Identifying priorities for pollution mitigation in Scotland using national and local scale cost:effectiveness and cost:benefit analysis – a case study for phosphorus pollution of rivers.

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Abstract
Scottish catchments are expected to provide a range of 'ecosystem goods and services' including clean drinking water, diverse habitats, recreational opportunity (eg fishing), visual beauty, and a resource for a range of industries. They face many pressures, including point and diffuse pollution. There will be direct costs in implementing measures to improve water quality. The EU Water Framework Directive (WFD) makes provision that if costs are disproportionate, water quality mitigation can be derogated, at least temporarily. Strategic analysis can identify proportionate and adequate measures for water management. In this paper, we explore the use of data from a national scale screening tool, combined with estimates of cost:efficacy of P pollution mitigation, and loss of value caused by P pollution, to assess priority catchments for action under the Water Framework Directive. This analysis shows the cost:effectiveness of treating sewage treatment work sources of P, and identifies Scottish catchments (such as Lunan Water, the Angus Esk rivers, the lower Tweed, the River Tyne and the River Eden), which have a high benefit:cost ratio for mitigation.

Keywords
Screening, Phosphorus, Water quality, Water Framework Directive, Water valuation, Fishery

Introduction
The value of catchment water can take the form of provisioning services, e.g. drinking water, water for a distillery or irrigated agriculture, or as a means of transport; regulating services, e.g. the assimilation of waste products; cultural services, e.g. a place to swim or boat; and supporting services, e.g. a home for aquatic species. Similarly, the costs of using water can be seen as both direct, e.g. the financial costs of taking water out of a lake or river (abstraction) and putting water and other materials into water bodies (discharge) on fishery value, and indirect, e.g. the environmental impacts on aquatic species from the introduction of pollutants or the loss of recreation opportunities from the diversion of water from a river.

Decisions on how to manage water quality could be based on private, social or ecological considerations depending on specific management goal(s) and the level of resourcing for implementation of improvement. In each case, classical economic theory suggests that resources should be deployed such that the marginal costs of pollution abatement equate marginal benefits of improvement at a relevant scale. For example, at a local level it might be expedient for regulatory Agencies (e.g. SEPA) to focus on considering only private costs of pollution (see point Q_1 in Figure 1). The WFD describes the costs of poor ecological status of water as made up of three components: financial costs, resource costs and environmental costs" (Article *9)* and notes that these are made up of both '*use*' and *'non-use' values* (see *Annex IV.I.48).* This description presents alternative points of where the marginal social cost=marginal abatement cost (Figure 1) depending on the extent to which such "externalities" of pollution are considered, and thus how holistically the term "Good Ecological Status (GES)" is defined in a particular water body. Taking into account wider values (underlying public perceptions, preference structures, and attitudes of importance to society) will lead to alternative social optima. For illustration these alternative social optima are set further to the left in the diagram (eg. Q_2 if only social costs of pollution are considered, Q_3 if the full ecological costs are considered). The shapes and positions of these points are often not well known, especially with respect to diffuse pollution and morphological pressures. However, it is important to recognise that ecological response to pollution control is highly non-linear, and aiming at inflexible thresholds may be inappropriate for achieving cost:effective environmental management (Statzner et al., 1997). In such cases, there is a need to establish more clearly the position of the optimum point for GES to be set, requiring a more comprehensive analysis of social benefits, through integrated valuation of water resources (including use and non-use values), as well as consideration of the ecological impact of pollution. If such an analysis leads to the conclusion that costs incurred at point Q_3 (ecological optimum) >costs incurred at point Q_2 (social optimum) there may be a case for

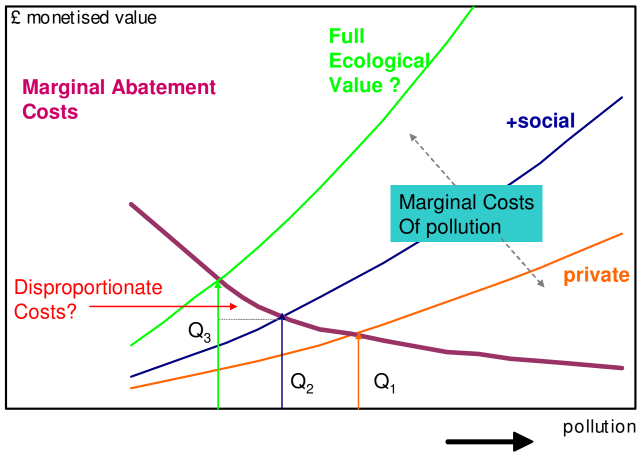


Figure 1. Framework for assessing cost disproportionality for programmes of measures.

derogation, based on disproportionate costs. The exploration of the conditions for disproportionality, and the magnitude of these disproportionate costs, requires a judicious use of a range of techniques, such as integrated value mapping, Multicriteria Analysis (MCA), Cost Benefit Analysis (CBA); Cost Effectiveness Analysis (CEA), and other qualitative and participatory techniques. Such strategic analysis could identify proportionate and adequate measures for water management. However there are significant transaction costs in exploring this approach. It is not therefore feasible for Agencies charged with delivering policy such as WFD to pursue such a full analysis, across all sectors, pressures and impacts. An expedient alternative is to set measures of Good Ecological Status (GES) for water bodies, using a chemical, morphological and ecological criteria, derived from best current understanding, which are then subject to modification as this understanding improves. The effectiveness of alternative programmes of measures to achieve this status can then assessed by Cost:Effectiveness Analysis. Only if it emerges from Stakeholder response and interaction, that significant funding issues arise from meeting these standards, would disproportionality need to be explored more deeply. This pragmatic approach to decision making is often used by the regulator and other Agencies, but it should be supported by the longer term strategic approach which assesses the values giving a basis for refining the pragmatic methods used. The framework described above forms part of a 5-year program of research funded by the Scottish Executive to explore the effective management of catchments to enhance water quality. As a first step in this program, we are undertaking a national screening exercise to assess where costs of mitigation of diffuse pollution are disproportionate and where expenditure will provide the best returns of increased value of water.

Materials and Methods

Good status definition and Screening Tool description of P loads and concentrations in streams. The Water Framework Directive requires member states to achieve good water status by 2015. UKTAG (an advisory body charged with providing technical advice on the Water Framework Directive to the regulatory agencies) define good status for P in streams as an annual mean soluble P concentration of 40 µg/l in siliceous streams (alkalinity < 100 mg /L $CaCO_3$) and 100 µg/L in calcareous streams (UKTAG, 2006), although revised standards are currently under consultation. A recent national scale exercise has led to the development of a screening tool to identify water bodies vulnerable to specific pressures (SNIFFER, 2006). In this work, estimates of the total load of sediment and phosphorus lost from agricultural and forestry areas were made by integration of land use derived estimates of pollutant concentrations in soil with model-based calculations of soil and water movement using the the Phosphorus and Sediment Yield CHaracterisation In Catchments (PSYCHIC) model (DEFRA Project PE0202). The structure of PSYCHIC is centred on a physically based calculation that operates at the plot scale using published erosion calculations, modified and added to through experimental knowledge, to produce a load of soluble phosphorus and phosphorus bound to eroded sediment. The main drivers for the model are the climate and water balance values of monthly rainfall and rain days, surface runoff and drainage. These plot scale calculations can be used with statistical physical characteristics, land use and land management information at the landscape scale (1 km^2) to provide monthly estimates of sediment and phosphorus loss that may be summed to give annual loads. The soil erosion algorithms have been validated for lowland, arable and grassland soils, but the soils data on the erodibility of humose and peaty upland in Scotland soils is lacking. Hence outputs from this model may not be reliable for the highland and upland areas of Scotland. In addition to diffuse sources of particulate P, the Screening Tool provides estimates of soluble P transport, incidental transport from manures, and point source contributions from septic tanks (0.3 kg TP/person/day) and sewage treatment works (0.44 kg TP/person/day) using principles developed in the project DEFRA PE0106 (Haygarth, 2003).

The modelling framework allows estimates to be made of total P loads from land and consequent mean stream soluble P concentrations with 80% risk of exceedance for both diffuse and point sources at 3 scales: 1 km^2, local catchments (LC – ie land area contributing directly to an identified 10 km reach of stream) and total catchments (TCA ie the total contributory area to a given point in the stream). These figures are corrected for retention by the watercourse, so the gross load from the landscape (needed to calculate reductions in P load required to achieve good status) is:

$$\text{P load (gross) (kg/ha)} = \text{P load (kg/ha)} / R_f \qquad (1)$$

Where R_f = retention factor. Retention factors were calculated based on catchment hydraulic load. From this load calculation, a "Perfect Mixer Average P concentration" in the stream is calculated:

$$\text{PMAC (mg/L)} = 100 * \text{P load (kg/ha) /HER (mm)} \qquad (2)$$

These predictions have been compared with observed data provided by SEPA to obtain a regression equation which both predicts observed soluble P concentrations in stream , and gives a measure of the uncertainty of predictions. This allows a likelihood of exceedance of a given concentration to be determined (SNIFFER, 2006):

$$\text{Ln[P]} = 0.714 * \text{lnPMAC} - 1.0478 \ (n=597, r^2 = 0.49) \qquad (3)$$

where P = observed soluble P concentrations (mg/L).

Estimation of reduction in loading required and mitigation cost/effectiveness of BMPs. These equations allow us to obtain an estimate of how much P load reduction is needed for achieving a given water quality status. An equation of the form below has been fitted to output from the Screening Tool:

[P] = a(1-exp(-k.PMAC)) (4)

For STW P load only, a=0.216 and k =-1.101. For diffuse P load only, a=0.0657 and k=-6.999. For Total P load a=0.235 and k=-1.377. To achieve an 80% likelihood of 0.04 mg/L a target [P] of 0.019 mg/L is required. To achieve an 80% likelihood of 0.10 mg/L a target [P] of 0.048 mg/L is required. For estimation of costs of reduction of P inputs from sewage treatment works, we have used the data of Hutchinson et al. (2005) who give a fixed marginal cost of £6.36/kg P for large STWs. For estimation of costs of reduction of P inputs from farming, two cost curves have been derived, one for arable area and one for improved grassland area using information from Haygarth et al. (2003). This lists a series of measures that contribute to P loading from farmland and estimates the marginal cost of mitigation (£/kg TP mitigated), which is assumed constant for a given measure, and the amount of P mitigation per ha that is mitigated. It further assumes that a fixed proportion of the landscape is affected by each measure. We have selected from these measures, those which we considered most appropriate and ranked them in order of increasing marginal cost. A curve has then been fitted to these data for both arable and improved grassland related measures:

Arable: CP = 0.87*(exp((PLR/11.0)-1) (5a)
Managed Grassland: CP = 0.22*(exp((PLR/0.74)-1) (5b)

Where CP = costs of P mitigation (£/ha of arable or managed grassland) and PLR = P loss reduction (kg/ha of arable or managed grassland). Any further residual P loss from unimproved grass, septic tanks, urban sources or forestry are not yet considered.

Estimation of change in fishery value due to reduced P inputs to rivers. Hilton et al (2006) identify two kinds of river environment (a) those with a retention time from river source of > 4 to 6 days, in which the transition from oligotrophic to eutrophic conditions will have the same character as transition in Lochs ie phytoplankton will become dominant as eutrophication proceeds. These will normally be lower reaches, or rivers with significant impoundment upstream; (b) those with a lower retention time, in which there is insufficient retention for algae in the water column to proliferate. In these, the trophic succession is: slow growing macrophyte dominance -> fast growing, light efficient macrophytes eg filamentous algal growth (Cladophora) on stream bed -> to epiphyte coverage of macrophtyes -> prolific benthic algal growth.

Taking the second case as most relevant to restoration of Scottish rivers, Hilton et al (2006) consider nutrients, stream velocity (through spatial and temporal variation and grazing), stream substrate (affecting rooting, which can often be shallower and less stable in sediment or nutrient rich environment) and shading (leading to floating spp dominance) as co-determinant in the development of eutrophic plant growth. Under summer low flows and well lit conditions (which we assume, as achieving shading along stream reaches is likely to be one of the most expensive BMPs, and it would probably also achieve P mitigation), we can assume that nutrients are the main limitation. Hilton et al., (2006) consider that the median growing season soluble [P] may well be a better indication of nuisance eutrophication than total P load. However at the low end of the scale, loads may be more important, as P supply becomes limiting. Westlake (1981) suggests nutrients will not be limiting in waters with P concentrations >30 ug/L and N concentrations > 1 mg/l. Since the prospect of getting N concentrations below 1 mg/L in agricultural areas is very low, it seems appropriate to focus on P as a potential limiting nutrient. For water resources management, and indeed managing the enrichment of streams, it is the periods of peak biomass that are most important (Biggs et al., 2000). Dodds et al. (1997) provide a relationship between maximum chlorphyll a in streams and TP and TN:

$$\text{Log (max chl a)} = 0.00652 + 1.100671\log(TP) - 0.1929\log(TP)^2 + 0.3129 \log TN \quad (r^2 = 0.370) \quad (6)$$

Biggs et al (2000) also provide a relationship between chlorophyll a and Ash Free Dry Weight (AFDM):

$$\text{Ln Chlorophyll a } (mg/m^2) = 0.338 + 1.396 \text{ X Ln AFDM } (g/m^2) \quad (r^2 = 0.790) \qquad (7)$$

Biggs et al. (2000) also provide a relationship for New Zealand streams between %Ephemeroptera/Trichopetera/ Plecoptera species in invertebrate samples (ie clean water species) and Ash Free Dry weight (see figure 29 in Biggs et al (2000)). Using these three relationships, we can come up with a relationship between clean water invertebrate status and average TP concentration in the stream. Assuming there is a linear relationship between fishery value and clean water invertebrate status, this allows us to predict the loss in value as a fishery as mean soluble P content of the stream changes. For example to mitigate from 100 to 40 µg soluble P/L would increase the relative value of the fishery from 47% to 59% of the pristine value. The River Dee is a near pristine water course in NE Scotland, and the fishery value has been estimated at £6,000,000 per year (1998 figs – needs update) over a catchment area of 210,000 ha. This gives an average value per ha of catchment of around £30 per ha. These relationships thus give a basis for estimating the benefit to fisheries of mitigating P pollution. This is certainly not the whole value of the river, but it gives a basis for an initial screening of the benefit to cost ratio across the country, to establish priority catchments.

Results.

The calculations described above have been combined with spreadsheet output from the Screening Tool (2006) to give estimated requirements for P mitigation, contributions to the costs of this mitigation, and estimated benefit:cost ratios, considering just the effects of mitigation on fishery value. Figure 2 shows the spatial distribution of the modelled likelihood of river water body phosphorus concentrations from both diffuse and point sources meeting good status standards (adjusted for catchment geology) (Screening Tool, 2006):

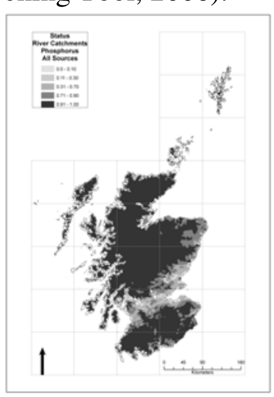


Figure 2. Spatial distribution of the likelihood of (a) river water body P concentrations achieving good status for P (considering both diffuse and point sources of P).

Comparison of Screening Tool predictions with other risk evaluations. Table 1 gives the percentage frequency failure of river water bodies to achieve good status (with 80% likelihood) with respect to P for rivers using local catchment data for the whole of Scotland. The data are categorised according to the four categories of water body risk as defined by the Pressures and Impacts report, considering all pressures (SEPA 2006). It is notable that 19% of class 2b water bodies (those considered to have little or no risk of failure) are identified as having <80% chance of good status with respect to P. This may reflect the uncertainty about P delivery to water from organic soils, which is an acknowledged weakness of the current modelling approach. What is clear, however, is that the number of water bodies with a <80% likelihood of achieving good status increases, the higher the risk category.

Table 1. Comparison of the SEPA designation of river water bodies with the Screening Tool (2006) estimated proportion of water bodies failing to achieve 80% likelihood of good status.

WFD Diffuse Pollution Risk Category	No. of River water bodies	% of Water Bodies predicted to have <80% probability of good status by the Screening Tool
1a (at risk of failure)	558	64
1b (probably at risk)	1003	42
2a (probably not at risk)	342	40
2b (not at risk)	1096	19
Total	2999	37

Cost:effectiveness and Cost:benefit maps Fig 3 shows the estimated load reductions per ha of catchment required to achieve an 80% likelihood of good status, with good status defined as either 100 or 40 μg/L annual mean soluble P in streams. Figure 4 shows the estimated cost of achieving either the 40 or 100 ug/L compliance level across Scotland. Figure 5 shows the cumulative costs for each element of mitigation, applying first mitigation of agricultural measures costing less that sewage treatment, then sewage treatment, then arable, then managed grassland.

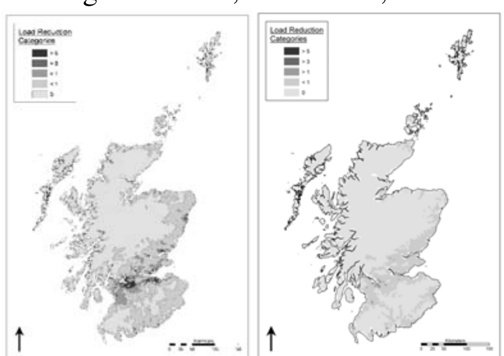
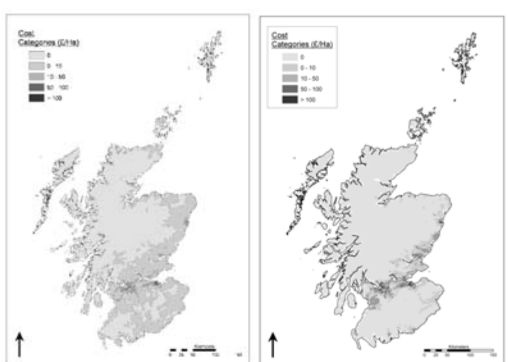


Fig 3. Estimated reduction in (a) P loads required from all sources (b) after removal of sewage treatment work derived P. In units of kg P per ha of local catchment.



(a) (b)

Figure 4.. Estimated costs per ha of mitigating the effects of P pollution from diffuse agricultural and sewage treatment works sources, to two environmental quality standards (a) 100 μg/L and (b) 40 μg/L.

Benefit/cost maps. Figure 6 shows the ratio of estimated benefit to cost of mitigation, based on loss of value as a fishery only. This clearly highlight the high benefit of mitigating P pollution in Angus (rivers North and South Esk, Lunan Water),Fife (Eden), East Lothian (Tyne), and Berwickshire (Tweed) where there are high value fisheries and erodible soils under arable farming, giving a very different assessment of priorities for acton. This is because water bodies that are very expensive to fix (eg urban influenced water bodies, with high sewage inputs) tend to have a lower benefit to cost ratio.

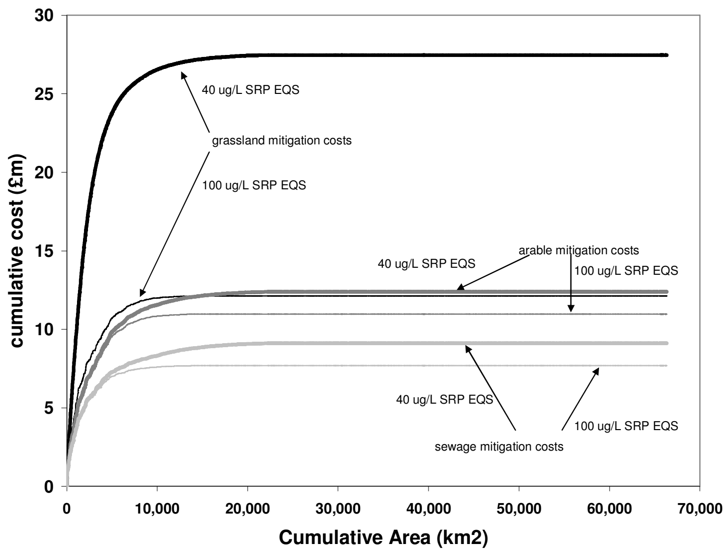


Figure 5. Estimates of mitigation costs for achieving mean SRP standard of 40 or 100 µg/L in rivers in Scotland: grassland, arable and sewage treatment costs, by cumulative area.

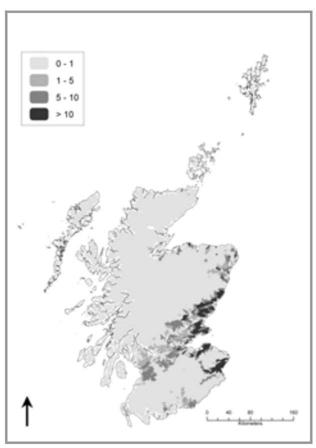


Figure 6 Estimated benefit to cost ratio classes for P mitigation to a 40 ug/L standard across Scotland.

Discussion

Hilton et al. (2006) present an argument that P loads/concentrations from septic tanks and sewage treatment works have about 4 times as much impact on filamentous algae and epiphyte growth as P loads from agriculture. This is because :

a. There is a larger proportion of soluble P in STW/septic tank effluent (ca 80%) compared to agricultural (ca 50%) sources of P pollution.
b. The proportion of STW/septic tank effluent loading occurring in the growing season (ca. 60%) is more than the proportion of agricultural loading occurring in the growing season (ca. 25%).

This suggests that we should weight the mitigation of P more strongly in favour of septic tank and sewage treatment works, if mitigation cost:benefit of P in rivers is to be optimised. Biggs et al. (2000) note that several other factors control the occurrence of filamentous algae in streams. These include riparian shading, artificial flushing events in regulated rivers, optimising benthic invertebrate habitat to increase losses through grazing activity. Hence considering only P mitigation as influencing the occurrence of filamentous algae is not correct, and more effective restoration of ecological status may not necessarily need mitigation of P loads. On could however argue, that as the factor with the lowest status determines the class, it is relevant to WFD to consider P in isolation.

Conclusions

The data and maps presented here are preliminary and provisional, with many assumptions and approximations. The map of benefit:cost ratio is therefore very imprecise. Nonetheless it is clear that certain areas of Eastern and South Eastern Scotland (such as Lunan Water, the Angus Esk rivers, the lower Tweed, the River Tyne and the River Eden), appear to present the highest priority for action, based on the criterion that a high benefit to cost ratio should occur. A new collaborative project, exploring the potential for mitigation of diffuse pollution at a catchment scale, called the Monitoring Priority Catchments Project, has recently been set up with collaboration between Macaulay Land Use Research Institute, SEPA and Scottish Agricultural College. The two catchments Lunan Water and Cessnock water/river Irvine, both show a high benefit to cost ratio associated with mitigation, so are suitable for high priority for action. The assumptions and approaches described in this paper will be developed and improved over the course of the 5 year research program (2006-2011).

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