkshire Derwent) hydrological responses are also likely to result in further significant delays before the impact of a change is seen in the waterbody of interest and it is highly unlikely that a steady state will be reached within 10 years (Stalnacke and others 2004). In addition it should be noted that the modeling approach does not account for changes in the concentration of groundwater sources of nitrate-N. Therefore, the concept of steady state does not extend to include these very long term changes. Although the representation of point sources is important for determining in-river N dynamics, in the entire Derwent they only account for an input load of approximately 1.1 kg N ha^{-1} per year. In terms of nitrate-N in the Derwent, there is very little scope for improving water quality by reducing point source inputs. This option is not considered any further in the analysis. Reductions in the mean annual load and concentration of nitrate-N under the alternative scenarios were presented by Fezzi and others (2008a). The relative effects on loads and concentrations are similar but it is important to consider both.

Results

In water quality terms, to summarize, the model results indicated that:

- Scenario A would be least effective, yielding reductions of less than 8% at all sites.
- Scenario B, though considerably more effective at Sites 1–3 (approx 10% reduction) was only slightly more effective than A at Sites 4 and 5.
- Scenarios C1, C2 and C3 are clearly the most effective, not surprising given their more drastic nature. Scenario C1 gave changes of greater than 20% at all sites. For Site 5, C1 would prove marginally more effective than C2 or C3.

From these results, used in conjunction with an economic analysis, it is possible to compare the cost-effectiveness (ratio of change in farm gross margin to reduction in mean N concentration) at Site 5. Again, summarizing from Fezzi and others (2008a), these allow us to rank the options in the following order of decreasing suitability: C1, C3, B, C2, A. However, much information is masked by these summaries of the modeling exercises. The detailed output from the model is a time-series plot of nitrate-N at daily resolution (e.g., Fig. 3 for Scenarios C1, C2 and C3) which contains a lot of information. Plots summarizing the main features of these time-series help to illustrate important features more clearly.

Percentile plots of nitrate-N concentration at Site 5 (Fig. 4) indicate the goodness of fit of modeled data to observations (underestimates of extreme concentrations but otherwise good correspondence) and the impact of introducing mitigation measures. A lack of sufficient observations makes presentation of the most extreme percentiles somewhat meaningless; therefore they are not shown. Importantly, the plot shows that, whilst the overall impact in terms of median concentrations may be similar (e.g., for C1, C2 and C3), the shape of the percentile distribution plot may be very sensitive to the geographic area where the measure is spatially targeted.

Figure 5 shows seasonal effects, summarizing data as mean monthly concentrations. Table 2 depicts the effectiveness of the policy options at each site not only in terms of mean concentration but also in terms of 95th percentile concentration and mean value for April–June. In the eastern and southern parts of the catchment, point sources, though low, are more significant than elsewhere. They represent over 8% of the load at Site 4 and over 3% at Site 5. At other sites levels are typically 1%. This is reflected in nitrate-N concentrations, which are highest at Sites 4 and 5.

The effect of the policy options in the context of change in Farm Gross Margins are shown in Table 3. The C1 option is most cost effective for all indicators.

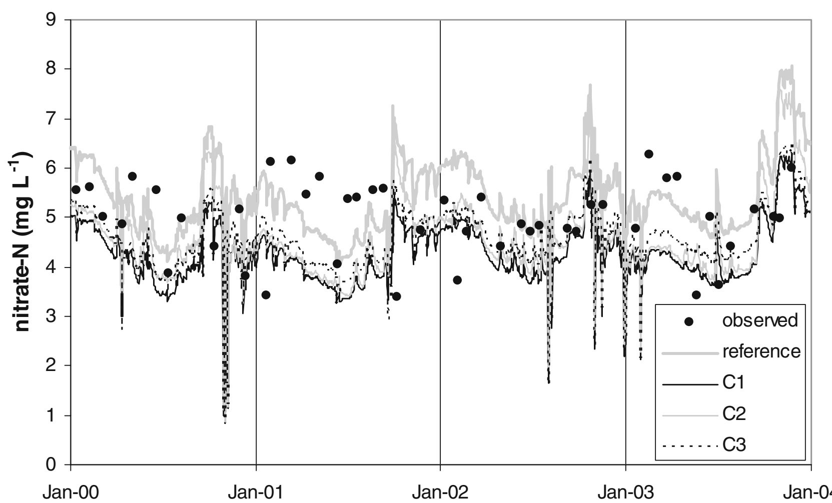

Fig. 3 Time series of nitrate-N concentration at Site 5 on a daily resolution, showing modeled responses using 2000–2003 climate and land-use (reference) and under Scenarios C1, C2 and C3. Observed data are also shown

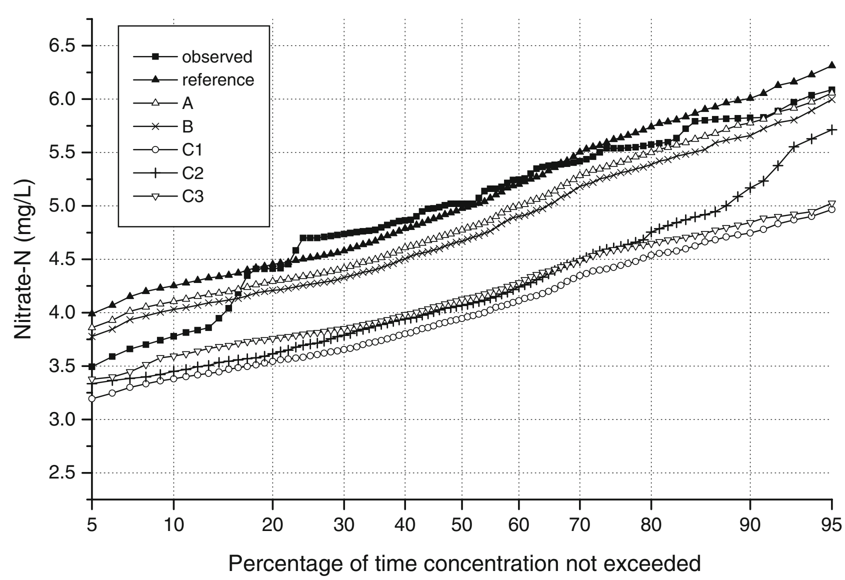

Fig. 4 Cumulative frequency plots of nitrate-N concentration illustrating the predicted efficacy of the scenarios at Site 5. Observed and reference (*modeled*) lines represent the 2000–2003 period

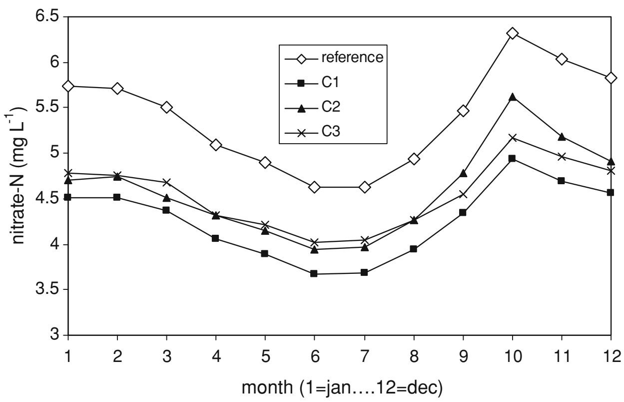

Fig. 5 Monthly mean nitrate-N concentration at Site 5 for the reference condition (*modeled*) and under scenarios. These are averages for each month over the 4 year period that simulation and scenario analysis was carried out

The Environment Agency (EA) GQA Nitrate classification, based on mean concentrations, has not been directly taken up as part of the UKTAG recommendations for standards to support WFD implementation. However, this classification is helpful to summarize the effectiveness of the proposed measures. Under baseline conditions the model predicts the following distribution of grades across the QUESTOR reach network: 0 reaches at grade 1 (very low: < 1.13 mg NO_3-N L^{-1}), 2 at grade 2 (low: 1.13–2.26 mg NO_3-N L^{-1}), 25 at grade 3 (moderately low: 2.26–4.52 mg NO_3-N L^{-1}), 34 at grade 4 (moderate: 4.52–6.77 mg NO_3-N L^{-1}), 4 at grade 5 (high: 6.77–9.03 mg NO_3-N L^{-1}) and 0 at grade 6 (very high: > 9.03 mg NO_3-N L^{-1}). The boundary between Grade 4 and 5 roughly corresponds to the EU Drinking Water Directive threshold of a 95th percentile value of 50 mg nitrate L^{-1} (11.3 mg NO_3-N L^{-1}). The spatial distribution is shown (Fig. 6a) Table 4 illustrates the catchment-wide effectiveness of the policy options addressed. This is achieved by looking on a reach by reach basis at the difference in GQA nitrate grade between the simulated baseline reference conditions and as predicted under the respective scenarios. Despite being most effective in terms of lowering the three measures of nitrate-N concentration at Site 5, C1 is inferior to both C2 and C3 in this respect. When compared with Fig. 6a, the spatial distribution of these improvements can be seen for the examples of C1 and C3 (Fig. 6b and 6c).

Table 2 Baseline simulated 2000–2003 nitrate-N concentrations (mg L^{-1}) and change in concentration under the selected policy options

Site	Nitrate-N concentration criterion	Baseline	Scenario A	Scenario B	Scenario C1	Scenario C2	Scenario C3
1 (Rye)	Mean annual	3.70	−0.21 (6)	−0.38 (10)	−0.54 (15)		−2.19 (59)
	95th %ile	4.99	−0.29 (6)	−0.51 (10)	−0.77 (16)		−2.94 (59)
	Apr–Jun mean	3.50	−0.20 (6)	−0.38 (11)	−0.50 (14)		−2.02 (58)
2 (Dove & Hodge Beck)	Mean annual	2.89	−0.19 (7)	−0.37 (13)	−0.40 (14)		−1.51 (52)
	95th %ile	3.79	−0.26 (7)	−0.48 (13)	−0.53 (14)		−2.00 (53)
	Apr–Jun mean	2.72	−0.18 (7)	−0.37 (13)	−0.36 (13)		−1.37 (51)
3 (Pickering Beck)	Mean annual	4.49	−0.19 (4)	−0.34 (8)	−0.71 (16)		−2.88 (64)
	95th %ile	5.34	−0.22 (4)	−0.48 (7)	−0.84 (16)		−3.48 (65)
	Apr–Jun mean	4.29	−0.18 (4)	−0.33 (8)	−0.67 (15)		−2.72 (63)
4 (Derwent: LM)	Mean annual	6.28	−0.19 (3)	−0.25 (4)	−1.47 (23)	−3.25 (52)	
	95th %ile	8.39	−0.30 (4)	−0.34 (4)	−2.09 (25)	−4.53 (54)	
	Apr–Jun mean	5.23	−0.14 (3)	−0.20 (4)	−1.16 (22)	−2.59 (50)	
5 (Derwent: B)	Mean annual	5.40	−0.21 (4)	−0.30 (6)	−1.14 (21)	−0.81 (15)	−0.85 (16)
	95th %ile	6.82	−0.27 (4)	−0.34 (5)	−1.51 (22)	−0.60 (9)	−1.29 (19)
	Apr–Jun mean	4.87	−0.18 (4)	−0.27 (6)	−0.99 (20)	−0.73 (15)	−0.68 (14)

The figures in brackets represent the percentage improvement (*Derwent LM* Derwent at Low Marishes, *Derwent B* Derwent at Buttercrambe, the catchment outlet)

Table 3 Change in Farm Gross Margin (£m) and the Cost-effectiveness (C.E.) (£m per mg L^{-1} nitrate-N reduction at Site 5, the catchment outlet) for each of the policy options under each of the three criteria of nitrate-N concentration

	Scenario A	Scenario B	Scenario C1	Scenario C2	Scenario C3
ΔFGM	−2.39	−1.89	−5.53	−5.53	−5.35
C. E. (annual mean)	−11.3	−6.3	−4.8	−6.8	−6.2
C. E. (95th percentile)	−8.9	−5.6	−3.6	−9.2	−4.2
C. E. (Apr–Jun mean)	−13.7	−7.0	−5.5	−7.5	−8.1

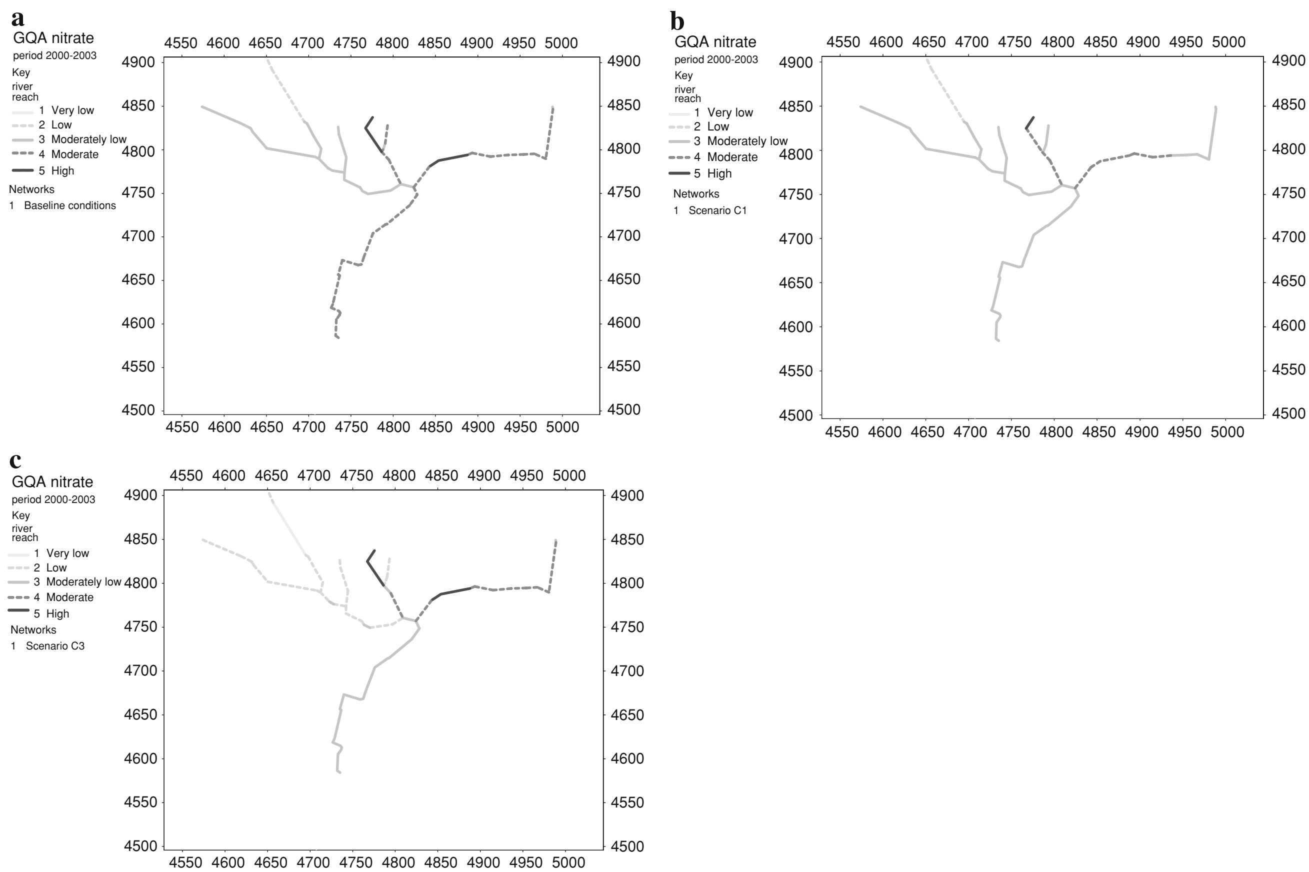

Fig. 6 Simulated GQA nitrate class for (**a**) the baseline model (reference) period, (**b**) under Scenario C1 and (**c**) under Scenario C3

Table 4 Degree of improvement in GQA nitrate grade under each of the policy options for the 65 individual reaches in the Derwent modeled river network

Improvement in grade	Scenario A	Scenario B	Scenario C1	Scenario C2	Scenario C3
No change	60	57	35	33	19
+ 1	5	8	30	30	44
+ 2	0	0	0	2	2

Discussion

Four main points arise. These are stated and then discussed in more detail below.

- At any location, temporal response is controlled by the timing of diffuse inputs (cropping/fertilizers etc) and point sources.
- Land-use assemblage, rainfall input and catchment hydrology determines the geographic range of effectiveness of the policy options.
- Relative cost-effectiveness of the policy options is very sensitive to the indicator of nitrate-N chosen (Fig. 7). Scenario B, though of small environmental benefit, is worth considering from an economic perspective.
- Groundwater storage delays the effects of land-management change, especially upstream of Sites 3 and 4, compromising the worth of the policy options.

Dynamics of River Nitrate Concentration

At the catchment outlet (Site 5) the combined impacts of various different sources give an integrated signal. Model applications indicate late autumn maxima of nitrate-N concentration (Fig. 3) reflecting the predominance of agricultural diffuse sources in rural catchments. Late autumn maxima are due to accumulation of soil mineral N, at its highest following crop harvest. Soil hydrological processes moderate and delay the delivery of diffuse

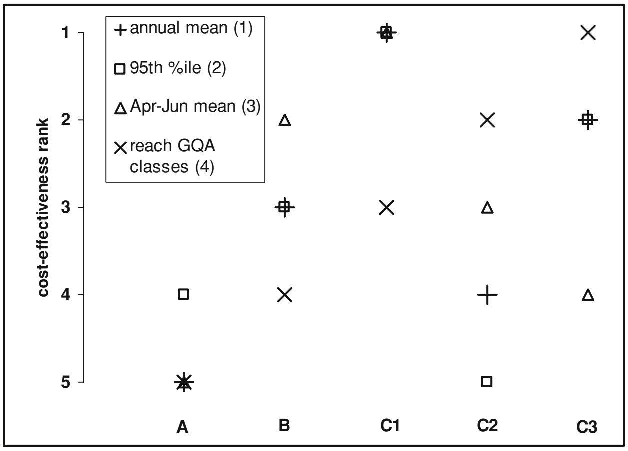

Fig. 7 Cost-effectiveness rank under different nitrate-N criteria. Criteria 1, 2, and 3 refer to site 5. Criterion 4 refers to mean improvement across all reaches in GQA nitrate class

pollution to the river. These delaying and dampening factors may be greatly enhanced in areas where groundwater contributions are high. There is spatial variability in the importance of groundwater sources across the Derwent catchment. Observations at the Howsham site (a few km upstream of Site 5) suggest a delay towards late winter in the occurrence of these maximum values. The differences between observed and simulated peak nitrate-N concentrations may suggest that these delaying factors may be of greater significance than currently represented in the model. Point source inputs are generally assumed constant throughout the year. Although of low magnitude in the Derwent, they are higher in nitrate-N concentration than the diffuse sources. This is partly because a significant part of the catchment is under low intensity agricultural grassland or non-agricultural land. In the river, point sources will be at their most significant in relative terms during the summer when flow dilution is low.

Influence of Geographic Variability

In the east (Sites 4 and 5) there is a higher groundwater contribution to stream flow than in other sub-catchments and this moderates and delays the impact of a change in land management more than elsewhere. Furthermore the slightly higher annual rainfall observed in the west may result in a more marked response to measures. This is illustrated by the differing effectiveness (as seen at Site 5) of Scenarios C2 and C3. When looking within sub-catchments themselves rather than at Site 5 however, Scenario C1 shows greatest effectiveness in the eastern part of the catchment. This is due to the greater prevalence of arable land both as a proportion of total agricultural area and total land area.

Table 5 Hypothesized annual diffuse N loads (kg ha^{-1}) in the Yorkshire Derwent if all non-agricultural land remains unchanged and all agricultural land is converted to a single land-use category (either permanent grassland or winter wheat)

Land-use	Site 1	Site 2	Site 3	Site 4	Site 5
Permanent grassland	8.2	8.0	7.2	6.9	8.0
Winter wheat	34.2	34.3	29.2	29.0	34.3
Reference	16.3	13.6	15.3	19.2	19.7

Reference loads (modeled using 2000–2003 climate/land-use) are included for comparison

The areas where streamflow source has greater surface water dominance (Sites 1 and 2) are clearly showing a higher sensitivity to land-use change than elsewhere. Riskiness across the entire catchment appears to more closely reflect the character of Sites 1 and 2. This is reflected in Scenario C3 appearing relatively more effective as an option than C2, particularly in terms of the higher concentrations of nitrate-N (e.g., 95th percentile). If the purpose of policy were to be best supported by specifically preventing the highest river nitrate-N concentrations a recommendable way forward would be to target mitigation in these more responsive areas.

The spatial variability of inherent vulnerability to diffuse pollution of rivers can be illustrated by a hypothetical normalization of agricultural land use across the catchment. Table 5 displays predicted diffuse N loads at steady state, if in turn we hypothesize that all agricultural land is assumed as, either, permanent grassland or winter wheat.

Which Policy Option and Impact of Choice of Nitrate-N Criteria

A change from arable to low intensity grassland has a dramatic effect upon water quality. Of the other measures, reductions in fertilizer inputs clearly would need to be very substantial (>50%) to yield any tangible benefits in water quality. Similarly, a reduction in stocking density by 20% is insufficient to yield appreciable benefits.

In terms of the EA GQA Nitrate classification, it is evident (from Fig. 6b, c and Table 4) that only the scenarios involving large scale conversion of arable land to low intensity grassland (C1, C2, C3) are powerful enough measures to improve the GQA Nitrate class at the catchment outlet (Site 5). Nevertheless Scenario B, being less costly relative to the others, is at least as cost effective as C2 and C3, and is worth considering if only a minor improvement is deemed necessary. Whilst C1 yields improvements in class in fewer reaches than C3 it is the only one showing improvements in all the reaches currently at class 5. This is a noteworthy factor in its effectiveness.

Regardless of the nitrate-N concentration criterion used, the C1 option leads to the most marked improvements. This suggests that reductions in nitrate-N concentration at Site 5 are most sensitive to changing the land management in the downstream parts of the catchment outside the areas targeted under C2 and C3 (Fig. 1). Alternatively, priorities could be set to achieve benefits at downstream sites (i.e., Site 5) in conjunction with tangible improvements in specific headwater areas. In the Derwent this might reflect Defra interest via CSF. Consequently the most cost-effective option (at Site 5) depends on the choice of nitrate-N criterion (Table 3). Scenario C3 is more attractive when considering the peak concentrations whereas Scenario C2 is the better option if interest is focused on late-spring concentrations.

The contrast between the hydrochemical effectiveness of measures and their cost-effectiveness highlights an interesting difference of perspectives between biological and financial assessments. However, it is important to note that neither assessment considers the vital issue of the benefit value generated by these differing measures. From an economic perspective it is a weighing of both costs and benefits, which determines the optimal policy. Economics defines humans as the sole arbiters of value and this is likely to radically change the choice of policy measure and the spatial and temporal incidence of their application. For example, research into the value of improving river quality shows that there is a strong distance decay in values across space with these being highest for populations near to rivers and progressively lower as distance to that river increases (Bateman and others 2006b). This would mean that improvements in highly populated areas will generate higher total values than those in remote locations. Such consideration could substantially alter the optimal spatial application of policies and indeed could change their ranking. Ongoing research, such as that under the EU Aquamoney programme (http://www.aquamoney.ecologic-events.de/), seeks to assess the benefits of WFD-inspired water quality improvements and the spatial incidence of those benefits.

The Moderating Impact of Groundwater Responses

There is evidence that groundwater nitrate levels in major aquifers in NE England have been rising in recent decades (Smedley and others 2004). The Jurassic Corallian limestone aquifer outcrops across much of the northern part of the Derwent catchment. Data are not available for the Derwent but boreholes in an unconfined aquifer of the comparable Corallian in Oxfordshire show median concentrations of 8.1 mg N L^{-1} (Cobbing and others 2004) with many samples exceeding the EC limit of 11.3 mg N L^{-1}. Such values exceed those observed in river water throughout the Derwent. The National Rivers Authority (1994) has reported increasing N concentration in the Yorkshire Corallian, although such increases are not reported for the Oxfordshire boreholes. Groundwater discharge via springs can be significant volumetrically and have been reported in the Derwent catchment (Carey and Chadha 1998). These predominantly occur along the Ebberston-Filey Fault with the most significant being the Brompton Springs (located in the catchment of Site 4) where flows of between 0.06 and 0.7 $m^3 s^{-1}$ have been observed. The long term Q95 at Site 4 is 0.91 $m^3 s^{-1}$. Where groundwater discharges are significant the effects of land-use change on river water quality can take a matter of decades to occur. It is likely that rising groundwater N concentrations due to land use intensification in the 1970s will in the future continue to influence many river reaches, in particular upstream of Site 4 but also in northern tributaries with significant Corallian outcrop such as Pickering Beck (reaches upstream of Site 3).

The issue of extreme temporal lags, such as those arising within groundwater catchments, is of course key to the economic assessment of policy options. Economic theory and practice progressively discounts the present value of future costs and benefits such that a scheme delivering modest water quality improvements within the short term may very well be preferred over one which provides more substantial gains far into the future. The incorporation of discounting effects within our analysis would be a trivial extension. That we have not undertaken this is a deliberate decision to weigh all improvements equally in this initial analysis.

Conclusions

There is considerable variation in estimates of economic costs for the land management scenarios tested. Furthermore, the case-study shows that when combined with water quality assessment in an analysis to determine cost-effectiveness, ranking of the options is highly sensitive to the choice of water quality (in this case nitrate-N) criterion (Fig. 7). This focuses attention on the importance of defining the nature of the links between chemical and biological parameters to determine the most effective chemical indicators of ecosystem health. In this respect, whilst nitrate-N is of concern for eutrophication, more broadly under the WFD, ecological status of waterbodies will be linked with indicators representing a wider range of chemical parameters including pH, BOD, DO, ammonium and soluble reactive phosphorus to name but a few. In spatial terms, the WFD requires all waters to be protected and an appraisal of the relative improvements in water quality under our different scenarios reveals considerable geographic variability. For example, cost-effectiveness ranking is likely to differ depending on where predictions of water quality improvement are assessed, be they at (i) the catchment outlet, (ii) the most polluted reaches, or (iii) equally across all reaches. In such assessments it is also necessary to account for the retarding effects on mitigation in areas of significant groundwater discharge. Relic effects of past land-use change must be recognized when interpreting catchment water quality model predictions; and the negative value of such delays should be quantified in cost-effectiveness assessments. Ongoing research seeks to incorporate the benefit value of water quality improvements within wider assessments of policy options.

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