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Managing Rural Diffuse Pollution

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Contents

Foreward

JC Gemmell, SEPA

Theme 1: Total Maximum Daily Loads (TMDLs):
A Means of Managing Diffuse Pollution?

Chaired by C Spray, SEPA

Agricultural Diffuse Pollution: Are we on the
Right Track to Successful Abatement? 3
V Novotny

Matching Water Framework Directive Controls
with Diffuse Pollution Challenges in Scotland 13
RC Morris, J MacDonald, SM Greig and C Preston

Total Maximum Daily Loads: the USEPA Approach to
Managing Faecal Indicator Fluxes to Receiving Waters:
Lessons for UK Environmental Regulation? 23
D Kay, C Stapleton, M Wyer, A McDonald and J Crowther

Decreasing the Nitrogen Soil Surface Surplus in the
Danube River Basin by Applying Agricultural Measures:
A Comparison of Cost Effectiveness Ratios 34
L Fröschl, R Pierrard and W Schönbäck

High Resolution Phosphorus Transfers in Rural Catchments:
The Hidden Importance of Rural Point Sources 41
P Jordan, J Arnscheidt, S McCormick and C Ward

An Initial Assessment of the Suitability of Total Maximum
Daily Loads (TMDLs) as a Means of Managing Diffuse
Pollution Under the Water Framework Directive 45
RJ Cooper, RC Ferrier, RD Harmel, SJ Langan, AJA Vinten and MI Stutter
Theme 2: Cost-effectiveness of Best Management Practices (BMPs)?

Chaired R Heath, Pulles Howard & de Lange (Inc), South Africa

A Statewide Approach to Identifying, Quantifying and Mitigating Diffuse Pollution-Related Problems 53

BM Evans

The Use of Ponds to Reduce Pollution from Potentially Contaminated Steading Run-off 62

KV Heal, AJA Vinten, F Gouriveau, J Zhang, M Windsor, B D’Arcy, A Frost, L Gairs and SJ Langan

Cost-effective Programmes of Measures: Theory Versus Reality 71

S Blacklocke, A Hooper, M Rosenberg and R Earle

Assessing the Cost-effectiveness of Integrated Measures to Decrease Loss of Nitrate, Phosphorus and Faecal Indicator Organisms 77

M Shepherd, S Anthony, P Haygarth, D Harris, P Newell-Price, S Cuttle, B Chambers and D Chadwick

Economic Implications of Minimising Diffuse Nitrogen Pollution from Livestock Manures 84

BJ Chambers, JR Williams, E Sagoo, KA Smith and DR Chadwick

A Measure-centric Approach to Diffuse Pollution Modelling and Cost-curve Analysis of Mitigation Measures 93

DR Chadwick, BJ Chambers, S Anthony, S Grainger, P Haygarth, D Harris and K Smith

Theme 3: How Should BMPs be Targeted and Prioritised Within a Catchment?

Chaired by G Lawrie, NFUS

Assessing the Significance of Diffuse Pollution Risks in Order to Target and Prioritise Best Management Practices 100

AH Sinclair, A Frost, A Vinten, P Chapman and J MacDonald

Retention of Pollutants by the Sink Structures in Catchments - Studies to Reduce Diffuse Pollution in China’s Rural Areas 111

C Yin, X Wang and B Shan
<table>
<thead>
<tr>
<th>Title</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>Catchment Characterisation and Targeting of Best Management Practices using PSYCHIC</td>
<td>121</td>
</tr>
<tr>
<td>PJA Withers, El Lord, J Strömqvist and PS Davison</td>
<td></td>
</tr>
<tr>
<td>A Risk Assessment and Mitigation Strategy to Minimise Livestock Pollution to Surface Waters</td>
<td>128</td>
</tr>
<tr>
<td>DW Merrilees, JW Dickson, WA Jeffrey and D Kay</td>
<td></td>
</tr>
<tr>
<td>Environment Sensitive Farming - Practical Advice for Land Managers</td>
<td>137</td>
</tr>
<tr>
<td>AD Carter, S Groves, R Bailey and J MacLeod</td>
<td></td>
</tr>
<tr>
<td>Managing Diffuse Pollution from a Forestry Perspective</td>
<td>144</td>
</tr>
<tr>
<td>HM McKay and TR Nisbet</td>
<td></td>
</tr>
<tr>
<td>The Use of Fish to Determine Impacts of Diffuse Pollution on Rivers and Human Health</td>
<td>153</td>
</tr>
<tr>
<td>RGM Heath, HH du Preez and B Genthe</td>
<td></td>
</tr>
<tr>
<td>Theme 4: How Can Land Users be Persuaded to Adopt BMPs?</td>
<td></td>
</tr>
<tr>
<td>Chaired by Prof WAC McKelvey, SAC</td>
<td></td>
</tr>
<tr>
<td>Delivering Environmental Benefits Through Land Management Contracts</td>
<td>163</td>
</tr>
<tr>
<td>I Clayden</td>
<td></td>
</tr>
<tr>
<td>Carrots, Sticks, Sermons or Hugs?</td>
<td></td>
</tr>
<tr>
<td>Designing Co-ordinated Policy Measures for the</td>
<td></td>
</tr>
<tr>
<td>Uptake of Environmental Management Options</td>
<td>165</td>
</tr>
<tr>
<td>B Davies</td>
<td></td>
</tr>
<tr>
<td>Agriculture Policy Reform: Opportunities to Reduce Diffuse Pollution</td>
<td>173</td>
</tr>
<tr>
<td>C Davies, M Gloyer and A Johnstonova</td>
<td></td>
</tr>
<tr>
<td>Farmer Uptake of Nutrient Management Best Practice</td>
<td>179</td>
</tr>
<tr>
<td>G Goodlass</td>
<td></td>
</tr>
<tr>
<td>A Case Study: Adoption of Best Management Practice in Brittany (France) Using Economic Instruments and Regulation</td>
<td>186</td>
</tr>
<tr>
<td>P Newell Price and M-L Daumer</td>
<td></td>
</tr>
<tr>
<td>Regulatory Options for the Management of Rural Diffuse Pollution</td>
<td>192</td>
</tr>
<tr>
<td>BJ D’Arcy, K Schmulian and R Wade</td>
<td></td>
</tr>
</tbody>
</table>
Poster Presentations

The Farm Soils Plan
  R Audsley

Phosphorus Storage in Fine Channel Bed Sediments
  DJ Ballantine, DE Walling, AL Collins and GJL Leeks

Red, Amber and Green for the PEPFAA Code
  C Christian, R Audsley and AH Sinclair

Estimating Diffuse Phosphorus Loads to Lakes:
  Implications for the Calculation of Total Maximum Daily Loads
  LH Defew

Sediment Loads and Sources in the Bush Catchment:
  A Move Towards Informed Management Strategies
  D Evans and C Gibson

Lound Catchment Project: Working with Stakeholders
  to Deliver Benefits for Water and Wildlife
  HG Gray and C Lorenc

Minimising the Pressures and Impacts on Freshwater
  from Agriculture in the Upper Ballinderry River SAC
  M Horton and A McGarel

A Framework for Valuing the Health Benefits of
  Improved Bathing Water Quality in the River Irvine Catchment
  EK Johnson, D Moran and AJA Vinten

Determination of the Veterinary Antibiotics Pollution
  in Soil from Agricultural Sources
  M Kaštelan-Macan, S Babic, D Ašperger, D Mutavdžić and AJM Horvat

Opportunities and Constraints for Using Best Management Practices:
  Some Lessons from the Tarland Catchment Initiative
  S Langan

Can We Improve Prediction of P Concentration in
  Lunan Lochs by Changing the Plus Model?
  I Papadopoulou, A Vinten and J DeGroote

Tackling Diffuse Nitrate Pollution:
  Swapping Eutrophication for Climate Change?
  D Reay

Catchment Sensitive Farming on the Hampshire Avon
  MC Robson, R Curtis, J Swain and C Routh
Ammonia Volatilisation from Cattle Slurry Applied to Grassland: Effects of Application Technique and Rate 240
   E Sagoo, JR Williams, BJ Chambers, J Lapworth and TH Misselbrook

Field Testing of Mitigation Options for Phosphorus and Sediment (MOPS) 244
   CJ Stevens and JN Quinton

Nitrate Contamination of Groundwater from Agriculture in Canterbury, New Zealand: Measurement and Management of a Hidden Problem 248
   KJW Taylor

Rapid Incorporation of Solid Manure as a Best Management Practice? 249
   RE Thorman, J Webb and S Yamulki

Methods to Improve Pollution Control Potential of Woodchip Corrals 253
   AJA Vinten, D Merrilees, G Sym, J Parker and C Crawford

Farmers’ Responses to Water Quality Problems in the Leet Catchment 259
   PE Widdison, TP Burt, DNM Donoghue and R Hudson

Nitrogen Losses After Cattle Slurry Applications to a Drained Clay Soil 267
   JR Williams, E Sagoo, BJ Chambers, RB Cross, J Short and RA Hodkinson

Soil and Crop Management Effects on Sediment and Phosphorus Concentrations in Run-off from Agricultural Land 272
   PJA Withers, RA Hodgkinson and A Bates
Foreword

It is my pleasure as Chief Executive of the Scottish Environment Protection Agency (SEPA) to extend a warm welcome to this conference on behalf of the Scottish Agricultural College (SAC) and SEPA.

As I am sure you are aware, this conference has been actively promoted by the International Water Association (IWA). We are very grateful to the IWA for this support and recognition, as we continue in our search for answers to the many and varied questions about the pressures on the water environment nationally and internationally, now and in the future.

This is the 6th SAC-SEPA conference in Edinburgh and comes when SEPA itself is celebrating its 10th year of operation. The previous conference was in SAC’s Centenary Celebration Year in 2004. SEPA is a considerably younger organisation, so 2006 is a significant milestone in our history. It is a year in which we will be publishing a major State of Scotland’s Environment report and engaging with people in all walks of life to address the single biggest environmental threat - climate change. This conference brings together a very important audience for this message, one which can use its knowledge and influence to lead positive action. Climate change will increasingly govern the conditions under which land use is sustained and will have to be a major part of future thinking and future conferences. In Scotland, as in other parts of the UK, we talk a lot about the weather. This is perhaps no bad thing and will equip us well for the future challenges!

It has been eleven years since the first SAC-SEPA conference. The conference proceedings published to date have included over 220 papers from around 560 authors. This enormous output has come from the research community, consultants, regulators, industry representatives, non-governmental organisations, local authorities and government representatives from the UK and abroad. The shared knowledge and experience generated by all of these contributions has helped us make huge strides forward in dealing with diffuse pollution. Understanding the issues as well as the solutions is a pre-requisite for effective, well targeted management action.

This conference looks at the management of rural sources of diffuse pollution. Here in Scotland there is already a great deal of work underway, principally under the banner of Water Framework Directive, and we will be hearing about some of these actions and initiatives.

All of this is happening at a time when a number of other EU Directives are being implemented, on the control of agricultural waste, the pollution of water by agricultural nitrates, bathing water quality, groundwater protection, sewage sludge use and the operation of large scale pig and poultry installations. On top of this, Common Agricultural Policy (CAP) reform is just over a year old, a revised EU Rural Development Regulation has just been agreed and the key regulatory aspects of the Water Framework Directive are being implemented.

The challenge – the imperative - for us all is to achieve social and economic progress and an excellent environment. The costs and benefits of protecting the environment – and the costs and implications of not doing so - need to be well understood. Land
managers need support in understanding their part in tackling diffuse pollution, and the role of one-to-one advice is critical here.

New regulation will be required for the control of diffuse pollution - of that there can be no doubt. The Water Framework Directive requires EU Member States to manage diffuse sources of pollution. What is clear, however, is that legislation alone will not bring about the necessary change in practices, especially for sectors that are responsible for managing so much of Scotland's land area. Action will be required sooner by farmers in some areas and it will be important that the reasons why are well communicated and understood. The River Basin Management Planning process will play a key part in this.

In a post-CAP reform era, the incentive for farmers and growers to produce crops or livestock depends on the market place as opposed to the receipt of farm support. This is a crucial change and must now be accompanied by the re-direction of farm support payments to the protection and conservation of natural resources. In Scotland, we expect this to happen via Land Management Contracts. Delivering financial support to address the priority issues of diffuse pollution, biodiversity and landscape is paramount.

There is no doubt that modern farming and forestry practices have a major impact on air, soil and water quality. We must balance the positive and negative aspects of food and timber production and seek to maximise the 'win wins'. Consumer interest in how and where food is produced has never been greater and significant opportunities exist for farmers who follow the principles of Integrated Farm Management. The same applies to organic farming where consumer demand is outstripping supply and where particular attention is given to soil use and management as well as chemical inputs. Farmers are becoming more aware of these challenges and many are already successfully integrating environmental issues into their day-to-day decision making. The energy and drive to sustain and develop a business for the future is critical. Adapting to new demands is essential for business survival.

Clearly, if we want a better water environment, and we certainly do, we need to support land managers with their decisions. The evidence must be there to support action and we will need to demonstrate that the actions being taken are working. At this conference we will hear of first hand experience in using particular tools, techniques and measures that target and manage rural diffuse pollution effectively.

Making conferences like this happen involves an enormous amount of work. I would therefore like to say a big thank you to all of the SAC and SEPA staff who played a part in organising this event. I would also like to thank all of the speakers and poster presenters. We are very pleased to have speakers from all around the UK and the world. This is both a very exciting prospect and a fantastic achievement on behalf of all concerned.

My expectations of this conference are high and the partnerships we will forge through working together will, I hope, make our environment even better tomorrow than it is today.

**Dr J Campbell Gemmell**
*Chief Executive, Scottish Environment Protection Agency*
AGRICULTURAL DIFFUSE POLLUTION: ARE WE ON THE RIGHT TRACK TO SUCCESSFUL ABATEMENT?

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SUMMARY

In the US, the US Environmental Protection Agency and States are collaborating in a co-ordinated integrated effort to address agricultural point source (e.g. feedlots) and diffuse (crop production, pasture) pollution through the use of Total Maximum Daily Load (TMDL) programmes. In European Union Member States similar efforts are spearheaded by the Water Framework Directive (WFD) and Nitrate Directive. The majority of diffuse pollution abatement activities in watersheds impacted by anthropogenic land use focus on a single stressor, often related to a distant endpoint. However, impairment of the integrity of receiving water bodies may also be a result of other anthropogenic impacts such as stream modification, poor or no riparian zones, and water withdrawals for irrigation. Focusing on single stressors and pollutant inputs may therefore not lead to a successful restoration of integrity (good ecological status) of agricultural streams. Multi-stress modelling, identification of integrity clusters and Cluster Dominating Parameters concepts may help to increase the effectiveness of abatement strategies.

INTRODUCTION

It is well documented fact that water quality in watersheds where all or a significant part of land is devoted to agriculture and silviculture has deteriorated in the past 50 years. However, the connection between agricultural practices and deterioration of once clean streams was not made until the late 20th century. In the US in the late 1970s, farmers and agro-business resented any notion of connection between pollution and agriculture (Novotny and Chester, 1981) Farmers were considered stewards of the land and soil was considered as having an infinite capacity to accept all fertilizers (manure, wastewater applications and chemicals) and pesticides applied to it to grow crops or dispose pollutants (e.g. wastewater sludge or septage). However, agriculture was changing worldwide from small family farms to agricultural conglomerates. The change from family, mostly subsistence farming to agro-business also created a revolution in productivity.

The Specialty Workshop of the International Water Association (IWA) (Bendoricchio, 1999), conferences by SAC/SEPA in Scotland and many other meetings and scientific work, have established the link between agriculture and pollution of receiving water bodies. The problems were first noticed in lakes and this focused the scientific community towards studying and identifying the causes of eutrophication. However, later it was found that the problems occurred at all geographical scales, from small streams and lakes, through medium-scale, to large-scale rivers associated with large water bodies such as the Gulf of Mexico in the US and Black, Baltic and North Seas in Europe.
During the past 20 years, it became clear that problems other than eutrophication were also important and that inland lakes were not the only water bodies affected. A National Science and Technology Council (2000) publication dealing with the effects of nutrients on coastal waters noted that unusually high numbers of deaths of sea lions and manatees, unusual patterns of coral reef destruction, widespread fish kills, outbreaks of certain shellfish poisoning, disappearance of sea grasses, and occurrence of so called ‘dead zones’ could all be linked in one way or another to subtle changes in the absolute abundance of certain organisms near the very base of the food web. The abundance of these organisms is related, directly or indirectly, to nutrients flowing into the system from upstream watersheds. The water quality problems of these water bodies have one unifying cause: they have been linked, to some degree, to agricultural practices.

In the US and elsewhere, nitrate levels in high river flows in spring, after the application of fertilizers, were closely correlated to the percent of the watershed under agriculture (Figure 1). Because the concentration scale is logarithmic, the chart indicates that nitrate concentrations begin to increase exponentially when the agricultural land use percentage exceeds 40%.

![Figure 1: Nitrate concentration vs. % agricultural land use in the Willamette River Basin, Oregon (from US Geological Survey, 1999). Forest was the other major land use in this watershed](image)

Currently, the 1991 Nitrate Directive of the European Community requires all Member States to designate Nitrate Vulnerable Zones for protection of water resources from agricultural pollution. The methodology for vulnerability assessment identifies surface waters that are excessively polluted or could become polluted by nitrates. The strategy is then a reduction of excessive current loads and prevention of future increases that could impair the water body. The Nitrate Directive applies mostly to agricultural point (animal husbandry) and non-point sources.

In the US, the US Environmental Protection Agency (USEPA) uses nutrient criteria to assess vulnerability and target action. The Total Maximum Daily Load (TMDL)
programme, embedded in Section 303(d) of the US Clean Water Act, is the flagship programme in the US for implementing water quality standard in watersheds in which mandatory point source discharge controls are not sufficient to meet the water quality goals specified in the Act. However, because the programme controls mostly identifiable point source discharges only, implementing agricultural land-based measures is very difficult. Consequently, there are only a few TMDL studies that have been successful for controlling agricultural pollution.

ROOT CAUSES OF AGRICULTURAL POLLUTION AND WATER BODY DETERIORATION

Green Revolution

To feed the ever increasing human and livestock population, agricultural production and productivity has had to increase. This has been accomplished first by expanding the amount of cultivated land by converting pristine lands – prairies, forests, deserts and wetlands – to agricultural lands by deforestation, drainage, irrigation and cultivation of idle grasslands. The land use conversion process itself creates pollution (Novotny, 2003). However, traditional farming was incapable of satisfying the rapidly increasing need for food to feed rapidly growing urban and rural populations. Yet, today, there may be more food per capita than 50 years ago and some countries that were food deficient and had severe famines not long ago (e.g. China) have become food exporters. This dramatically increased agricultural production and productivity has been a result of intensification of agriculture by changing farming practices and by the introduction of agricultural chemicals – fertilizers, herbicides and insecticides. This was the result from a planned international effort called the ‘Green Revolution’ funded by the Rockefeller and Ford Foundations and governments of many developing and developed countries. The yields were dramatically increased by: (a) new crop cultivation methods; (b) developing new crop varieties; (c) irrigation; (d) dramatically increased chemical fertilizer and pesticide applications; and (e) mechanization.

Although the invention of chemical fertilizers and man-made toxins (pesticides) capable of eradicating unwanted weeds and insects occurred more than 80 years ago, only after the 1950s did chemical industries switch to promoting and manufacturing these chemicals for the use in agriculture and growing suburban lawns (Ongley, 1996; Novotny, 2003). In the past 40 years, the Green Revolution has increased food production per hectare by more than 1000%, famine has decreased by 20% and caloric consumption per capita, in spite of the accelerated population growth, has increased 25%. The ecological impact of the Green Revolution has been a severe degradation of water quality and eutrophication (Rosset et al., 2000).

Best Management Practices (BMPs) to the Rescue

After the impact of agricultural pollution was recognized and quantified between 1970 and 1980, a plethora of structural and non-structural Best Management Practices (BMPs) have since been developed. These are described in numerous manuals and books. BMPs are generally classified into four categories: (1) source controls (erosion control, soil conservation, targeted fertilizer applications); (2) hydrologic modifications of source area (less used in agriculture); (3) reduction of delivery of pollutants between the sources and receiving water body (e.g. riparian buffers,
infiltration); and (4) capture, storage and treatment (e.g. ponds, wetlands).

Traditionally, BMPs control the load of pollutants (sediment, nutrients, organic carbon) from the agricultural sources to the receiving water bodies. Consequently, the objective of many agricultural diffuse pollution abatement efforts is to minimize the loads of nitrogen and the endpoint is the reduction of nitrate concentration in the large receiving water bodies that have a continental economic and ecological significance (e.g. Baltic Sea, Gulf of Mexico, Chesapeake Bay). The focus of control (endpoint) in the watershed of the Gulf of Mexico is Total Nitrogen causing hypoxia in the Gulf (Rabalais et al., 1999). Other common endpoints are phosphorus concentrations of lakes or chlorophyll, sediment, bacterial pollution caused by livestock, or dissolved oxygen depletion caused by manure applications or algal respiration.

Defining the objectives of management and endpoints is obviously important, the concern with large subcontinental or regional water bodies is justified, and nitrogen and phosphorus may be the most obvious villains under such circumstances. The problem is that the sources of the pollution loads are often quite far from the points of impact. For example, the main sources of nitrogen causing hypoxia in the Gulf of Mexico are farm operations in mid-west US states that are more than one thousand kilometers away (Burkart and James, 1999). Farmers in these states are therefore detached from the point of impact and have no economic and often no regulatory incentives to reduce fertilizer applications or change their farming practices. The same logistical dilemma is typical for many other large water bodies.

The EU Nitrate Directive attempted to resolve the regulatory problem by putting limits on nitrogen loads and, indirectly, on the use of fertilizers. However, focusing on reducing inputs of nitrogen only may not improve the integrity of those streams draining agricultural watersheds that have been heavily modified by drainage, straightening and loss of riparian vegetation. The anthropogenic modifications of streams to serve agriculture, and overloading these water bodies with sediments and nutrients from surrounding fields have also caused these streams to lose their buffering (waste assimilative) capacity and have increased their vulnerability.

**Objectives (Goals) of Management**

The cases of large regional water quality deterioration such as those mentioned above could be characterized as single stressor problems. Single stressor management has been the core of most management programs, i.e. identify the problem (endpoint) and the most dominant cause of impairment (single stressor) and then devise the abatement to reduce or eliminate the dominant stress, hoping that other stressors, if any, will also be reduced. However, the Clean Water Act (CWA) and the EU Water Framework Directive (WFD) also put achievement and maintaining the integrity of the water body (CWA) or the best ecological status (WFD) as the main goal and focus of abatement. Both have similar meanings.

Integrity has been defined as the capacity of a receiving water body to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organization comparable to that of the natural habitat of the region (Karr et al., 1986; Barbour et al., 1999). Water body integrity is assessed using multidimensional metrics of indicator organisms (Barbour et al., 1999).
The fish indices of biotic integrity (IBIs) has 12 metrics and the IBI is a summation of the metrics. IBIs have also been developed for benthic macroinvertebrates.

“Integrity” has three dimensions: physical/habitat, chemical and biological (US Environmental Protection Agency, 1994). ‘Pollution’ is then understood according to its definition in the CWA, i.e. any human action that downgrades the water body integrity. Watersheds and their water bodies impacted by anthropogenic effects are subjected to external and internal stresses. These stresses can lead to impairment and a loss of sustainability.

Folke et al. (2002) investigated the concept of resilience which was defined as the state of a system that tends to maintain its integrity when subject to disturbance (Holling, 1973). Figure 1 shows the simplest effect of the impact of agriculture on the concentrations of nitrogen in the receiving waters. N concentrations in the watersheds in Oregon exponentially increased (resilient state) after the agricultural land use exceeded 40 % threshold.

Vulnerability is the opposite of ‘resilience’ (Folke et al., 2002). Water body and watershed vulnerability refers to the expectation that near future water quality and ecological conditions may reach an impaired status. Vulnerability is synonymous to the ‘threatened’ status of a water body defined by the TMDL regulations. The ‘triggers’ of change to an impaired status are Batjes (2000): land use changes, global climatic changes, acid rain, excessive water use and withdrawals, stream anthropogenic modifications, pollutant discharges and excessive unsustainable application of chemicals.

Watershed Retention Capacity (WRC) defines watershed resilience. WRC (resilience) for pollutants is related to the Capacity Controlling Parameters (CCPs) that include (Salomons and Stol, 1995; Batjes, 2000; Novotny, 2003):

- Organic matter content of soil and vegetation
- Acid Neutralization Capacity (ANC)
- Sulfide content of soils and sediments
- Soil structure and texture
- Microbial activity
- Cation Exchange Capacity
- Redox Potential (Eh)
- Soil adsorption capacity
- Salinity
- Shallow bedrock geology

Most significant loads of nitrogen from Midwest US, in addition to fertilizers, are caused by drainage of wetlands (Burkart and James, 1999).

Recovery. Stålnacke et al. (2003) analysed the impact of dramatic reductions of nutrient fertilizer inputs in Eastern Europe following the political changes in 1989. For example, in Estonia and Latvia, sales of mineral fertilizers dropped by a factor of 15 from 1987 to 1996, and the number of livestock decreased by a factor of four. Yet, in the short run of about 5 to 10 years after these changes, decreases of nutrient inputs into east European receiving waters and resulting concentrations were far less than expected. In some cases, improvements in water quality were statistically insignificant.
DOMINATING PARAMETERS OF IMPAIRMENT AND RECOVERY

Streams draining agricultural basins are impaired by many stressors among which channel alteration, loss of riparian buffers, siltation, reduction of watershed buffering capacity, and nutrient loads are most important. Identifying the dominant stress may not be simple and the abatement activities may be directed to a stress that may not bring about the desired improvement of integrity to the best ecological status. New models quantitatively identifying the impact of multiple stressors and threshold of impairment and recovery are needed.

The current STAR (Science To Achieve Results) project conducted at Northeastern University (Boston, Massachusetts) is developing non-linear models of watershed biotic integrity and loading by extracting knowledge from large data containing indices of biotic integrity and their metrics and stresses from several states. The basic model concept is a hierarchical, four-layer progression of risks from landscape and hydrologic/hydraulic stresses and diffuse and point source pollutant inputs to instream impacts causing risks to aquatic biota. Four risks can be considered: habitat degradation, water pollution by pollutants, sediment contamination and fragmentation (Novotny et al., 2005). Artificial Neural Net (ANN) non-linear layered models are highly compatible with the hierarchical risk propagation modelling concept.

The ANN models identified clusters (states) of the fish Index of Biotic Integrity (IBI) (Virani et al. 2005) and its metrics. Data sets containing more than 50 parameters measured several times at about 2000 sites in Maryland and Ohio were analysed. The Self Organizing Maps (SOM) of the fish IBI and its metrics were developed by unsupervised ANN learning. SOM (Kohonen, 2001) is a data clustering and visualization technique which converts complex, non-linear relationships between high-dimensional data vectors into simple geometric relationships on an easy to visualize low-dimensional display (usually a 2-dimensional space). In SOM analysis, each neuron unit has a different weighted connection to each and every one of the SOM output layer. These weights model the influence of an input variable (fish IBI metrics) to the sites patterned in an SOM neuron. SOM is an effective data clustering tool with its output emphasizing the salient features of the data and allowing the subsequent automatic formation of clusters of similar data items. SOMs of environmental variables (habitat, chemistry, and macro-invertebrates) were then overlaid over the SOMs of the metrics and overall fish IBI to identify the parameters that showed a similar SOM pattern (Virani et al., 2005).

The three clusters of fish IBI metrics recognized in Ohio reflect the quality of the fish community. The overall fish IBIs in the clusters indicated that sites in Cluster I had ‘superior’ fish composition, sites in Cluster II were intermediate, and sites in Cluster III were inferior. However, overall IBIs varied within each cluster and there was a minor overlap because the overall IBI is a summation of scoring of metrics. Hence the same IBI can be achieved by many variants of metric scores. Because each neuron of SOM contains several physical monitoring sites, it was possible to locate the clusters regionally and put them on a map (Figure 2). It can be seen that most Cluster III sites were located in the highly agricultural northwest corner of the state (dominated by monocultural corn growing) and around the Cleveland – Akron industrial area. The best Cluster I sites were in the hilly more pristine eastern and southern parts of the state.
Non-linear Canonical Correspondence Analysis (CCA) was subsequently used to link the stressors, not used in the unsupervised ANN learning, to the SOM and quantitatively rank the stressors as to their impact on IBIs and their metrics (Figure 3) (Virani et al., 2005). CCA is a powerful method for the multivariate exploration of large-scale data (Legendre and Legendre, 1998). This is a weighted average ordination technique providing simultaneous ordering of sites and species, rapid and simple computation and very good performance when species have non-linear and unimodal relationships to environmental gradients (Palmer, 1993). CCA can also be used to identify the Cluster Dominating Parameters (CDP). From Figure 3 it can be seen that sites in Cluster I (with superior fish IBIs) are most sensitive to habitat quality parameters and forested land (including riparian land) while sites in Cluster III (with inferior IBIs) are most impacted by pollutants and intensive land uses by humans.

Figure 2: Distribution of Clusters of IBI in Ohio. Cluster III (poor) sites are located in the northwest (agricultural), north (Cleveland – Akron), central (Columbus) and southeast (Dayton – Cincinnati) (Virani et al., 2005)
Using the results of the unsupervised learning of SOMs and CCA, the top 25 CDP parameters affecting fish IBIs for Ohio were identified. Nitrogen and phosphorus concentrations were less important. Indeed P concentrations were not identified in the top 25 parameters impacting biotic integrity. Using this identification of CDPs, an accurate input/output ANN model was then developed by supervised ANN learning wherein the overall fish IBI was the dependent output and the 12 most important CDPs were inputs (Brooks and Novotny, 2005).

**CONCLUSIONS**

A great majority of abatement plans for diffuse pollution in watersheds impacted by anthropogenic land use and activities focus on a single stressor, often related to a distant endpoint. These efforts and regulative approaches are justified because they involve large water bodies of great economic and ecological importance. However, many local diffuse pollution problems are caused by multiple stresses and a single stressor approach focusing on one nutrient may not necessarily help improve the integrity (ecological) status of the immediate small or medium size water body. Using this analysis of Ohio sites and a similar analysis for Maryland (Virani et al., 2005), it was found that sites with superior Cluster I fish IBIs were impacted mostly by habitat parameters. Using the results of the unsupervised learning of SOMs and CCA, the top 25 CDP parameters affecting fish IBIs for Ohio were identified. Nitrogen and phosphorus concentrations were less important. Indeed P concentrations were not
identified in the top 25 parameters impacting biotic integrity. Using this identification of CDPs, an accurate input/output ANN model was then developed by supervised ANN learning wherein the overall fish IBI was the dependent output and the 12 most important CDPs were inputs (Brooks and Novotny, 2005) within which embeddedness was the most important CDP. Embeddedness is a consequence of siltation, gradient (affecting watercourse velocity), channelization and impounding (affecting watercourse depth). Other impacts that are correlated with it are shown on Figure 3. Increasing embeddedness drives the site towards Cluster III. Cluster III is dominated by more traditional pollutants and poor substrate.

Identification of Clusters and their dominating stressors (CDPs) is important in the choice of abatement strategies. Focusing on a single ‘popular’ stressor and pollutant input may not lead to a successful restoration of integrity (good ecological status) of agricultural streams. Multi-stress modeling and identification of CDPs may therefore serve to help put the abatement activities on the right track.

ACKNOWLEDGMENTS

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MATCHING WATER FRAMEWORK DIRECTIVE CONTROLS WITH DIFFUSE POLLUTION CHALLENGES IN SCOTLAND

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SUMMARY

The Water Framework Directive (WFD) requires regulatory controls be established to prevent or control the input of pollutants to surface or groundwater. For the first time this provides potential for a regulatory approach to mitigating diffuse pollution in Scotland, currently the most significant chemical pressure on Scotland’s water quality. A regulatory approach on its own will not achieve the required water quality targets; rather, a combination of measures is required including education, economic incentives and regulation. Key to the proposed approach and integral to the WFD is the concept of control measures being proportionate to risk. This paper focuses on how best to match emerging WFD controls with diffuse pollution pressures on the water environment in Scotland. To do so requires an understanding of how and where diffuse pollution occurs as well as an understanding of the pollutant loading, associated short- and long-term impacts and the effectiveness of remedial measures.

INTRODUCTION

The nature of diffuse pollution has important implications for its control. Diffuse pollution typically arises from a multitude of small sources, such as poaching of river banks by livestock or run-off from forestry tracks, that are individually insignificant in terms of environmental impact but at a catchment scale can have a detrimental effect. The transport of diffuse pollutants is also important. Pollutants often travel overland, mainly driven by rainfall events before reaching surface waters. For groundwater, pollutants are transported through the soil column (Scottish Executive, 2005).

For the control of diffuse pollution these characteristics mean practical measures need to focus on land and run-off management practices rather than traditional ‘end-of-pipe’ regulation. It is the activities which cause diffuse pollution rather than the individual pollutant losses which must be addressed in order to see an improvement in water quality. The impacts of diffuse pollution are related to certain geographic, climatic and geological conditions and may differ greatly from place to place, resulting in the need for a targeted approach to pollution mitigation. The nature of diffuse pollution also means that environmental impacts and the success of control measures to address those impacts are measured on a catchment scale. This provides opportunities for community ownership of the problems and their solutions. A regulatory approach on its own will not achieve the required water quality targets and a combination of measures is required including education, economic incentives and regulation.
WATER FRAMEWORK DIRECTIVE IMPLEMENTATION IN SCOTLAND

The WFD is being progressively implemented in Scotland. In an annual report to the Scottish Parliament (Scottish Executive, 2003), the Scottish Executive set out the progress on implementing the Water Environment and Water Service (Scotland) Act 2003 (WEWS Act). At this time, a National Stakeholder Forum was established to inform implementation, a single River Basin District for Scotland was designated with a separate cross-border area for the Solway-Tweed and the process for developing regulatory regimes was described.

Point source, engineering and building works, water abstraction and impoundment controls are now all in force under the Water Environment (Controlled Activities) (Scotland) Regulations 2005 (CAR). These came into force on 1 April 2006. In December 2005, the Scottish Executive published a consultation on proposals for a strategy to address diffuse water pollution from rural land use. A considerable amount of detailed work will be required in 2006-2007 to develop the shape of a diffuse pollution control regime. A package of measures is envisaged, both regulatory and supportive, using voluntary programmes and farm support payments whilst at the same time proposing that ‘national’ and ‘targeted’ GBRs be developed. SEPA strongly supports the phasing in of ‘national’ GBRs in 2007 and of ‘targeted’ GBRs in 2008. It will be important to align the conditions for the receipt of farm support to WFD objectives and not to lose, or dilute, any of the existing requirements under ‘Good Farming Practice’ for payments made under Rural Development Regulation (RDR) Schemes. D’Arcy et al. (2006) outlines how this legislation will work in practice and what regulatory options there are for the management of diffuse pollution, in particular.

In 2004, SEPA carried out the most detailed characterisation assessment of pressures and impacts on the water environment ever undertaken for both the Scotland and Solway-Tweed River Basin Districts (SEPA, 2005a). Diffuse pollution places up to 45% of the water bodies in Scotland at risk of not meeting the WFD’s environmental objectives. Agriculture was clearly shown to be the major pressure although not invariably the most severe. The results will be used to prioritise both environmental monitoring and those water bodies where management action is required. Further characterisation to refine the risk assessments is on-going and will be published by SEPA in the Significant Water Management Issues Report, as required by WFD, in mid-2007.

SEPA recently published its River Basin Management Planning Strategy (SEPA 2005b) which describes how SEPA plans to produce Scotland’s first and subsequent RBMPs. The first River Basin Management Plan will be published in 2009. This will establish the programme of measures to meet the objectives of WFD such that this programme is fully operational by 2012. The deadline for achieving environmental objectives, including those for diffuse pollution, is 2015.

NATURE OF THE PROBLEM – CHARACTERISATION OF DIFFUSE POLLUTION

The characterisation results (SEPA, 2005a) clearly showed that diffuse pollution, from a range of sources, is a significant pressure on Scotland’s water resources. A total of 488 (24.3%) river, 57 (18.4%) loch, 18 (45%) transitional, 59 (13.1%) coastal and 21 (19.8%) groundwater bodies are at risk of not achieving the WFD’s environmental
objectives due to diffuse pollution. Diffuse pollution is now a more significant source of pollution than point sources in most water bodies. This significant shift reflects the improvements in point source discharges brought about by more effective regulation. Agriculture is the most significant cause of diffuse water pollution in Scotland although it is not invariably the most severe. Urban run-off is responsible for some of the most polluted waters in Scotland. Other important pressures are forestry and acidification. Figure 1 shows the relative pressure on river waters by industry sector.

Figure 1: Diffuse pressures on rivers in Scotland by sector. The category ‘agriculture and forestry’ is where further characterisation is required to split their relative contributions. ‘Other’ includes recreation, water transport and refuse disposal activities (Scottish Executive, 2005)

A screening tool for diffuse pollutants has been developed (Anthony et al., 2005) to supplement the risk assessment described above. This has allowed sources of pollutants from agriculture, forestry, urban run-off, roads, septic tanks and sewage discharges to be estimated (Table 1). Agriculture clearly dominates the losses of N, P and soils, contributing 74%, 52% and 88% of the total load, respectively. Sewage discharges dominate the faecal pathogen load, but with agriculture making a significant 23% contribution. Forestry is a significant contributor to soil losses because of the sensitivity of some areas, e.g. upland nutrient-poor lochs, where P losses are locally significant.
Table 1: Modelled total annual losses (for Escherichia coli 1014 cfu per year, otherwise tonnes per year) to surface and groundwater by source

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<td>Urban</td>
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<td>2724</td>
<td>470</td>
<td>45569</td>
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<td>P</td>
<td>289</td>
<td>36</td>
<td>2833</td>
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<td>SS</td>
<td>46820</td>
<td>29598</td>
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<td>FIO</td>
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Information on agricultural practices causing water bodies to be placed at risk has been compiled from farm scale diffuse pollution audit work (Frost et al., 2000). Major activities posing a risk to the water environment include excessive use of nutrients in many cases caused by not taking into account the nutrient content of manures and slurries, the use of contractors resulting in spreading of slurry in inappropriate conditions, cropping in high-risk locations e.g. arable cultivation of fields that flood regularly, cultivation of slopes next to watercourses, and the access of stock to watercourses.

SEPA will continue to refine its risk assessment data. A new monitoring programme will be underway in 2006. For diffuse pollution, monitoring needs to be more intensive and it is planned to monitor in a range of ‘priority catchments’ which represent a selection of land use, soil type and climatic conditions in Scotland. More details of this and the results of further characterisation will be described in the Significant Water Management Issues Report due to be published in 2007.

**MONITORING AND CLASSIFICATION**

The WFD requires SEPA, and other responsible organisations in Scotland, to develop new monitoring and classification systems by December 2006. The monitoring and classification systems will cover all surface water and groundwater bodies, and be based on a new Ecological Classification system, with five quality classes. The classification system will be underpinned by a range of biological quality elements, supported by measurements of physico-chemistry, hydrology and morphology. As a prerequisite to remediation accurate estimates of the relative sources of diffuse pollutants must be made. For diffuse pollutants this presents a particular challenge due to the flow proportional nature of diffuse loads and concentrations (Figure 2).
Figure 2: Total phosphorus concentration during a storm event in the Loch Leven catchment (South East Scotland) (Greig et al., 2005)

SEPA’s approach to monitoring and assessing diffuse pollution is being extended and the requirement to focus on quantifying diffuse pollution loads across a range of flows has been identified. There remains a need to develop new tools for assessing diffuse pollution and identifying and targeting remediation actions.

As a mechanism for assessing diffuse pollution inputs and informing selection of appropriate remediation targets, and options for achieving these targets, the Total Maximum Daily Load (TMDL) concept, described elsewhere in this volume, which has been developed and applied successfully in North America, may have relevance to water pollution management in Scotland, particularly in respect to meeting WFD objectives.

Based on SEPA’s preliminary assessment of TMDLs, a summary of how elements of the TMDL approach could contribute to diffuse pollution assessment and management in Scotland are provided below:

- TMDLs differ from existing approaches to achieving water quality targets in that they shift attention to pollutant sources, rather than impact. This ensures that subsequent actions focus on cause as well as symptom.
- The TMDL approach allows quantification and assessment of both pollutant concentrations and pollutant loads.
- Managing pollutant loads promotes the development of mechanisms to control the cumulative impact of diffuse, as well as point source, pressures in a catchment. This might include sedimentation of riverbeds, pollution of estuarine sediments, delivering bathing water compliance and eutrophication of lochs.Degradation of many freshwater and estuarine habitats occurs as a consequence of the gradual accumulation of materials or pollutants. This accumulation is controlled by loads rather than concentrations.
- It should be possible to apply Water Quality Standards, as required by the WFD, within a TMDL approach. The development of load thresholds would be required to assess those pollutants potentially resulting in a cumulative impact on water quality or ecosystem health. This is system specific (i.e. to discharge point and sensitive species’ etc.).
- Diffuse pollution ‘hotspots’ can be targeted using the TMDL approach. This is consistent with the risk-based approach identified within the WFD.
- Source apportionment is central to the successful implementation of a TMDL and is identified within the WFD directive as a means of managing pollution.
- Many elements of the TMDL process are similar to those that are applied with River Basin Management Plans (RBMPs), thus it may be possible to integrate TMDL targets and procedures into RBMPs.
- SEPA’s diffuse pollution screening tool and proposed diffuse pollution monitoring strategy are broadly compliant with the assessment and monitoring tools promoted with the TMDL approach.

In summary, elements of the TMDL approach may have relevance to managing diffuse pollution, particularly in relation to providing a planning framework for identifying and implementing measures to address diffuse pollution pressures. Subsequent papers in this conference provide further details of the TMDL approach.

**RIVER BASIN MANAGEMENT PLANNING**

A RBMP must be produced for each river basin district. These plans set out the characteristics, pressures and impacts of the district, including a list of the water-based protected areas and an economic assessment of water services. They must also provide a summary of monitoring carried out and the consultation and participation opportunities given to interested parties. Finally, they should list the environmental objectives to be met and summarise a programme of measures designed to deliver those objectives.

The programme of measures within each RBMP will consist of a combination of the most cost effective measures that will include:

- controls under the CAR regime delivered by SEPA;
- regulatory regimes operated by other responsible authorities;
- economic instruments such as agricultural support and forestry grants; and
- voluntary measures such as initiatives to improve habitat quality and education initiatives, for example, to address urban diffuse pollution.

The river basin planning process is an ongoing one which protects, improves and promotes sustainable development and use of the water environment between each published RBMP (Figure 3). There is flexibility within river basin planning which will ensure that objectives set within each RBMP are achievable and affordable, and that the plans are able to consider strategic trends, drivers and issues affecting the water environment (SEPA, 2005).
The WEWS Act requires that Advisory Groups are to be formed to assist and advise SEPA in the RBMP production. To meet this requirement and facilitate the active involvement of stakeholders a river basin planning advisory group network is to be established from April 2006. This will consist of a National Advisory Group and a network of Area Advisory Groups. Each Area Advisory Group will also establish a forum with a wide and open membership of interested stakeholders. This tiered system of stakeholder involvement will allow a bottom-up and top-down approach to the production of RBMPs.

AGRICULTURAL SUPPORT PAYMENTS, PARTNERSHIP AND WFD

Approximately £450 million is paid to Scottish farmers under the Common Agricultural Policy (CAP) and RDR Single Farm Payment and Agri-Environment Schemes, respectively. Making sure that this funding provides an incentive to deliver a protected and improved environment, including meeting WFD objectives, is critical. Changing from a system of support that encourages production, and progressive intensification, to one that delivers multiple benefits is needed. The significance of diffuse water pollution from agriculture is now well recognised and SEPA believes that the economic incentives now need to be directed to the uptake of measures to address it.

The 2003 reform of the Common Agricultural Policy (CAP) took effect in Scotland in January 2005. As well as decoupling farm support from the need to produce crops or livestock, a system of cross-compliance was introduced. This includes a limited range of environmental legislation such as the EC Groundwater, Nitrates and Sludge Directives as well as a requirement to keep land in ‘Good Agricultural and Environmental Condition’ (GAEC). SEPA is involved as a ‘specialist agency’ in operating cross-compliance in Scotland. In Scotland, GAEC is assessed by the
Scottish Executive against 18 measures that relate to the protection of soil and habitats. Breaches of GAEC or the Statutory Management Requirements will lead to penalties being applied to farm support payments paid by the Scottish Executive.

SEPA relies on partnerships to deliver on its environmental objectives. With such diverse sectors such as farming, it is difficult to know how best to communicate on issues such as diffuse pollution. No single representative organisation covers all types of farm business in Scotland. The Scottish Crofting Foundation, Scottish Rural Property and Business Association and NFU Scotland are key stakeholder groups and will continue to be closely involved in the RBMP process in Scotland. For sewage and water-related priorities, SEPA informs a forward investment programme based on EC Directive requirements and agreed water quality objectives. It is not as well structured as that for agriculture.

Since SEPA was created in 1996, we have set out to build a constructive relationship with farmers at both a national and local level. Working closely with SAC and the Farming and Wildlife Advisory Group (FWAG) on buffer strips, as well as on waste minimisation and habitat enhancement work has been central to this approach. Since 2001, SEPA has also worked with farming, environmental, crop protection companies and the Government in supporting The Voluntary Initiative, the alternative to the introduction of a pesticides tax. This 5-year programme is to continue and is expected to align itself more closely with the delivery of WFD objectives, especially for diffuse pollution. SEPA has also recently commenced dialogue with the Agricultural Industries Confederation (AIC) as the representative body of suppliers of fertilisers and feedstuffs to UK agriculture.

Agricultural interests will be represented throughout the RBMP process. Advisory groups will identify priorities for environmental protection and an associated cost-effective programme of measures. It is vital that land managers engage in this process as this will then maximise the opportunity to target effective management action. It will be possible for individual farmers to participate actively in the RBMP process through the RBMP Area Advisory Group Fora. Farm advisors will also be an important link between farmers and river basin planning.

REGULATORY CONTROLS

The newly introduced CAR legislation will be covered in detail in the final session of this Conference. The CAR approach will apply to a number of industry sectors. A light touch of regulation is proposed for agriculture and forestry compared with other industries; the emphasis being on the use of GBRs as the mode of Authorisation.

Currently, although there is a significant amount of guidance available for farmers and the economic drivers are now more aligned towards improving water quality, there is no single piece of legislation tackling diffuse pollution per se. For rural land use the proposed approach will follow a similar structure to CAR but will be considerably less stringent than that applied to point source discharges. A combination of measures encompassing economic incentives, education, guidance and regulation will be used as part of a supportive framework. This is in view of the fact that the mitigation of diffuse pollution requires a different regulatory approach to traditional point source controls. The nature of diffuse pollution is such that controls must be focused on the activities causing the pollution rather than quantitatively regulating pollutant
losses. The proposed national GBR will ensure a level playing field for the industry. The targeted approach will ensure that measures are only applied where they are required, in accordance with risk.

It is proposed that the GBRs for rural land use will be structured around the main pressures, as follows:

- fertiliser and manure management;
- land and livestock management;
- management and use of pesticides and veterinary medicines;
- control of surface water run-off; and
- planning tools (e.g. diffuse pollution audits).

CONCLUSIONS

Managing rural diffuse pollution will depend on a good understanding of land use within catchments. The use of the Total Maximum Daily Loads could play a valuable part in River Basin Management Planning so as to address point and diffuse source pollution collectively. This will require SEPA to implement suitable monitoring strategies and learn from US experience. It will also require SEPA to work collaboratively through partnerships with key stakeholders.

A combination of measures will be required to bring about an improvement in the water environment in respect of diffuse pollution pressures. While the WFD requires legislative controls to be introduced, this cannot work in isolation from voluntary programmes, education, promotion of guidance and the provision of financial incentives via farm support payments.

A targeted approach that focuses on ‘priority catchments’ is proposed such that the accumulative benefit of adopting Best Management Practices can be gained (as well as effectiveness be measured).

The River Basin Management Planning process is crucial to the delivery of WFD objectives and stakeholder engagement is central to this. The Programme of Measures developed for the Scotland River Basin District in relation to diffuse rural pollution will need to founded on a supportive approach.

About this Text

The opinions expressed in this paper are those of the authors and do not necessarily reflect the view of the Scottish Environment Protection Agency (SEPA).
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TOTAL MAXIMUM DAILY LOADS: THE USEPA APPROACH TO MANAGING FAECAL INDICATOR FLUXES TO RECEIVING WATERS: LESSONS FOR UK ENVIRONMENTAL REGULATION?

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SUMMARY

In the European Union, the Water Framework Directive (WFD) (CEC, 2000) requires the EU environmental regulators to design a ‘programme of measures’ (Article 11) to maintain at least ‘good’ status in ‘protected’ areas which are defined in Annex VI. The first protected area mentioned is that covered by the Bathing Water Directive 76/160/EEC which is currently under revision (CEC, 2002; CEC, 2004). The original 1976 Directive, and the revisions which now define ‘good’ (or possibly the newly defined ‘sufficient’) status must be achieved or a member state could be subject to legal action, or infraction, proceedings, by the Commission. Several UK studies have been conducted to provide empirical data describing the impacts of catchment-derived faecal indicator loadings on recreational water compliance locations.

In the United States of America, the Total Maximum Daily Loads (TMDLs) concept enshrines similar principles into water quality management (USEPA, 2002). Section 303(d) of the Federal Water Pollution Control Water Act (page 103) requires that States identify water bodies that do not meet defined water quality standards. The TMDL process investigates these water quality problems and designs actions in consultation with stakeholders to effect remediation. In defining how much of a pollutant a water body can tolerate, whilst complying with water quality standards, a TMDL investigation should quantify all pollutant fluxes to include effluent discharges from wastewater treatment facilities, diffuse source pollution from agriculture and surface drainage from streets or highways. Many TMDL studies are addressing catchment derived faecal indicator loadings to recreational waters and shellfish harvesting areas (USEPA, 2002).

This paper provides an analysis of the TMDL and WFD approaches in the area of faecal indicator control. Common problems of (i) empirical data gaps; (ii) the absence of operationally appropriate modelling tools and (iii) inadequate data on the effectiveness of remedial measures are noted and some research priorities are suggested.

INTRODUCTION

This paper examines emerging issues surrounding implementation of the European Union (EU) Water Framework Directive (WFD) (CEC, 2000) and the parallel US concept of Total Maximum Daily Loads (TMDLs).

The WFD is the most significant piece of environmental legislation so far produced by the EU. The WFD defines a new context and approach for water quality regulation
based on the ‘drainage basin’ in which the member states are required to manage both point and diffuse sources of pollution to achieve ‘good’ ecological status and water quality by 2015. The approach represents a radical change from traditional point-source effluent quality regulation towards ambient water quality control at the point where the water is used for ecosystem maintenance, water supply, recreation or fisheries. Whilst correction of poor effluent quality at point source discharges can be addressed through traditional ‘engineered’ solutions at treatment plants, maintenance of ambient water quality criteria, or good ecological status, requires integrated river basin management (IRBM) involving a much wider community of stakeholders covering industrial dischargers, the farming community and land managers. Consultation with stakeholders and the wider public is required by the WFD which presents further challenges for those involved in implementation. Thus, engineers, social scientists and environmental professionals throughout Europe face new challenges and opportunities as the implications of the new approach become apparent.

In a recent review of WFD implementation issues in Austria (Achleitner et al., 2005) defined seven principal requirements that the WFD places on EU member states, namely:

1. Achievement of good ecological status (and/or potential) in all water bodies within the EU by 2015 through integrated catchment management with no deterioration of the current status;
2. Establishment of coordinated river basin management in the EU across borders for transboundary catchments (e.g. the Danube, Rhine, Elbe, etc.);
3. Development of a full cost recovery system for water supply and wastewater services which applies the ‘polluter pays’ principle and covers environmental and resource costs (although there may be a divergence between investments required by the regulator to treat ‘all flows’ and the charge based on ‘actual flows’ to the polluter by the water and sewage undertaker);
4. Formulation of river basin management plans which will be periodically updated;
5. Integrated point and diffuse source pollution control;
6. Reduction and subsequent elimination of defined priority hazardous substances;
7. Develop a legally binding ‘programme of measures’ (PoM) and water quality and quantity monitoring programmes to underpin management programmes for control and planning.

The TMDL Approach

Parallel developments in the United States of America are seen in the Federal Water Pollution Control Act (USEPA, 2002) and the Total Maximum Daily Loads (TMDLs) concept. This encapsulates very similar principles of public consultation and catchment-wide management to those in the WFD (USEPA, 2005). Section 303(d) of the Federal Water Pollution Control Water Act (page 103) requires that States identify ‘impaired’ water bodies that do not meet defined water quality standards.
The TMDL process investigates these water quality problems and designs actions in consultation with stakeholders to effect remediation (i.e. similar to a WFD ‘programme of measures’ outlined below). In defining how much of a pollutant a water body can tolerate, whilst complying with water quality standards, a TMDL investigation should quantify all pollutant fluxes to include effluent discharges from wastewater treatment facilities, diffuse source pollution from agriculture and surface drainage from streets or highways. Some 59,356 water quality impairments have been reported between January 1996 and 23 February 2006 and 18,663 TMDLs were approved by USEPA over the same period. The top five reasons for water quality impairment leading to an agreed TMDL, accounting for 69% of all TMDLs to date have been: heavy metal pollution (3,940 TMDLs); microbial pollutants and pathogens (3,630 TMDLs); nutrients (2,232 TMDLs); sediments and siltation (1,722 TMDLs); and organic enrichment and low DO (1,274 TMDLs) (USEPA, 2005). Some 4043 TMDLs, for all impairment causes, were approved by USEPA in fiscal year to 30 September 2005.

Thus, in the United States, where ‘pressure and impacts’ on water resources similar to the EU are evident, the two principal ‘impairment’ reasons leading to a TMDL are ‘metal’ pollution and ‘pathogens’ (the pathogens themselves are rarely measured but are indexed by faecal indicator organisms (FIOs) which are used as the main compliance parameters for recreational and shellfish harvesting waters).

The TMDL procedure defines ‘allocations which specify the amount (or concentration) of a pollutant that can be discharged to a waterbody such that standards are attained in both the receiving water body and all downstream waters’ (Anon, 2005a). This policy driver underpins the need for (i) information on the fluxes of catchment-derived point and diffuse source microbial loadings and (ii) the likely remediation efficacy of alternative control measures applied to the sewerage infrastructure and/or agricultural diffuse pollution sources (Kay et al., 2005a; Lewis et al., 2005; Kay et al., 2006; Dickson et al., 2005; Kay et al., 2005b).

A CASE STUDY: TOMALES BAY, CALIFORNIA

The Problem Statement

In Tomales Bay, California, the principal ‘beneficial uses’ are recreation and shellfish harvesting. The bay has been defined as ‘impaired by pathogens’ due to diffuse source pollution from agriculture and a range of other sources including human effluents, wildlife, recreational boating, equine sources, urban storm drainage and septic tanks (Anon, 2004). The ten small sewage treatment works and sewage holding ponds were not permitted to discharge to the Bay directly, rather, they spread any effluents onto fields. However, accidental spills were considered an ongoing possibility. This combined loading had caused a ‘threatened’ classification under the State Shellfish Protection Act and a prohibition on commercial harvesting during rainfall periods by the California Department of Health Services. In addition, the Bay’s shellfish have been implicated in an illness outbreak (Anon, 2003).

The Sources

A source analysis suggested that agricultural diffuse pollution produced the principal FIO loading to the Bay. The TMDL process sought to rectify this through targeted development of non-point pollution control strategies including on-farm ‘best
management practices’ (BMPs) together with an education and outreach programme driven by a catchment-wide stakeholder participation exercise including shellfishery and watershed user groups. The principal non-point source control measures for FIOs were outlined in a Basin Plan amendment of September 2005 (Anon, 2005a). This identified the three main tributaries for the bay as ‘impaired’ producing four clearly impaired locations, (i) Tomales Bay itself, (ii) Laquntas Creek, (iii) Walker Creek and (iv) Olema Creek.

**Numeric targets**

The targets are an interpretation of water quality standards and those set for the Bay were:

i. <30 shellfish harvest closures per year;

ii. in the Bay, median faecal coliform (FC) <14 100 ml-1 and 90%ile <43 100 ml-1;

iii. in the three tributaries geometric mean faecal coliform (FC) <200 100 ml-1 and 90%ile <400 100 ml-1.

**Load Allocations**

The load allocations set the highest FIO concentrations allowable in specified tributaries to ensure that the numeric targets are achieved in the receiving waters. These relate to contributions from ‘discharging entities’ and intermittent wildlife discharges are not accommodated through this mechanism. The pathogen TMDL states:

“Discharging entities will not be held responsible for uncontrollable coliform discharges originating from wildlife. If wildlife contributions are determined to be the cause of exceedences, the TMDL targets and allocation scheme will be revisited as part of the adaptive implementation program. The discharge of human waste is prohibited. All sources of human waste have an allocation of zero. Non-point source run-off containing coliform bacteria of animal and wildlife origin, at levels that do not result in exceedences of water objectives, does not constitute wastewater with particular characteristics of concern to beneficial uses. Therefore, animal and wildlife-associated discharges, in compliance with the conditions of this TMDL, do not constitute a violation of applicable discharge prohibitions.” (Anon, 2005a.)

The concentration based pollutant wasteload allocations for Tomales Bay are outlined in Table 1.

Approximately three years have been allocated to operators of waste management facilities, dairies, equestrian areas, urban wastewater schemes and managers of dairy farms to submit plans for FIO flux remediation through compliance with appropriate ‘Waste Discharge Requirements’ to achieve the limits identified in Table 1.

The Agricultural Water Quality control programme required on the Tomales Bay watershed has been costed, as required by the California Water Code, to between $0.9 to $2.0 million per year for the next ten years. This cost derives from technical assistance and evaluation, provision of water toughs and on-farm measures such as stock exclusion from stream banks and is shared between the grazing lands.
operators, which number approximately 150. The operators may be eligible for state and federal water quality grants and federal agricultural support grants, although the extent of grant aid is not specified.

Table 1: Concentration-based pollutant load allocations for dischargers of pathogens in Tomales Bay watershed

<table>
<thead>
<tr>
<th>Pollutant source</th>
<th>Waste load allocation faecal coliform 100 ml⁻¹</th>
<th>For direct discharge to the Bay</th>
<th>For discharge to tributaries</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>median</td>
<td>90th percentile</td>
<td>geometric mean</td>
</tr>
<tr>
<td>Onsite Sewage Disposal Systems</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Small Wastewater Treatment Facilities</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Boat Discharges</td>
<td>0</td>
<td>0</td>
<td>N/A</td>
</tr>
<tr>
<td>Grazing land</td>
<td>&lt;14</td>
<td>&lt;43</td>
<td>&lt;200</td>
</tr>
<tr>
<td>Dairies</td>
<td>&lt;14</td>
<td>&lt;43</td>
<td>&lt;200</td>
</tr>
<tr>
<td>Equestrian Facilities</td>
<td>&lt;14</td>
<td>&lt;43</td>
<td>&lt;200</td>
</tr>
<tr>
<td>Municipal Run-off</td>
<td>&lt;14</td>
<td>&lt;43</td>
<td>&lt;200</td>
</tr>
<tr>
<td>Municipal Run-off Open space lands</td>
<td>&lt;14</td>
<td>&lt;43</td>
<td>&lt;200</td>
</tr>
<tr>
<td>(terrestrial wildlife)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>In-Bay Background (marine wildlife)</td>
<td>&lt;14</td>
<td>&lt;43</td>
<td>N/A</td>
</tr>
</tbody>
</table>

Pollution load allocations for Tomales Bay tributaries as a geometric mean faecal coliform 100 ml⁻¹

| Walker Creek                          | 95     |
| Laqunitas Creek                        | 95     |

Geometric mean, percentile and mean values are based on no less than 5 samples collected in a 30 day period and no more than 10% of samples are to exceed the 90th percentile value.

A 5 year rolling review of water quality monitoring data is required to track improvement achieved in response to these measures and to:

- Evaluate spatial and temporal water quality trends in the Bay and its tributaries;
- Further identify significant pathogens source areas;
- Evaluate coliform levels and loadings to the Bay at the terminus of major tributaries;
- Collect sufficient data to calibrate and validate the Bay hydrodynamic model to observed coliform levels; and
- Collect sufficient data to prioritize implementation efforts and assess the effectiveness of implementation actions (Anon, 2005a).
It is envisaged that these data will allow the quinquennial review process to answer the following questions (Anon, 2005a):

1. Are the Bay and the tributaries progressing toward TMDL targets as expected? If progress is unclear, how should monitoring efforts be modified to detect trends? If there has not been adequate progress, how might the implementation actions or allocations be modified?

2. What are the pollutant loads for the various source categories (including naturally occurring background pathogen contributions and the contribution from open space lands), how have these loads changed over time, how do they vary seasonally, and how might source control measures be modified to improve load reduction?

3. Is there new, reliable, and widely accepted scientific information that suggests modifications to targets, allocations, or implementation actions? If so, how should the TMDL be modified?

4. The allocations assume a conservative bacterial die-off rate of 0.02 per hour. This value is based on rates reported for San Francisco Bay in 1970. If bacterial die-off is found to be higher, higher allocations may be considered. What are bacterial die-off rates in the water column and stream sediments? Do they vary by season? What are bacteria transport times from sources to the Bay?

5. How does estuarine mixing and dilution of tributary waters vary by flow and season?

6. What is the relationship between precipitation, run-off, tributary loads, Bay coliform levels, and water quality exceedances and shellfish harvesting closures?

7. Are there bacteria in Tomales Bay sediments that enter the water column during storm events? If yes, how should this process be accounted for?

A baseline faecal coliform tributary and Bay water quality monitoring programme is also planned from January 2006 with weekly sampling intervals for five weeks, thence monthly, sampling from March to December. Clearly, this programme would not be adequate to address the questions above and further investigations would be required to address the detailed scientific issues implied in these questions.

**Parallels with WFD**

There are many parallels between TMDLs and WFD ‘programmes of measures’, both require significant stakeholder engagement, they operate at the catchment or drainage basin scale and commence by defining the ‘pressures and impacts (WFD)’ or ‘problem statement’ (TMDL).

The existing TMDL studies are clearly ahead of the implementation of WFD ‘programmes of measures’ in the area of faecal indicator flux modelling and control but they do not appear to have, at this early stage, to have developed a set of operational tools to assist UK authorities with implementation of WFD principles at ‘protected areas’ where compliance with FIO standards is to be achieved via WFD implementation.
UK Implementation of FIO Control

The most significant policy challenge presented by the implementation of the WFD is the quantification and management of diffuse source pollution which derives from the farming community and urban sources (Defra, 2002; Defra, 2003). Examination of the available policy instruments for diffuse pollution control have been undertaken in the United Kingdom by Defra and others (Oxera, 2003; Defra, 2004; Dampney, 2002) and cost estimates have been published (RPA Consulting, 2003; WRc 1999). Clearly, the nutrient parameters, such as phosphorus and nitrogen, have sewage (point source) and agricultural (diffuse source) components. In 1999, the Water Research Centre (WRc) undertook an early scoping study based on the Commission’s proposals for a WFD first published in 1997 (COM 97(49) FINAL) (WRc, 1999). This centred principally on phosphorus reduction and calculated an additional cost to the farming sector at the time of £35,000 for each tonne of phosphorus not discharged to receiving waters, or £175 per ha per year.

Tackling diffuse water pollution from agriculture (DWPA) in England and Wales is being taken forward through the Catchment-Sensitive Farming (CSF) programme within Defra. Some £25 million is being spent over the next 2 years on a network of Catchment-Sensitive Farming Officers (CSFOs) for collective and individual engagement with farmers in priority catchments where the problems are most acute. Assessment of priority catchments includes, amongst other parameters, catchments where failures in bathing water criteria have been recorded due to DWPA. In parallel, Defra is also working to identify the most cost-effective additional policy options likely to be required to meet WFD objectives. Options being considered include the extension of existing Defra policies as well as the development of new approaches for tackling DWPA and investigation of the principal DWPA problems (Haygarth, 2005).

Considered overall, the nutrient parameters have received by far the most coordinated European attention, to date, within both the policy and academic communities with major EU projects such as EUROHARP (Borgvang, 2005) developing a ‘toolkit for nutrient management and assessments of nutrient mobility in river basin districts (Neal et al., 2005). In the UK, nutrient pressure ‘risk maps’ for controlled waters have been produced by the UK Environment Agency using an approach reported by Heathwaite et al. (2005) The analysis sponsored by Defra and that in Haygarth et al. (2005) provide a provisional inventory of diffuse pollution losses in England and Wales. They reinforce this assessment and conclude:

“Nitrogen, because of its sheer volume of usage and early legislative controls (The Nitrates Directive), is the most researched and understood pollutant. However, its impact in some regions is now thought to be of less importance than phosphorus, which controls the productivity of many inland freshwater lakes and waterways. Sediment and pathogen transfers (sic. i.e. in fact faecal indicators) represent the ‘newer’ challenge for the 21st century.”

and suggest the following forward strategy for Defra:

- Modelling to be at the centre of a future research strategy, but this needs to be backed up with long term and well co-ordinated platform catchments. Scenario
testing (including climate change) to quantify effects of measures is most cost-effectively done using appropriate policy models. We have identified that these exist (albeit better for some contaminants than others).

- There is still a need for field/catchment scale experimentation to address some issues. We suggest that the best way forward is to assess what models are now capable of achieving in support of programmes of measures. Where a lack of knowledge limits progress (e.g. the effectiveness of mechanism of a mitigation method), then experiments should then be commissioned to provide the data.

- This integrated, multi-pollutant ‘model centric’ approach, that attempts to integrate all previous models that have previously focussed on individual final goal of a robust policy model for diffuse pollution.

In terms of the suggested ‘21st Century Challenge’ of microbial modelling to limit pathogen risk as indexed by compliance with the faecal indicator criteria set in relevant Directives, source apportionment budget studies and some empirical black box models predicting faecal indicator concentration and flux from land use parameters have been reported for UK catchments. Most recently, the UKs sentinel catchment for WFD research, the Ribble catchment was modelled in this manner (Kay et al., 2005c). Additionally, empirical studies seeking to quantify the remediation potential of on-farm measures to reduce diffuse pollution have been completed in Scotland and reported in Dickson et al. (2005) and Kay et al. (2005d). Other interventions to reduce faecal indicator flux in UK catchments have been reported in Kay et al. (2005b) (i.e. flood retention wetlands) and ‘natural treatment systems’ for effluents including reed beds, a lagoon and integrated constructed wetland (ICWs) (Kay et al., 2005). One of the most promising interventions reported to date are on-farm Integrated Constructed Wetlands (ICWs) used in Ireland for treating contaminated water from farm hardstanding areas and roofs (Harrington, 2005). Initial nutrient and faecal indicator removal efficiencies seem very encouraging and the systems appear more robust to flow alterations than engineered ‘natural treatment’ systems such as reed beds (Kay et al., 2005a). However, regulatory concerns have been expressed concerning the downward translocation of pollutants to groundwater and these are the subject of current investigation with EU support in Ireland and the UK.

The conclusions of Haygarth et al. (2005) identify key research requirements of the policy community. There is a particular need to progress the field of catchment microbial and sediment modelling. The microbial component has received more international attention in Canada, Australia, the United States (i.e. via the TMDL approach outlined above) than to date in the UK and EU as part of the WFD implementation. When this emerging area has a similar science base to the nutrient parameters, the type of multi-parameter model needed to inform the policy community should be possible. There is a need, however, for pan-European coordination. This might be implemented through amendments to the current remits of the WFD ‘Common Implementation Strategy’ and, in the UK, through attention to microbial and sediment modelling needs by the UK Technical Advisory Groups addressing WFD implementation. Detailed examination of this area will be required if ‘protected areas’, such as bathing and shellfish harvesting waters, are to be managed effectively through the implementation of the principles enshrined in the WFD as clearly envisaged in early drafts (CEC, 2002) of the new Bathing Water Directive published in November 2005 (Anon, 2005b).
Whilst US regulatory approaches exemplified in the TMDL concept have sought to set water quality criteria in ‘inputs’ as seen in Table 1. There is little evidence to date that the US studies have progressed to the stage of distributed white box modelling which could underpin evidence-based advice on remediation strategies to the farming community. Current criteria specified in Table 1 are little more than ‘ambient’ water quality criteria for recreational and shellfish harvesting waters. This ‘precautionary’ approach is understandable but only serves to illustrate the basic lack of hard scientific information on catchment microbial dynamics. This suggests that Haygarth’s (Hagarth et al., 2005) definition of microbial dynamics as the 21st Century Challenge is just as appropriate to the agencies and research scientists responsible for TMDL application as it is to those EU colleagues designing programmes of measures for WFD implementation.

ACKNOWLEDGEMENTS

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REFERENCES


DECREASING THE NITROGEN SOIL SURFACE SURPLUS IN THE DANUBE RIVER BASIN BY APPLYING AGRICULTURAL MEASURES: A COMPARISON OF COST-EFFECTIVENESS RATIOS

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SUMMARY

Agricultural production is one of the main sources of diffuse nitrogen emissions. The major issue of the present work is the economic assessment of agricultural measures to reduce the nitrogen surplus in agricultural land of selected countries in the Danube River Basin. Hence it aims at the development of a method, based on cost-effectiveness analysis, to assess the potential of ‘best available techniques’—measures in agricultural production to reduce the nitrogen surplus in surface soils. The national nitrogen soil surface balance is used to generate an indicator for the effectiveness value for measures. Internal costs considering the induced change in direct and indirect production costs as well as the change of gross output are calculated. Cost-effectiveness ratios are derived and used to rank the different measures considered. Although the majority of ecologically favourable measures induce costs, ecologically and economically beneficial ones are also identified.

INTRODUCTION

The European research project daNUbs, funded under the 5th framework programme, aims at the reduction of nutrients flowing via the Danube River into the Black Sea in order to establish respectively an ecologically good condition in the western Black Sea area. The daNUbs project is based on the common understanding that all countries in the Danube River Catchment influence the Black Sea ecology and can contribute to coping with these problems. The Danube River Catchment area comprises 13 countries: Germany, Austria, Czech Republic, Slovak Republic, Hungary, Slovenia, Croatia, Bosnia and Herzegovina, Serbia and Montenegro, Romania, Bulgaria, Moldova and Ukraine, either totally or partly. At present the nitrogen load flowing into the Black Sea, originating in the Danube River Catchment, amounts to 386,816 t N per year (81% from anthropogenic sources, from which 46% points originate from agricultural production). The most important diffuse nutrient source considered here is nitrogen, originating from agricultural production processes. However, the states of nitrogen emissions from agricultural production vary widely in the countries.

In the daNUbs project, it is assumed that measures to reduce the contribution from agricultural production to the nitrogen load flowing into the Black Sea will be implemented by the year 2015. A general ‘High Production Scenario (HP 2015)’ was defined, in which the agricultural production levels of Austria and Germany were assumed to stay constant by 2015. The agricultural production of the other countries forming the Danube River Catchment area were assumed to regain the level of 1990 by 2015 because their agricultural production will pass an intensification and reach
western productivity standards. Consequently, in scenario HP 2015 the nitrogen loads from agriculture in all of the countries forming the Danube River Catchment will increase by 29% on average, compared with the level of 2000. Using the HP 2015 scenario as a reference, the impact of implementing Best Available Techniques (BAT) in agricultural production, in particular measures to minimise nitrogen emissions from agricultural production, were studied.

The main objective of the authors’ contributions to the daNUbs project (IFIP, 2005) was the development of a suitable assessment methodology for the nitrogen load flowing into the Black Sea. The present paper summarises the first step in this work, which consists of the methodology as well as their exemplarily use to assess the agricultural nitrogen surplus on the soil level. The method is based on cost-effectiveness analysis and uses the OECD nutrient balance calculation scheme (OECD, 2001) to assess the effects of the measures considered to reduce nitrogen emissions from agricultural production.

**COST-EFFECTIVENESS ANALYSIS**

The work presented here considers nine of the 13 countries mentioned. For Croatia, Bosnia/Herzegovina, Serbia/Montenegro and Moldova, no reliable agricultural production data were available and consequently no nitrogen soil surface balance could be calculated. However, the nine countries considered form 78% of the Danube River Basin. The shares of the area of the countries forming part of the Danube River basin are: Germany 16%, Austria 96%, Slovenia 81%, Czech Rep. 28%, Hungary 100%, Slovak Rep. 96%, Bulgaria 50%, Romania 93% and Ukraine 6%.

First, in cost-effectiveness analysis, the annual average internal costs of a measure, for utilised agricultural area (UAA) where it is applied, have to be identified. Second, the effect of a measure in terms of changed nitrogen surplus in the surface soil has to be calculated. Finally the cost-effectiveness ratios in terms of annual cost occurring per annually avoided unit of nitrogen surplus in the surface soil can be calculated for the measures considered.

The long-term internal costs of a measure consist of the difference in direct production costs, indirect production costs and change of gross output of the agricultural producer, induced by the measure. Costs of an activity are measured by what has to be given up in terms of real resources for what is currently being done, by implementing a measure corresponding to the opportunity cost concept (Schleiniger, 1999). The analysis of costs is based on the price level in 2002/03. In addition, it is assumed that the real price level in Central and Eastern European countries (CEEC) increases by 2% annually up to 2015 compared with Austria and Germany (IFIP, 2005). In the present work, governmental costs (such as subsidies) are not taken into consideration.

To calculate the effects of selected measures their impacts on the soil surface are determined by calculating the national nitrogen soil surface balance before and after carrying out the measure. This calculation is done according to the OECD-calculation scheme (OECD, 2001). The nitrogen soil surface balance is calculated as the difference between the total quantity of nitrogen inputs entering the soil and the total quantity of nitrogen outputs leaving the soil annually, a positive balance is called surplus.
Table 1: Overview of the measures considered to be realised in the Danube River Catchment Countries and their share of implementation

<table>
<thead>
<tr>
<th>Objective of measure</th>
<th>Quantitative Objective</th>
<th>Best available technique</th>
<th>Share of UAA where M is applied</th>
</tr>
</thead>
<tbody>
<tr>
<td>M1 Accurate application of fertilisers regarding amount and time related application rates</td>
<td>Reduction of mineral fertilisers use by 10%</td>
<td>(1) Timely application rates (2) Chemical soil analysis (3) Soil surface balance on field level (4) Ban on application of fertilisers during winter</td>
<td>Austria, Germany: 30% CEEC: 10%</td>
</tr>
<tr>
<td>M2 Reduction of nitrogen emissions from manure</td>
<td>Reduction of ammonia emissions from manure by 25%</td>
<td>(1) Use of hose spreader (2) Accurate manure storage capacity (3) Accurate straw bedding in animal housing</td>
<td>100% in all countries</td>
</tr>
<tr>
<td>M3 Increase of plant productivity by applying capital intensive production techniques</td>
<td>Increase in plant productivity Austria, Germany: +10% CEEC: +20%</td>
<td>(1) Demand-oriented irrigation (2) Demand-oriented plant protection (3) Improvement of plant nutrition</td>
<td>Austria, Germany: 15% CEEC: 30%</td>
</tr>
<tr>
<td>M4 Reduction of direct nitrogen emissions to the hydrosphere</td>
<td>Reduction of erosion by 75 % and surface run-off by 20%</td>
<td>(1) Minimum soil tillage (2) Zero tillage (3) Mulch seeding (4) Cover crops</td>
<td>20% 3% 10% 50% in all countries</td>
</tr>
</tbody>
</table>
For the calculations of production data, fertiliser input, nitrogen emission coefficients for livestock and rates of nitrogen fixation by crops were taken from the FAOSTAT (2004) databases as well as national statistics and adjusted to the scenarios. To calculate the nitrogen balances, after carrying out a specific measure, the induced changes in production and fertiliser input of this measure were estimated. The difference in the nitrogen surplus before and after carrying out the measure represents the effect in cost-effectiveness analysis.

**RESULTS OF THE COST-EFFECTIVENESS ANALYSIS**

Using the method described above, the costs of the measures specified as well as their effects on the nitrogen soil surface surplus (see Table 2) were calculated. Subsequently, cost-effectiveness ratios for each of the measures regarding the nitrogen soil surface surplus (see Table 2) were determined.
Table 2: Costs (C), effects (E) and cost-effectiveness ratio (CER) of measures regarding the nitrogen soil surface surplus in million Euro per year, 1,000 tons per year and €/kg in 2015

<table>
<thead>
<tr>
<th></th>
<th>M1</th>
<th>M2</th>
<th>M3</th>
<th>M4</th>
<th>Bundle</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>CER</td>
<td>C/E</td>
<td>CER</td>
<td>C/E</td>
<td>CER</td>
</tr>
<tr>
<td>Germany</td>
<td>2.97</td>
<td>24.5</td>
<td>8.3</td>
<td>19.91</td>
<td>268.5</td>
</tr>
<tr>
<td>Austria</td>
<td>8.43</td>
<td>30.1</td>
<td>3.6</td>
<td>24.21</td>
<td>291.6</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>0.71</td>
<td>0.9</td>
<td>1.3</td>
<td>14.83</td>
<td>65.4</td>
</tr>
<tr>
<td>Slovak Republic</td>
<td>1.02</td>
<td>1.6</td>
<td>1.6</td>
<td>12.62</td>
<td>98.0</td>
</tr>
<tr>
<td>Hungary</td>
<td>0.87</td>
<td>5.9</td>
<td>6.7</td>
<td>12.57</td>
<td>217.7</td>
</tr>
<tr>
<td>Slovenia</td>
<td>1.38</td>
<td>0.4</td>
<td>0.3</td>
<td>27.69</td>
<td>42.6</td>
</tr>
<tr>
<td>Romania</td>
<td>1.01</td>
<td>6.6</td>
<td>6.5</td>
<td>10.68</td>
<td>364.7</td>
</tr>
<tr>
<td>Bulgaria</td>
<td>0.56</td>
<td>1.7</td>
<td>3.0</td>
<td>9.69</td>
<td>68.9</td>
</tr>
<tr>
<td>Ukraine</td>
<td>1.28</td>
<td>0.7</td>
<td>0.6</td>
<td>5.47</td>
<td>31.0</td>
</tr>
</tbody>
</table>

Notes: M1, accurate application of fertilisers regarding amount and time-related application rates; M2, Reduction in nitrogen emissions from manure; M3, increase of plant productivity by applying capital intensive production techniques; M4, reduction in direct nitrogen emissions to the hydrosphere.

Costs

The internal costs accruing to the agricultural producer when carrying out a measure differ strongly depend on the area on which it is applied (considering only those areas being part of the Danube River Basin), on the structure of production regarding the kinds of agricultural products as well as on the cost levels in the countries examined. Both, measures imposing additional costs to the agricultural producers and measures increasing net profits, are identified. Measure M3 is commercially profitable in Germany, Austria, Czech Republic, Slovak Republic, Hungary and Slovenia. However, in Romania, Bulgaria and Ukraine this measure imposes costs and thus decreases net profits, because the cost-level for machinery which is involved in applying the measure is high in comparison to the general cost-level. On the contrary, the labour intense measure M4 is profitable in Romania, Bulgaria and Ukraine because of labour costs being low, while its implementation imposes costs to the agricultural producers in the remaining countries analysed.
**Effects of the Measures**

If all the measures are put into effect, the nitrogen surpluses are on average cut by a third compared with scenario HP 2015, the highest reduction compared with the initial level of the nitrogen surplus is achieved in the Ukraine (46%), the lowest in the Czech Republic (23%). The absolute reductions achieved are by far the highest in Hungary and Romania, primarily because of the size of the area on which the measures are applied. In these two countries together, the nitrogen surplus is reduced by more than 400,000 tons, which is 56% of the total reduction achieved.

**Cost-effectiveness Ratios**

The basic variables are the costs of measures on the one hand and the effects of measures regarding the difference of the nitrogen soil surface surplus on the other hand: from these, the cost-effectiveness ratios of single measures were calculated. From the viewpoint of cost-effectiveness, the most satisfactory measures are ones that are commercially profitable and at the same time beneficial to the environment, like M3 in all the EU member states as well as M4 in Romania, Bulgaria and Ukraine. Because of the high investment necessary, the highest cost-effectiveness ratios have been calculated for M2 in all of the countries examined, hence being the least preferable measure. If all of the measures considered are carried out simultaneously, the overall cost-effectiveness ratios will be positive in each of the countries. The highest cost-effectiveness ratios occur in Austria, Germany and Slovenia and the lowest in Hungary. Hence they are, as a whole, economically a burden for the agricultural producers.

**Ranking of Measures Based on Cost-effectiveness Ratios**

Figure 2 shows the costs and the effects of the measures. The ranking of the measures according to their CER reflects that profitable measures shall be implemented first. The costs of implementing all the measures considered in the different countries show a large difference regarding the expected effects. In many of the countries, a significant effect may be obtained by profitable measures, e.g. in Romania 80% of the reduction of nitrogen surplus may be achieved in a profitable way. Although the long-term profitability of specific measures is shown, high investment costs may prevent their implementation.
Figure 2: Combination of ‘best available techniques’ – measures for the different countries (ranking according to CER in brackets)

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REFERENCES


HIGH RESOLUTION PHOSPHORUS TRANSFERS IN RURAL CATCHMENTS: THE HIDDEN IMPORTANCE OF RURAL POINT SOURCES

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SUMMARY

Monitoring nutrient transfers from land to freshwater is generally limited to labour intensive discrete and/or automatic sampling regimes and, when estimating mass fluxes, some form of interpolation to fill gaps in the data series. Short-term patterns may be missed completely and storm events inappropriately sampled. Here we present a high-resolution dataset of total phosphorus (TP) concentrations in a stream draining a 5-km² agricultural catchment in Northern Ireland. The TP data are collected on-site every 10 min using homogenisation, digestion and colorimetric phases in continuous flow instrumentation. Concurrently, rainfall and stream discharge data are collected at 5 and 15 min, respectively. The data from this new technology indicate that TP depletion curves initiate prior to storm peaks. A period of baseflow also indicates the importance of rural point sources that maintain the stream in a eutrophic state between storm events.

INTRODUCTION

The Blackwater River drains approximately 1,480 km² of predominantly grassland agriculture in the wider Lough Neagh basin (4,500 km²) and the catchment straddles the Irish border region where the topography is dominated by drumlins and inter-drumlin swales. Previous research in this area has shown that diffuse and point source phosphorus (P) transfers from Blackwater sub-catchments combine to deliver an annual load of approximately 2 to 3 kg TP/ha/yr (Jordan et al., 2005a). In smaller sub-catchments, diffuse, storm-dependent P transfers also combine with chronic P transfers during non-storm periods even though there is an absence of major point sources. These chronic transfers are a minor proportion of the total annual load delivered to Lough Neagh in any year but do, however, maintain streams and larger tributaries in a eutrophic state between storms. This is particularly important during the summer when warmer temperatures promote eutrophic impacts in the river channel. In the current study, a 5-km² agricultural sub-catchment was monitored with a new in-stream TP analyser to investigate the patterns of both acute and chronic P transfer. The 10 min resolution of this analyser transcends all previous catchment-based P monitoring that is either, at least, time-based grab sampling or, at most, grab sampling augmented with some form of flow-based automatic water sampling during storm events (Kronvang and Iversen, 2002). Either method is labour intensive, and does not adequately sample the dynamics of P transfers over extended time periods. Stone et al. (2000) and Scholefield et al. (2005), for example, have demonstrated the importance of sampling both acute and chronic P transfers to investigate both storm and non-storm P transfer dynamics, respectively. The methods employed could not, however, be used to sample extensively over all flow ranges and for extended periods.
MATERIALS AND METHODS

At the 5-km² catchment outlet, a Dr Lange Sigmatax and Phosphax suite of instrumentation was installed to extract and analyse a water sample for TP on a 10-min cycle. The Sigmatax extracts a sample from the stream water column using positive air pressure and pumps into a 100-mL homogenisation chamber. This ultrasound homogenisation cycle is included to break up larger organic particles prior to digestion. A 10-mL aliquot is delivered to a digestion chamber where the sample is reduced by boiling using the sulphuric acid–persulphate method. Phosphorus is determined photometrically using the molybdate antimony–ascorbic acid method and conforms to DIN EN 38405 D11. Data are stored by internal and external dataloggers and downloaded by telemetry. A weekly cleaning cycle is performed to ensure that intake tubing is free from detritus. Reagents are changed every 3 months. Concurrent with the TP monitoring, stream discharge and rainfall data are collected by a Starflow depth–velocity recorder on a 15-min cycle, and an ARG100 tipping bucket rain gauge set to measure events on a 5-min cycle, respectively. In this paper, all data are averaged to a 1-h resolution for comparison.

RESULTS

Results are presented for April to October 2005 and incorporate several large storm events and periods of baseflow (Figure 1). Three TP transfer ‘event-types’ were observed during this period. First, large TP transfers were coincident with storm discharges, although lower TP peaks were observed with succeeding storms and TP depletion during storm events occurred prior to discharge peaks. This indicates periods of P flushing where build-up from, for example, channel attenuation occurs during smaller events. This acute diffuse transfer is likely to be from a combination of soil P and channel sediment P sources. Second, as baseflow established over the summer period, non-storm TP transfers became greater, indicating a concentration effect of one or more chronic point sources. Furthermore, during periods of zero rainfall, these chronic TP transfers had a diurnal pattern with a recovery between 23:00 and approximately 11:00 hours each day followed by a rapid increase that was sustained for the rest of the day (Jordan et al., 2005b). The third ‘event-type’ was characterised by sudden TP increases that did not correspond with similar increases in stream discharge. These events were attributed to discrete pollution episodes unrelated to storm diffuse or chronic point sources.

DISCUSSION

The identification of three TP transfer ‘event-types’ in an agricultural mini-catchment was only possible by implementing a high resolution, time-based sampling regime, in this case using a bank-side automated analyser. Even with the data limited to a single summer, the results have important implications for monitoring and management of freshwater eutrophication in rural rivers. The diffuse, storm dependent transfers accounted for most of the TP load transferred from this grassland catchment. The chronic and discrete TP transfers that were independent of stream discharges did, however, present an ongoing eutrophication impact and maintained the stream system in the hypertrophic range for the summer of 2005. This is a pattern observed in many Northern Ireland rivers. In the absence of waste-water treatment works and other industrial point sources, it will be important to define and mitigate these
sources. They are most likely to be a combination of single dwelling septic tanks and farmyard infrastructure and the impact will be reach specific depending on agricultural intensity and rural population density.

Figure 1: Hourly rainfall, discharge and TP concentration time-series from April to October 2005 in a 5-km² agricultural mini-catchment

CONCLUSIONS

Routine monitoring of eutrophic streams on a weekly to monthly basis may largely sample non-diffuse P transfers. In some agricultural catchments, where diffuse sources are assumed to transfer most of the annual P load, monitoring on such a coarse resolution will not adequately assess the effects of managing P loss from soils after mitigation measures and are more likely to continue measuring the impacts from rural point sources.

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AN INITIAL ASSESSMENT OF THE SUITABILITY OF TOTAL MAXIMUM DAILY LOADS (TMDLs) AS A MEANS OF MANAGING DIFFUSE POLLUTION UNDER THE WATER FRAMEWORK DIRECTIVE

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SUMMARY

In the United States, the control of diffuse pollution is addressed by the Total Maximum Daily Loads programme. In this paper, we assess whether this programme is suitable for use under the Water Framework Directive, and identify two key issues concerning the setting of water quality standards and uncertainty.

INTRODUCTION

The Water Framework Directive (WFD) raises a number of challenges for water quality regulation in the UK, one of which concerns the identification, quantification and control of diffuse pollution inputs to water bodies. With the initial diffuse pollution characterisation and impacts analysis now complete, and the monitoring programmes about to be implemented, attention must now turn to the setting of environmental objectives and the design and implementation of programmes of measures (POMs) to control diffuse pollution inputs. While the WFD provides a basis for the development of legislation in this respect, an appropriate regulatory framework has yet to be established. In the United States (US), the control of diffuse pollution is addressed by the Total Maximum Daily Loads (TMDL) programme, which provides a framework within which linkages between diffuse pollution sources and impacts are assessed, and permissible loads accordingly defined. The purpose of this paper is to introduce the TMDL concept, demonstrate its application using the Tarland Burn in NE Scotland as a case study, and discuss its suitability as a means of managing water quality under the WFD.

THE TMDL CONCEPT

Over the past 30 years, water quality management in the US has focussed mainly on the control of point source pollution, via effluent-based water quality standards established under the Clean Water Act (CWA). Although this has resulted in a general improvement in water quality, in 2000 over 36 % of water bodies still did not meet the CWA goal of being ‘fishable, swimmable’, due to inputs of pollutants such as nutrients and sediment from diffuse sources such as agriculture and urban run-off (Boyd, 2000). Consequently, there has been a recent shift in the focus of water quality management from effluent-based to ambient–based water quality standards. This is the context within which the Environmental Protection Agency (EPA) is obligated to implement the TMDL programme, the objective of which is attainment of ambient water quality standards through the control of both point source and diffuse pollution.
Although the TMDL programme originated in the 1970s, it was largely overlooked until the early 1990s, when the EPA was forced to develop guidance in response to increasing litigation concerning the status of many water bodies. Under new regulations issued by the EPA in 1992, all states and tribes were tasked with assessing the condition of their water bodies every two years, establishing priorities for pollutant load reductions and implementing improvements. As a result, over 21,000 water bodies have since been identified as being impaired, which will necessitate the development of more than 40,000 TMDLs over the next 10–15 years. To address this problem, the EPA published proposed changes to the TMDL rules in 1999, which were intended to strengthen the programme’s ability to achieve clean water goals, by offering clearer guidance, new tools and improved stakeholder engagement via knowledge transfer. However, while the proposed rules were seen by some as a welcome step towards a more holistic approach to managing water quality, it was also clear that they would significantly alter the politics, economics and implementation of water quality regulation. As a result, the proposed rules attracted significant criticism, particularly in terms of the timetable for implementation and the adequacy of the science underpinning the control of diffuse pollution. The new rules were finally approved in 2000, but Congress delayed their implementation, and the EPA has since withdrawn them, stating that their implementation would be untenable. For now, the 1992 regulations remain in effect.

Under the 1992 regulations, all states and tribes must assess the condition of their water bodies in relation to numeric and narrative water quality standards set for designated uses (e.g. aquatic life protection, drinking water supply, recreation). It is recognised that such standards may not always be met, due to background variability and measurement errors, and as a result a 10% violation rate is allowed (up to 10% of samples collected may exceed set standards). However, in instances where the violation rate exceeds 10%, the affected water body must be classed as impaired, and a TMDL developed for the pollutant causing the impairment (EPA, 1999). The first step in the process of TMDL development is the identification of the key factors and background information that describe the nature of the impairment and the context for the TMDL. The second step is the identification of numeric or measurable indicators and target values that can be used to evaluate attainment of water quality standards in the impaired water body. Often the TMDL target will be the numeric standard for the pollutant of concern, but in some cases TMDLs must be developed for parameters that do not have numeric standards. In these situations impairment is determined using narrative standards or identifiable impairment of designated uses (e.g. no fish), which are then interpreted to develop quantifiable target values to measure attainment or maintenance of the desired level of water quality.

The third step in the process is a source assessment, in which the sources of pollutant loading to the impaired water body are identified and characterised by type, magnitude and location. In the fourth step of the process, a linkage must be defined between the selected indicator(s) or target(s) and the identified sources. This linkage establishes the cause and effect relationship between the pollutant of concern and the pollutant sources, and enables the total loading capacity to be estimated. Acceptable pollutant loadings that will not exceed the total loading capacity and will lead to attainment of the desired level of water quality are then calculated in step
five. These loadings are distributed among the significant sources of the pollutant of concern, with wasteload allocations containing the allowable loadings from point sources, and load allocations containing the allowable loadings from diffuse sources. A margin of safety is identified at this stage to take account of uncertainty in the analysis, although this is often arbitrarily determined rather than rigorously quantified. The margin of safety may be applied implicitly, using conservative assumptions, or explicitly, by setting aside a portion of the total loading capacity. Finally, in step six an implementation and monitoring plan is devised to oversee changes on the ground and determine whether the TMDL has resulted in attaining the desired level of water quality. A conceptual overview of how the TMDL framework could be integrated with river basin management planning under the WFD is presented in Figure 1.

Figure 1: Conceptual overview of how water quality could be managed under the Water Framework Directive using the Total Maximum Daily Loads framework (in grey) as a means of regulation

CASE STUDY: TARLAND BURN

Introduction

The Tarland Burn is a tributary of the River Dee in NE Scotland, and drains an area of intensive mixed land use. Agriculture in the lower zone of the catchment comprises a mosaic of arable fields and improved grassland, supporting cattle, sheep, barley and potato. The headwaters of the catchment feature steeper slopes comprising rough
grassland and heather moorland, supporting widespread plantation forestry. The village of Tarland has a population of 520, and is served by a waste water treatment plant (WWTP) situated on the main stem of the burn. The remainder of the population is rural, with on-site sewerage in the form of septic tanks. Under the recently completed characterisation and impacts analysis, the Tarland Burn was classified as being at significant risk (1a) of not achieving good status by 2015, due to diffuse pollution and morphological pressures. A significant amount of research on diffuse pollution impacts and best management practices (BMPs) has been conducted within the catchment over the past 5 years, making it an ideal study site for an assessment of the suitability of TMDLs as a means of managing diffuse pollution.

**Data Acquisition**

During the 353-day period from the 3rd of July 2004 to the 20th of June 2005, intensive stream monitoring was conducted on the main stem of the burn at Coull Bridge. It should be noted that this location is several kilometres upstream of the true catchment outflow, meaning that only 72% (51.6 km²) of the true catchment area (71.6 km²) was monitored. Stage was measured continuously using an SR50 sonic stage sensor, and logged on a CR10X datalogger. The record of stage was subsequently converted into a time series of discharge using a rating relationship derived from discrete measurements of discharge over the full range of flows (n = 10, \( r^2 > 0.99 \)). The burn was sampled on a daily basis using an automatic pump sampler. Additional samples were acquired on a four hourly basis whenever flows exceeded Q10 (2.756 m³ s⁻¹, 1999-2002). All samples were stored in the sampler until collection, and returned to the laboratory for analysis on a weekly basis. Total phosphorus (TP) concentrations were then determined colorimetrically (Skaler SAN++) on the unfiltered samples after persulphate digestion. For days when only a single sample was collected, the daily TP flux was calculated by multiplying the total daily water flux by the sample concentration. On days when flows exceeded Q10, the daily TP flux was calculated by summing fluxes determined for each four hourly period (in turn determined by multiplying the four hourly water flux by the corresponding sample concentration).

**TMDL Development**

As new physico-chemical standards for use under the WFD have yet to be published, we opted to use a target concentration value for TP of 0.05 mg/L, which is recommended by the EPA as an indicative value of impairment in many rivers and streams in the US (EPA, 1999). Over 24% of samples collected during the period of monitoring had TP concentrations in excess of this standard; under EPA guidelines this would necessitate the development of a TMDL (EPA, 1999). The total water flux recorded in the period of monitoring (2.27 x 10⁷ m³) was then multiplied by the target concentration value for TP (0.05 mg/L) to give the target loading capacity for TP (1135 kg). This value was then divided by the number of days in the period of monitoring (353) to give a TMDL of 3.2 kg (presented against actual daily fluxes of TP in Figure 2). The target loading capacity for TP was then compared with the actual flux of TP recorded during the period of monitoring (1428 kg), in order to determine the amount by which the actual flux of TP must be reduced in order to meet the target concentration value for TP (= 293 kg).
The major sources of P in the catchment were then identified and their respective contributions to the actual flux of TP recorded during the period of monitoring determined using GIS-based tools (Ferrier et al., 2004). These tools enable human population density data from the 2001 census to be used in conjunction with user-entered per-capita rates for specified pollutants to estimate annual pollutant fluxes from waste water treatment plants (WWTPs) and septic tanks. The estimated annual P loadings from these sources in the Tarland catchment were adjusted for the length of the monitoring period and then subtracted from the actual flux of TP recorded during the period of monitoring to give an estimate of P loading from diffuse sources (Figure 3). Relationships between loadings of P from major sources in the catchment and violations of the target concentration value of TP were then examined in order to determine acceptable load allocations to meet the desired level of water quality. Analysis of the field data revealed that the export of P during the period of monitoring was positively correlated with discharge (r = 0.47, P = <0.001). Although the strength of this relationship is statistically weaker than might be expected, almost all violations of the target concentration value were associated with run-off events.

Figure 3: Source assessment of total phosphorus loads at Coull Bridge during the period of monitoring, using per-capita rates of 0.7 and 0.5 kg/a for the WWTP and septic tanks, respectively.
Given that the release of P from the WWTP and septic tanks would have been relatively consistent during the period of monitoring, we concluded that loadings from diffuse sources were probably responsible for the majority of violations of the target concentration value for TP. We therefore set wasteload allocations for the WWTP and septic tanks at the estimated values for the period of monitoring, and set the load allocation for diffuse sources as the estimated value for the period of monitoring minus the amount by which the actual flux of TP must be reduced in order to meet the target concentration value for TP. In order to allow for a margin of safety in our load allocation budget, we derived mean probable errors for best and worse case data acquisition scenarios using results published in the literature (Figures 4 and 5). Sources of error in load calculations may be grouped into four main categories, namely errors associated with the measurement of flow, errors associated with sample collection, errors associated with sample storage and errors associated with laboratory analysis. Further sources of error also arise when using process-based models and GIS tools to estimate source loadings, but these have yet to be satisfactorily quantified. Therefore, the error ranges presented should be viewed as minimum estimates.

Figure 4: Time series of best case uncertainty (± 17 %) in daily phosphorus loads at Coull Bridge during the period of monitoring. At this level of uncertainty, a 49% reduction in diffuse P loading is required to achieve the target concentration value of 0.05 mg/L for TP.
Figure 5: Time series of worst case uncertainty (± 67 %) in daily phosphorus loads at Coull Bridge during the period of monitoring. At this level of uncertainty a 79% reduction in diffuse P loading is required to achieve the target concentration value of 0.05 mg/L for TP.

After recalculating the required reduction in diffuse P loading under the best and worst case error scenarios, we finally calculated the cost of implementing best management practices (BMPs) to meet the desired level of water quality. A number of BMPs have been shown to be effective at reducing P losses from agricultural land, particularly when used as part of a treatment train (Vinten et al., 2005). However, for simplicity, we opted to consider only a single control measure, namely the installation of buffer strips. Removal efficiencies of between 50 - 80 % for TP have been reported in the literature for properly sized and managed buffer strips (Novotny and Olem, 1994). Under the best case error scenario, we opted to install 6-m wide buffer strips along 29.3 km of streams lying within the zone of arable fields and improved grassland using Land Management Contracts Menu Scheme (LMCMS) rates for buffer strip creation and management (£200 per ha), at a total cost of £35,160 over 5 years. However, in order to meet the desired level of water quantity under the worst case error scenario, we opted to install 20 m wide buffer along the same stream length using Rural Stewardship Scheme (RSS) rates for fencing (£3 per m) and the creation and management of species-rich grassland (£250 per ha), at a total cost of £322,300 over 5 years.

DISCUSSION

The TMDL programme forms the backbone of the EPA's current watershed level approach to water quality regulation, and is in many ways similar to the framework established for river basin management planning under the WFD. One important difference arises when considering the establishment of environmental objectives, and that is that the WFD requires all water bodies to achieve both good ecological and chemical status, whereas the TMDL programme requires only the latter. However, providing that indicators of ecological status are measurable using numeric targets, then there is no reason why the TMDL process could not be implemented under the WFD. Indeed, we believe that the TMDL process offers a robust framework for controlling both point source and diffuse pollution, as it integrates the variable effects of flow and targets both acute and chronic impacts. Moreover, it also shifts the focus of water quality regulation away from symptoms of impairment and on...
to the underlying causes, and thus provides a more holistic approach to managing water quality at the catchment scale. The case study presented above highlights two areas of critical importance to the success or otherwise of the TMDL process. The first area concerns the setting of the environmental standard; where do we draw the line? If we had used a target concentration value of 0.03 mg/L for TP instead of the guideline value recommended by the EPA, the required reduction in diffuse P loading would have been 79%, even before the effects of uncertainty were considered (adding in the effects of uncertainty under the worst case scenario would have required the complete eradication of diffuse P loading, and even then the desired level of water quality would not have been met). The second area concerns the level of uncertainty itself; who pays for this? It is not unreasonable to assume that the worst case scenario presented here is probably typical of spatially distributed, investment-limited monitoring programmes. The cost of regulation under the worst case scenario is an order of magnitude greater than under the best case scenario, and the difference should not be classed as an agricultural externality. Rather, the cost of additional uncertainty should be considered in its own right, and perhaps targeted ahead of remediation actions in order to make regulation more cost effective in the long term.

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A STATEWIDE APPROACH TO IDENTIFYING, QUANTIFYING AND MITIGATING DIFFUSE POLLUTION-RELATED PROBLEMS

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SUMMARY

The State of Pennsylvania has implemented a number of strategies and programs to mitigate surface water quality problems throughout the state. To address these problems, the state has developed an integrated approach to identify and quantify water quality problems using rapid bio-assessment and simulation modeling protocols, respectively. In the latter case, watershed and/or ‘in-stream’ computer models are used to estimate loads as part of ‘total maximum daily load’ assessments required by the Clean Water Act. The results of such assessments are used to prioritize areas for future restoration activities. These restoration activities, which are normally co-ordinated by local watershed groups, are funded by the State’s ‘Growing Greener’ programme that combines money from both federal and state sources.

INTRODUCTION

Under Section 303(d) of the Clean Water Act, all states within the United States are required to conduct Total Maximum Daily Load (TMDL) assessments for all water bodies within their jurisdiction that are not attaining water quality standards for their designated use. To address this requirement, the Pennsylvania Department of Environmental Protection (PaDEP) has developed a multi-tiered approach to identifying, quantifying and rectifying water quality problems within the state. This paper describes PaDEP’s integrated approach to addressing those problems primarily related to diffuse or non-point source pollution impacts on streams.

This integrated approach incorporates the following components: (1) identification of water quality impacts via the use of assorted protocols, (2) inclusion of identified problem areas on a statewide ‘impaired waters’ list, (3) quantification of specific pollutant loads via the use of in-stream water quality data or simulation modelling, (4) development and implementation of mitigation plans, (5) evaluation of results through future assessment activities, and (6) development and implementation of new mitigation strategies as needed.

PROBLEM IDENTIFICATION

PaDEP’s assessment and listing methodology constitutes the ‘decision rules’ that the Department uses when assessing the quality of waters and identifying water bodies that do not meet designated and existing uses. The primary ‘uses’ for water bodies include aquatic life use, human health use and recreational use. Problems associated with impacts to aquatic life use are identified primarily through stream biological community assessments. The Department identifies impacts to human health using fish tissue analyses and evaluations of raw (intake) water monitored by water purveyors. Recreational use impairments are identified using bacteriological data.
All water body use attainment information is organized by State Water Plan Sub-basin, PaDEP five-digit stream code, and a segment identifier which identifies multiple unique stream segments within each five-digit stream code (see Figure 1). PaDEP’s five-digit stream code system is based on surface waters that appear on United States Geological Survey, 1:24,000-scale, 7.5-Minute Quadrangle Maps. Streams are segmented at the confluence of tributaries, and each segment is identified by the PaDEP five-digit stream code and upstream and downstream river miles (distance above the mouth of the stream).

Figure 1: Mapping protocol for organizing stream assessment data
State water quality standards assign designated beneficial uses to waters, and the PaDEP assesses waterways to determine if uses are being attained. A variety of physical, chemical, and biological water quality and habitat indicators are measured, and this information is used to decide if water uses are attained. The water quality indicators used by Pennsylvania for making use attainment decisions are interpreted within the context of the standards, including designated uses, narrative or numeric criteria, and anti-degradation policies. Other factors that influence the state’s selection of indicators include: sampling effort, the cost of collecting and analysing samples, the variability of the indicator in the environment, the level of precision desired by decision makers, and the sampling frequency required to meet data quality objectives. In reality, the bulk of the problems identified for future mitigation activities are a result of rapid bio-assessment stream surveys.

*Aquatic life* use data are used to assess the ability of Pennsylvania’s water bodies to maintain and/or propagate fish species and additional flora and fauna that are indigenous to state aquatic habitats. Aquatic life use attainment surveys of the state’s wade able streams and rivers (systems that do not exceed one meter in depth or one meter/second velocity) are conducted through the DEP’s Statewide Surface Waters Assessment Program (SSWAP). This SSWAP involves a statewide assessment of habitat and biological (benthic macroinvertebrate) indicators that are used to measure the ability of surface waters to support expected aquatic life uses.

*Human health use* attainment surveys of Pennsylvania’s surface waters are conducted through the Pennsylvania Fish Tissue Sampling and Fish Advisories Programme. The purpose of this multi-agency programme is to sample for toxins potentially harmful to humans. Target species usually consist of water body-specific, recreationally important species that are commonly taken by anglers for consumption. Fish tissue samples are generally collected during periods of low flow between August and October when reproduction is complete and a full summer of exposure to potential toxins has occurred.

Currently, this programme includes a mixture of risk assessment-based methods and U.S. Food and Drug Administration (FDA) Action Levels that are used as the basis for issuing or lifting advisories. Risk assessment methods form the basis for meal-specific advisories due to PCBs and mercury. Advisories for other compounds use FDA Action levels to issue ‘Do Not Eat’ advice. Once advisories are issued, the affected stream segments are put on the State’s ‘impaired water body list’. Additional Human Health use attainment evaluations, which may result in subsequent ‘listing’, are conducted through the review of raw (intake) water quality as described through self-monitoring efforts of drinking water facilities. Physical, chemical, or bacteriological quality over extended periods of time is compared to potable water supply criteria outlined in Pennsylvania’s Water Quality Standards regulations to determine use attainment status.

*Recreational use* attainment decisions for Pennsylvania’s surface waters are made using bacteriological indicator data collected by government agencies (including the PaDEP, the Pennsylvania Department of Health, and the United States Geological Survey) and citizen/volunteer groups. Faecal coliform bacteria are used as indicators of possible sewage contamination because they are commonly found in human and animal feces. Although faecal coliforms are generally not harmful themselves, they
indicate the possible presence of pathogenic (disease causing) bacteria, viruses and protozoa that also live in human and animal digestive systems. Therefore, their presence in a water body suggests that pathogenic microorganisms may be present as well, and that water contact recreation such as swimming may be a health risk.

Important recreational areas and aquatic life use impaired water bodies with obvious potential sources of bacteria (e.g. municipal point sources, combined sewer overflows, and agricultural sources relating to manure application, livestock grazing, and animal feeding) are targeted for bacteriological sampling. Sampling is conducted during the swimming season (May 1 through to September 30) and consists of at least two sampling groups collected per site per season.

Based on the use of various measures, the water quality status of Pennsylvania’s waters is summarized using a five-part categorization of waters according to their use attainment status. Each five-digit water body segment (as shown in Figure 1) is placed into one of these categories. Different segments of the same stream may appear on more than one list if the attainment status changes as the water flows downstream. The listing categories are as follows:

**Category 1:** Waters attaining all designated uses.

**Category 2:** Waters where some, but not all, designated uses are met. Attainment status of the remaining designated uses is unknown because data are insufficient to categorize a water consistent with the state’s listing methodology.

**Category 3:** Waters for which there are insufficient or no data and information to determine, consistent with the State’s listing methodology, if designated uses are met.

**Category 4:** Waters impaired for one or more designated uses but not needing a TMDL because a TMDL has been completed, use attainment is expected to occur within a short time frame (within 3 years), or the impairment is not due to a pollutant.

**Category 5:** Waters impaired for one or more designated uses by any pollutant.

**LISTING OF IMPAIRED WATERS**

The results of all assessments completed within the state are reported on a publicly-available ‘Integrated List’ (www.depweb.state.pa.us/dep/cwp/view.asp?a=3&q=461149&depNav=l).

On this site, streams can be displayed on top of topographic maps and queried via the use of an interactive database. Among other things, streams can be queried as to the reason for their being given an ‘impaired’ (i.e. Category 5) listing. Such reasons are described in terms of the ‘source’ (e.g. municipal point source, agriculture, urban run-off, surface mining, etc.) and ‘cause’ (e.g. nutrients, siltation, organic enrichment/low DO, etc.) assignments given to each impaired stream segment.

Waters included under Category 5 essentially constitute the Section 303(d) list (i.e. ‘impaired waters list’) that the U.S. EPA will approve or disapprove under the Clean Water Act. Where more than one pollutant is causing the impairment, the water
remains in Category 5 until all pollutants are addressed in a completed/EPA-approved TMDL, or until the water is ‘de-listed’ via other approved administrative means.

**QUANTIFICATION OF PROBLEM**

Water bodies identified on the State’s 303(d) list as being impaired require the completion of a TMDL assessment. Such assessments can be done for individual impaired stream segments. More typically, however, such assessments are done on a watershed or sub-watershed basis, with the assessed areas ranging in size from about 2–200 square kilometres. Development of a TMDL implies development of a report with specific mandated elements. At a minimum, this report must describe the water quality problem and identify allowable pollutant loads to the water body from both point and non-point sources that would prevent a violation of water quality standards in the future. This latter requirement dictates that some means be used to quantify pollutant loads in a watershed for which a TMDL is being prepared. Given the general lack of sufficient in-stream water quality data, many states in the United States have decided to use computer simulation models to derive the necessary load estimates. In Pennsylvania, a GIS-based modelling approach has been developed to support TMDL assessments in watersheds where nutrients and/or sediments have been determined to be the primary cause(s) of stream impairment. This approach involves the use of AVGWLF, a calibrated modeling tool that facilitates the use of the Generalized Watershed Loading Function (GWLF) model (Haith and Shoemaker, 1987) and customized statewide datasets and model parameterization routines within an ArcView GIS software interface.

The general approach used for non-point source TMDLs in Pennsylvania is to: (1) derive GWLF input data for use in the impaired watershed; (2) simulate nutrient and/or sediment loads within the impaired watershed; (3) compare simulated loads within the impaired watershed against loads simulated for a nearby ‘reference’ watershed that exhibits similar landscape, development and agricultural patterns, but that also has been deemed to be unimpaired; and (4) identify and evaluate pollution mitigation strategies that could be applied in the impaired watershed to achieve pollutant loads similar to those calculated for the reference watershed.

With AVGWLF, a customized interface developed by Penn State for the ArcView GIS package is used to parameterize input data for the GWLF model (Evans et al., 2002). In utilizing this interface, the required GIS files are loaded automatically, and various tools are provided to create and select watersheds for simulation purposes. Once the selected watershed is identified (see Figure 2), the user is prompted to specify the desired file location for model output and to supply other information related to “non-spatial” model parameters (e.g. beginning and end of the growing season; and the months during which manure is spread on agricultural land). Critical terrain and other watershed data are extracted based on the watershed boundary and subsequently used to automatically derive values for required model input parameters that are then written to the *transport.dat* and *nutrient.dat* input files needed to execute the GWLF model. Also accessed through the interface is a statewide weather database that contains 25 years of temperature and precipitation data for 78 weather stations across Pennsylvania. This database is used to create the necessary weather.dat input file for a given watershed simulation. Subsequent to input file creation, the GWLF model is executed directly within the GIS interface. An example of some of the loading output generated by the model is shown in Figure 3.
DEVELOPMENT AND IMPLEMENTATION OF MITIGATION PLANS

As part of the TMDL report, some description of potential strategies for mitigating water quality problems in the watershed must be provided. However, little detail is required at this stage since more in-depth analyses are usually undertaken as part of 'watershed restoration' activities subsequent to TMDL development. In Pennsylvania, the completion of a TMDL typically triggers the release of state funds from the State's 'Growing Greener' programme to develop and implement a restoration plan. This particular programme is the largest environmental investment in modern Pennsylvania history, and represents a dramatic restructuring of state spending policy to address critical environmental concerns. To date, the Growing Greener programme has provided nearly $650 million to protect watersheds, preserve farmland open space, invest in parks and outdoor recreation, reclaim abandoned mines and wells, and make improvements to the state's water and sewer infrastructure. This programme has distributed grants to local governments, watershed groups and others for the protection of Pennsylvania's water resources, including the management of non-point sources of pollution.

Figure 2: GIS interface
One of the first activities typically completed during the restoration planning and implementation process is a more thorough evaluation of potential mitigation strategies. One tool that many groups in Pennsylvania use to support this activity is a software programme that operates in tandem with AVGWLF. More specifically, nutrient and sediment loadings derived via the use of AVGWLF can be used as input in a companion software tool called PRedICT (Pollution Reduction Impact Comparison Tool). This tool allows the user to create various ‘scenarios’ in which current landscape conditions and pollutant loads (both point and non-point) can be compared against ‘future’ conditions that reflect the use of different pollution reduction strategies (best management practices) such as agricultural and urban BMPs, stream protection activities, and upgrading of wastewater treatment systems. It includes pollutant reduction coefficients for nitrogen, phosphorus and sediment, and also has built-in cost information for an assortment of pollution mitigation techniques. The user specifies desired conditions in terms of such things as acres of agricultural BMPs used, number of septic systems to be converted to centralized wastewater treatment, types of plant upgrades, percentage of urban areas to be treated by wetlands and detention basins, etc. Built-in reduction coefficients and unit costs are then utilized to calculate resultant nutrient and sediment load reductions and scenario costs.

Figure 4 illustrates the type of output that can be obtained for a given watershed via the use of AVGWLF and PRedICT. Shown in this figure are ‘existing’ and ‘future’ loads calculated on the basis of a suite of proposed pollution mitigation strategies. Calculations of pollutant load reductions and associated costs within PRedICT are accomplished via a series of data handling algorithms and mathematical expressions written in Visual Basic. The general approach used in most cases is to calculate load reductions for each pollutant based on the number of additional ‘units’ (e.g. acres, stream miles, per capita septic system conversions, etc.) for which the particular
BMP is being implemented and the appropriate pollutant reduction coefficients and unit costs specific to that BMP. These additional 'units' are based on the difference between 'existing' and 'future' values (e.g. acres, stream miles) specified by the user for each BMP or pollutant reduction strategy. More specific information on the usage of PRedICT is provided in Evans (2003).

POST-RESTORATION EVALUATION

After restoration activities have been implemented in a watershed, the success of such activities is evaluated on the basis of in-stream surveys conducted to verify if the health of the stream has improved. If no improvements are noted, additional funds are provided by the State to implement more or better corrective measures in the watershed until the desired results are achieved. This is an iterative process that may take many years to complete in significantly degraded watersheds. However, this approach is predicated on the belief that moving towards a corrective solution is better than not moving at all.

To date, few restoration projects funded through the Growing Greener programme have been in place long enough to warrant re-examination of affected streams. However, limited data from some fixed water quality monitoring stations maintained by PaDEP that are located at the mouth of watersheds where mitigation measures have been taken suggests that improvements in stream health are beginning to occur. Within the next 2–5 years, PaDEP plans to begin the process of re-surveying Pennsylvania’s streams to evaluate the effectiveness of mitigation measures that have been implemented over the past few years.

Figure 4: Example of predicted load reductions via combined use of AVGWLF and PRedICT
REFERENCES


THE USE OF PONDS TO REDUCE POLLUTION FROM POTENTIALLY CONTAMINATED STEADING RUN-OFF

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SUMMARY

Ponds/wetlands are being increasingly encouraged as a tool for reducing diffuse pollution from farm steadings in Scotland to comply with the EC Water Framework Directive. However, guidance on their design and maintenance is limited and their effectiveness for different farm types in Scotland is not known. Results from continuous flow and water quality monitoring (May–July 2005) of a mature onstream pond/wetland, constructed on a dairy farm prior to current best practice guidelines, and from vegetation and macroinvertebrate surveys of this and two other farm ponds/wetlands highlight some aspects to be considered in monitoring the performance of these systems. Future work will focus on testing and improving design guidance by systematically monitoring and modelling flow, water quality, habitat and factors affecting performance and maintenance in representative farm ponds/wetlands, including new systems constructed with funding from the Scottish Executive and SEPA's Diffuse Pollution and Habitat Enhancement Initiatives.

INTRODUCTION

The EC Water Framework Directive (WFD) (EC, 2000) requires the introduction of co-ordinated programmes of measures to achieve good ecological status in surface waters. The control of diffuse, land-based sources of pollution to these waters is of increasing concern to regulatory agencies seeking to implement the Directive. In Scotland, nutrients [nitrogen (N) and phosphorus (P)], suspended solids (SS) and faecal indicator organisms (FIOs) arising from diffuse agricultural run-off are a major cause of downgrading of rivers, lakes and coastal bathing waters. The use of farm ponds to mitigate the collectively significant pollution arising from steadings has recently been promoted in Scotland. The aim of installing ponds is to treat run-off from farms, particularly from areas such as access tracks and roads, roofs, road crossings and peripheral yards with only minor or occasional pollutant loading. While many studies report good removal of nutrients, SS and FIOs in ponds and constructed wetlands (e.g. Borin et al., 2001; Hunt and Poach, 2001; Dunne et al., 2005a), it is necessary to evaluate the likely performance of farm ponds/wetlands, not only immediately after installation, but also over time. Performance of these systems has been shown to vary seasonally (Thorén et al., 2004; Dunne et al., 2005b), with changing hydraulic and pollutant loadings (Fink and Mitsch, 2004) and may decrease with increasing age as sediment accumulates and vegetation develops.

The processes involved in water treatment in farm ponds vary between pollutants. P removal occurs mainly by sedimentation, adsorption to substrate and plant uptake.
(including algae) whereas for N nitrification/denitrification can be a significant sink. Conversely, farm ponds may also be sources of N and P, released from sediment and winter die-back of vegetation and algae. Pond performance may therefore vary between farm types, depending on the nature and timing of pollutants received, and this may need to be reflected in design guidance. For instance, it is hypothesised that ponds could be less effective in retaining nutrients (particularly P) from livestock farm run-off compared with arable farms for a number of reasons, including:

- Much of the P will be associated with organic waste in particulate, colloidal and soluble form, and hence be less quick to sediment than inorganic soil aggregates.
- Inorganic P may be sorbed onto organic colloids and desorb to form soluble P when run-off is diluted strongly by water.
- Organic rich sediment accumulated in ponds may be highly susceptible to the formation of anaerobic conditions, leading to potential P release from any iron oxides present.
- Organic sediment will also be more prone to resuspension than inorganic sediment because of its lower density.

Guidance on farm steading ponds in Scotland is currently limited. SEPA’s interim guidance states that: ‘All livestock farms will have existing drainage systems in place to allow roof water and run-off from ‘clean’ roads and yards to discharge to local watercourses... it is this existing drainage that steading ponds are intended to address... Such an approach would also be consistent with what is currently defined as ‘good agricultural practice’ in The PEPFAA (Prevention of Environmental Pollution from Agricultural Activity) Code’. To provide demonstration sites and increase understanding of the factors affecting pond performance, the Scottish Executive and SEPA’s Diffuse Pollution and Habitat Enhancement Initiatives have funded the design and construction of four farm ponds throughout Scotland. This will be augmented by ongoing SEERAD-funded work at SAC and the Macaulay Institute evaluating best management practices (BMPs) on farms, supported by a new PhD project at The University of Edinburgh that aims to investigate the factors (including design) which influence the long-term performance of farm ponds and the relationships between pollutant removal efficiency, biodiversity enhancement and costs. This paper presents results from a SEERAD-funded project on the performance of a mature farm pond/wetland (constructed prior to current best practice guidance) and from a biodiversity survey of three existing farm pond/wetland systems. A discussion on plans for future work is also presented.

MATERIALS AND METHODS

Performance of an Established Onstream Farm Pond/Wetland

The pond studied was on Langside dairy farm (110 dairy cows plus followers) in the Cessnock catchment, Ayrshire. The main watercourse draining the farm steading and 221 ha of land passes through an abandoned 5000-m³ reservoir and associated wetland, with a total area of 0.63 ha. The system is ‘onstream’ and not designed in accordance with current best practice guidelines [as outlined, for instance, by Campbell et al. (2004)], but it provided the opportunity to assess the efficacy of an
established pond/wetland for nutrient removal and FIO mitigation. In summer 2005, water samples from the inlet and outlet of the pond were collected by ISCO 6712 samplers equipped with an Area Velocity Flow 750 module and WAVECOM Modem Control, serviced by a FLOWLINK data retrieval package. Composite water sampling (10 shots per bottle) was triggered by discharge at the inlet and by time (hourly) at the outlet because the stream cross-section below the outlet structure was too wide for flow-based sampling. Water samples were analysed for total N, total P and ammoniacal N (NH₄-N) using automated colorimetric methods and SS (mass of dried solids retained on filter paper from a known volume of sample). Although bottles from the water samples were collected on a weekly basis, comparison of fresh versus aged samples showed < 5% decline in ammoniacal N content. Discharge at the inlet was estimated by the area-velocity method from 5-minute measurements of water level and velocity. Outlet water level data showed that there was little hydraulic delay in the system (about 1 hour) and that the water residence time during storm events is about 1 day. From the water sample analysis results, hourly interpolated water quality data were calculated for the monitoring period using a double-parabolic interpolation function (XlXtrFun, Advanced Systems Design and Development, 2003). These were then combined with hourly discharge data from the inlet to provide a continuous estimate of loads.

Comparison of Biodiversity Between Existing Farm Pond/Wetland Systems

To provide a measure of the habitat value of farm ponds/wetlands, vegetation and macroinvertebrates were surveyed in three systems in Scotland in July 2005: (i) Langside farm, Ayrshire (described above); (ii) Black Loch wetland, Lunan Lochs, Angus – an established wetland of ~ 1.5 ha area and ~ 20,000 m³ volume (again constructed prior to best practice guidelines) that receives run-off from fields amended with poultry manure; (iii) a pond/wetland system designed in accordance with best practice guidelines and constructed in 2004 at Old Castle mixed arable farm, Berwickshire. Each system was arbitrarily divided into six compartments from inlet to outlet and within each compartment plant species were identified and counted in five randomly located 1-m x 1-m quadrats. Macroinvertebrates were sampled and identified to species level in each system using the National Pond Survey method (Biggs et al., 1998).

RESULTS

Performance of an Established Onstream Farm Pond/Wetland

Inlet and outlet total P and discharge for Langside pond are shown in Figure 1. Total P concentrations at the inlet rise with inlet discharge, peaking during storm events, while outlet total P concentrations often exceed inlet concentrations. When compared with total P water quality standards for Scottish freshwater lochs (SEPA, 2002), the outlet water quality would be classified as ‘hypertrophic’ (total P > 0.1 mg/L). To assess the performance of the pond/wetland in pollutant removal, mean hydraulic load, and total N, total P, NH₄-N, and SS concentrations at the inlet and outlet were used to calculate the overall mass removal efficiency (as % of inlet load) for five periods where continuous inlet–outlet data exist (Table 1). Removal of SS by the pond/wetland is good. The overall efficiency of removal of both N and P is poor, but
removal varied between periods from –84% to 20% and –183% to 46%, respectively. The area is therefore at times a source of nutrients, particularly during low flows. The removal of FIOs is good (e.g. 83% for Escherichia coli bacteria; Vinten, pers. comm.). A positive (but not significant) relationship was observed between % removal of SS, total P and total N and hydraulic load which was attributed to the entry of larger particles (with faster settling velocity and containing total P and N) into the pond at higher flows. Marginally significant relationships between total P (P = 0.088) and total N (P = 0.075) removal and SS removal suggest that designing a pond/wetland to maximise SS removal should also result in good total P and N removal. In order to elucidate P retention within farm ponds/wetlands, the fractionation of P needs to be explored in more detail, although this was not possible at Langside because of the field storage of water samples.

Figure 1: Inlet discharge and total P concentrations at the inlet and outlet to Langside pond/wetland, days 138-180 of 2005. Horizontal bars on total P values show the time period represented by each sample.
Comparison of Biodiversity between existing Farm Pond/Wetland Systems

Measures of the abundance, richness and diversity (the Shannon diversity index, commonly used to describe the value of a habitat) for vegetation and macroinvertebrates are shown in Table 2 for the three farm wetland/pond systems surveyed in July 2005. For vegetation, the highest species richness is found at Langside, the highest abundance and Shannon diversity at Old Castle, while Black Loch has the lowest abundance, richness and diversity. Although the Old Castle system is considerably younger than Langside, vegetation is clearly very well established. For macroinvertebrates, the highest richness and Shannon diversity occurs at Old Castle and Langside, while the lowest diversity is at Black Loch and the lowest abundance at Langside.

Table 2: Measures of vegetation and macroinvertebrate abundance, richness and diversity from sampling of three Scottish farm wetland/pond systems, July 2005

<table>
<thead>
<tr>
<th>Measure</th>
<th>Pond/wetland system</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Old Castle</td>
</tr>
<tr>
<td><strong>Vegetation</strong></td>
<td></td>
</tr>
<tr>
<td>Total abundance</td>
<td>22337</td>
</tr>
<tr>
<td>Density (individuals m^{-2})</td>
<td>745</td>
</tr>
<tr>
<td>Richness (No. of species)</td>
<td>17</td>
</tr>
<tr>
<td>Shannon diversity index</td>
<td>2.01</td>
</tr>
<tr>
<td><strong>Macroinvertebrates</strong></td>
<td></td>
</tr>
<tr>
<td>Total abundance</td>
<td>814</td>
</tr>
<tr>
<td>Richness (No. of families)</td>
<td>20</td>
</tr>
<tr>
<td>BMWP (Biological Monitoring Working Party) score</td>
<td>37</td>
</tr>
<tr>
<td>Shannon diversity index</td>
<td>2.30</td>
</tr>
<tr>
<td>ASPT (average score per taxon)</td>
<td>4.63</td>
</tr>
</tbody>
</table>

Most measures of abundance and diversity are similar at Old Castle and Langside (apart from macroinvertebrate abundance and the Shannon diversity index for vegetation which are both considerably lower at Langside) but the species...
compositions in the two systems are quite different. Apart from various species of water boatmen that occur in both systems, the most abundant macroinvertebrates at Old Castle were chironomids, midge larvae (Orthocladiinae (subfam.)) and common darter (dragonfly) (Sympetrum striolatum), compared with bivalves (Anodonta spp.), caddis larvae (Diceranota spp.), shrimps (Gammarus spp.) and flies (Leptocera humida) at Langside. The dominant plant species were floating sweet-grass (Glyceria fluitans) (28%) and Yorkshire fog (Holcus lanatus) (22%) at Old Castle, common reed (Phragmites australis) (55%) at Langside and water horsetail (Equisetum fluviatile) (49%) at Black Loch, where the figures in brackets are the relative abundances for the overall sample.

Black Loch appears impoverished for most indicators compared with the other two systems; its high total macroinvertebrate abundance is due to the dominance of water fleas (Bosmina sp.), accounting for 66% of individuals. Comparison of the ASPT values with SEPA's Scottish River Classification System indicates that water quality is ‘fair’ (Class B) in the Old Castle and Langside systems and ‘poor’ (Class C) in Black Loch.

**DISCUSSION**

Detailed flow and water quality monitoring at Langside suggest that mature onstream ponds/wetlands can be sources of nutrients as well as sinks at certain times of year. This variable efficiency of wetlands has been observed in other pond/wetland systems receiving agricultural run-off (Table 3).

**Table 3: Performance of other pond/wetland systems treating agricultural run-off/wastewater**

<table>
<thead>
<tr>
<th>Reference and study information</th>
<th>% mass removal: mean (min – max values)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Braskerud (2002): instream wetlands receiving arable and dairy run-off, Norway</td>
<td>21-44</td>
</tr>
<tr>
<td>Dunne et al. (2005b): constructed wetland treating dairy farmyard water, Ireland</td>
<td>(5–84)*</td>
</tr>
<tr>
<td>Fink and Mitsch (2004): restored wetland receiving arable and forest run-off, USA</td>
<td>41†</td>
</tr>
<tr>
<td>Geary and Moore (1999): pond-wetland system for dairy wastewater, Australia</td>
<td>26 (3.2–45)</td>
</tr>
<tr>
<td></td>
<td>49 (7–99)</td>
</tr>
<tr>
<td></td>
<td>(–15–45)</td>
</tr>
<tr>
<td>Reddy et al. (2001): experimental marsh-pond system for swine wastewater, USA</td>
<td>66–69</td>
</tr>
<tr>
<td></td>
<td>43–60‡</td>
</tr>
<tr>
<td></td>
<td>37–51</td>
</tr>
<tr>
<td></td>
<td>31–44</td>
</tr>
<tr>
<td>Thorén et al. (2004): constructed pond-marsh treating agricultural/urban run-off, Sweden</td>
<td>17 (6–36)§</td>
</tr>
</tbody>
</table>

*SRP: 5% removal in winter; 81–84% removal rest of year. †Nitrate N. ‡Most removed in warmer months. §Annual removal, but 40% of annual N removal exported February to March 2001.
Comparable SS removals have been reported in other systems, but the lower nutrient removal rates at Langside probably reflect a number of factors: the onstream nature of the system and short residence time (1 day compared with the 14–21 days preferred for offstream ponds); the often organic nature of the pollution; the presence of large amounts of sediment which releases nutrients as the water warms in spring; the residual effect of large winter loadings; and the likelihood that the vegetation is no longer acting as a sink. In terms of pond design, this means that offstream pond/wetland systems are preferable because, even if nutrient remobilisation occurs, input to the receiving watercourse would be small during low flow conditions.

The results emphasise the need to evaluate new ponds carefully and to establish maintenance protocols that ensure maximum retention capability is retained. Continuous flow and water quality monitoring is essential for assessing treatment efficiency as performance varies over short timescales (hours and days) and infrequent water quality analyses may miss rapid changes in concentration during storm events that can impact on receiving watercourses. Vegetation and macroinvertebrate survey results show that considerable variability in biodiversity and conservation value exists between systems, probably due to the interaction of many factors, including: system age, pollutant and hydraulic loading, habitat heterogeneity, microclimate, localisation within the landscape and proximity of other wetlands/ponds for colonisation. There is a need to investigate the relationships between faunal and floral diversity and water treatment efficiency, and therefore quality, in order to propose recommendations for design and maintenance that maximise both pollutant removal and habitat value in farm ponds/wetlands.

**FURTHER WORK**

The issues identified above will be investigated in systematic assessments of established and newly-constructed farm ponds/wetlands in Scotland during the next 2 years. The research will include continuous flow and water quality monitoring of systems of different ages, representing the main farm types in Scotland that present a risk of diffuse pollution from steading run-off. Sites will include ponds designed in accordance with current best practice and constructed with funding from the Scottish Executive and SEPA’s Diffuse Pollution and Habitat Enhancement Initiatives. Factors relating to pond maintenance requirements and costs (e.g. vegetation growth, sediment accumulation) and their habitat and surroundings will also be considered. Although these pond/wetland systems are highly complex, it is hoped that the key processes and influencing factors can be identified and modelled to provide answers to questions, including:

- What are the relationships between hydraulic and pollutant loadings and treatment performance?
- How does design of a pond/wetland (e.g. water depth, extent of margins, presence of a treatment train, treatment of first flush) affect treatment performance and habitat value?
- How compatible are water quality treatment and habitat functions of ponds, and do trade-offs need to be made?
- How does pond/wetland treatment performance and habitat value change over time?
Through addressing the above questions, the ultimate aim is to improve guidance on the design and maintenance of ponds and wetlands to reduce the contribution of farm steading run-off to diffuse pollution of Scottish waters.

ACKNOWLEDGEMENTS

We acknowledge the support of SEERAD in funding the Langside research, Colin Crawford (Scottish Agricultural College) for field support and the farmers at Old Castle, Langside and Black Loch for access to their land. Jill Lancaster (The University of Edinburgh) and SEPA advised on the macroinvertebrate surveys.

REFERENCES


COST-EFFECTIVE PROGRAMMES OF MEASURES:
THEORY VERSUS REALITY

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SUMMARY

The European Union Water Framework Directive (WFD) requires that river basin management authorities identify cost-effective (i.e. least-cost) combinations of best management practices and other measures – programmes of measures (POMs) – to achieve at least ‘good’ status of surface and ground waters in Member States. However, the cost-minimisation requirement does come with an allowance for accommodating unique cost-distribution and other constraints or objectives the implementing river basin management authorities may have. In conducting preliminary cost-effectiveness analysis of POMs for pilot catchments in Ireland’s Eastern River Basin District (ERBD), it has become evident that the estimated cost-effective POMs for catchments throughout the District may often come with unobtainable or highly disproportionate marginal costs among the three major sectors. In response to this reality, the ERBD project team is developing a tool that will allow ERBD management officials to essentially conduct their own constrained cost-effectiveness analysis, thereby allowing them to identify cost-effective POMs subject to their own unique cost-distribution/limitation constraints.

INTRODUCTION

In 2003, the Eastern River Basin District (ERBD) project team commenced with a biophysical characterisation of four hydrometric areas in 12 local authorities that comprise the ERBD in the Republic of Ireland. A socioeconomic characterisation was completed in 2004, and in 2005 the project team began distilling this information into a series of scientific and economic analyses. This included a cost-effectiveness analysis of ERBD water resources protection and remediation strategies – known as POMs. This paper (1) details the cost-effectiveness analysis requirements under the WFD, (2) explains the theory behind identifying cost-effective POMs, and (3) shares experiences from and developments in formulating and implementing piloted cost-effective POMs in the ERBD.

COST-EFFECTIVE POMS UNDER THE WFD

Annex III (b) of the WFD (European Union, 2000) stipulates that a cost-effectiveness analysis of water resource management measures be conducted as a prerequisite in formulating POMs.

‘The economic analysis shall contain enough information in sufficient detail (taking account of the costs associated with collection of relevant data) in order to make judgements about the most cost effective combination of measures in respect of water uses to be included in the programme of measures under Article 11 based on estimates of the potential costs of such measures.’
In general, the combinations of measures that comprise a programme of measures must be adequate to yield at least ‘good ecological status’, as defined in Annex V (1.2):

‘The values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate slightly from those normally associated with the surface water body type under undisturbed conditions.’

Quantitative evaluation of aquatic flora and fauna is required to determine ‘biological quality elements’. Numeric estimates of hydromorphological, chemical and physicochemical elements supporting these biological elements are required under Annex V (1.1).

Thus the WFD-required cost-effectiveness analysis entails identifying least-cost combinations of measures that, if undertaken over a pre-specified time period (typically from 2010 to 2015), will yield some pre-specified quantifiable water resource management objective. Given that the information and data generation resources necessary to determine current water body conditions are typically scarce in Member States, as are the resources to determine the effectiveness of various management measures in reducing the impacts of the various pressures causing these conditions, Annex II (1.5) of the WFD provides that:

‘Member States may utilise modelling techniques to assist in such an assessment…to optimise the design of…the programmes of measures required under Article 11.’

However, whereas the identification of cost-effective POMs is a requirement of the WFD, the implementation of the least-cost combinations of measures is not a rigid requirement. With regard to the establishments, economic sectors or subsectors, or jurisdictions that pay the costs of water services such as potable water supply, sewage treatment, or diffuse pollution mitigation, Article 9 of the WFD states that:

‘Member States shall ensure by 2010 an adequate contribution of the different water uses, disaggregated into at least industry, households, and agriculture, to the recovery of costs of water services, based on the economic analysis conducted according to Annex III and taking account of the polluter pays principle. Member States may in doing so have regard to the social, environmental and economic effects of the recovery…’

Further, explicit measures that have already been transcribed into State or jurisdictional administrative law from the 11 EU Directives enumerated in Part A of Annex VI of the WFD (termed ‘basic measures’) are ‘to be included within the programmes of measures’, regardless of their relative cost-effectiveness. Thus, technically, the WFD implicitly requires that a constrained cost-effectiveness analysis be undertaken as a prerequisite to the formulation of POMs, where constraints to the cost-minimisation might include:

1) full implementation of measures that are not comparatively cost effective; and

2) limitations on the proportions of total costs of water services each establishment, economic sector or subsector, or jurisdiction is required to contribute via implementation of measures.
THEORY IF IDENTIFYING COST-EFFECTIVE POMs

In theory, estimating ‘optimal’ programmes of measures is a simple proposition. In other words, deriving algorithms to yield least-cost combinations of measures implemented at various levels, subject to a series of constraint variables with pre-specified maximum or minimum relative or absolute values, is not a difficult exercise. In fact, the ERBD project team approached its pilot study to identify a constrained cost-effective programme of measures to achieve the anticipated water quality standard for phosphorous in the Athboy Catchment in the ERBD in this very way.

Barring hydromorphological and groundwater pressures and having determined that phosphorous is likely to be the sole surface water quality limiting pollutant in the Athboy Catchment, with

- data from simulations of phosphorous loadings to surface waters in the Catchment via the MIKE BASIN water quality modeling suite;
- an assessment of basic measures and their respective levels of implementation in the jurisdictions that overlap the Athboy Catchment; and
- some literature-based costs and effectiveness estimates of a relatively comprehensive list of phosphorous reduction measures;

an adequate collection of variables were on hand to estimate the ‘optimal’ programme of measures for the Athboy Catchment via application of relatively simple linear algebra. However, since

- the ERBD project team is not charged with the ultimate selection of programmes of measures to be implemented in the Athboy Catchment (and since the responsibility does lie with the lead local authorities); and
- ground-truthing the cost-effectiveness analysis inputs and communicating the outputs must be done in direct consultation with local authority officials in the Athboy Catchment (and since this action is to be taken as a follow-up to the initial cost-effectiveness analysis);

the ERBD project team considers it unrealistic to expect those who will make the final determinations of cost-effectiveness analysis variable inputs, including and especially the constraint variables for the cost-minimisation calculations, to simply work from a series of such algorithms.

Hence a user-friendly, web-based POMs selection feature with basic optimization functionality is being developed and integrated into the project’s decision support tool. With this feature, those who are charged with making these final determinations should be able to draw upon the most recent and precise information relevant to optimising their final preferred POMs to be implemented.

REALITY OF SELECTING AND IMPLEMENTING COST-EFFECTIVE POMs

Through its Athboy Catchment and other pilot applications, the ERBD project team has come to realise that identifying constrained cost-effective POMs and ultimately implementing them over a 5-year or longer period cannot take place in an analytical vacuum, especially considering the fact that the project’s analysts do not have the
final word on the POMs selection and long-term implementation. Table 1 contains the Athboy Catchment’s phosphorous loadings that were determined via modelling, summed across categories of pressures, and then compared against the loading consistent with the anticipated standard at average annual flow.

**Table 1: Estimated P loadings in the Athboy Catchment (Preliminary)**

<table>
<thead>
<tr>
<th>P Loads (kg/year)</th>
<th></th>
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</thead>
<tbody>
<tr>
<td>Manure fertilised land</td>
<td>2,186</td>
</tr>
<tr>
<td>Chemical fertilised land</td>
<td>2,255</td>
</tr>
<tr>
<td>Septic systems</td>
<td>35</td>
</tr>
<tr>
<td>WWTPs</td>
<td>811</td>
</tr>
<tr>
<td><strong>Total current</strong></td>
<td>5,288</td>
</tr>
<tr>
<td><strong>Good status</strong></td>
<td>2,299</td>
</tr>
<tr>
<td><strong>Target reduction</strong></td>
<td>2,989</td>
</tr>
</tbody>
</table>

Table 2 shows the literature-based costs and effectiveness estimates of analysed measures.

**Table 2: Cost-effective measures for phosphorous for Athboy Catchment (preliminary)**

<table>
<thead>
<tr>
<th>P Sources</th>
<th>Management measures</th>
<th>Effectiveness (kg/year P red)</th>
<th>Costs(€/year – 10 years)</th>
<th>Cost-Effective (€/kg P red)</th>
<th>CE Rank</th>
</tr>
</thead>
<tbody>
<tr>
<td>Manure fertilised land</td>
<td>Manure management plans</td>
<td>437</td>
<td>250,000</td>
<td>572</td>
<td>6</td>
</tr>
<tr>
<td></td>
<td>25% stocking reduction</td>
<td>546</td>
<td>22,800,000</td>
<td>41,758</td>
<td>15</td>
</tr>
<tr>
<td></td>
<td>Sheltered manure storage</td>
<td>218</td>
<td>150,000</td>
<td>688</td>
<td>7</td>
</tr>
<tr>
<td></td>
<td>1.5 km² riparian buffers</td>
<td>328</td>
<td>580,000</td>
<td>1,768</td>
<td>9</td>
</tr>
<tr>
<td></td>
<td>Feed optimisation plans</td>
<td>219</td>
<td>3,200</td>
<td>15</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>500 m³ retention ponds</td>
<td>164</td>
<td>60,000</td>
<td>366</td>
<td>5</td>
</tr>
<tr>
<td>Chemical fertilised land</td>
<td>Fertiliser management plans</td>
<td>564</td>
<td>82,500</td>
<td>146</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>50% grassland conversion</td>
<td>1,128</td>
<td>408,500</td>
<td>362</td>
<td>4</td>
</tr>
<tr>
<td></td>
<td>1.5 km² riparian buffers</td>
<td>338</td>
<td>580,000</td>
<td>1,715</td>
<td>8</td>
</tr>
<tr>
<td></td>
<td>500 m³ retention ponds</td>
<td>451</td>
<td>60,000</td>
<td>133</td>
<td>2</td>
</tr>
<tr>
<td>Septic systems</td>
<td>Inspections and upgrades</td>
<td>32</td>
<td>652,408</td>
<td>20,388</td>
<td>14</td>
</tr>
<tr>
<td></td>
<td>Treatment plant tie-ins</td>
<td>35</td>
<td>244,678</td>
<td>6,991</td>
<td>12</td>
</tr>
<tr>
<td></td>
<td>Education programme</td>
<td>9</td>
<td>75,000</td>
<td>8,333</td>
<td>13</td>
</tr>
<tr>
<td>WWTPs</td>
<td>MLE without filtration</td>
<td>561</td>
<td>2,649,000</td>
<td>4,722</td>
<td>10</td>
</tr>
<tr>
<td></td>
<td>MLE with filtration</td>
<td>686</td>
<td>3,788,000</td>
<td>5,522</td>
<td>11</td>
</tr>
</tbody>
</table>

Figure 1 is the transformation of this information into the web-based POM selection feature.
Figure 1: The ERBD project decision support system’s web-based POM selection feature (v.ß)

The POM selection feature is designed around the premise that local authority officials making final decisions regarding POMs will be primarily interested in (1) whether existing legal requirements will be met (hence the basic and supplemental measures categories); (2) what the cost-effective combination of measures is and how much this POM will cost (see x-marked measures for selections and note totals in the first view); and (3) how the marginal cost burden is distributed among the major sectors (8% agriculture and 92% domestic for the constrained cost-effective selection shown in the second view). Note the utility allows the user to identify the cost differential between the cost-effective POM (first view) and any constrained cost-effective POM (second view).
CONCLUSIONS

The ERBD project team is currently in the process of refining its estimates of phosphorous loadings and water resources protection and restoration measures’ effectiveness and costs in response to face-to-face consultations with the local authority engineers and procurement officers that manage the Athboy Catchment. This process that will be undertaken for each of the catchments in the ERBD throughout 2006. The project team will also continue to modify and improve the POM selection feature of its decision support system in consultation with the ultimate users, which are these same local authority officials. Although the cost-effectiveness analytical inputs (and thus outputs) and user interface will undoubtedly change as each of the catchments are analysed in further detail throughout 2006 and beyond, the ERBD project’s basic cost-effectiveness analysis methodology will not change.

Under the ERBD project’s cost-effectiveness analysis methodology, least-cost POMs for each catchment will be identified along with the variety of real-world constraints that must be applied when making final selections of POMs and proceeding to implementation. With this information, as it is to be incorporated into this highly flexible and dynamic process, WFD objectives for the maintenance and achievement of good or excellent status water bodies will be achieved pragmatically, transparently, inclusively, expeditiously, and with the highest possible level of fiscal responsibility.

ACKNOWLEDGEMENTS

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REFERENCES

ASSESSING THE COST-EFFECTIVENESS OF INTEGRATED MEASURES TO DECREASE LOSS OF NITRATE, PHOSPHORUS AND FAECAL INDICATOR ORGANISMS

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¹ADAS, Gleadthorpe, Meden Vale, Mansfield, Notts, NG20 9PF, UK, E-mail: mark.shepherd@adas.co.uk; ²IGER, North Wyke, Okehampton, Devon, EX20 SB, UK

SUMMARY

The timetable for implementation of the Water Framework Directive requires best available information to be synthesised now. Cost-effectiveness is an important consideration when deciding what actions should be taken to control diffuse pollution losses from agriculture. This paper presents preliminary results from a toolkit for assessing the cost-effectiveness of combinations of mitigation methods invoked by a range of policy options. It is a mix of simplified diffuse pollution models (to determine baseline losses of nitrate, phosphorus and faecal indicator organisms), best available information on cost-effectiveness drawn from other projects and, using these building blocks, a cost-curve approach. The approach relies on expert judgement.

INTRODUCTION

Implementation of the Water Framework Directive and addressing diffuse water pollution from agriculture means identifying and implementing practical on-farm methods for mitigating losses of pollutants from land to water. Agricultural sources of diffuse pollution include nutrients (nitrogen and phosphorus), agrochemicals (plant protection products, veterinary medicines and biocides), sediment and pathogens (faecal indicator organisms and FIOs). Generally, we have a good understanding of the mitigation methods available to farmers (e.g. Vinten et al., 2005); recent UK projects have listed these methods, which can number over 50 (RPA, 2005). They range in complexity from something as simple as using a fertiliser recommendation system through to more complex (and expensive) approaches, such as installing a constructed wetland.

Research shows that combinations of mitigation methods will be needed to reduce losses to acceptable limits (Shepherd and Chambers, 2006). The challenge is now to ‘encourage’ land managers to implement these mitigation methods. Measures (or ‘policy options’) to encourage uptake range from voluntary to regulatory (Table 1).

To inform the debate, an understanding of the cost-effectiveness of these policy options is also important. This paper puts forward a methodology for determining the cost-effectiveness of combinations of mitigation methods (which could be brought about by different policy options). This novel approach allows an assessment of combinations of measures and their effectiveness for controlling losses of nitrate, phosphorus and FIOs. Here, we present some preliminary results.
Table 1: A summary of the measures available for implementing changes in farm practices, with examples (based on Dampney et al., 2002)

<table>
<thead>
<tr>
<th>Measure</th>
<th>Examples</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Voluntary without external pressure</td>
<td>Advice to follow good agricultural practice</td>
</tr>
<tr>
<td>2. Voluntary with external pressure</td>
<td>Compliance with Farm Assurance Schemes</td>
</tr>
<tr>
<td>3. Voluntary but compensated</td>
<td>Environmental Stewardship schemes</td>
</tr>
<tr>
<td>4. Mandatory, not compensated</td>
<td>Nitrate Vulnerable Zone (NVZ) regulations</td>
</tr>
<tr>
<td>5. Mandatory, compensated</td>
<td>Compulsory purchase of land</td>
</tr>
</tbody>
</table>

MATERIALS AND METHODS

Impact of Mitigation Methods

Previous ‘Cost-Curve’ projects have separately identified mitigation methods and their likely effectiveness in controlling nitrate (Scholefield, 2005), phosphorus (Haygarth, 2003) and FIO (Haygarth, 2005) losses from agricultural land. From these projects, we identified a list of 44 methods with potential to decrease losses of at least one of the three chosen pollutants (Table 2). To move towards a quantitative assessment of the effectiveness of mitigation methods, two separate activities were undertaken. First, mitigation methods were defined in sufficient detail that users of the work would understand what was meant, for example, by ‘establish a cover crop before spring sown crops’. The cost of each mitigation method was then assessed. Second, the efficacy of each mitigation method in reducing losses of nitrate, phosphorus and FIOs was estimated. This was done using quantitative data collected from model runs during the series of previous Cost-Curve projects (as described above), and was supplemented by literature review data where methods had not previously been modelled. Method effects were expressed as absolute reductions in pollutant loss, and did not consider any interactions between methods. To prevent over-estimation of the effectiveness of multiple methods, the loss reductions were re-expressed as a percentage of the loss due to specific sources, namely external (fertilisers), internal (soil) and recycled (manure and excreta) sources. The net efficacy of multiple methods could then be calculated using a multiplicative model as:

\[
\text{Net Efficiency} = 1 - (1 - E_1) \times (1 - E_2) \times (1 - E_n)
\]

where En is the proportional efficacy of an individual method. The source apportionment draws upon a conceptual Cost-Cube model (Haygarth, 2005; Chadwick et al., 2006). However, as the approach does not explicitly represent the different modes of pollutant mobilisation and transport, it is still possible for the effectiveness of method combinations to be over-estimated.

Model Farms and Baseline Losses

Assessments were based on ‘typical’ farm systems. The model farm systems were defined to be representative of current practices and were characterised by an area
of arable or grassland, a number of livestock, and associated inorganic fertiliser and managed organic manure inputs (Table 3). Pollutant losses from each model farm were calculated for combinations of soil texture (clay loam or sandy loam) and climate conditions (net soil drainage 170–620 mm) to represent the range of baseline pollutant losses across England and Wales.

Table 2: An example of mapping of mitigation methods (not an exhaustive list presented here) against potential policy options for invoking change. A ticked box indicates that this policy option has the potential to implement this mitigation method. The extent to which the method is implemented was not considered.

<table>
<thead>
<tr>
<th>Nutrient Management Plan</th>
<th>Farm Assurance Scheme</th>
<th>NVZ Action Programme</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduce overall stocking rates on livestock farms</td>
<td></td>
<td>✓</td>
</tr>
<tr>
<td>Reduce dietary N and P intakes</td>
<td>✓</td>
<td></td>
</tr>
<tr>
<td>Adopt phase feeding of livestock</td>
<td>✓</td>
<td></td>
</tr>
<tr>
<td>Use a fertiliser recommendation system</td>
<td>✓ ✓ ✓</td>
<td></td>
</tr>
<tr>
<td>Integrate fertiliser and manure nutrient supply</td>
<td>✓ ✓ ✓</td>
<td></td>
</tr>
<tr>
<td>Do not apply P fertilisers to high P index soils</td>
<td>✓ ✓</td>
<td></td>
</tr>
<tr>
<td>Do not apply fertiliser to high-risk areas</td>
<td>✓ ✓ ✓</td>
<td></td>
</tr>
<tr>
<td>Avoid spreading fertiliser to fields at high-risk times</td>
<td>✓ ✓ ✓</td>
<td></td>
</tr>
<tr>
<td>Increase the capacity of farm manure stores</td>
<td>✓ ✓</td>
<td></td>
</tr>
<tr>
<td>Site manure heaps from watercourses and field drains</td>
<td></td>
<td>✓</td>
</tr>
<tr>
<td>Do not apply manure to high-risk areas</td>
<td>✓ ✓</td>
<td></td>
</tr>
<tr>
<td>Do not spread manure to fields at high-risk times</td>
<td>✓ ✓</td>
<td></td>
</tr>
<tr>
<td>Incorporate manure into the soil</td>
<td>✓</td>
<td></td>
</tr>
<tr>
<td>Transport manure to neighbouring farms</td>
<td></td>
<td>✓</td>
</tr>
<tr>
<td>Incinerate poultry litter</td>
<td></td>
<td>✓</td>
</tr>
</tbody>
</table>
Table 3: A summary of the representative farm types used to calculate costs and effectiveness of mitigation methods

<table>
<thead>
<tr>
<th>Farm system</th>
<th>Animal count</th>
<th>Excreta (t/year)</th>
<th>Managed manure (%)</th>
<th>Field area (ha)</th>
<th>Fertiliser (kg N/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Grass (dairy)</td>
<td>270</td>
<td>5040</td>
<td>60</td>
<td>150</td>
<td>190</td>
</tr>
<tr>
<td>Grass (suckler beef)</td>
<td>220</td>
<td>2288</td>
<td>60</td>
<td>100</td>
<td>60/100</td>
</tr>
<tr>
<td>Breeding Pigs (indoor)</td>
<td>1330</td>
<td>2125</td>
<td>100</td>
<td>71</td>
<td>145</td>
</tr>
<tr>
<td>Broilers</td>
<td>150000</td>
<td>2500</td>
<td>100</td>
<td>437</td>
<td>145</td>
</tr>
<tr>
<td>Arable</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>300</td>
<td>165</td>
</tr>
<tr>
<td>Arable plus manure</td>
<td>-</td>
<td>2700</td>
<td>100</td>
<td>300</td>
<td>145</td>
</tr>
</tbody>
</table>

The baseline pollutant losses were calculated using a suite of ‘tier-one’ diffuse pollution tools for N, P and FIOs, dealing with losses from soil, from fertiliser or from manure. These baseline data, combined with assessments of the cost and effectiveness of each mitigation method were used to calculate the likely cost benefit of combinations of mitigation methods that could be invoked by a policy option (described below).

**Policy Option Mix**

Different policy options that might be considered for use in encouraging uptake of combinations of mitigation methods can be tested. For each option mix, we are able to produce a mitigation method-policy option matrix and map out mitigation methods likely to be invoked by a policy option. Table 2 provides information on this, as an example.

To calculate the effectiveness of a policy option in reducing pollutant losses, we then developed a tool to calculate a Cost-Curve for the list of methods that were potentially applicable to a farm under a particular policy option. Using the examples in Table 2, a Nutrient Management Plan could potentially impact on seven mitigation methods, NVZ regulations on 11 mitigation methods. Other policy options (not presented here) invoked fewer or more mitigation options, depending on the nature of the option.

A Cost-Curve is defined as the relationship between emission abatement and marginal cost. The function is continuous and has a positive gradient, i.e. the marginal cost always increases with increasing emission reduction, thereby satisfying the law of diminishing returns. Cost-curve optimisation is a numerically intensive calculation that scales exponentially with the number of potential methods. The optimal Cost-Curve can be determined only by simulating all possible orders of method implementation, as the marginal cost is dependent on the methods already implemented. For this work, we adopted a pragmatic approach in which the tool iteratively selects and implements the method with the least cost-benefit ratio at each cost step. At each step, each method from the pool of currently unimplemented methods is implemented separately and the cost-benefit of implementation is calculated. The method with the least ratio of additional cost and emission reduction is implemented. Mutually exclusive methods that have not yet been implemented will not be considered on subsequent steps.
The Cost-Curve calculation tool optimises simultaneously on the percentage reduction in phosphorus, nitrate and FIO losses. If preferred, options allow for a weighting to be given to methods that are effective against a specific pollutant. This would enable optimisation against, for example, differential costs of clean up per unit of pollutant loss. Additionally, the tool can be used to optimise on only cost or benefit, and to limit either the maximum spend or number of methods implemented. The maximum spend limit can be used to restrict method implementation to advisory policy options that either save money or are relatively low cost.

RESULTS AND DISCUSSION

We show two examples of model farms: arable with manure and a dairy farm. Modelled baseline losses are shown in Table 4. Preliminary results for the assessment of cost-effectiveness of two policy options, as examples, are presented; a Nutrient Management Plan and a Farm Assurance Scheme. Table 2 shows which mitigation methods are invoked.

Table 4: Modelled baseline losses of nitrate-N and phosphorus (kg) and FIO (relative units) for two representative farm types and two soil textures. Annual average rainfall was assumed to be 850 mm

<table>
<thead>
<tr>
<th>Farm type</th>
<th>Sandy loam</th>
<th>Clay loam</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>N</td>
<td>P</td>
</tr>
<tr>
<td>Arable (+ manure)</td>
<td>Soil + dissolution</td>
<td>10898</td>
</tr>
<tr>
<td></td>
<td>Manure + excreta</td>
<td>1649</td>
</tr>
<tr>
<td></td>
<td>Mineral fertiliser</td>
<td>2267</td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td>14814</td>
</tr>
<tr>
<td>Average across farm (kg/ha):</td>
<td>49</td>
<td>&lt;0.1</td>
</tr>
<tr>
<td>Dairy</td>
<td>Soil + dissolution</td>
<td>3958</td>
</tr>
<tr>
<td></td>
<td>Manure + excreta</td>
<td>1926</td>
</tr>
<tr>
<td></td>
<td>Mineral fertiliser</td>
<td>1695</td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td>7579</td>
</tr>
<tr>
<td>Average across farm (kg/ha):</td>
<td>51</td>
<td>&lt;0.1</td>
</tr>
</tbody>
</table>

Baseline phosphorus (and FIO) losses are small (Table 4) from the sandy soils because the loss processes are dominated by surface pathways (Haygarth, 2004). Baseline FIO losses are small from the arable farm because manure is stored before application and this causes considerable pathogen die-off before spreading (Haygarth, 2005).
Figure 1: The estimated efficiency (% reduction against baseline) and cost (£'000/farm) of two example policy options applied to two representative farm types

The results from the cost-effectiveness assessment (Figure 1) show several points, with more scope for decreasing nitrate loss on the dairy farm than on the arable farm. The Nutrient Management Plan had no effect on FIOs; neither policy option affected FIOs on the arable farm. This is because control is influenced by storage practices and it was assumed that there was no storage on the arable farm: manure was imported and spread. Effectiveness in decreasing P loss was similar between farm types. The Farm Assurance Scheme was more effective than the Nutrient Management Plan, particularly on the dairy farm. This is because the Farm Assurance Scheme included measures that controlled timing of manure applications and storage, in our example. The costs to the farm business of implementing the two example options differ between farm types, with a net saving (negative cost) on the arable farm, but a cost of c. £7,000 to the dairy farm. This comes down to interpretation of which measures are invoked. We have assumed that both example policy options result in farms having to source alternative protein sources (‘dietary manipulation’) for the dairy herd, which bears a substantial cost. Without this, both policy options would be close to cost neutral on the dairy farms. Suggested mitigation methods would clearly benefit diffuse pollution losses, but have implications for the economic sustainability of individual farm businesses. A major benefit of the cost-effectiveness tool under development is that it allows a wide range of scenarios to be tested and compared.
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REFERENCES


ECONOMIC IMPLICATIONS OF MINIMISING DIFFUSE NITROGEN POLLUTION FROM LIVESTOCK MANURES

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SUMMARY

The economic implications of additional financial investment in extra slurry storage capacity and improved manure spreading equipment to reduce diffuse nitrogen pollution from livestock manure management was assessed for six ‘model’ livestock (two pig, three cattle and one broiler) farms. On the pig and dairy farms, the investment in additional slurry storage required to avoid the need to apply slurry in the autumn was between £3,500 and £5,880/year (amortised over 20 years), compared with an increased fertiliser N replacement value of the slurry (as a result of reduced leaching losses) of between £180 and £760/year. The cost of the bandspreading equipment was c. £3,250/year (amortised over 10 years) compared with the increased fertiliser N replacement of the slurry N (as a result of reduced ammonia losses) of between £230 and £330/year on the pig and dairy farms. On the broiler farm, reducing N leaching losses by changing the application timing from autumn to spring increased the fertiliser N replacement value of the broiler litter by £3,860/year, compared with the cost of the improved spreading equipment at £2,860/year (amortised over 10 years). Studies quantifying both nitrate leaching and ammonia volatilisation losses have clearly shown that an integrated approach to slurry N management is needed, so that management policies that aim to reduce nitrate leaching do not exacerbate other N loss pathways (so called ‘pollution swapping’).

INTRODUCTION

In the UK each year, approximately 90 million tonnes of livestock manures (47 million tonnes of slurry and 43 million tonnes of solid manure) supplying 450,000 tonnes of nitrogen (N) are applied to agricultural land (Williams \textit{et al.}, 2000). Efficient utilisation of manure nitrogen is essential to reduce diffuse air (e.g. ammonia) and water (e.g. nitrate) pollution, and to maximise crop N utilisation. The UK is committed to reducing ammonia emissions and nitrate leaching losses to comply with EU Directives. The current Nitrate Vulnerable Zone Action Programme (which covers 55%, 14% and 3% of agricultural land in England, Scotland and Wales, respectively) restricts the application of high available N manures (i.e. pig/cattle slurries and poultry manures) on sandy and shallow soils in the autumn/early winter period.

On many farms, changing manure application practices to reduce nitrate leaching losses will require significant financial investment in extra slurry storage capacity to avoid the need to apply slurry in the autumn. In addition, investment in slurry bandspreading equipment (e.g. trailing hose and trailing shoe machines; Plate 1) is likely to be required to apply slurry evenly to growing arable and grassland crops in spring/summer, without causing damage to soils and reducing crop quality. However,
the cost of these financial investments will be partly offset by increased manure fertiliser N replacement values (resulting from reduced nitrate and ammonia losses) and increased spreading opportunities for slurry throughout the cropping season.

Plate 1: Trailing hose slurry application to winter wheat (left) and trailing shoe slurry application on grassland (right)

This paper quantifies the economic implications of changing manure management practices to minimise nitrate leaching and ammonia volatilisation losses on model pig, dairy and broiler farms on contrasting soil types and under different climatic conditions.

MATERIALS AND METHODS

The impacts of changing manure application timings from autumn (August–October) to spring/summer (February–July) on nitrate leaching losses and ammonia emissions were quantified for six model livestock farms (Table 1). The amount of nitrogen applied to land was estimated based on standard livestock N production figures (Anon, 2000). For the dairy farms, it was assumed that 60% of the estimated N production was spread as slurry, with the remainder deposited during spring/summer grazing. For the pig and broiler farms, it was assumed that all of the estimated N production was returned to land via manure applications.

N Losses

Data from experiments carried out on a range of soil types (Chambers et al., 2000; Williams et al., 2005a), where nitrate leaching losses were measured after autumn surface broadcast applications of slurry or broiler litter, were used to estimate nitrate leaching losses as a percentage of the total N applied. Ammonia emissions were estimated using MANNER (Chambers et al., 1999) assuming a slurry dry matter content of 4% for pig slurry and 6% for cattle slurry. Slurry bandspreading was assumed to reduce ammonia losses by 30% compared with surface broadcasting (Chambers et al., 2001).

Storage Capacity and Spreading Equipment

The existing slurry storage capacity was assumed to be 3 months for the model dairy farms and 4 months for the model pig farms, which is typical of current commercial practice (Smith et al., 2000, 2001). On the dairy farms, it was assumed that slurry could be spread in spring from early February to late March and to silage aftermaths in the summer (i.e. an August to January no spread period). To avoid the need to
spread cattle slurry in the autumn period, an extra 3-months slurry storage (2,100 m³) was assumed to be needed on the dairy farms. On the pig farms which were in arable production, cropping patterns were assumed to limit slurry applications to a 3-month period between early February and late April (i.e. a May to January no spread period) as a result, it was assumed that the slurry storage capacity needed to increase from 4 to 9 months (1,000 to 2,250 m³).

Table 1: Description of model farms

<table>
<thead>
<tr>
<th>Farm type and cropping (location)</th>
<th>Soil texture</th>
<th>Average annual rainfall (mm)</th>
<th>Animal numbers</th>
<th>Quantity of manure spread (t)</th>
<th>Manure N spread (kg)</th>
<th>Nitrate leaching losses following autumn spreading (% total N applied)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Breeding pigs – arable (Cambs)</td>
<td>Clay</td>
<td>550</td>
<td>1,330 places</td>
<td>3,000</td>
<td>12,000</td>
<td>9*</td>
</tr>
<tr>
<td>Breeding pigs – arable (East Yorks)</td>
<td>Silty clay loam over chalk</td>
<td>750</td>
<td>1,330 places</td>
<td>3,000</td>
<td>12,000</td>
<td>19†</td>
</tr>
<tr>
<td>Dairy – grass (North Yorks)</td>
<td>Sandy loam</td>
<td>700</td>
<td>270 (150 dairy cows, 120 followers)</td>
<td>5,040 (6% dry matter)</td>
<td>15,120</td>
<td>11‡</td>
</tr>
<tr>
<td>Dairy – grass (Devon)</td>
<td>Sandy loam</td>
<td>1,100</td>
<td>270 (150 dairy cows, 120 followers)</td>
<td>5,040 (6% dry matter)</td>
<td>15,120</td>
<td>30‡</td>
</tr>
<tr>
<td>Dairy – grass (Shropshire)</td>
<td>Sandy loam</td>
<td>650</td>
<td>270 (150 dairy cows, 120 followers)</td>
<td>5,040 (6% dry matter)</td>
<td>15,120</td>
<td>17§</td>
</tr>
<tr>
<td>Broilers – arable (Notts)</td>
<td>Loamy sand</td>
<td>640</td>
<td>150,000 places</td>
<td>2,475</td>
<td>74,250</td>
<td>13¶</td>
</tr>
</tbody>
</table>

*4 site years between 1995/96 and 2000/01, †1 site year 2002/03, ‡4 site years 1990/91 to 1993/94, §3 site years 1991/92 to 1993/94, ¶3 sites years 1990/91 to 1992/93.

The financial costs of increasing slurry storage capacity were calculated using typical industry figures of £35/m³, with the capital investment costs amortised over 20 years at an interest rate of 5%, giving an annual repayment cost of £80 per £1,000 borrowed. For the broiler litter, it was assumed that the solid manure was stored in field heaps, so there would be no substantial extra capital investment required to store the manure from autumn through to spring.
The financial costs of purchasing improved spreading equipment (i.e. a tanker and trailing hose/shoe bandspreader for slurry applications, and a moving floor spreader with vertical rear beaters for broiler litter, to allow cattle/pig slurry and broiler litter to be applied evenly to growing crops in spring/summer with minimal crop damage) were taken from manufacturers’ list prices and amortised over 10 years, with an interest rate of 5% (annual repayment cost £130 per £1,000 borrowed).

The fertiliser N replacement value of the manure applications was assumed to be 40 p/kg N. The annual net costs (additional amortised capital cost – the increased fertiliser N value) of the improved manure management practices were calculated so that the costs of reducing nitrate-N and ammonia-N losses could be quantified on a £/kg basis.

RESULTS AND DISCUSSION

Pig Slurry

After autumn pig slurry applications (Table 2) on the Cambridgeshire clay soil farm, estimated nitrate leaching losses were 450 kg N (5,000 kg slurry N from 5 months slurry production x 9% of total N applied leaching loss factor) and 950 kg N (5,000 kg slurry N from 5 months slurry production x 19% of total N applied leaching loss factor) on the East Yorkshire shallow soil over chalk farm (see Table 1 for model farm base data). The greater leaching losses from the East Yorkshire farm reflect the higher over winter rainfall and more nitrate ‘leaky’ nature of the shallow soil over chalk compared with the clay soil. Ammonia losses from the surface broadcast applications were estimated at 1,920 kg (i.e. 12,000 kg slurry N x 16% of total N applied) on both farms, compared with 1,340 kg N after the trailing hose applications (30% reduction compared with surface broadcasting).

Table 2: Summary of N losses from model pig and cattle farms

<table>
<thead>
<tr>
<th>Farm (location)</th>
<th>Nitrate-N loss (kg/farm)</th>
<th>Ammonia-N loss (kg/farm)</th>
<th>Cost/kg nitrate-N abated (£/kg/year)</th>
<th>Cost/kg ammonia-N abated (£/kg/year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Autumn Surface spread Bandspread slurry</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Breeding pigs – arable (Cambs.)</td>
<td>450</td>
<td>1,920</td>
<td>1,340</td>
<td>14.09</td>
</tr>
<tr>
<td>Breeding pigs – arable (East Yorks)</td>
<td>950</td>
<td>1,920</td>
<td>1,340</td>
<td>6.46</td>
</tr>
<tr>
<td>Dairy – grass (North Yorks.)</td>
<td>690</td>
<td>2,720</td>
<td>1,900</td>
<td>12.35</td>
</tr>
<tr>
<td>Dairy – grass (Devon)</td>
<td>1,890</td>
<td>2,720</td>
<td>1,900</td>
<td>4.25</td>
</tr>
<tr>
<td>Dairy – grass (Shropshire)</td>
<td>1,070</td>
<td>2,720</td>
<td>1,900</td>
<td>7.85</td>
</tr>
</tbody>
</table>

The capital costs of increasing the pig slurry storage capacity from 4 to 9 months were £43,750 and the capital costs of purchasing a tanker and trailing hose machine
£25,000. The investment in trailing hose application equipment was assumed necessary to allow the pig slurry to be applied evenly to growing crops without causing soil compaction and crop damage. The annual amortised repayment cost was £3,500 for the extra slurry storage capacity (over 20 years) and £3,250 for the improved application equipment (over 10 years). Changing the slurry application timing from autumn to spring increased the slurry fertiliser N replacement value (by reducing N leaching losses) of the pig slurry by £180 on the Cambridgeshire farm and £380/year on the East Yorkshire farm. Also, the fertiliser saving by reducing ammonia emissions through the use of bandspraying equipment was equivalent to £230 per year on both farms. Hence, the net annual cost of changing the slurry application timing from autumn to spring was £6,340 on the Cambridgeshire farm and £6,140 on the East Yorkshire farm. The costs of abating nitrate loss were equivalent to £14.09 and £6.46/kg of nitrate-N saved on the Cambridgeshire and East Yorkshire farms, respectively. The cost of abating ammonia loss was £5.60/kg of ammonia-N saved (Table 2).

**Cattle Slurry**

After autumn cattle slurry applications, estimated nitrate leaching losses were 1,890 kg N (i.e. 6,300 kg slurry N x 30% of total N applied) on the Devon farm, 1,070 kg N (6,300 kg slurry N x 17% of total N applied) on the Shropshire farm and 690 kg N (6,300 kg slurry N x 11% of total N applied) on the North Yorkshire farm (Table 2). The differences in nitrate losses between the model farms largely reflected differences in rainfall volumes (Table 1) and to a lesser extent seasonal differences in grass growth and drainage volumes. Ammonia emissions from the surface broadcast applications were estimated at 2,720 kg (18% of N applied), compared with 1,900 kg from the bandspraying applications (30% reduction compared with surface broadcasting).

At all three farms, the capital investment costs of increasing slurry storage from 3 to 6 months (2,100m³ to 4,200m³) were £73,500 and the capital costs of purchasing a tanker and trailing shoe machine were £25,000. The investment in trailing shoe application equipment will increase the number of days slurry can be spread without causing soil compaction and sward damage, and will reduce sward contamination. The annual amortised repayment cost was £5,880 (over 20 years) for the increased storage capacity and £3,250 (over 10 years) for the improved spreading equipment. The value of the nitrogen saved (through reduced N leaching losses) by changing the application timing from autumn to spring was £760/year on the Devon farm, £430/year on the Shropshire Farm and £280/year on the North Yorkshire farm. The saving in ammonia loss from bandspraying slurry application on the three farms was £330/year. The net costs of changing application timing practices from autumn surface broadcast spreading to spring bandspraying were £8,040, £8,370 and £8,520/year on the Devon, Shropshire and North Yorkshire farms, respectively. The costs of abating nitrate loss were equivalent to £4.25, £7.85, £12.35/kg of nitrate-N saved on the Devon, Shropshire and North Yorkshire farms, respectively. The cost of abating ammonia loss was £3.96/kg of ammonia-N saved on all three farms.
INTEGRATED SLURRY N MANAGEMENT

If Only Life Was So Simple…

In our previous calculations, we assumed that there were no interactions between the changes in manure management practices and nitrate leaching or ammonia volatilisation losses. However, there is emerging strong evidence that changing slurry application timings from autumn to spring/summer can alter the balance of N losses via ammonia volatilisation. For example, Williams et al. (2005a) showed in measurements on the East Yorkshire model pig farm that the balance of N losses changed when pig slurry applications were moved from autumn to spring (Figure 1). After the March topdressed pig slurry application to a growing wheat crop (26% of total N applied), ammonia emissions were higher (P<0.05) than after the September application to cereal stubble (6% of total N applied), with emissions from the May timing intermediate between the other two (15% of total N applied). After the September application, the low ammonia emissions were due to rapid infiltration of the slurry into the dry cereal stubble compared with the slow infiltration of slurry into the soil after the March application. After the September application, nitrate leaching losses were equivalent to 19% of total N applied (Table 1), so that total N losses were similar from the autumn and spring application timing at 25% and 26% of total N applied, respectively. Also, Williams et al. (2005b) reported that ammonia losses after bandspread cattle slurry applications to first cut silage aftermaths in early June 2004 (summer) were c. threefold greater (P<0.05) than after autumn/spring application timings (Figure 2). The higher losses from the summer application reflected warmer temperatures and reduced slurry infiltration rates into the dry soil, in contrast to the cooler temperatures and moist soil conditions in autumn/spring.

Figure 1: Balance of ammonia volatilisation and nitrate leaching losses after pig slurry applications to winter cereals (Williams et al., 2005a)
Studies on a drained clay soil at Brimstone Farm in Oxfordshire (Williams et al., 2006) showed that when cattle slurry applications timings were moved from autumn to spring to reduce nitrate leaching losses, elevated ammonium-N concentrations of up to 3.9 mg/L NH$_4$-N (c. fivefold greater than the EC Freshwater Fish Directive limit of 0.78 mg/l NH$_4$-N) were measured in the spring drainage water flows. This was because rainfall after the spring slurry application caused contaminated water to move rapidly from the ‘wet’ soil surface to pipe drains, via the network of soil macropores and mole drains in the heavy clay soil.

![Figure 2: Ammonia emissions after cattle slurry applications to grassland (Williams et al., 2005b)](image)

These data indicate that there is a need to recognise that slurry management practices that aim to reduce nitrate leaching losses by moving from autumn to spring/summer application timings, may exacerbate ammonia volatilisation losses under warmer and drier soil conditions, and increase the risks of other nutrient pollution of drainage waters from clay soils.

**Broiler Litter**

Assuming that all of the broiler litter was previously applied in the autumn, nitrate leaching losses after surface broadcast applications on the sandy soil Nottinghamshire farm were estimated at 9,650 kg N (equivalent to 13 % of total N applied). Changing the application timing from autumn to spring increased the fertiliser N replacement value of the broiler litter by £3,860, as a result of reduced nitrate leaching losses. The capital cost of purchasing a spreader that was capable of applying broiler litter evenly and at agronomically required rates was £22,000, giving an annual amortised repayment cost of £2,860 (over 10 years). Overall, the net benefit to the farm was £1,000/year. However, the practical problems of topdressing broiler litter to growing crops in spring (i.e. soil traffiability problems, non-compatibility of machine spread widths with tramline spacings and potential odour/fgy nuisance) is likely to make such an application policy impractical on most farms. On suitable soil types, such as the sandy soils on the Nottinghamshire model farm, the rapid soil incorporation of broiler litter before the establishment of spring sown crops would reduce
odour/fly nuisance and further increase the fertiliser N value of the broiler litter by minimising ammonia volatilisation losses. MANNER estimated that ploughing broiler litter into the soil within 24 hours of application would reduce ammonia emissions by c. 85% compared with leaving the manure on the soil surface. This increased the potential fertiliser N replacement value of the broiler litter by a further £3,550/year. After autumn application, rapid soil incorporation would have little impact on the fertiliser N replacement value of the broiler litter because any of the saved N would subsequently be lost by overwinter nitrate leaching from the ‘leaky’ sandy soil (Chambers et al., 2001).

CONCLUSIONS

Considerable capital investment (between £3,500 and £5,880/year over 20 years) in extra slurry storage capacity was required to change slurry application timings from autumn to spring on the model pig and dairy farms. The increased fertiliser N replacement of the slurry as a result of the reduced nitrate leaching losses was modest at between £180 and £380/year on the pig farms, and £280 and £760/year on the dairy farms. The amortised (over 10 years) repayment cost of the bandspreading equipment was c. £3,250/year on both the pig and dairy farms, compared with an annual benefit in increased fertiliser N replacement value as a result of reduced ammonia losses of £230/year on the pig farms and £330/year on the dairy farms. The net costs of reducing nitrate leaching losses through the bandspreading of slurry in spring compared with autumn surface broadcast applications ranged from £6.46 to £14.09/kg of nitrate-N abated on the pig farms, and £4.25 to £12.35/kg of nitrate-N abated on the dairy farms. The ammonia abatement costs were £5.60/kg of ammonia-N saved on the pig farms and £3.96/kg of ammonia-N saved on the dairy farms.

On the broiler farm, reducing nitrate leaching losses by changing the manure application timing from autumn to spring increased the fertiliser N value of the manure by £3,860/year. Moreover, the increased fertiliser N value of spring applied broiler litter was around £1,000 greater than the annual amortised repayment costs of a solid manure spreader capable of applying the broiler litter evenly to growing crops in spring, although this policy would not be a practical option in most cases. Rapid soil incorporation of the spring applied broiler litter reduced ammonia volatilisation losses and increased the fertiliser N replacement value by a further £3,550 compared with leaving the broiler litter on the soil surface.

Recent research has shown that changing slurry application timings from autumn to spring/summer is likely to affect other N loss pathways (so called ‘pollution swapping’). This highlights the need for an integrated approach to manure N management, so that management practices which aim to reduce nitrate leaching losses (i.e. moving from autumn application timings to spring/summer) do not exacerbate other N loss pathways, either through ammonia losses to air or ammonium-N and other nutrient losses in drainage waters from clay soils.

ACKNOWLEDGEMENT

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REFERENCES


A MEASURE-CENTRIC APPROACH TO DIFFUSE POLLUTION MODELLING AND COST-CURVE ANALYSIS OF MITIGATION MEASURES

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SUMMARY

A new model framework has been developed that explicitly represents the mode of action of individual mitigation measures for reducing diffuse water pollution. This measure centric framework is termed the ‘Cost-Cube’. The model represents the functional behaviour of pollutants and the associated processes and pathways that can be affected by remediation measures. It is based on an export coefficient approach but extended to provide explicit fractions of the total pollutant loss by unique aspects of the source, mobilisation and transport dimensions. This framework was used to explore mitigation options for ammonium, nitrite, biological oxygen demand, and pathogens in terms of their economics and applicability for a number of model farm systems. A cost benefit analysis was used to prioritise measures that work within the bounds of current agricultural practice.

INTRODUCTION

Protection of water quality requires implementation of land management options that reduce the totality of contributors to diffuse pollution including transfers of nutrients, sediment, organic matter, agrochemicals and potentially pathogenic organisms. To do this effectively, we require an understanding of the key behavioural characteristics of diffuse pollutant groups and the potential for targeting of measures. We have developed a new conceptual modelling framework that represents the mode of action of mitigation measures for reducing diffuse pollution, and we have coupled this to a cost-curve analysis. In this paper, we present the parameterisation and application of this ‘measure-centric’ framework, as well as the ultimate deliverable, a cost-curve analysis of mitigation measures.

MATERIALS AND METHODS

Model Framework

A new conceptual modelling framework was developed to explicitly represent the mode of action of individual measures. This measure centric framework was centred on the ‘Cost-Cube’, an enhanced export coefficient model with three dimensions of pollutant source, mobilisation and transport (see, for example, RPA, 2003). The Cost-Cube explicitly represents the proportions of pollutant loss due each type of source, process of mobilisation and path of transport (Figure 1). These are referred to as the aspects of each dimension. For example, the mobilisation dimension is separated into the processes of solubilisation, detachment and contingent losses.
Haygarth and Jarvis, 1999). The volume of the cost-cube is proportional to the total pollutant loss. If each combination of dimension and aspect is equally important in contributing to the observed pollutant loss, then the Cost-Cube is made up of 27 equally sized cubes.

Figure 1: Dimensions and aspects of the Cost-Cube model

A control measure that reduces pollutant losses can be visualised as targeting one or more of the cubes, reducing their volume and hence the magnitude of total pollutant loss, i.e. the sum of all the cube volumes. For example, a measure that affects animal diet to reduce the quantity of pollutant excreted will reduce the volume of all cubes on the recycled-source aspect dimension. A measure that involved careful spreading of animal manure to minimise losses by surface run-off will reduce the volume of all cubes on the recycled-source and surface-transport aspect dimensions. A farm system can be represented by one or more Cost-Cubes, making explicit the total pollutant loss and the relative importance of the pollutant pathways.

Data in the literature were used to characterise control measures by their costs and percentage efficiencies. Each measure was assigned a target area on the Cost-Cube, using the dimension and aspect co-ordinate system, explicitly representing their mode of action in the measure centric framework. This information was then used by a mathematical sub-model to calculate the best order of implementation of the measures with the objective of achieving maximum pollution reduction for minimum cost. Such a calculation of the ‘cost-curve’ assumes that measure effects are multiplicative.

Parameterisation

The Cost-Cube model can be parameterised using the output from mechanistic models or by summarising empirical data and expert opinion. In our approach, experts were asked, for example, to estimate the proportion of the total pollutant loss due to detachment or solubilisation processes of mobilisation. Careful and iterative questioning, together with group discussions and review, allowed us to define expert expectations of the relative importance of each aspect of the pollutant source,
mobilisation and transport dimensions of the Cost-Cube model. This can potentially become an involved process when investigating many combinations of environmental conditions. Therefore, a simple mathematical sub-model of the pollutant vectors (the mobilisation or transport mechanisms, such as over land flow and soil erosion) was also used to interpolate data from a reference environment condition to other site situations. For example, the quantity of pollutant lost in overland flow due to soil detachment was scaled in proportion to the ratio of the modelled quantity of eroded soil at the reference and target locations.

<table>
<thead>
<tr>
<th>Cube Name:</th>
<th>Cube Description:</th>
<th>Cube Export Coefficient:</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ammonium</td>
<td>Ammonium Loss at [Heavy : Arable : Dry : Drained]</td>
<td>1.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>TRUE</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Cost-Cube Dimensions / Aspects (Relative Proportions)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>SOURCE</strong> Internal</td>
</tr>
<tr>
<td>---------------------</td>
</tr>
<tr>
<td>0.10</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><strong>MOBILISATION</strong> Internal</th>
<th>External</th>
<th>Recycled</th>
</tr>
</thead>
<tbody>
<tr>
<td>Detachment</td>
<td>0.05</td>
<td>0.00</td>
</tr>
<tr>
<td>Solubilisation</td>
<td>0.95</td>
<td>0.10</td>
</tr>
<tr>
<td>Contingent</td>
<td>0.00</td>
<td>0.90</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><strong>PATHWAY</strong> Detachment</th>
<th>Solubilisation</th>
<th>Contingent</th>
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</thead>
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<tr>
<td>Surface</td>
<td>0.23</td>
<td>0.20</td>
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<tr>
<td>Preferential</td>
<td>0.77</td>
<td>0.75</td>
</tr>
<tr>
<td>Through</td>
<td>0.00</td>
<td>0.05</td>
</tr>
</tbody>
</table>

Figure 2: Expert assessment of the contribution of each Cost-Cube dimension and aspect to the total loss of ammonium under the reference environment condition indicated under Cube Description

For this study, the reference environment condition was defined as a drained clay loam soil, under arable cultivation, located in the east of England. The field was characterised as having a gentle slope (1\(^\circ\)) and subjected to an average annual rainfall of 650 mm and potential evapotranspiration of 580 mm. The vector model calculated that there would be 12 mm of surface run-off (at the field scale), 69 mm of preferential flow and 122 mm of through flow annually (total drainage equals 204 mm). This was associated with an average total sediment loss of 220 kg annually. The reference pollutant source was defined by inputs of a spring application of 150 kg/ha of ammonium nitrate fertiliser and 50 t/ha of stored cattle slurry. The slurry was described as having a dry matter content of 6% and an available nitrogen content (after volatilisation losses) of 50 kg of nitrogen.

Figure 2 shows the results of the expert parameterisation for this reference for ammonium loss. A similar parameterisation was also carried out for BOD, pathogens and nitrite. For each pollutant, the principal sources were fertilisers (external aspect) and manure (recycled aspect), and the principal mechanisms of mobilisation were solubilisation and contingent or incidental losses. The dominant transport pathway was preferential flow in the cracks and tile drainage.
Application

Representative farm systems were used for application of the Cost-Cube framework. For each farm system, the total livestock numbers, crop areas and manure production were defined (Table 1). These farm type definitions were based on those used in a previous project (Chambers, 2002). The vector model was then used to calculate total pollutant losses for different combinations of climate, soil drainage and soil texture acting upon each farm system. These combinations were chosen to represent the range of environment conditions in England and Wales, including sandy loam and clay loam soils, and low (650 mm) and high (1100 mm) annual rainfall locations.

For each farm system, a list of applicable pollutant control measures was drawn from a review of potential mitigation measures, and the annual cost of implementation calculated in proportion to land area, livestock numbers and quantity of managed manure. Whole farm costs for the dairy system were calculated as being as low as £500 per year for measures that reduced pollutant losses by activities that can be equated with good agricultural practice. For example, the avoidance of slurry spreading on hydrologically well-connected field margins, or avoiding grazing high-risk fields on wet days. However, the field evidence for the efficiency of these measures in reducing pollutant mobilisation and delivery is variable and confidence in their effect is low. In contrast, there are other measures which if implemented are known to be guaranteed to reduce potential pollutant inputs but which are also very costly. For example, the reduction of dietary nitrogen intake or the removal of mineral fertiliser from the system costs £10,000 and £50,000 per year, respectively.

For the arable system, the costs of mobilisation and delivery control measures were generally calculated to be higher due to the greater land area, and the greatest costs were associated with either the removal or change in type of mineral fertiliser applications.

Table 1: Summary of the attributes allocated to each Representative Farm System in the modelling process

<table>
<thead>
<tr>
<th>Farm System</th>
<th>Animal count</th>
<th>Excreta (t/year)</th>
<th>Managed manure (%)</th>
<th>Field area (ha)</th>
<th>Fertiliser (kg N/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dairy (grass)</td>
<td>270</td>
<td>5,040</td>
<td>60</td>
<td>150</td>
<td>190</td>
</tr>
<tr>
<td>Breeding pigs (indoor)</td>
<td>1,330</td>
<td>2,125</td>
<td>100</td>
<td>70.5</td>
<td>145</td>
</tr>
<tr>
<td>Broilers</td>
<td>150,000</td>
<td>2,500</td>
<td>100</td>
<td>436.7</td>
<td>145</td>
</tr>
<tr>
<td>Arable</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>300</td>
<td>165</td>
</tr>
</tbody>
</table>

For the broiler system, with the exception of feed antibiotics, the greatest costs were taken to be associated with the changes in management of the large arable land area required for the spreading of the litter. However, it was also considered that there was the potential to save £42,700 per year by the proper accounting of the nutrient value of the litter and reducing the mineral fertiliser applications. Such a saving could potentially off-set some of the lesser measure costs, such as improved manure application timing, and thereby produce a net environmental and economic benefit. A similar situation was taken to exist for the indoor breeding pig system. Due to manure being managed as slurry rather than solids, it was considered that there
was a potentially valuable option involving the aeration of slurry to provide medium (good) efficiency for ammonium, nitrite and BOD and also low (poor) efficiency for pathogens. However, it should be noted that some of the slurry management options (such as slurry injection) that might be implemented to reduce gaseous ammonia losses have also been shown to have negative effects on the loss of all pollutants. This would need to be taken account in any cost-curve analyses that considered a broader range of pollutants.

RESULTS

The cost-curve sub-model was used to identify the optimal suite of measures for each representative farm system. Table 2 provides a sample set of results for the control of ammonium losses from the dairy system. Table 2 shows the generic form of a cost-curve, with increasing ratio of cost to benefit as more measures are implemented on the farm. In this example, money is initially saved by taking advantage of the nutrient content of manure to reduce fertiliser applications. Measures controlling the timing and placement of manure are next, with a modest but cheap reduction in pollutant loss. The most expensive measures, such as reducing stock count (and hence farm revenue), are implemented last. They apparently have minimal effect in reducing pollutant loss because the preceding measures have already significantly reduced losses.

Table 3 provides a summary of the overall results of the cost-curve calculations and key measure-centric options for diffuse pollution mitigation. Table 3 highlights for each Representative Farm System the abatement measures that were most frequently implemented first (in the top five) in the ordered cost-curve. The measures implemented first were those with greatest ratio of benefit to cost, i.e. they were the most cost effective. They are characterised by relatively low cost and common sense solutions that work within the bounds of current agricultural practice. For example, the integration of manures with fertilisers when planning nutrient applications and avoiding spreading fertiliser at times of high risk. The measures that were least attractive (implemented last) were those that have a fundamental impact on the income and raison d’être of the farm system, for example, reduction of stocking rates or taking land out of production.
Table 2: Modelled cost-curve for the control of ammonium pollution from the model dairy system (Model 1) averaged across each of the climate, drainage and soil texture Cost-Cube scenarios

<table>
<thead>
<tr>
<th>Cost-step</th>
<th>Measure description</th>
<th>Pollutant loss (%)</th>
<th>Annual Cost (£)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>Baseline</td>
<td>100</td>
<td>0</td>
</tr>
<tr>
<td>1</td>
<td>Integrate fertiliser with manure</td>
<td>77</td>
<td>-8,467</td>
</tr>
<tr>
<td>2</td>
<td>Introduce clover to grassland system</td>
<td>72</td>
<td>-10,782</td>
</tr>
<tr>
<td>3</td>
<td>Do not apply slurry to well connected areas</td>
<td>62</td>
<td>-10,499</td>
</tr>
<tr>
<td>4</td>
<td>Avoid grazing high-risk fields when wet</td>
<td>58</td>
<td>-10,349</td>
</tr>
<tr>
<td>5</td>
<td>Establish artificial wetland</td>
<td>44</td>
<td>-8,515</td>
</tr>
<tr>
<td>6</td>
<td>Change fertiliser type</td>
<td>34</td>
<td>-6,622</td>
</tr>
<tr>
<td>7</td>
<td>Allow drainage to deteriorate</td>
<td>30</td>
<td>-5,122</td>
</tr>
<tr>
<td>8</td>
<td>Avoid slurry spreading at times of high risk</td>
<td>25</td>
<td>-1,816</td>
</tr>
<tr>
<td>9</td>
<td>Reduce dietary nitrogen intake</td>
<td>17</td>
<td>9,014</td>
</tr>
<tr>
<td>10</td>
<td>Do not apply fertiliser to well-connected areas</td>
<td>16</td>
<td>10,949</td>
</tr>
<tr>
<td>11</td>
<td>Export 50% of slurry</td>
<td>14</td>
<td>16,593</td>
</tr>
<tr>
<td>12</td>
<td>Aeration of slurry</td>
<td>12</td>
<td>22,994</td>
</tr>
<tr>
<td>13</td>
<td>Install hedges and reduce field size</td>
<td>11</td>
<td>33,288</td>
</tr>
<tr>
<td>14</td>
<td>Establish a riparian strip</td>
<td>10</td>
<td>38,054</td>
</tr>
<tr>
<td>15</td>
<td>Reduce stock count</td>
<td>8</td>
<td>88,663</td>
</tr>
<tr>
<td>16</td>
<td>Avoid fertiliser spreading at times of high risk</td>
<td>8</td>
<td>103,663</td>
</tr>
<tr>
<td>17</td>
<td>Reduce field stock rates when wet</td>
<td>8</td>
<td>111,983</td>
</tr>
<tr>
<td>18</td>
<td>Batch storage of slurry</td>
<td>8</td>
<td>132,093</td>
</tr>
<tr>
<td>19</td>
<td>Use slowly available nitrogen fertiliser</td>
<td>8</td>
<td>187,098</td>
</tr>
</tbody>
</table>

DISCUSSION

The Cost-Cube framework is a highly flexible and valuable tool for ranking potential control measures in the development of catchment sensitive management plans. The explicit representation of the mode of action of measures, and the potential to use both empirical data and expert opinion, gives it potential value as a decision tool for reaching a consensus about pollution source and mitigation in stakeholder discussions. The use of this approach has also highlighted the need for further field investigations to refine the efficiency values of the ‘top five’ measures, as these are considerably less precise than their financial costs and especially sensitive to location. Further details of the effects of artificial wetlands and management of the hydrological connectivity of drained cracking clay soils would be clear priorities in this regard. This modelling approach also has the potential to be linked to a Geographic Information System (GIS) to provide national coverage and could also be used to optimise for mitigation of multiple pollutants simultaneously.

Further development of such cost benefit analyses should also take account of the potential externalities of pollution control. For example, there would be additional benefits to reducing nitrogen and phosphorus fertiliser use in terms of reducing greenhouse gas emissions through a reduction in energy required to manufacture...
inorganic fertiliser. Or there would be the potential of a more bio-diverse landscape as a result of reduced nutrient input and reduced animal stocking rate with an associated increase in visitor numbers making use of the cleaner environment for recreation. The costs of such additional benefits should be included in the final cost-benefit analyses used to select measures to reduce diffuse water pollution to agriculture.

**Table 3:** List of abatement measures that were most frequently in the first five to be implemented in the calculation of each representative farm system cost-curve. These are drawn from across all the soil, drainage and climatic conditions used in the calculations for all pollutants

<table>
<thead>
<tr>
<th>Dairy system</th>
<th>Arable system</th>
<th>Broiler system</th>
<th>Pig system</th>
</tr>
</thead>
<tbody>
<tr>
<td>Establish artificial wetland</td>
<td>Establish artificial wetland</td>
<td>Batch store litter</td>
<td>Export 50% of manures to other farms</td>
</tr>
<tr>
<td>Do not apply slurry/manure to well-connected hydrological areas</td>
<td>Allow drainage to deteriorate</td>
<td>Do not apply slurry/manure to well-connected hydrological areas</td>
<td>Establish artificial wetland</td>
</tr>
<tr>
<td>Allow drainage to deteriorate</td>
<td>Change fertiliser type</td>
<td>Avoid manure spreading at times of high risk</td>
<td>Do not apply slurry/manure to well-connected hydrological areas</td>
</tr>
<tr>
<td>Integrate fertiliser with manure</td>
<td>Cultivate land to reduce hydrological connectivity</td>
<td>Export 50% of manures to other farms</td>
<td>Integrate fertiliser with manure</td>
</tr>
<tr>
<td>Introduce clover to grassland system</td>
<td>Do not apply fertiliser to well-connected hydrological areas</td>
<td>Integrate fertiliser with manure</td>
<td>Change fertiliser type</td>
</tr>
</tbody>
</table>

**ACKNOWLEDGEMENTS**

We acknowledge the financial support provided by Defra.

**REFERENCES**


ASSESSING THE SIGNIFICANCE OF DIFFUSE POLLUTION RISKS IN ORDER TO TARGET AND PRIORITISE BEST MANAGEMENT PRACTICES

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SUMMARY

A targeted approach that focuses on sensitive catchments, such as Nitrate Vulnerable Zones and Bathing Water catchments, is described. Farm audits that consider diffuse pollution issues as a three-stage process, and a treatment train approach to BMP implementation at each stage in the process are recommended. A simple conceptual framework for evaluating pollution control measures is set out and an example of cost-effectiveness and cost benefit analysis of a buffer strip to reduce total P loading to Loch Leven is given. Setting up a database giving information for each farm type, on source strength per pollutant, potential mitigation measures, cost-effectiveness, loadings mitigated and time scale for efficacy is recommended. A policy matrix, i.e. a summary of recommended measures, prioritised and categorised into proposed appropriate levels of regulation, for mapping of on-farm measures against policy instruments, voluntary measures and potential General Binding Rules (GBRs), etc. is also proposed.

INTRODUCTION

Agriculture is the predominant land use in Scotland, and it has been identified as presenting a significant threat to achieving good water status. Forestry is also a major land use, and it may also place certain water bodies at risk. The consultation paper ‘Diffuse Water Pollution from Rural Land Use’ (Scottish Executive, 2005a) outlines the current strategy for the control of diffuse pollution. A tiered system of controls is proposed, developed along similar lines to the Water Environment (Controlled Activities) Regulations 2005 http://www.scotland.gov.uk/Consultations/Current/Q/rowId/-1#conRow0National. General Binding Rules (GBRs) provide national general authorisation for those farming activities posing a threat to water quality and will be based on basic levels of good practice. While this is expected to bring about improvements, the paper indicates that there would still be problems associated with certain pollutants in some catchments. For these areas the Executive envisages that the national Rules would be supplemented by a set of targeted GBRs to apply to particular land use activities. At the same time, through Land Management Contracts involving more targeted supportive measures, the Executive is expected to make available incentives to promote good practice aimed at achieving good water quality in Scotland’s rivers, lochs and coastal waters. In some cases change is already taking place to bring about improvements in the protection of the water environment through good practice, developed and promoted via the PEPFAA Code (Scottish Executive, 2005b) and The 4 Point Plan (Scottish Executive, 2004), and by regulation
within sensitive catchments. The aim of this paper is to review recent work assessing the significance of diffuse pollution risks, and to propose a framework for carrying out farm-scale audits and evaluating and selecting pollution control measures and identifying opportunities for knowledge transfer.

**Assessing the Significance of Diffuse Pollution Risks**

A detailed characterisation assessment of pressures and impacts on the water environment has been carried out for both the Scotland and Solway-Tweed River Basin Districts (SEPA, 2005a). Diffuse pollution placed up to 45% of the water bodies in Scotland at risk of not meeting the Water Framework Directive (WFD)'s environmental objectives. Agriculture was clearly shown to be the major pressure although not invariably the most severe. SEPA will use these results to prioritise both environmental monitoring and those water bodies where management action is required. Further characterisation to refine the risk assessments is ongoing and will be published by SEPA in the Significant Water Management Issues Report, as required by WFD, in mid-2007 (Morris et al., 2006).

A screening tool for diffuse pollutants has been developed (Anthony et al., 2005) to supplement the risk assessment described above. This has allowed sources of pollutants from agriculture, forestry, urban run-off, roads, septic tanks and sewage discharges to be estimated. The methods used were generally based upon existing indicators of relative pollution risk that are appropriate for application at the regional and national scale, rather than detailed mechanistic modelling. The models also developed approaches that had previously been applied in the UK, including elements of the NIRAMS (Nitrogen Risk Assessment Model for Scotland) model of nitrate leaching and the PSYCHIC model (Phosphorus and Sediment Yield Characterisation in Catchments) (Withers et al., 2006). Phosphorus losses resulted in the greatest land area designated at risk and were primarily associated with agriculture. The report concluded ‘the screening tool outputs can be used to target additional monitoring in areas at high risk’ and ‘ultimately the development of targeted and effective catchment management plans to meet the needs of the WFD’.

With the initial diffuse pollution characterisation and impacts analysis now complete, and the monitoring programmes about to be implemented, attention must now turn to the setting of environmental objectives and the targeting and prioritising of a programme of best management practices (BMPs) at a farm scale to minimise the significance of diffuse pollution.

**Targeting and Prioritising BMPs at a Farm Scale**

Determining the siting and suitability of BMPs requires a structured approach:

1. Most importantly the diffuse pollution risks must be prioritised in light of a diffuse pollution audit, because appropriate BMPs will first and foremost be determined by the problem and will be very different, for example, for eutrophied lochs than for bathing water catchments.

2. BMPs appropriate to the risks identified must be selected.

3. Other factors must be taken into account, such as cost-effectiveness, habitat benefits and the likely level of BMP participation across a catchment.
In the absence of data on the source strengths of pollutant loads prioritisation will undoubtedly be subjective to some degree. However, the methodology of impact prediction developed for Environmental Impact Assessment (EIA) provides a relatively objective framework to support the decision-making process. This method aims to assess the significance of impacts on the environment based on most notably the sensitivity of the environmental resource, and the perceived magnitude of the impacts on that resource. The process avoids any sort of numerical scoring, and still requires a good deal of reasoned professional judgement, but can be used to determine broad significance ‘thresholds’ which allow prioritisation of mitigation measures (Table 1).

Table 1: Significance based on sensitivity of receptor and magnitude of impact

<table>
<thead>
<tr>
<th>Magnitude</th>
<th>Low</th>
<th>Medium</th>
<th>High</th>
</tr>
</thead>
<tbody>
<tr>
<td>High</td>
<td>Moderate</td>
<td>Substantial</td>
<td>Substantial</td>
</tr>
<tr>
<td>Medium</td>
<td>Slight</td>
<td>Moderate</td>
<td>Substantial</td>
</tr>
<tr>
<td>Low</td>
<td>Slight</td>
<td>Slight</td>
<td>Moderate</td>
</tr>
</tbody>
</table>

SEPA recently commissioned the preparation of three linked documents in order to provide a systematic methodology for identifying farm scale diffuse pollution and for implementing strategies to alleviate problems (SEPA, 2006). The first part, ‘Farm scale diffuse pollution audits’, sets out a methodology for carrying out diffuse pollution audits on farms and includes standard audit forms to aid this process. The second part, ‘Siting and suitability of BMPs’ sets out guidance on selection of suitable mitigation measures for problems identified in the audit. The third part, the ‘BMP Manual’ contains descriptions of a wide range of possible BMPs to address diffuse pollution issues organised in a way to facilitate selection. The guidance is meant to help those charged with addressing agricultural diffuse pollution issues in practice. It will prove useful to catchment management planners in priority or problem catchments, to regulators, and to advisors and consultants to farmers as they try to comply with a support regime that will increasingly stress environmental protection in compliance with government policy. Launching the guidance on the web will allow field advisers to pilot and amend where necessary, before publishing the hardcopy and CD.

Sensitivity

For the purposes of assessing diffuse pollution impacts, the sensitivity of the environmental resource and the likely magnitude of the impact must be considered for each pollutant in turn. Sensitivity may be determined at a catchment scale or by some factor specific to the farm (e.g. the presence of a neighbouring designated site or the soils present on the farm). Before undertaking any work in the field, an assessment should be made of the catchment containing the farms in question. Certain catchments will be more sensitive to certain pollutants and it is important that an appreciation of such sensitivity is built into the subsequent field work. Some catchments will be more prone to certain types of diffuse pollution and this should be borne in mind when surveying. A targeted approach that focuses on ‘priority catchments’ is proposed such that the accumulative benefit of adopting BMPs can be gained and effectiveness measured. Examples are given below:
Nitrate Vulnerable Zones (NVZs)

In Scotland, 14% of the land area is designated as NVZs. These are located in Aberdeenshire, Moray, Banff and Buchan; Strathmore and Fife; Lothians and the Borders; and Lower Nithsdale. The European Nitrates Directive 91/676/EEC requires legally binding rules to be put in place for NVZs to reduce nitrate loss from agricultural land when nitrate levels exceed, or are likely to exceed, the levels set in the Directive. These rules are known as Action Programmes. Action Programme measures in NVZs in Scotland require that a fertiliser and manure plan to assess N fertiliser requirement for each crop and field is prepared and implemented; N must not be applied in excess of crop need; organic manure applications must not exceed specified N limits; and closed periods when slurry and poultry manure shall not be applied to any land that has a sandy or shallow soil (Scottish Executive, 2003). SAC estimated in the Regulatory Impact Assessment that these measures could reduce nitrate leaching in the NVZs by up to 3,580 tonnes N/annum (Oglethorpe et al., 2002).

The Nitrates Directive requires the Action Programme to be reviewed periodically. Ahead of the next review (2006), the European Commission has challenged the UK’s implementation of the Nitrates Directive. The Commission has identified several aspects of the existing Action Programme that it wishes the UK to address. The major issues affect livestock farms, and relate to the farm-based limit on livestock excretal returns; and the extension of the closed period for spreading slurry and poultry manure with associated requirement for increased storage. The Commission has also identified the need to be sure what the upper N fertiliser limits are and there needs to be some maximum. The Scottish Executive is preparing a consultation paper, expected in February 2006, proposing changes to the 2003 NVZ Action Programme Regulations. Recent changes in economics including CAP reform have already introduced changes to agricultural practice that on balance are expected to reduce nitrate leaching independently of the provisions of the Action Programme. The expected increase in conversion of arable land to extensively-managed, permanent green cover; and the reductions in economic optimum N fertiliser requirement due to higher N fertiliser costs and continuing low commodity price will also contribute.

Designated Bathing Waters

Faecal coliforms and faecal Streptococci from agriculture contribute too many of the 60 designated bathing waters in Scotland failing the EC Bathing Water standards, especially in the South West of the country (Ayrshire, Dumfries and Galloway). Merrilees et al. (2004) and Aitken et al. (2004) reported on a SEERAD-funded project to deliver capital works measures and farm-specific BMPs to reduce the potential risk of faecal indicator organism (FIO) contamination from livestock farms to rivers. Forty-eight farms were surveyed in four sensitive river catchments draining into bathing waters in the south, west and north coast of Scotland. FIO bacteria from farmland potentially impacted all the rivers. Waste storage facilities, farming practices, field conditions, grazing management and risks of FIO contamination to watercourses were assessed on each farm and BMPs were designed, costed, implemented and monitored. With only nine farms, Brighouse Bay was an ideal place to establish and monitor BMPs and assess any improvement in water quality. The main aim was to determine to what extent improved farm practices and BMPs could contribute to improved water quality in the catchment (Dickson et al., 2005; Merrilees et al., 2006).
**Magnitude of Impacts**

The magnitude of each diffuse pollution impact is less likely to be determined by overall catchment characteristics and more by individual farming practice and dependant on the problems and other features specific to the farm that will be highlighted by a farm scale diffuse pollution audit.

**Significance**

Once sensitivities and magnitudes have been assessed, the matrix approach shown in Table 1 allows an appraisal of the overall significance of different pollution risks, e.g. nitrate.

A farm with predominantly sandy or shallow soils within an NVZ (HIGH sensitivity), where more than 50% of land is ploughed and cropped in autumn (HIGH magnitude), would be expected to have a SUBSTANTIAL risk of nitrate diffuse pollution into groundwater.

A mixed beef and spring cropping farm (LOW magnitude), with predominantly heavy soils outwith an NVZ (LOW sensitivity), would be expected to have a SLIGHT risk of nitrate diffuse pollution and would therefore be a lower priority for nitrate-targeted BMPs.

By carrying out this type of appraisal for each diffuse pollutant, the pollutants can be prioritised and BMP selection can proceed.

**Farm Audits**

An audit of potential sources of diffuse pollution on the farm is carried out. This is done partly by interviewing the farmer and collecting data from him and partly by walking the farm and farm steading and observing any problems. These may include problems related to fertiliser and manure usage, problems related to pesticide usage and problems related to soil management including soil erosion. The aim of the audit is to identify all potential problems that can then be prioritised. A useful way of considering diffuse pollution issues is to regard it as a three-stage process. Without each stage being present, diffuse pollution does not occur. A treatment train approach to BMP implementation may address the problem at each stage in the process.

First, there must be a source of the pollutant. This might be excess nitrogen in soil or the nutrients in slurry spread on the soil surface. A source in itself does not constitute pollution however.

Second, there must be a pathway. Excess nitrogen in agricultural soil is not a pollutant although it may be one in the soil of adjacent natural or semi-natural habitats. It only becomes a pollutant if there is a mechanism or pathway, determined by landscape characteristics such as slope, soil type and vegetation, for the excess nutrient to move from the soil, where it may be a desirable component of soil fertility, to a receptor. The pathway might be leaching of nitrates to a groundwater receptor, transfer of nitrates in drain flow to a surface water receptor or denitrification and transfer of oxides of nitrogen to the atmosphere acting as a receptor.
Third, there must be a receptor. This is some component of the environment where the substance being considered has undesirable consequences and is a pollutant. Phosphorus in agricultural soil is not usually a pollutant. It is an essential plant nutrient and a component of soil fertility. Phosphorus in loch water may be a pollutant as it may cause eutrophication. Phosphorus in the soil in an area of land containing interesting assemblages of wild plants may also be a pollutant as it may tend to reduce the biodiversity of the habitat. Diffuse pollution therefore consists of the transfer of substances from a source via a pathway to a receptor where that substance has undesirable consequences.

**Critical areas**

On some farms, most of the transfer of certain pollutants arises from a relatively small proportion of the total farm area. For example, soil erosion can transfer silt and adsorbed nutrients and pesticides from land to water. On farms where in-field soil erosion occurs, it is typically present only in limited areas, usually moderate or steep slopes with susceptible soils that are in arable cropping. Control measures may be targeted at such areas leaving the rest of the farm unaffected.

Certain even more limited areas are sometimes termed ‘hot spots’. These may include such areas as stock feed rings, areas where stock habitually shelter and areas where sprayers are filled. The pollution arising from such hot spots may be out of proportion to their small area.

**Cost-effectiveness and cost benefit analysis of BMPs**

Where Environmental Quality Standards (EQS) for a water body are agreed to have a sound environmental basis, and where several alternative measures are to be considered the preferred method of evaluation is cost-effectiveness analysis (CEA). This attempts to evaluate the least cost way of achieving a desired standard. Cost Benefit Analysis (CBA) is appropriate for evaluating investment projects where substantial non-market benefits or costs occur. In the SEERAD-funded project on ‘Evaluation of BMPs for diffuse pollution control’ (Vinten et al., 2006), the efficacy of specific BMPs with respect to two main pollutants, P and FIO, have been studied. Details of this work will be reported elsewhere (Vinten et al., 2006), but Table 2 gives an example of the analysis framework used.

Loch Leven is a lowland Loch prone to eutrophication by soluble and particulate P derived from field run-off and drainage water. This results in economic loss for downstream users and from recreational users. Much of the non-farm input of P has been controlled and attention is now focused on control of inputs from farmland. In 1997, a buffer strip was installed on the Green’s Burn to limit soil erosion losses to the stream. A simple conceptual framework for evaluating pollution control measures is set out below:

A. Identify Environmental Quality Standards for receiving waters.

B. Estimate the pollutant loading and concentration from the farm before pollution control measures are installed.

C. Estimate reduction in pollution loading and concentration after installation of pollution control measures.
D. Use relative area of farm and whole catchment to assess contribution of single farm to and control measures to compliance with EQS.

E. Use information on costs of measures and scaled up environmental benefits to do simple Cost-Effectiveness and Cost Benefit Analysis.

The results of this analysis show that the cost-effectiveness of £32/kg TP of the buffer strip for P removal is quite low for arable, compared with other work (e.g. Hutchinson et al., 2005 have figures of £2.70 for arable buffers, and £75/kg TP for grassland buffers in Northern Ireland). This may reflect the high risk nature of the soils to erosion and topography in the Green’s Burn catchment. Recent work looking at a soil bund to contain eroded soil from a field in this catchment showed a much greater cost-efficiency. On a cost benefit analysis basis there is still a clear positive benefit to Loch Leven of the buffer strip. The Northern Ireland values are based on a national scale evaluation.

**Buffer strip width and management**

At the simplest level, buffer strips are effective against diffuse pollutants by disrupting the pathway between the source (agricultural land) and the receptor (watercourses), and in the case of pesticides this is the principal way in which they work. The vegetation in buffer strips can also physically trap pollutants such as soil particles (and phosphate) and absorb nitrates through their roots. The effectiveness of all of these mechanisms is likely to be enhanced by increasing the width of the buffer strip, and so the general guidance should be to make buffer strips as wide as possible. In practice however, the width of buffer strips is likely to be a compromise determined by a number of other factors. If the vegetation in the buffer strip is being established by sowing seed, then the width of the seed drill is likely to be an important factor. If a 3-metre drill is used to sow grass, then a 6- or 12-metre buffer strip is likely to be preferable to a 5- or 10-metre strip. The guidance in the PEPFAA Code (Scottish Executive, 2005b), NVZ regulations (Scottish Executive, 2003) and LERAPs will often be a starting point in determining buffer strip widths.

Field topography is a major consideration. Watercourses at the foot of steep slopes that are prone to erosion or run-off should have wider buffer strips than watercourses in flat fields. A particular problem that is often encountered when establishing buffer strips is where a steep slope has a short, flat run-out at the bottom, between the slope and the burn. If this flat area is converted into a buffer strip, and particularly if it is fenced, then it can create problems for the farmer having to turn machinery on the steep slope. In these situations the farmer must decide whether to reduce the buffer strip or increase it to take in the whole slope. Such decisions are likely to depend on the perceived value of the land on the slope, but set-aside may be an acceptable option for such slopes. Buffer strips are unlikely to be effective in controlling suspended solids from eroding soils where the topography concentrates run-off. In-field erosion control measures should be applied in these situations.

The effectiveness of buffer strips will depend on the way the strip is managed as well as its width. Vigorous growth of grass or tall-herb vegetation is likely to be the most effective vegetation. Although such vegetation will often develop by natural regeneration on bare ground, quicker establishment and buffer strip effectiveness may be brought about by sowing a suitable seed mix (and may be required under
certain agri-environment options). Fuller details of the methods used for siting fenced areas and buffer strips are described and illustrated in the Farming and Watercourse Management Handbook (2000).

Table 2: Use of Green’s Burn Buffer strip to reduce total P loading to Loch Leven

<table>
<thead>
<tr>
<th>Environmental quality objective</th>
<th>Eutrophic restored to mesotrophic Loch</th>
<th>Notes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Required pollutant mitigation</td>
<td>Reduction of annual TP loading to Loch Leven from 0.5 to 0.2 g/m² (=4.20 tonnes TP pa over 13.9 km² loch area)</td>
<td>Based on OECD (1982)</td>
</tr>
<tr>
<td>BMP contribution</td>
<td>Green’s Burn buffer strip reduces TP load by estimated 0.38 tonnes TP pa</td>
<td>Vinten et al. (2005)</td>
</tr>
<tr>
<td>BMP cost</td>
<td>£12k pa (fencing and management)</td>
<td>£5/m² for fencing and £100/ha for RSS species rich grassland</td>
</tr>
<tr>
<td>Cost-effectiveness</td>
<td>(£12k/380kg) = £32/kg TP</td>
<td></td>
</tr>
<tr>
<td>Value of Loch Leven restoration</td>
<td>User value: £243k pa</td>
<td>Frost and McTernan (1997)</td>
</tr>
<tr>
<td></td>
<td>Non-user value: £290k pa</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Total: £533k pa</td>
<td></td>
</tr>
<tr>
<td>Benefit:cost ratio</td>
<td>(£533k/£12k) (0.38t/4.2t) = 4.0 (Good)</td>
<td></td>
</tr>
</tbody>
</table>

FUTURE DEVELOPMENTS

Database of Farm Scale Sources of Diffuse Pollution

SEPA holds a significant amount of data on water quality, but data on farm scale sources of pollution are lacking. A database giving information for each farm type, on source strength per pollutant, potential mitigation measures, cost-effectiveness, loadings mitigated and time scale for efficacy is proposed, building on aspects of the screening tool (Anthony et al., 2005). The database would draw on the literature, but also on audits from the six farm types in SEPA’s survey (Frost et al., 2000), the SAC BMP farms in SEERAD-funded project SAC/348/03 (Table 3), and relevant catchments where farm scale data are available. These catchments include Tarland (MI), Glensaugh (MI), Ythan (MI); Lunan (MI/SAC), Piltanton (SAC/MI), Leven (SAC/MI), Cessnock (SAC/MI/CREH), Nairn (SAC/CREH), Brighouse (SAC/MI/CREH), Ettrick (SAC/CREH), Sandyhills, Bush (SAC/MI). The database would dovetail with SEPA’s ‘BMP Manual’ and could be put on a website. The output would form the focus for ongoing improvement of estimates, as information from future projects, particularly the results of the SEERAD work packages (WP3.4 Measurement of water quality; WP 3.5 Management of water quality) and Defra Catchment Sensitive Farming, etc. becomes available.

Policy Matrix

A policy matrix, i.e. a summary of recommended measures, prioritised and categorised into proposed appropriate levels of regulation, for mapping of on-farm measures against policy instruments, voluntary measures and potential GBRs etc., is required.
ADAS has developed a matrix for Defra to map on-farm measures against policy instruments and potential policy instruments (Shepherd et al., this volume). This quickly provides an assessment of the likely effectiveness of a policy instrument in invoking particular measures, and identifying gaps, i.e. measures that have not been picked up by current policy instruments. We propose this approach be extended to the above range of categories.

**Knowledge Transfer**

Although considerable research has been undertaken on sustainable farming systems and BMPs for good environmental management, there has been a dearth of practical case studies illustrating good practice and how land managers can benefit from CAP Reform measures supporting good practice. Monitor farms in catchments where farm scale data are available could provide the focus for farm-based BMP events. The target audience should involve land managers, policy makers, contractors and supply industries, e.g. fertiliser/pesticide suppliers and partner organisations, as well as the public. There are already good examples within Scotland of successful partnership projects, e.g. The Tarland Burn (Cooper et al., 2006).
**Table 3:** Summary of catchments, farms and BMPs in current SAC project (SAC/348/03)

<table>
<thead>
<tr>
<th>Area</th>
<th>Catchments</th>
<th>Farms and enterprise</th>
<th>‘BMPs’</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Ayrshire</td>
<td>Cessnock</td>
<td>A: Dairying</td>
<td>Dirty water separation</td>
</tr>
<tr>
<td>Pollutants:</td>
<td></td>
<td></td>
<td>Roofing</td>
</tr>
<tr>
<td>FIOs, P, NH$_4$-N, SS</td>
<td></td>
<td></td>
<td>Pond</td>
</tr>
<tr>
<td>Issue:</td>
<td></td>
<td></td>
<td>Culvert</td>
</tr>
<tr>
<td>Bathing Waters</td>
<td></td>
<td></td>
<td>Off stream drinking</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>RAMS</td>
</tr>
<tr>
<td></td>
<td></td>
<td>B: Dairying</td>
<td>Pond</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Upstream RSS buffer</td>
</tr>
<tr>
<td></td>
<td></td>
<td>C: Dairying</td>
<td>Fencing</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Timing of slurry</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Management</td>
</tr>
<tr>
<td></td>
<td></td>
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<td>Grazing management</td>
</tr>
<tr>
<td>2. Perth and Kinross</td>
<td>Loch Leven</td>
<td>D: Mixed arable</td>
<td>Buffer strip</td>
</tr>
<tr>
<td>Pollutants:</td>
<td></td>
<td></td>
<td>Bund for run-off</td>
</tr>
<tr>
<td>P, SS</td>
<td></td>
<td></td>
<td>Capture</td>
</tr>
<tr>
<td>Issue:</td>
<td></td>
<td></td>
<td>Horseshoe wetland</td>
</tr>
<tr>
<td>Eutrophication</td>
<td></td>
<td>Lunan Lochs</td>
<td>P budgeting</td>
</tr>
<tr>
<td></td>
<td></td>
<td>E: Mixed arable</td>
<td>Pond</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Ochre P removal</td>
</tr>
<tr>
<td></td>
<td></td>
<td>F: Arable and beef</td>
<td>Buffer strip</td>
</tr>
<tr>
<td>3. Bush</td>
<td>Glencorse Burn</td>
<td>G: Beef outwintering</td>
<td>Nil</td>
</tr>
<tr>
<td>Pollutants:</td>
<td></td>
<td>G: arable and grass</td>
<td>Buffer strip</td>
</tr>
<tr>
<td>FIOs, NH$_4$-N, SS</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Issue:</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>4. Stranraer</td>
<td>Piltanton Burn</td>
<td>H: Beef outwintering</td>
<td>Effluent treatment from woodchip</td>
</tr>
<tr>
<td>Pollutants:</td>
<td></td>
<td></td>
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<tr>
<td>FIOs, NH$_4$-N, SS</td>
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<td></td>
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<tr>
<td>Issue:</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>nitrate in Gw and</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>surface water BOD</td>
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**ACKNOWLEDGEMENTS**

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**REFERENCES**


RETENTION OF POLLUTANTS BY THE SINK STRUCTURES IN CATCHMENTS – STUDIES TO REDUCE DIFFUSE POLLUTION IN CHINA’S RURAL AREAS

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SUMMARY

Eutrophication of lakes and reservoirs is a serious problem in China. Diffuse sources from agricultural lands account for a large portion of the load. Besides reducing the pollution from the sources, the retention strategy during the transport process is important and it plays an important role in Best Management Practice in China. The retention strategy consists of using different sink landscape structures in watershed scales, such as multipond systems, small stone dams, swales, vegetative belts, dry ponds, waterside buffer zones and others. During continuous run-off events, the concentration reductions of total suspended solids (TSS), total phosphorus (TP), total dissolved phosphorus (TDP) and dissolved reactive phosphorus (DRP) by the system ranged from 40% to 70%. During discontinuous run-off events, removal rates of pollutants by the whole system were higher as there was no or little surface water and pollutants exported from the watershed; removal rates of pollutants all exceeded 99%. The statistical analysis results of run-off events indicated that a dry pond was the steadiest structure for controlling diffuse pollutants export. A multipond system is a wetland system composed of many tiny ponds connected by ditches and the ponds are scattered among the villages and crop fields. We discovered many mechanisms, such as decreasing the flow velocity with volume change, settling particulate matters using sedimentation, removing dissolved pollutants using adsorption and recycling nutrients back to the crop fields. These structures can effectively reduce diffuse pollution and enhance nutrient cycling in rural watersheds with wet and dry weather. These structures are made of natural materials and are very environmentally friendly. They are cheap to construct and usually do not need energy for operation.

INTRODUCTION

Eutrophication in regional water bodies is being attributed to increased nutrients inputs (Artola, 1995; Sharpley, 1998). Diffuse sources of sediment and nutrients, primarily from agricultural lands, have been identified as the major cause of water quality degradation (Carpenter, 1998; Arheimer, 2000; Dzikiewicz, 2000; Sharpley, 2000). Village area, farmland and breed industries in agricultural watersheds may be the cause of main sensitive sources for nutrients loss. When unsuitable management measures of these lands are employed, contamination of water resources may result. Thus, elevated concentrations of nutrients from rural areas pose a potential hazard of eutrophication and a consequent cost for water treatment (Sanderson, 2001).

Yuqiao Reservoir is the major source of drinking water to Tianjin City. Preliminary research has identified diffuse polluted run-off from agriculture as the largest contributor of nitrogen and phosphorus to the reservoir, which accounts for about
60% phosphorus and 50% nitrogen of the total load (Wang, 1999). Approximately 1300 tons per day of litter can be generated. Whereas, the use efficiency of these wastes to farming is lower than 60%. During rainfall events, surface run-off is often seriously polluted through washing out these nutritional matters. Results from sampling and pollutants surveillance of polluted run-off showed a serious pollution with a TSS content of 6.6–14.8 g/L, TN of 91.7–252.3 mg/L, TP of 13.0–34.6 mg/L, COD of 1128–9124 mg/L, respectively.

Agricultural diffuse pollution is a landscape problem that requires an integrated approach involving implementation of both on-field and off-field control measures (Tim, 1995). Many studies have been conducted on the control of nutrients flow from agricultural lands by landscape structures (Pomogyi, 1993; Uusi-Kämppää, 2000; Kuusemets, 2002). But most of these studies are about control effect on nutrient transport processes by a single type of structure, relatively few have documented combination control effect of multiple buffer/detention structures. Furthermore, in most previous studies, the focus was on the final effect of a treatment system and treated the transport process of nutrients as a black or grey box. The objects of this study were: (1) to explicate the detailed transport processes of diffuse P-pollutants crossing a typical complex agricultural landscape in North China; and (2) to compare control behaviors of different buffer/detention structures and to assess their integrated effects under different hydrological conditions.

MATERIALS AND METHODS

Study Area and Monitoring Installation

Taohuasi watershed is in the north bank of Yuqiao Reservoir with a surface area of 249 ha. The relief is strongly differentiated – there are many ridges, hills, and depressions accompanying cliffs, channels, and ephemeral stream valleys. Land use in this watershed is predominantly hilly land and cropland (Figure 1). Long-term average precipitation is 749 mm, of which 80% falls from June to September. Streams distributed in the watershed are almost all ephemeral streams. Most of the rainfall events are storms. Surface run-off is the main mechanism by which nutrients are exported from the watershed. During two study years, precipitations were lower than the long-term average and we integrally monitored eight rainfall-run-off events. Continuous flow generates only during a heavy rainfall event, of which water flow can transport from the upstream to the mouth of the watershed. Discontinuous flow usually generates during a gentle rainfall, of which water flow is intersected with little or no outflow at the watershed mouth. Monitoring sites were composed of 28 sampling sites for surface flow (S1 to S28) and five for tributary (T1 to T5) flow from various land uses along the stream channel. All analyses of pollutants were done using standard methods (APHA, 1985).
Calculation of the sediment and P-pollutants removal rate

The mass removal rate of sediment and P-pollutants of single buffer/detention structure or the whole system they constituted can be calculated from flow-weighted mean concentration and run-off volume of waterflow when it enters and leaves the system or structure. The formula can be established as follows:

\[
\begin{align*}
M_{PI} &= \sum Q_{PIi}C_{PIi} \\
M_{PO} &= \sum Q_{POi}C_{POi} \\
M_{PT} &= \sum Q_{PTi}C_{PTi} \\
R &= \left[1 - \frac{M_{PO}}{M_{PI} + M_{PT}}\right] \times 100\% \\
\end{align*}
\]

in which \(Q_{PIi}\), \(Q_{POi}\) and \(Q_{PTi}\) are inflow, outflow and tributary flow volume of the whole system or single structure (m³); \(C_{PIi}\), \(C_{POi}\) and \(C_{PTi}\) are mean pollutants concentration in inflow, outflow and tributary flow of the whole system or single structure (g/L or mg/L); \(M_{PIi}\), \(M_{POi}\) and \(M_{PTi}\) are total pollutants mass in inflow, outflow and tributary flow of the whole system or single structure (kg); \(i\) is buffer/detention structure number, from 1 to 5; \(R\) is the removal rate of the whole system or single structure.
RESULTS AND DISCUSSION

Landscape Structures and Flow Velocity Variation

Diversified artificial and natural buffer/detention landscape structures, included stone dams (SDs), roadside grassed ditch (RGD), grassed filter strip (GFS), dry ponds (DPs) and riparian buffer zone (RBZ), were found distributed along the ephemeral stream channel upslope to downslope in the experimental watershed. These structures had many different characteristics (Table 1). All the landscape structures constituted the whole buffer/detention structure system to control the transport process of diffuse polluted run-off.

Table 1: Characteristics of the buffer/detention structures along the ephemeral stream channel

<table>
<thead>
<tr>
<th>Type*</th>
<th>Constituting Parts</th>
<th>Elevation (m)</th>
<th>Height or depth (m)</th>
<th>Length† (m)</th>
<th>Width (m)</th>
<th>Area (m²)</th>
<th>Adjacent land use</th>
</tr>
</thead>
<tbody>
<tr>
<td>SDs</td>
<td>Dam 1</td>
<td>48.3</td>
<td>1.2</td>
<td>19.0</td>
<td>16.0</td>
<td>304.0</td>
<td>Hilly land</td>
</tr>
<tr>
<td></td>
<td>Dam 2</td>
<td>47.2</td>
<td>0.9</td>
<td>39.0</td>
<td>14.0</td>
<td>546.0</td>
<td>Hilly land, village</td>
</tr>
<tr>
<td></td>
<td>Dam 3</td>
<td>46.0</td>
<td>1.1</td>
<td>59.0</td>
<td>21.0</td>
<td>1.24 × 10³</td>
<td>Hilly land, village</td>
</tr>
<tr>
<td></td>
<td>Dam 4</td>
<td>40.6</td>
<td>0.8</td>
<td>86.0</td>
<td>11.0</td>
<td>946.0</td>
<td>Village, orchard</td>
</tr>
<tr>
<td>RGD</td>
<td>–</td>
<td>26.2–27.1</td>
<td>0.4</td>
<td>260.0</td>
<td>2.0</td>
<td>520.0</td>
<td>Road, cropland</td>
</tr>
<tr>
<td>GFS</td>
<td>–</td>
<td>24.3–25.4</td>
<td>–</td>
<td>182.0</td>
<td>7.0–7.5</td>
<td>1.27 × 10³</td>
<td>Cropland</td>
</tr>
<tr>
<td>DPs</td>
<td>Pond 1</td>
<td>24.1</td>
<td>0.9–1.0</td>
<td>6.3</td>
<td>–</td>
<td>31.5</td>
<td>Cropland</td>
</tr>
<tr>
<td></td>
<td>Pond 2</td>
<td>22.6</td>
<td>1.2</td>
<td>23.5</td>
<td>–</td>
<td>212.0</td>
<td>Cropland</td>
</tr>
<tr>
<td>RBZ</td>
<td>–</td>
<td>16.8–17.3</td>
<td>–</td>
<td>302.0</td>
<td>–</td>
<td>2.87 × 10⁵</td>
<td>Cropland, reservoir</td>
</tr>
</tbody>
</table>

*SDs, stone dams; RGD, roadside grassed ditch; GFS, grassed filter strip; DPs, dry ponds; RBZ, riparian buffer zone.

†Length refers to distance from inlet to outlet of the structure.

The change of flow velocity in the water pathway plays an important role in detention time and export load of pollutants. From the view of the whole system, the flow velocity presented a fluctuant pattern with many instances of increase and decrease (Figure 2). During a continuous run-off event, the mean flow velocity was 36.9 cm/s at the inlet of the system and decreased to 13.6 cm/s at the outlet. Due to the junction of large tributary flow from the village area (resulting from the low infiltration rate of the ground), there was a sharp elevation of flow velocity from 10.1 to 26.2 cm/s. But soon it decreased to 12.0 cm/s when the flow encountered a stone dam. Because of transporting in the sloping stream channel, the flow velocity was gradually increased and reached 32.3 cm/s at the inlet of roadside grassed ditch. Due to the enhancement of surface roughness by vegetation, the flow velocity gradually decreased. But it increased again at the inlet of the grassed filter strip before entering the tributary
Due to the flourishing vegetation growing in the strip, flow velocity decreased sharply from 28.5 to 10.9 cm/s. In the ponds zone, the flow velocity at the outlet was very low because of decreased of kinetic energy resulting from a huge water-storage volume of ponds. The velocity increased between the two ponds for the sloping stream channel. The riparian buffer zone also presented better capacity of slowing flow velocity through vegetation and flat relief. The variability of flow velocity under a discontinuous run-off event was similar to that of a continuous event. But the change was relative gentle for the intermittent transport of water and relative small run-off volume. The multiple buffer/detention structure system provided a good opportunity for sediment and particulate nutrients to settle and transform due to the prolonged detention time.

Figure 2: The spatial variation of surface flow velocity in the flow pathway during the continuous and discontinuous run-off event. The dashed line means run-off transporting in the stream channel between two buffer/detention structures

Sediment and P-pollutants Concentration Variation in the System

During transporting in the flow pathway, the sediment and P-pollutants concentration were affected greatly by the multiple buffer/detention structures (Figure 3). In this research, the typical variation characteristic of pollutants concentration in the flow path during both continuous and discontinuous run-off events was that the concentration elevated as input of tributary flow from various agricultural sources (especially village area) and gradually decreased as run-off passing through each buffer/detention structure. During a continuous run-off event, the concentrations of TSS, TP, TDP, DRP in inflow of the whole system were 2.70 g/L, 0.31 mg/L, 0.07 mg/L, 0.05 mg/L, respectively. But when mixed with tributary flow from the village, the concentrations of TSS, TP, TDP, DRP were elevated to 3.68 g/L, 3.69 mg/L, 0.42 mg/L, 0.38 mg/L, respectively. After passing through the four stone dams, particulate materials (TSS, TP) were retained markedly for the formation of better settling conditions by dams. The proportion of dissolved P-pollutants (TDP, DRP) retained was small. This possibly
was contributed to the low vegetation coverage on the structure surface resulting in little uptake quantity of dissolved forms of P. When the polluted run-off reached and crossed through the zone of the roadside grassed ditch, the concentrations of TSS decreased from 1.12 to 0.62 g/L. But for P pollutants, the concentrations presented a pattern of fluctuation as waterflow transported through the ditch. This result suggested that the grassed ditch was under a highly disturbed condition. Lack of effective management measures and destroy events between times and some trash cumulation in the bottom of the ditch maybe the answer to this. Comparatively, the concentrations of sediment and P-pollutants decreased sharply as run-off was transported in a grassed filter strip. The concentrations of TSS, TP, TDP, DRP in inflow and outflow were 0.92 g/L, 2.14 mg/L, 1.32 mg/L, 1.23 mg/L and 0.52 g/L, 1.01 mg/L, 0.39 mg/L and 0.32 mg/L, respectively. The result suggested that a grassed filter strip is effective for removal of various P forms. This removal effect also presented during polluted run-off transporting through dry ponds and riparian buffer zone. The concentration of pollutants was reduced further and reached 0.71 g/L TSS, 0.29 mg/L TP, 0.08 mg/L TDP and 0.07 mg/L of DRP in outflow of the whole system, respectively.

During a discontinuous run-off event, transport processes of water and pollutants were intermittent (Figure 3). Thus, little or no water and pollutants are exported from the watershed. The reduction of pollutants concentration was more rapid due to discontinuous transport processes which could cause low kinetic energy of run-off to carry out pollutants and longer contact time between soil matrix and pollutants increasing sorption and infiltration.

![Figure 3: The spatial variation of pollutants concentrations along the flow pathway (declining with elevation from 48.3m to 16.8m upslope and downslope) during continuous run-off event and discontinuous run-off event](image-url)
Table 2: The removal rates of sediment and P-pollutants by the multiple structures system during different types of run-off events

<table>
<thead>
<tr>
<th>System input</th>
<th>System output</th>
<th>Removal rate (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>SRH*</td>
<td>SRV</td>
<td>SRO</td>
</tr>
<tr>
<td>Continuous run-off events</td>
<td></td>
<td></td>
</tr>
<tr>
<td>TSS (kg)</td>
<td>2784.3</td>
<td>821.1</td>
</tr>
<tr>
<td>TP (kg)</td>
<td>0.127</td>
<td>0.812</td>
</tr>
<tr>
<td>TDP (kg)</td>
<td>0.047</td>
<td>0.300</td>
</tr>
<tr>
<td>DRP (kg)</td>
<td>0.037</td>
<td>0.235</td>
</tr>
<tr>
<td>Discontinuous run-off events</td>
<td></td>
<td></td>
</tr>
<tr>
<td>TSS (kg)</td>
<td>2249.2</td>
<td>2582.6</td>
</tr>
<tr>
<td>TP (kg)</td>
<td>0.218</td>
<td>1.908</td>
</tr>
<tr>
<td>TDP (kg)</td>
<td>0.081</td>
<td>0.706</td>
</tr>
<tr>
<td>DRP (kg)</td>
<td>0.063</td>
<td>0.551</td>
</tr>
</tbody>
</table>

*SRH, surface run-off from hilly land; SRV, surface run-off from village; SRO, surface run-off from orchard; SRC, surface run-off from cropland.

**SEDIMENT AND P-POLLUTANTS REMOVAL CONTRAST**

**Removal Efficiency of the Multiple Structures System**

The loads of sediment and P-pollutants exported from the outlet of the watershed were reduced to a great degree after gradual removal by each buffer/detention structure upslope to downslope. But the removal effectiveness was different due to the different hydrological characteristics of continuous and discontinuous run-off. For example, during continuous run-off events, the total input loads of TSS, TP, TDP, DRP by surface run-off were 3710.9 kg, 1.102 kg and 0.407 kg, 0.319 kg, respectively. Their exports from the system were 1235.2 kg, 0.433 kg, 0.210 kg and 0.181 kg, respectively (Table 2). The removal rates were 66.7%, 60.7%, 48.4% and 43.3%, respectively. Comparatively, during discontinuous run-off events, the total input loads of TSS, TP, TDP, DRP by surface run-off were 5123.3 kg, 2.482 kg, 0.918 kg and 0.717 kg, respectively, with total export loads only of 20.6 kg, 0.014 kg, 0.005 kg, 0.004 kg, respectively. Their removal rates all exceeded 99%. These high removal rates of pollutants mainly resulted from the intermittent transport process of polluted run-off.

**Removal Efficiency of Single Structure**

Due to differences in physical characteristics, spatial location in the watershed, main removal mechanisms and degree of disturbance by humans among these structures, removal efficiency of each single structure was different too. During continuous run-off events, due to the huge storage capacity and long detention time of dry ponds, they had the highest removal rate of TSS, TP, TDP and DRP, of which were 34.6%, 34.3%, 20.5% and 17.3%, respectively. For dissolved forms of P-pollutants (TDP, DRP), a grassed filter strip had higher removal rates (Table 3). This mainly contributed to the higher infiltration rate of water resulting from dense roots of vegetation and
biological uptake in a grassed area. Because of slowing run-off velocity and forming detention conditions, stone dams have a higher removal rate of TSS and TP. During discontinuous run-off events, there was often little or no outflow from these buffer/detention structures and the removal rates of pollutants were improved markedly. For dry ponds, the removal rate of pollutants could reach 100% because of zero outflow. For a grassed filter strip and riparian buffer zone, the removal rates of pollutants were all over 80% and were higher than that of stone dams and a roadside grassed ditch, of which were about 40–70%.

The statistical analysis results of rainfall-run-off events (n = 8) indicated that removal effectiveness of various pollutants by dry ponds was the steadiest, of which the average coefficient of variation (CV) of removal rate was 29.5%. Comparatively, the average CV of a roadside grassed ditch was the highest (49.0%). This result suggested that the purification function of a roadside grassed ditch was more instable because of disadvantageous influences from human disturbances.

**Table 3:** The removal rate (%) of sediment and P-pollutants in surface run-off by different buffer/detention structures during different types of run-off events

|          | SDs* | RGD | GFS | DPs | RBZ | SDs | RGD | GFS | DPs | RBZ |
|----------|------|-----|-----|-----|-----|-----|-----|-----|-----|-----|-----|
| **TSS (kg)** | 32.1 | 9.6 | 12.0 | 34.6 | 7.8 | 68.5 | 54.7 | 85.5 | 100.0 | 81.0 |
| **TP (kg)**  | 23.7 | 6.5 | 15.4 | 34.3 | 9.2 | 63.4 | 52.4 | 83.7 | 100.0 | 84.1 |
| **TDP (kg)** | 12.1 | 8.8 | 13.6 | 20.5 | 11.4 | 52.1 | 50.4 | 81.8 | 100.0 | 84.2 |
| **DRP (kg)** | 10.3 | 7.1 | 13.3 | 17.3 | 9.6 | 49.7 | 48.2 | 83.5 | 100.0 | 83.3 |

*SDs, stone dams; RGD, roadside grassed ditch; GFS, grassed filter strip; DPs, dry ponds; RBZ, riparian buffer zone.*

**CONCLUSIONS**

Diversified artificial and natural buffer/detention landscape structures played an important role in transport processes of diffuse P-pollutants mainly from rural areas. Significant amounts of sediment and P-pollutants were removed from run-off by the multiple structures system. During continuous run-off events, the removal rates of TSS, TP, TDP, DRP by the system were about 66.7%, 60.7%, 48.4%, 43.3%, respectively. During discontinuous run-off events, removal rates of pollutants by the system were higher due to little or no surface water and pollutants exported from the watershed, of which all exceeded 99%. The statistical analysis results of rainfall-run-off events (n = 8) indicated that dry ponds were the steadiest structure for controlling diffuse P-pollutants export. When controlling diffuse polluted run-off from agricultural lands, local buffer/detention landscape structures can be used and managed as an effective and low cost measure to improve water quality downstream.
REFERENCES


CATCHMENT CHARACTERISATION AND TARGETING OF BEST MANAGEMENT PRACTICES USING PSYCHIC

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SUMMARY

PSYCHIC is a recently-developed prototype decision support system for identifying the spatial distribution of sediment and phosphorus (P) export risk. As part of Defra’s Catchment Sensitive Farming initiative, PSYCHIC was applied to four priority catchments with varied landscape type and land management activities. Total catchment export of sediment estimated by PSYCHIC varied from 0.03–0.32 t/ha/year. Predicted total P export varied from 0.59–1.34 kg/ha/year of which between 15% and 73% was predicted to originate in run-off from agricultural land. Spatial distribution analysis indicated that the largest agricultural contributions were associated with catchment ‘hotspots’ with combinations of high rainfall, steep slopes, erodible soils, frequent manure applications and presence of underdrainage systems. The results suggest that if used in combination with local knowledge and an appreciation of the main water quality issues in the catchment, PSYCHIC has the potential to play a useful role in river basin management planning. Implications for the targeting of mitigation measures in the four catchments are discussed.

INTRODUCTION

The EU Water Framework Directive (WFD) requires the preparation of river basin management plans to implement a programme of measures to achieve good water quality based on a range of ecological, and associated chemical, indicators. These measures will need to control both point source and diffuse pollutants, including suspended sediment (SS) and phosphorus (P) export from agricultural land. Phosphorus is the nutrient of most concern in freshwaters, and there is generally a close association between sediment and P in land run-off. Conceptually, the transport of sediment and P in land run-off and drainage can be considered to occur in two fundamental steps: mobilisation and delivery. Soil particles, and the P attached to them, must firstly be mobilised by processes of dispersion and detachment, before they can be delivered in rapid surface and sub-surface (drainflow) hydrological pathways to the waterbody. Additional dissolved P may also be mobilised when rain follows soon after the application of fertilizers and manures to the land surface (‘incidental’ P transfers).

There is considerable variation in the vulnerability of land to sediment and P transfer depending on landscape factors including rainfall patterns, soil type, slope, stream density and road/track networks. This underlying pattern of vulnerability can be heavily modified by land use management including land use, N and P inputs, livestock density, cultivation practices, crop management, manure management and presence of underdrainage systems. In order to utilise resources effectively, measures
to control SS and P loads from agricultural land must therefore be targeted within catchments to those areas where combinations of landscape and land management generate the highest pollution risk. To facilitate the cost-effective targeting of a range of land management options to mitigate SS and P export in catchments, a prototype decision support tool called PSYCHIC (Phosphorus and Sediment Yield Characterization In Catchments) has been developed within two catchments in England and Wales (Hampshire Avon and Wye) with siltation and eutrophications problems (Withers and Sharpley, 2006). This paper describes the further application and testing of the PSYCHIC prototype (Tier 1 screening tool) to four catchments with varied landscapes and land management. Its potential for use as part of river basin management planning is discussed.

MODELLING APPROACH USED IN PSYCHIC

PSYCHIC is used to quantify the export of sediment, particulate P and dissolved P in surface and sub-surface run-off from agricultural land and assess their contribution relative to other ‘point’ sources of P in the catchment. PSYCHIC involves a process-based modelling approach that is sensitive to those land management practices which have a large influence on the mobilisation and delivery of sediment and P. The model takes account of climate, landscape and land management factors and utilises current knowledge on the processes of sediment and P export. Predictions of the spatial and temporal distribution of flow, and SS and P loads, are provided on a monthly time step and at both 1 km² and field scales within a GIS environment. The decision support tool integrates model predictions of catchment areas with a high risk of export or ‘hotspots’ (Tier 1 screening tool) with field scale risk assessment of contributing areas on farms (Tier 2) to help identify appropriate best management practices (BMPs) to improve nutrient and land management. PSYCHIC then assesses the impact of the selected BMPs on sediment and P export at the field and catchment scale. Further description of the PSYCHIC model structure and its parameters is given by Davison et al. (2006).

APPLICATION OF THE TIER 1 SCREENING TOOL TO PILOT CATCHMENTS

The Tier 1 screening tool within PSYCHIC was applied to the four pilot catchments (Hampshire Avon near Salisbury, Teme in Shropshire/Worcestershire, Wensum in Norfolk and Bassenthwaite Lake in Cumbria) selected as part of the governments Catchment Sensitive Farming Initiative (Defra, 2004). These are all priority catchments suffering from sediment and/or P pollution and catchment officers have been designated to encourage farmers to adopt Good Agricultural Practice and additional targeted measures chosen from a range of options under Defra’s countryside stewardship schemes. PSYCHIC is used to inform which measures would be most cost-effective to tackle local water quality issues and where they need to be targeted within the catchment area. PSYCHIC input data (derived from the MAGPIE system, Lord and Anthony, 2000) and results were mapped, and risk factors and resulting mapped estimates of sediment and P transfer presented to catchment stakeholders.

The catchments differ widely in their climate, topography, land use and farming practices (Table 1). The Hampshire Avon is a chalkland catchment with mixed farming, variable relief and very low river drainage density; the Wensum catchment is largely
arable but with low rainfall, low relief and mainly sandy soils; the Teme catchment is dominated by clay loam and silty soils with a high proportion of grassland farming; whilst the Bassenthwaite catchment is almost wholly in grass with shallow clay loam soils on steep slopes and in a high rainfall area.

Table 1: Selected catchment characteristics, estimated P inputs, total P exports and the amounts of P mobilised and delivered in each of the four study catchments

<table>
<thead>
<tr>
<th></th>
<th>Avon</th>
<th>Wensum</th>
<th>Teme</th>
<th>Bassenthwaite</th>
</tr>
</thead>
<tbody>
<tr>
<td>Area (km²)</td>
<td>1714</td>
<td>685</td>
<td>1623</td>
<td>626</td>
</tr>
<tr>
<td>Average annual rainfall (mm)</td>
<td>844</td>
<td>669</td>
<td>833</td>
<td>1886</td>
</tr>
<tr>
<td>Land use</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Agricultural land (% of total)*</td>
<td>76</td>
<td>79</td>
<td>79</td>
<td>83</td>
</tr>
<tr>
<td>% arable</td>
<td>48</td>
<td>81</td>
<td>12</td>
<td>2</td>
</tr>
<tr>
<td>% managed grass</td>
<td>32</td>
<td>12</td>
<td>70</td>
<td>44</td>
</tr>
<tr>
<td>% rough grazing</td>
<td>20</td>
<td>7</td>
<td>18</td>
<td>54</td>
</tr>
<tr>
<td>P inputs‡</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total P input (kg/ha/year)</td>
<td>13</td>
<td>15</td>
<td>15</td>
<td>6</td>
</tr>
<tr>
<td>Fertiliser (% of total P)</td>
<td>34</td>
<td>32</td>
<td>24</td>
<td>20</td>
</tr>
<tr>
<td>Manure (% of total P)†</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cattle</td>
<td>45</td>
<td>12</td>
<td>38</td>
<td>63</td>
</tr>
<tr>
<td>Sheep</td>
<td>2</td>
<td>1</td>
<td>15</td>
<td>14</td>
</tr>
<tr>
<td>Pig</td>
<td>6</td>
<td>22</td>
<td>3</td>
<td>&lt;0.5</td>
</tr>
<tr>
<td>Poultry</td>
<td>13</td>
<td>33</td>
<td>20</td>
<td>3</td>
</tr>
<tr>
<td>Outputs‡</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total HER (mm)</td>
<td>319</td>
<td>221</td>
<td>326</td>
<td>400</td>
</tr>
<tr>
<td>Total sediment (t/ha/year)</td>
<td>0.03</td>
<td>0.08</td>
<td>0.19</td>
<td>0.32</td>
</tr>
<tr>
<td>Total P export (kg/ha/year)</td>
<td>0.59</td>
<td>1.34</td>
<td>0.86</td>
<td>1.11</td>
</tr>
<tr>
<td>% diffuse</td>
<td>15</td>
<td>13</td>
<td>65</td>
<td>73</td>
</tr>
<tr>
<td>Mobilisation of diffuse P‡</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total sediment (t/ha/year)</td>
<td>0.11</td>
<td>0.10</td>
<td>0.28</td>
<td>0.47</td>
</tr>
<tr>
<td>Incidental (kg/ha/year)</td>
<td>0.09</td>
<td>0.06</td>
<td>0.38</td>
<td>0.51</td>
</tr>
<tr>
<td>Soil (kg/ha/year)</td>
<td>0.11</td>
<td>0.13</td>
<td>0.28</td>
<td>0.44</td>
</tr>
<tr>
<td>Farmyards (kg/ha/year)</td>
<td>0.08</td>
<td>0.03</td>
<td>0.14</td>
<td>0.09</td>
</tr>
<tr>
<td>Delivery of diffuse P‡</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Surface run-off (kg/ha/year)</td>
<td>0.05</td>
<td>0.03</td>
<td>0.33</td>
<td>0.19</td>
</tr>
<tr>
<td>Drainflow (kg/ha/year)</td>
<td>0.04</td>
<td>0.14</td>
<td>0.23</td>
<td>0.63</td>
</tr>
</tbody>
</table>

*Agricultural land includes arable, managed grass and rough grazing.
†Includes P excreted by grazing animals.
‡Figures are expressed per ha of agricultural land
Total P inputs in fertilisers and manures were very similar in the Avon, Wensum and Teme catchments (13–15 kg/ha/year), but dropped to 6 kg/ha/year in Bassenthwaite due to the high proportion of rough grazing. In all catchments, > 70% of the total P inputs to land were derived from spread manure and excreta deposited during grazing. The manure derived largely from pig and poultry enterprises in the Wensum catchment, and from cattle in the other catchments.

The predicted estimates of total sediment and P export at the outlet of each catchment varied from 0.03–0.32 t/ha/year and from 0.59–1.34 kg/ha/year, respectively (Table 1). Of the total P export, 13–73% was predicted to be derived from diffuse agricultural sources. The very high diffuse P contributions (c. 70%) in the Teme and Bassenthwaite catchments were associated with larger amounts of P attached to eroding soil particles and mobilised from fresh applications of manures (incidental P). Hence sediment exports were at least two- to three-fold greater in these two catchments than in the Avon and Wensum catchments. Farmyards contributed most P in the Teme catchment. In the Wensum and Bassenthwaite catchments, most P is delivered by land underdrainage systems, whilst in the Avon and Teme there is a significant surface run-off component (Table 1). Knowledge of the relative proportions of surface and sub-surface export helps identify where sediment and P mitigation options will be most cost-effectively targeted.

Figure 1: Spatial distribution of sediment export in the Hampshire Avon catchment

Total catchment export represents only the average of a range in 1 km² export values and can be misleading without an analysis of the spatial distribution of export rates across the catchment. A spatial distribution analysis also provides a means of targeting measures to areas where the risk of sediment and P loss risk is greatest. For example, in the Avon catchment the total export of sediment is very low (30 kg/ha/year) due to low stream density (Table 1). However, an analysis of the pattern
of sediment export shows there are ‘hotspots’ within the catchment which have much higher sediment export rates up to 0.5 t/ha/year (Figure 1). For example, one area in the western part of the catchment shows high concentrations of suspended sediment in headwaters, and significant diatom activity in spring, associated with dairy farming including forage maize on underdrained heavy clay soils. Mitigation options here will need to focus on careful application of manures to avoid incidental P transfers, avoidance of poaching damage to grassland swards and improved husbandry of the maize crop to prevent sediment mobilisation after harvest (such as growing early harvesting varieties, establishment of winter cover crops and rotating the crop around the farm on less risky fields).

In the headwaters of the Wensum catchment, problems of siltation are associated with surface run-off from the light-textured and relatively erodible soils. Measures here therefore need to be focussed on minimising surface run-off and erosion risk. Heavy clay soils, of which a significant proportion would be expected to be drained, occur in the centre of the catchment and estimated P export from this area was therefore higher. Mitigation would require reduction in P inputs (for example by reducing livestock manure inputs and by more careful adjustment of fertiliser inputs for P derived from manures), and avoiding applications of P fertiliser or manures during the winter. Although PSYCHIC indicated that a high proportion of the P input was as pig and poultry manures, subsequent discussion indicated that a great deal of the poultry manure was incinerated at a local power station, and it was thought that pig numbers had fallen. Such local information is therefore necessary in the formulation of the risk assessment as well as the estimated impact of any proposed mitigation strategy.
Within the Teme catchment, largest sediment and P transfer was predicted to be via surface pathways in the central area, where soils have a substantial silt content and the proportion of arable land is increasing relative to the uplands (Figure 2). The stream is important as a fishery, and again siltation is of major concern. Here measures which impinge on both grassland and arable management including erosion control measures such as early sowing will be relevant.

In the grass dominated Bassenthwaite catchment, most diffuse P export occurs in the western part of the catchment, with hotspots associated with more intensive management (P inputs) on underdrained soils. Lack of opportunity to incorporate P inputs on the steeper slopes and under the high rainfall in the area means incidental P transfers are high (Table 1) and need to be controlled by careful fertiliser and manure management.

**DISCUSSION**

The PSYCHIC decision support tool is only a prototype at this stage and requires further modification and refinement as our knowledge of sediment and P mobilisation and delivery in catchments improves. However, its application is already proving very useful in helping to (a) rapidly assess the relative contribution of diffuse P sources relative to point P sources; (b) characterize the land use and farming activities in catchments in a consistent and spatially distributed manner and (c) quantify the associated risks of sediment and P transfer in surface run-off and sub-surface run-off across catchments. The separate delineation in P mobilisation between fertilizers/manures, the soil and farmyards, and in P delivery between surface and sub-surface pathways, provides a means of identifying what measures might be most appropriate and where they need to be targeted. Application of the Tier 2 PSYCHIC model within the hotspots identified by Tier 1 is now required to further predict the impact of more specific measures identified through river walks, wet weather surveys and farm visits by the respective catchment officers. We have found that the presentations, and especially the maps, promote stakeholder discussion and are appreciated as a contribution to the development of mitigation strategies. As with the example of incineration of poultry manure, local factors frequently come to light which require further consideration. It is planned to apply PSYCHIC to other catchments and to adapt the approach to maximise its usefulness to stakeholders.

**ACKNOWLEDGEMENTS**

PSYCHIC was developed by a research consortium including ADAS UK Ltd, National Soil Resources Institute, Centre for Ecology and Hydrology, Exeter University, Reading University and Lancaster University with funding from Defra, the Environment Agency and English Nature.
REFERENCES


A RISK ASSESSMENT AND MITIGATION STRATEGY TO MINIMISE LIVESTOCK POLLUTION TO SURFACE WATERS

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SUMMARY

The aim of the project was to determine to what extent best management practices (BMPs) could contribute to improved water quality in Brighouse Bay, a Designated Bathing Water in the South-west of Scotland. A range of BMPs was installed, mostly field rather than steading based, which prevented animal access to watercourses or water margins and treated yard run-off by retention ponds. Comparison with an unremediated ‘control’ catchment indicates a relative improvement in faecal indicator organism (FIO) flux to the bathing waters during the bathing season. This improvement is observed under high flow conditions, critical in terms of bathing water compliance. The most significant reduction in high flow FIO concentration was in watercourses where stock access was prevented from > 30% of the stream length and water margins.

INTRODUCTION

The report of the SEERAD Agriculture and Environment Working Group concluded that the main priority environmental issue for Scottish agriculture for the next 5–10 years was diffuse pollution to water (Scottish Executive, 2002). Large volumes of livestock manures and dirty water are generated on livestock farms and these materials can contain great numbers of FIO including many potential pathogens (Scottish Executive Prevention of Environmental Pollution from Agricultural Activity (PEPFAA) Code, 2005).

In 2004, Merrilees et al. reported on a SEERAD-funded project to deliver capital works measures and farm-specific advice to reduce FIO pollution from 48 farms in four sensitive bathing waters catchments across Scotland. The total works included drainage (1,600 m), fencing (44,000 m), gates (290), guttering/downpipes (900 m), waterpipes (25,000 m), troughs (130), cow tracks (4,900 m), concrete aprons/floors (1,100 m²), water breaks (500 m), steading roofs (12), plus ancillary pumps, sumps, tanks and concrete works. A number of constraints have been recognised and lessons learned about the undertaking of this sort of work in the farm context. A number of aspects about ongoing maintenance have been identified also. Reducing faecal pollution risks from farms requires a knowledge of the pollution pathways, promotion of good farming practices, capital works for manure and slurry reduction, containment, storage or treatment and integrated delivery of advice and financial support to ‘at risk’ farms as outlined in the ‘Forward Strategy for Scottish Agriculture’.

Brighouse Bay is a Designated Bathing Water and a Site of Special Scientific Interest (SSSSI) in the South-west of Scotland, near Kirkcudbright. This bathing water has
exhibited compliance problems in recent years in achieving the mandatory and recommended microbiological standards specified in Directive 76/160/EEC (CEC, 2004, COM581). Water quality data does suggest the strong possibility of farm animal derived pollution affecting this compliance location and driving non-compliance.

**METHODS**

With only nine farms, Brighouse Bay was an ideal place to establish and monitor BMPs and assess any improvement in water quality through comparison with a control catchment. The main aim was to determine to what extent improved farm practices and BMPs could contribute to improved water quality in the catchment. By introducing a range of relevant BMPs, a further objective was to assess their relative merits in terms of pollutant removal.

The objective of environmental monitoring in the Brighouse Bay investigation was to measure, and if possible quantify, the effects of the steading and field-based measures on the quality of water impacting on bathing water compliance in the bay.

The project had the following specific objectives:

- To collect evidence of water quality change.
- To assess the effectiveness of measures.
- To assess the cost effectiveness of measures.
- To acquire knowledge of the relative importance of measures.
- To establish a protocol for conducting a catchment appraisal of BMPs.

Farm pollution audits (Scottish Executive 4 Point Plan, 2002) have been carried out at each of the nine farms wholly within the catchment. The following factors were considered:

- Waste systems currently in place for slurry, farm yard manure and dirty water, their capacity, operation and condition.
- An audit of the waste produced and the suitable land available for spreading based on the risk of run-off after a waste application.
- An audit of the spread able days according to monthly rainfall, dry and frost days.
- Farmyard drainage audit and plan.
- Calculations to demonstrate the actual capacity of the existing waste systems.

Other farm data prepared includes:

- Farm Nutrient Budget and Fertiliser Plan.
- Water Margin Management Audit.
**Environmental Monitoring**

The primary water quality parameters examined were the faecal indicator compliance parameters of total coliform organisms, faecal coliform organisms and intestinal enterococci. In addition, the following physico-chemical parameters were measured: nitrate, nitrite, ammoniacal nitrogen and orthophosphate.

The pre-remediation monitoring team of SAC, CREH and the Macaulay Institute was deployed in October 2003. The timing of this deployment was to acquire water quality data as close as possible to the ‘summer’ catchment condition. However, the period of pre-remediation monitoring was out with the bathing season of June to September. Complementary investigations suggested that there may be a strong seasonality in the high flow concentration of faecal indicators in streams draining areas of livestock farming, with the highest concentrations evident in the summer period when stock were present in the catchment fields.

![Figure 1: Subcatchment and monitoring stations](image)

**Figure 1: Subcatchment and monitoring stations**

Given the timing of the pre-remediation sampling two options were possible. First, the monitoring could be undertaken in the same season (i.e. October to November 2004) which should minimise the effects of seasonality in the temporal comparison in water quality pre- and post-remediation. This option was, perhaps, the most ‘conservative’ but it would result in no data having been acquired during the bathing season itself during either pre- or post-remediation phases with the inherent risk that it would fail to pick up the impacts of farming and other activities on the catchment which only occurred in the period of the bathing season. Second, the monitoring
could be completed within the bathing season which might make direct comparison between the pre- and post-remediation data sets difficult but would provide data on bathing season FIO flux to underpin future policies within this catchment (Crowther et al., 2003). The latter option was agreed by the project steering group and was implemented.

**Rationale for Selecting BMPs**

After the audit of the farms within the catchment, plans were devised to install as wide a range of BMP measures as possible. The area had many opportunities for improvements to reduce the risk of diffuse FIO flow to the main watercourse or its tributaries. However, there were relatively few different ways of implementation as most of the measures involved the installation of fencing and the supply of alternative water supplies. Prioritisation had to be applied to ensure that the most appropriate measures were tested and to keep within project budget. Priorities reflect equal weight to (i) bathing water compliance, (ii) introducing a range of BMPs including ponds and wetlands. These priorities were drawn up with reference to the following selection criteria:

- Whether there was a direct FIO pathway into the bay.
- Intensity of the likely FIO loading.
- Possibility of FIO reduction/treatment between the source and the bay.
- Known ‘hot spots’ from CREH monitoring in autumn 2003.

**Measures Adopted**

Based on the Pollution Audit, pollution impacts were prioritised on a Scale 1 (very high) to 4 (low) and the following measures adopted in mitigation:

1. Removal of stock from direct access to the watercourse.
2. Creation of buffer strips between the grazed land and the immediate riparian area.
3. Removal of stock from the natural wetlands adjacent to watercourses.
4. Creation of retention ponds for treatment of contaminated water.
5. Removal of shallow cattle crossings which are usually also watering points.
7. Provision of alternative water supplies where stock access to watercourses was prevented.

**RESULTS AND DISCUSSION**

**BMP Measures**

SAC was always aware of the fact that farmers had to be willing partners for this project to be successful. If measures proposed were inappropriate to the farm, or had been seen as a feature that imposed constraints on farming activities, farmers would have been unlikely to accept such measures.
BMP measure 1 (exclusion of stock from watercourses) was readily understood and appreciated from the aspect of reducing the risk of diffuse pollution so long as an alternative reliable water supply was able to be installed.

BMP measure 2 (buffer strips) were also acceptable in most circumstances, but with varying tolerance of the amount of land take involved. The theory is understood, but not always will the ideal width of buffer strip be allowed on all farms.

BMP measure 3 (stock removal from wetlands) will not be applicable on all farms, but within the catchment area where other BMPs were installed three wetland areas were identified for exclusion from stock access. There are other wetlands in other parts of the catchment that are still open to stock. Farmers are aware that grazing value is low, but are also concerned that these areas will have to be managed in a different way to become a wildlife habitat and to prevent the growth of unsightly or invasive vegetation.

Measure 4 (farm ponds) was by far the most difficult measure that SAC promoted within the catchment. Farmers are environmentally aware but some could not see the need for farm ponds to treat lightly contaminated water. Farmers want alternative systems to treat ‘dirty water’, but under current regulations ponds would not be permitted to treat such water. The amount of land (and number of ponds) required to treat such water to an acceptable discharge quality could be very large. However, advantage can be taken of existing wetlands by isolating them from livestock.

BMP measure 5 (removal of shallow crossings over watercourses) will involve either land take or involve a considerable amount of engineering to deepen the watercourse and avoid the land take.

Measure 6 (removal of short lengths of ditch by piping) will have been carried out on farms for different reasons, such as making parts of fields more accessible for silage making. Farmers will generally be willing to adopt this measure as an aid to production. However, this activity should not be used extensively as it contravenes SEPA’s position on avoiding new culverts.

Fundamental to BMP measure 7 is to provide water from a natural source. This makes the supply sustainable and should, in theory, result in no change in the amount of water being taken from the watercourses. Apart from minor maintenance it has a nil ongoing cost, unlike a metered water supply. An important consideration in designing such a BMP is the level of abstraction given the regulations flowing from the Water Environment and Water Services (Scotland) Act 2003.

**WATER QUALITY ANALYSIS**

**Low Flows**

Mean concentrations of faecal indicator compliance parameters under dry weather conditions, are generally low. There is a slight elevation in their concentrations during low flows in the summer bathing season. However, the low flow fluxes represent a very small proportion of the total flux in the bathing season and seem not to impact on compliance with Directive 76/160/EEC (Wyer et al., 1997)
High Flows

Under high flow conditions, faecal indicator concentrations exhibit a strong seasonal pattern with much higher levels in the summer period. In the unmodified ‘control’ catchment, the increase in mean bathing water compliance parameters between the pre- ‘autumn/winter’ and the post- ‘summer’ sampling periods were as follows:

a. total coliform  10.6 -fold increase  
b. *Escherichia coli*   6.8-fold increase  
c. Intestinal enterocci  3.5-fold increase.

Commonly, over 95% of the faecal indicator flux in the bathing season occurs during storm periods, thus, the elevation in concentration in high flows is very significant.

Analysis

This observed seasonality made direct comparison of the high flow water quality data acquired pre- and post-remediation meaningless. However, the control catchment facilitated comparisons between the pre-remediation difference between ‘test’ and ‘control’ catchments (a) and the post-remediation difference (b). The value (b) minus (a) gives the magnitude of water quality improvement in the post-remediation period.

Two measures of subcatchment remediation intensity were calculated:

i. the additional percentage of the stream bank length fenced and,  
ii. the additional percentage of subcatchment land area from which the stream was ‘protected’, by the new fencing.

While this inferred relationship will always be ‘confounded’ by subcatchment factors such as, (a) the exact positioning of the fencing in relation to the sampling point and (b) locations of hardstandings and their ‘connectivity’ within the subcatchment, there does seem to be a clear improvement in the catchments where a significant proportion of the stream bank has been fenced.
(a) With reference to the increase in bank length fenced

(b) With reference to the reduction in land areas with access to watercourses

Figure 2: Scatterplots of the percentage shift in flux per unit area compared with site 501 for presumptive coliforms in the post-remediation period compared with the pre-remediation period
Examination of Figure 2 (a) and (b) suggests that over 30% of new stream bank length fencing is required before an effect can be seen. However, it is important to note that:

i. there is considerable variability in this hypothesised ‘effect’ between the remediated catchments; and

ii. this observation is based on only four relevant data points and must be considered tentative.

**Effectiveness of BMP Measures**

The Brighouse Bay study was conducted in a small catchment where interventions, such as stream bank fencing, could be implemented to cover a significant percentage of the available stream bank length. The study therefore offers an exemplar of the effects of remedial measures in this type of livestock farming area in South-west Scotland.

The faecal indicator and nutrient results provide complementary evidence of beneficial remediation effects. In particular, a measurable improvement in FIO flux to the bathing waters is suggested when compared with the control catchment, although compliance with EU bathing water standards after storm events is still not guaranteed. Some validation of The Scottish Executive/SEPA’s bathing water signage project was provided on the 19th August 2004, when a ‘poor’ bathing water quality signal was confirmed by spot sea water samples that proved to have FIO levels in excess of mandatory standards.

Early indications suggest that a beneficial effect can be seen in the Brighouse Bay subcatchments (i.e. noticeable FIO reductions) when approximately 30% more of the catchment stream bank lengths are fenced, which also results in a ‘protection’ from approximately 30% more of the catchment fields draining to a subcatchment outlet.

The creation of farm ponds has yet to be shown as an effective BMP measure. However, the beneficial effects of ponds and wetlands is demonstrated by one small subcatchment (monitored at site 103) in the Brighouse Bay study. This exhibited uniformly low pre- and post-remediation FIO concentrations at the catchment outlet. The full beneficial effect of newly isolated wetlands and ponds is likely to be seen only after plants in the water margins become established and the floor of the pond partially seals, possibly taking 2 to 3 years to take full effect.

Owing to the project timescale and the necessity to initiate monitoring ahead of carrying out the farm audits and determining the BMPs, rigorous evaluation of the individual measures applied has not been possible. Seasonal factors and land use changes in some fields (e.g. from autumn grazing to summer silage) are major confounders in the pre- and post-remediation comparisons at the sub-catchment level. To obtain full value from the project, monitoring in successive bathing seasons over a suggested period of 5 years is advised to confirm the improvements seen thus far.
ACKNOWLEDGEMENTS

SAC acknowledges funding of this project by SEERAD and the inputs from the various members of the Steering Group, comprising the Scottish Executive, SEPA, SAC, CREH, MRCS and NFUS. We wish to acknowledge the willing participation of the landowners who were involved in this project. Their inclusion was purely a reflection of their location within the catchment and not due to any real or perceived diffuse pollution problem. We acknowledge their helpful comments and all the background farm information they provided.

REFERENCES


ENVIRONMENT SENSITIVE FARMING – PRACTICAL ADVICE FOR LAND MANAGERS

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SUMMARY

Environment Sensitive Farming (ESF) delivers free advice in England on waste management and reducing diffuse pollution to air, soil and water. The advice is targeted to priority areas/topics agreed with local stakeholders in each of the eight government regions. Delivery methods include conferences, workshops, farmer groups, one-to-one advice and farm walks that are promoted through regional press and the website (www.environmentsensitivefarming.co.uk). Evaluation of the campaign is key to tailoring the type of delivery and ensuring that changes in farming practices do actually happen. After 1 year, initial results demonstrate that the campaign strategy is effective in getting recipients to change their attitudes and behaviours particularly towards waste and to some extent diffuse pollution management. The total number of ESF event attendees has been low relative to the total number of farmers and advisers and a major challenge in 2006 will be to communicate with greater numbers and influence those who do not normally attend advice events.

INTRODUCTION

One of the key elements to the successful implementation of the Water Framework Directive will be the early establishment of effective diffuse pollution control measures in agriculture. It was recognised in the Defra consultation on ‘Developing Measures to Promote Catchment-Sensitive Farming’¹ that if a Programme of Measures (POMs) is to successfully achieve good ecological and chemical status by 2015, then a joint programme of point and diffuse pollution control would be necessary. Defra recognises that to achieve these targets an effective mechanism for diffuse pollution control is needed well in advance of the formal River Basin Management Plans and associated POMs in 2009.

Agricultural waste is currently excluded² from England, Wales and Northern Ireland’s waste management controls³ which were introduced in May 1994 (Waste Management Licensing Regulations 1994⁴) and apply to all other sectors of industry and types of waste. This exclusion contravenes the Waste Framework Directive⁵ and the Landfill Directive⁶. At present, section 75(7)(c) of the Environmental Protection Act 1990 excludes some wastes including those from agricultural premises. Most agricultural waste is currently disposed of on-farm by open burning, by burial or by disposal in ‘farm dumps or tips’ and these methods can cause pollution of the environment and harm to human health.

² Section 75(7)(c) of the Environmental Protection Act 1990 excludes waste from mines and quarries and agricultural premises.
³ In Scotland waste management controls have applied to agricultural waste since 22 January 2005 [The Waste (Scotland) Regulations 2005 (SSI 2005 Number 22)].
⁴ SI 1994 No. 1056 as amended.
⁵ On 9 December 2004, Defra issued a consultation on the draft Waste Management (England and Wales) Regulations which propose to repeal this exclusion, the consultation closed for comment on 18 March 2005. The Regulations will come into force in early 2006.
The Environment Sensitive Farming (ESF) advice contract, launched in February 2005, aims to encourage voluntary change in the behaviour and practices of land managers prior to the introduction of the POMs and to inform the new agricultural waste management requirements. The advice campaign is outcome focussed and it is expected that farmers, land managers and their advisors will better understand the impact of farming practices on the environment, identify which practices are detrimental, which are beneficial and change their attitude and practices in order to reduce pollution and enhance the environment.

At present there are a number of national and regional advice initiatives in addition to specific local projects in England, the incentive for their uptake by land managers being driven by grant schemes and legislation, e.g. Environmental Stewardship (ES) and Cross Compliance. A key challenge to the success of the ESF campaign is to ensure that the target audience is motivated to implement voluntary changes in waste and diffuse pollution management. This is being achieved demonstrating how existing legislation or schemes can be integrated with the requirements of improved management practices, thus ensuring compliance and financial benefits whilst at the same time managing waste and diffuse pollution. Figure 1 illustrates how the various directives, schemes and initiatives inform the content of the ESF advice campaign.

Many previous advice projects have not focussed on the success of their impact and identified whether actual changes in attitudes, behaviours or practices have taken place in response to the advice given. A significant component of the ESF project independently evaluates the success of the delivery methods and the impact delivery events have.

**METHODOLOGY**

ESF is a Defra-funded, Rural Development Service (RDS) managed, programme of free advice targeted at farmers, contractors, advisors (agricultural and environmental) and distributors. The 3-year programme commenced in December 2004 and is
delivered at the regional level by ADAS, The Arable Group (TAG) and The Dairy Group with support from the NFU, CLA and LEAF and working closely with the relevant RDS Regional Farm Advice Co-ordinator (RFAC). Advice is given to land managers and key influencers on nutrient, manure, soil, pesticide and waste management. Conferences, workshops/seminars, farmer discussion/self-help groups, farm walks, individual farm visits are all used as advice delivery methods. ActiveSolutions, a specialist public relations company with extensive experience with the agricultural industry, has developed a comprehensive publicity and promotional programme directed at farmers and advisors on environmental issues. The approach includes the use of national and regionally published topical articles in the farming and local press in addition to targeted advertising of events and through postal invitations to individuals to attend local events. The ESF website (www.environmentalsensitivefarming.co.uk) promotes the project, provides an events calendar and bookings facility, includes technical information, relevant legislation, PowerPoint presentations and links to collaborators and other relevant sites. All information is available for downloading and use without copyright restrictions. Events are planned in accordance with regional prioritisation plans that are approved annually with the RDS project manager and developed with the RFACs and in consultation with organisations such as the Environment Agency, the NFU and English Nature. The intention is to improve the targeting and prioritising of resources and advice and to use existing networks such as LEAF demonstration farms, and NFU and CLA, together with their facilities.

MacLeod Research Ltd has, independently evaluated events and their success to date. In the 8-month period, February 2005 to September 2005, 45 ESF events were held. Evaluation forms, completed at events, from a total of 348 land managers, 294 advisors and 105 other agricultural influencers have been processed. Baseline surveys (attitudes and practice prior to attending events) from a total of 286 land managers and 220 advisors attending these events have been processed. Follow-up surveys (to identify whether changes in practices have actually happened 3 to 5 months after each event took place) were conducted by telephone with 139 land managers and 136 advisors/influencers. This paper provides a snapshot from the early months of the three-year programme and provides some useful messages from the farmers and advisers influenced to date. The findings are based on the 747 event evaluation surveys, 506 baseline surveys and 275 follow-up surveys processed for this period.

RESULTS

Understanding and Attitudes to Diffuse Pollution

Two out of three advisors and influencers completing base-line surveys between February and September 2005 said that they did not fully understand the term ‘diffuse pollution’. Eighty per cent of the land managers and 82% of the advisors and influencers interviewed 3 and 5 months after the ESF events held in February to April 2005 said that their understanding of diffuse pollution had improved as a result of attending the event. 47% of land managers and one in three advisors/influencers said that their limited understanding of ‘diffuse pollution’ had improved as a result of attending an ESF event. A further 33% of land managers and half of the advisors/influencers attending said that although they had a good understanding of ‘diffuse pollution’ before the ESF event, their understanding of ‘diffuse pollution’ had
improved further as a result of attending the ESF event.

The ESF Campaign has influenced the attitudes of 59% of land managers and 40% of the advisors and influencers interviewed 3 and 5 months after the ESF events held in February to April 2005 towards diffuse pollution. Forty-four per cent of land managers and one in three advisors and influencers said that as a result of attending the ESF event, they now have a greater sense of personal responsibility towards tackling diffuse pollution at farm level and a further 15% of land managers and 7% of advisors and influencers said that they have been persuaded that agriculture does cause some pollution problems. One in ten of the attending land managers still do not accept that agriculture is a major cause of diffuse pollution.

**Intentions to Change**

Two-thirds (67%) of the land managers attending ESF events held in February to April 2005 were planning to instigate measures to tackle diffuse pollution or make further improvements and changes to their farm practices as a result of attending the ESF event. Fifty-eight per cent of land managers said that as a result of attending the ESF event they were planning to make further changes and/or improvements to their farm practices with regard to tackling diffuse pollution. A further 9% of land managers are planning to start tackling diffuse pollution as a result of attending an ESF event. Twenty-five per cent of the land managers attending ESF events held in February to April 2005 felt that they had had been reassured by the event that they were already fully addressing all of the diffuse pollution issues and have no need to make further changes.

**Motivators and Barriers to Change**

Sixty-seven per cent (94) of land managers attending an ESF event, indicated that concern for the environment, a greater understanding of diffuse pollution and legislation were the most frequently mentioned motivators behind the intention to change. Sixty per cent thought that financial considerations would motivate their change but only 22% thought peer pressure would change their practice. Analysis of weighted data shows that concern for the environment is by far the most important motivating factor for intention to change farming practice in order to tackle diffuse pollution (seven times more so than for peer pressure). The four most common barriers to change perceived by both land managers and advisors/influencers attending the ESF events in February to April 2005 were cost, insufficient time or labour, likely impact on profitability and too many other things to do. One in three advisors also saw lack of knowledge to be a barrier to change and 44% of advisors believed that land managers’ lack of conviction about the problem or the effectiveness of any actions taken could be a barrier to change. However, 38% of land managers asked this question said that they could not think of any barriers to implementing changes regarding the management of diffuse pollution on farm.

**Other Organisations and Initiatives Influencing Change of Farming Practice**

ESF has an important signposting role to other advice initiatives and farm advisors in particular, acknowledge the influence of the Single Payment Scheme (SPS), ES schemes and the Environment Agency in persuading land managers to change their
practices. One in four (27%) land managers attending the ESF events in February to April 2005 said that no other organisation or initiative apart from ESF had played a part in persuading them to change farming practice regarding diffuse pollution.

**Advisors’ Competence and Confidence in Delivering Advice on Diffuse Pollution**

Prior to attending the ESF events one in four of the advisors (25%) said that they had insufficient understanding of the causes of ‘diffuse pollution’ and one in four (27%) of the advisors said that they had insufficient understanding of the effects of ‘diffuse pollution’ on farms. Survey results indicated that prior to attending the ESF events held between February and September 2005, a considerable proportion of advisors (60% or more) were not ‘very confident’ in providing advice in a number of areas relating to diffuse pollution management. Even more (70–80%) were not ‘very confident’ to give advice on biobeds, composting, waste minimisation and safe disposal of waste chemicals (Table 1).

**Table 1: Advisors’ confidence in providing advice in order to reduce pollution**

<table>
<thead>
<tr>
<th>Advice area</th>
<th>% Very confident</th>
<th>% Quite confident</th>
<th>% Not confident</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nutrient planning</td>
<td>33</td>
<td>45</td>
<td>18</td>
</tr>
<tr>
<td>Manure management</td>
<td>32</td>
<td>50</td>
<td>15</td>
</tr>
<tr>
<td>Dirty water management</td>
<td>17</td>
<td>47</td>
<td>17</td>
</tr>
<tr>
<td>Water margin management</td>
<td>29</td>
<td>54</td>
<td>18</td>
</tr>
<tr>
<td>Buffer strips</td>
<td>22</td>
<td>48</td>
<td>11</td>
</tr>
<tr>
<td>Biobeds</td>
<td>7</td>
<td>27</td>
<td>60</td>
</tr>
<tr>
<td>Safe disposal of waste chemicals</td>
<td>24</td>
<td>41</td>
<td>30</td>
</tr>
<tr>
<td>Composting</td>
<td>8</td>
<td>34</td>
<td>53</td>
</tr>
<tr>
<td>Waste minimisation</td>
<td>10</td>
<td>49</td>
<td>40</td>
</tr>
<tr>
<td>Soil compaction / poaching</td>
<td>31</td>
<td>53</td>
<td>12</td>
</tr>
<tr>
<td>Soil erosion</td>
<td>30</td>
<td>51</td>
<td>13</td>
</tr>
</tbody>
</table>

Baseline no 220 (‘no responses’ not included).

**Management Changes**

The majority of land managers were taking some measures to manage soil damage and erosion prior to attending the ESF events but were further motivated to take further action towards soil management, e.g. using options in ELS to prevent soil erosion and to provide buffer strip margins next to ditches, water courses and roads. Almost half of the land managers attending the ESF events on nutrients were already producing a nutrient management plan but post event were now producing or intending to produce a nutrient management plan. All but 3% of advisors are now advising or planning to advise their clients to produce a nutrient management plan. Advisors appear to be the most confident about advising their clients on nutrient management and the majority said they were recommending most of the specific actions prior to attending the ESF event where appropriate. Seventy-five per cent of land managers attending the ESF events believe that farmers as a whole need...
to do more to reduce pesticide impact on the environment. Since attending the ESF events, a high proportion of land managers plan to look for opportunities to limit the impact of pesticides through ELS and also at disposing of spray containers through waste disposal contractors. Advisors were, at the time of surveying, largely unconfident about giving advice on the new waste management regulations. The land managers’ approach to managing waste, prior to the ESF events, appears to be focused on limiting the amount of waste coming on to farm. Some land managers had considered how their waste can be managed however the majority of land managers were disposing of plastic wastes by burning or incineration rather than considering recycling or more environmentally friendly ways of disposal. Very few land managers (prior to the ESF events) had changed enterprise management or considered using a waste contractor to minimise the impact of farm waste on the environment. Virtually none of the land managers prior to the ESF events had considered shared schemes or ventures with their neighbours/local community.

DISCUSSION

The survey for early 2005 indicates that most land managers and advisors view the advice campaign positively and recognise that farming can be a major contributor to the diffuse pollution problem. In addition, many admitted that prior to attending an ESF event they did not recognise that some of their practices caused problems or they were completely unaware of water quality problems in their catchment. Some attendees also went so far as to say that ESF had prompted them to actually change their farming practices while others were planning to. The survey findings indicate that actions by land managers and their advisors to minimise the impact of nutrient losses and pesticide residues have already been undertaken prior to attending the ESF events. However, it is not clear whether this is because the audience is self-selective towards those land managers and advisors who are already proactive in these areas or because other environmental initiatives and land stewardship schemes have already targeted diffuse pollution issues relating to pesticide and nutrient management. Advisor confidence to deliver advice on all of the topics was lower than anticipated and the campaign needs to continue to prioritise targeting delivery to these key influencers.

To date, the ESF campaign has made the most significant impact on changes to attitudes and behaviours in the area of waste management. Land managers and advisors prior to attending ESF events have held a passive attitude towards waste management and events have specifically impacted on their attitudes and intended behaviour towards recycling and joint community ventures and schemes. Many of the other changes, whether adapted prior to or after attendance at the ESF campaign events are being driven by the SPS and ES schemes such as ELS. While many land managers and advisors were already actively involved in or looking towards ELS prior to attending the ESF events, the ESF campaign has highlighted specific opportunities for tackling diffuse pollution within the ELS scheme. Both land managers and advisors giving feedback from the initial meetings held between February and April 2005 show a strong preference for practical based and on-farm events with formal seminars and interactive discussion groups being the least preferred format.
In 2006, the campaign will integrate with existing sector initiatives such as industry led soil management workshops for outdoor pig and the potato sectors. Actual changes in practice will continue to be monitored by further follow up communication with attendees. We will also be considering how this advice contract can support and compliment the England Catchment Sensitive Farming Delivery Initiative in priority catchments.

**ACKNOWLEDGEMENTS**

The authors acknowledge the support of the Defra ESF advice panel and the significant inputs to the advice campaign from the RDS project manager, Sarah Escott and the RDS Regional Farm Advice Co-ordinators.
MANAGING DIFFUSE POLLUTION FROM A FORESTRY PERSPECTIVE

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SUMMARY

Forest design and forest operations have the potential to both improve and degrade the water environment. The characterisation of rivers and lochs in Scotland in preparation for the implementation of the Water Framework Directive indicates that forestry activities are thought to be a contributory factor in about 16% of river and 40% of loch water bodies at risk from diffuse pollution. The Forest and Water Guidelines provide practical advice to be followed when planning and carrying out forest operations in order to minimise the risk of diffuse pollution from acidification, siltation, nutrient enrichment, high colour, iron, and manganese concentrations, pesticides, and fuel oils. Research has shown the Guidelines to be effective at controlling these threats. Attention is increasingly shifting from viewing forestry as a problem to exploring how it can be used to protect and improve water bodies. In particular, opportunities exist for forestry to help tackle diffuse pollution from more intensive land uses, especially agriculture.

INTRODUCTION

The UK Forestry Standard, first published in 1998 (Forestry Commission, 1998), outlines the Government approach to sustainable forestry. It sets out the criteria covering key aspects of sustainable forestry in the UK and identifies indicators at a national level and then the requirements that must be demonstrated at the level of the forest management unit.

Three of the 18 criteria concern hydrology: water quality, water yield and water discharge patterns. The associated requirements are that water quality is protected or improved, water yields are maintained above any critical level and water discharge patterns are disturbed only when unavoidable and options have been explored. As examples of evidence of indicators at a forest management unit, the Standard suggests that local liaison takes place when appropriate and agreements for water are respected; an acceptable standard of forest design is applied to water margins; opportunities are taken to improve riparian zones in the course of forest operations; all operations are planned and carried out to minimise disturbance to watercourses and to avoid pollution and siltation; and that emergency pollution control measures are in place when high-risk operations are carried out. The Standard was revised in 2004 (Forestry Commission, 2004) but the criteria and indicators are unchanged.

RIVERS AND LOCHS THOUGHT TO BE AT RISK

The preliminary characterisation and impact analyses of Scottish River Basin Districts, published in 2005, identifies rivers and lochs considered to be at risk (Category 1a) and probably at risk (Category 1b) from diffuse pollution due to forestry activities. The
database constructed for the characterisation exercise lists in Category 1a four rivers and eight lochs where forestry is the primary pressure. Overall, forestry is considered to be a risk factor in 110 rivers and 28 lochs, equivalent to 16% of the total number of river and 40% of loch water bodies at risk from diffuse pollution in Scotland.

**Table 1: Characterisation of rivers and lochs in Scotland assessed at risk of diffuse pollution due to forestry**

<table>
<thead>
<tr>
<th>Category 1a</th>
<th>Number</th>
<th>Length (km)</th>
<th>Category 1b</th>
<th>Number</th>
<th>Length (km)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Primary</td>
<td>River</td>
<td>4</td>
<td>Length (km)</td>
<td>18</td>
<td>181.026</td>
</tr>
<tr>
<td></td>
<td>Loch</td>
<td>8</td>
<td></td>
<td>4</td>
<td>–</td>
</tr>
<tr>
<td>Secondary</td>
<td>River</td>
<td>24</td>
<td>189.054</td>
<td>64</td>
<td>622.409</td>
</tr>
<tr>
<td></td>
<td>Loch</td>
<td>8</td>
<td>–</td>
<td>8</td>
<td>–</td>
</tr>
</tbody>
</table>

**MINIMISING THE RISK FROM FORESTRY**

How does the Forestry Commission prevent these potential risks of diffuse pollution becoming a reality? The Forests and Water Guidelines, which were first published in 1988 (Forestry Commission, 1988), form the key supporting document for the UK Forestry Standard and include the equivalent of Best Practice Guidance. These went through a substantial revision with the 4th edition, which was published in 2003 (Forestry Commission, 2003). The revision was done in collaboration with both SEPA and EA.

The Guidelines deal with the following aspects of diffuse pollution: acidification; siltation; nutrient enrichment; high colour, iron and manganese concentrations; pesticides; and fuel oils. All must be borne in mind by forest practitioners but some must also be tackled earlier at the planning stage and addressed at the catchment scale. These include: acidification, associated with both new planting and restocking; nutrient enrichment; and as a pollution control measure, the design and management of riparian buffer areas. The most important issues are acidification, nutrient enrichment and siltation.

**Acidification**

Acidification of fresh waters occurs where the inputs of sulphur and nitrogen pollutants exceed the buffering (neutralising) capacity of the soils and the underlying rocks through which water passes before entering streams, rivers and standing water. The buffering capacity of these receiving waters is also an important factor. The most acidified areas in the UK are in the uplands where catchments with base-poor, slow weathering soils and rocks coincide with high pollutant inputs in the form of large volumes of moderately acidic rainfall. Although most pollutant inputs are now declining due to emission control, surface water acidification remains a particular problem in parts of central and south-west Scotland.

The quantity of sulphur and nitrogen pollutants deposited at a given site is strongly influenced by the nature of the vegetation layer. Forest canopies can significantly increase the capture of some of these pollutants in the atmosphere. This increased capture, often termed scavenging, is a function of the stand structure which creates...
turbulent air mixing. The effect therefore becomes more important as trees grow and the height of the stand increases. The enhanced capture of mist, which can contain large concentrations of sulphur and nitrogen, is greatest at high altitude because of the increased duration of cloud cover and high wind speeds.

Forest growth is usually limited by nitrogen availability, and drainage water from forests and woodlands generally has very low nitrate concentrations. Consequently nitrogen deposition would not normally be expected to pass through undisturbed forest ecosystems and result in acidification of water. However, nitrate leakage from older forest stands has been identified in areas of high nitrogen deposition, e.g. in south Scotland. There is concern that forest soils are becoming increasingly saturated with nitrogen and that this could lead to a marked rise in nitrate losses and thus acidification.

Significant nitrate leakage is also known to result from harvesting operations. This is due to increased rates of mineralisation and nitrification in the soil in the absence of uptake by the trees. This pulse may last for 2–5 years, depending upon the rate of re-vegetation. While an increase in nitrate concentrations in soil and stream waters poses only a very small risk of exceeding environmental quality standards, of more concern is the short-term increase in hydrogen ion concentration which may contribute to acidification and increase aluminium solubility.

The increased capture of acidic pollutants by forests could delay the recovery of acidified waters or even lead to further acidification in the most sensitive areas. Large-scale conifer afforestation represents the greatest threat while the replanting of existing forests can also be a cause for concern. In order to protect the freshwater environment, the Guidelines require the Forestry Commission to take the scavenging effect into account when considering new planting or restocking plans. Both the Forestry Commission and applicants must identify which areas are most at risk.

An indication of a site's sensitivity is obtained by the maximum pollutant load that a given ecosystem can tolerate without suffering adverse change – the critical load. For fresh waters, critical loads can be calculated which, provided they are not exceeded, should ensure the maintenance of water chemistry suitable for the protection of populations of fish and other freshwater biota. Defra have calculated critical loads for fresh waters for 10-km² grid square samples incorporating the role of N as well as S. Having compared these with total pollutant inputs of S and N, maps have been derived that indicate where critical loads for total acidity for fresh waters are exceeded.

The Forestry Commission has built in three additional safety margins to the Guidelines. We have used the deposition data from 1995–97 (ECRC, 2001) even though pollutant depositions have reduced significantly since then and are expected to continue to do so up to 2010 and beyond. The impact of new planting and restocking plans must be considered not only in exceedance squares but also in all adjacent squares. Lastly in view of the requirement to protect cSACs), site-specific data, if available, are used to refine the assessments of acidification risks for designated river catchments.

The indicative nature of the 10-km scale critical load exceedance map means that a more detailed catchment-based assessment might be required for determining the
actual susceptibility of individual waters to a forest scavenging effect within both exceeded and adjacent critical load squares.

**Planting**

For new planting, the Forestry Commission, taking advice as necessary from the appropriate water regulatory authority, will determine the need for the more detailed assessment depending on the size of the planting scheme, species mix, altitude, the proportion of forestry already in the catchment, soils, geology, and the sensitivity of local water uses. Broadleaved woodland poses less of an acidification threat due to the smaller scavenging effect, but the impact of larger planting schemes merits consideration.

In some cases, assessment will be possible on the basis of existing information. Where sufficient information is not already available, assessment is likely to involve the collection of one or more water samples at high flow (preferably in January, February, or March, when conditions tend to be most acidic) from the main watercourse receiving drainage from the area proposed for new woodland. The results from the chemical analyses will be used to calculate the receiving water’s critical load. This will then be compared with the 1995–97 total pollutant deposition of S and N for the appropriate critical load grid square. **Where the deposition exceeds the critical load, approval of a planting grant is unlikely until there are further reductions in pollutant emissions.**

**Restocking**

Harvesting temporarily eliminates pollutant scavenging until restocked crops approach canopy closure at around 15 years of age. By this time, agreed emission reductions are predicted to protect many catchments from the risk of further acidification. The combination of these factors means that in the future forest replanting will be less likely to contribute to the exceedance of freshwater critical loads compared to the first rotation. Nevertheless there are some circumstances where restocking plans require greater scrutiny, particularly higher altitude stands (> 300 m) at which level the scavenging of pollutant cloud deposition is known to increase markedly. Where conifer forest occupies a large proportion of the catchment above this altitude, consideration should be given to selective deforestation, subject to a detailed catchment-based assessment as part of an Environmental Impact Assessment under environmental impact regulations applying in the different countries. Also in the case of catchments designated as cSACs in critical loads exceedance and adjacent squares, a detailed catchment-based assessment is required for forest replanting under the Habitats Directive regardless of altitude.

The short-term release of nitrate that can follow the large-scale harvesting of some forest sites may pose an additional acidification threat within acid-sensitive areas and there may be a need to carry out a site impact assessment depending on catchment size, the timing of felling operations, species mix, local soils and geology, and the presence of fish. **Sites found to be at risk will require the size of felling area to be reduced and the adoption of management practices designed to minimise nitrate losses.**
Although nitrate leaching can be reduced by whole-tree harvesting, a much larger, longer-term threat of soil and water acidification is presented by the greater removal of base cations in harvested produce. Consequently, whole-tree harvesting is not recommended in critical load exceedance or adjacent squares, except on steep sites where whole trees have to be extracted to roadside.

Biological recovery of streams showing chemical improvement in response to emission reductions may benefit from the opening-out of stream sides. Where practicable, these areas should be targeted for earlier clearance of stands casting heavy shade and cleared zones linked to aid upstream migration of fish and invertebrates.

**Nutrient enrichment**

There is concern that the nutrient, and hence the ecological, status of fresh waters, particularly standing waters, may be significantly changed following the aerial application of phosphate fertilisers in their catchments. Fertilisers may be accidentally sprayed or blown into watercourses, or may be transported indirectly via subsequent leaching or run-off. Waters vary in their sensitivity to nutrient enrichment from forestry, with nutrient-poor (oligotrophic) waters most at risk of nutrient pollution.

Aerial phosphate fertiliser must be carefully planned to ensure that phosphate losses from consecutive applications in a given catchment do not exceed environmental quality standards in receiving lakes or reservoirs. Applications exceeding a total area of 50 ha in any 3-year period may pose a problem; the effect will depend on the soil properties, timing of application, size of the catchment and the characteristics of the water body. Early consultation with the water regulatory authority will establish whether a more detailed site assessment is required. Assessments need to include phosphate releases from large-scale felling operations, which can be significant on some soil types.

Fertiliser usage is declining in forestry in line with the reduction in new planting and a shift to more fertile soils. A move to more hand applications and better targeting of aerial treatments through the use of granulated fertiliser and improved helicopter guidance systems, are also reducing the risk of nutrient pollution.

The generally higher nutrient status of lowland soils means that lowland woodlands rarely require fertiliser applications. Thus woodland planting on ex-agricultural land may help to reduce leaching losses and tackle nutrient pollution within sensitive areas such as Nitrate Vulnerable Zones (NVZs). Opportunities exist for maximising this benefit through the targeted planting of riparian and downslope buffer areas. Forest type is an important factor, with conifer forests less able to reduce nutrient concentrations in drainage waters. This is because of their ability to capture nitrogen pollutants from the atmosphere and concentrate nitrate levels in groundwater, particularly in areas of low rainfall. High nitrogen inputs can result where forests are downwind of local pollutant sources, such as intensive pig- and poultry-rearing units. This is likely to be an increasingly important consideration with the expansion of NVZs. The main areas at risk are those receiving low rainfall where the concentrating effect of evaporation is disproportionately large. **Consideration should be given to avoiding large-scale conifer planting within NVZs receiving < 650 mm annual rainfall.**
Siltation

Siltation principally results from soil disturbance associated with ground cultivation, drainage, road construction and maintenance, and harvesting operations. It is best managed at the site level and good site planning is an essential part of any forest operation. The Guidelines require a methodological approach, which involves consideration of available techniques and resources, the potential environmental impacts of the work, and early consultation with appropriate bodies to assess the sensitivity of the area. Usually there will be a need for a detailed site assessment and the production of a well-annotated map (site plan) showing pertinent information, particularly site constraints. Common aspects are:

- the need for meticulous planning and supervision of operations;
- the adoption of less disruptive practices;
- the careful matching of machinery to site conditions;
- varying the scale and timing of operations according to site sensitivity;
- the use of a wide range of protective measures; and
- the preparation of a contingency plan in case of the accidental spillage of chemicals or oils.

Effectiveness of the Forests and Water Guidelines

Research and monitoring have played a key role in assessing the effectiveness of best management practices (Nisbet, 2001). The preferred approach has been to form a partnership with the water regulatory authorities and others in undertaking this work. In general, sensitive catchments and large-scale treatments were selected to provide good test cases. Where possible both water quality and biology are examined. Results are published, often in peer-reviewed scientific literature, for general scrutiny. Examples relevant to afforestation are Nisbet et al. (1995), Nisbet (1997) and Nisbet et al. (2002). The effects of fertiliser applications are described in Nisbet and Stonard (1995) and Lees and Tervet (1994). Road construction, harvesting and restocking impacts are written up by Pratt et al. (1996), Neal and Reynolds (1998), and Nisbet et al. (2002). These studies confirm the effectiveness of the Guidelines in controlling the threat of diffuse pollution from forestry. Demonstration studies such as the EU LIFE UK182 Project in Galloway (CVCWT, 2004) have also played a vital part in promoting and communicating the success of best practice for protecting the freshwater environment.

THE USE OF WOODLANDS IN AGRICULTURAL LANDSCAPES

The success of the Guidelines has resulted in attention shifting from viewing forestry as a threat to exploring how it can be used to improve the ecological status of water bodies. In particular, opportunities exist for forestry to help tackle diffuse pollution from more intensive land uses such as agriculture. One example is the use of riparian woodland buffers to retain sediments, nutrients and pesticides in waters draining from adjacent farmland. Investigations of the use of forestry in largely agricultural catchments are beginning to demonstrate the potential value of woodlands. Work in the upper reaches of the Severn within the Pontbren catchment showed that
infiltration of water sheeting off grazed pasture disappeared into the soil when it passed into recently planted shelterbelts (Carroll et al., 2004). Infiltration increased dramatically under 6- to 8-year-old trees but there seemed to be an effect even under 2-year-old trees which could be due to the effect of tree roots and soil animal activity and also reduced grazing pressure. Replicated experiments have been set up to examine the effect in greater details.

Broadmeadow and Nisbet (2005) have shown how the width of the buffer, the structure of the vegetation and species choice can be manipulated to maximise this benefit. Another example is the use of woodland to protect erosion prone soils and control sediment losses. A guide has been produced based on studies at Bassenthwaite in the Lake District, which describes a catchment approach that is being used by a partnership of organisations to control sediment inputs to the lake (Nisbet et al., 2004).

**NEXT STEPS**

Waterbodies categorised as Category 1b are the focus of more detailed assessments to determine whether or not they are actually at significant risk. The Forestry Commission has participated in the SEPA project to develop a screening tool to identify and characterise diffuse pollution pressures.

Attention will focus on the rivers and lochs where forestry has been identified as a factor in placing these water bodies at risk. Forest design and management practices will be assessed to ensure that the Guideline measures are appropriate and sufficient to minimise any threat.

Effort will also be given to exploring how forestry could be better integrated with farming to benefit water quality and ecology. There are clear opportunities for developing win-win solutions using woodland to help reduce both diffuse pollution and the generation of flood flows, as well as providing a range of nature conservation, recreation and landscape benefits.

In parallel, research continues to test the Guidelines in practice and assess the effect of new developments, such as continuous cover forestry, short rotation forestry and urban woodlands. Climate change is another important area of work, which has the potential to affect most forest and water issues. One example of current research is a study in the New Forest of the role of riparian woodland shade in controlling summer maximum stream temperatures. The Forestry Commission will continue to review and update the Guidelines to ensure that they reflect the most recent research and experience. This will help to ensure that the forestry sector makes a positive contribution to safeguarding the sustainable use of the UK’s water resources.
REFERENCES


INTRODUCTION

Modern agriculture, industrialization and urbanization have negatively affected environmental quality and specifically aquatic systems (Förstner and Wittmann, 1983). In South Africa, the pollution of freshwater aquatic systems can be linked to point source discharges (waste water treatment works and industrial effluents) and diffuse surface run-off (agricultural, mining, informal settlements and urban). As a result of these anthropogenic activities, innocent people as well as other life forms may be exposed to harmful contaminants that may be released without adequate consideration of human health and the environmental effects (Tchounwou et al., 1996).

Effects on human health as a result of exposure to surface water contaminants can occur through contact recreation, drinking water and the consumption of contaminated food for example, fish and shellfish (USEPA, 1991). People consuming fish or shellfish are potentially at risk as these organisms have the potential to bioaccumulate harmful contaminants from the aquatic environment (USEPA, 1991; Bevelhimer, 1995). The contaminants that have been bioaccumulated by the fish or shellfish pose carcinogenic, genotoxic and non-carcinogenic health risks to consumers (Reinert et al., 1991; USEPA, 1991). It must be stressed, however, that the consumption of fish is generally beneficial as it provides a good source of protein, vitamins, omega fatty acids and basic minerals (USEPA, 1997), as well as many others benefits. It is evident that the consumption of fish is beneficial to humans, but if these fish are contaminated they pose a health risk to consumers.

As a result of the potential health risk associated with the consumption of chemically contaminated non-commercially caught fish, the United States of America has been issuing fish consumption advisories and bans that are designed to reduce the risk to fish consumers (USEPA, 1995a, 1999).

The results of metal surveys in South Africa linking human health and consumption of fish have indicated that in highly industrialised catchments and those with mining activities (mainly gold and coal), that the metals in the fish tissues are at levels that could cause human health risks under different exposure scenarios (Heath, 1999; Heath et al., 2004a,b). Similarly studies undertaken on biocides from diffuse pollution sources such as intensive agriculture, forestry and where malaria control is practiced have indicated that the levels of pesticides recorded in the fish could possible cause human health risks under different exposure scenarios (Bouwman et al., 1990; Heath, 1999; Heath and Claassen, 1999).

From the preceding it is evident that possible human health risks due to the consumption of contaminated fish from South African freshwater systems have received little attention. This is an unacceptable situation as pollutants from various
anthropogenic activities are polluting these systems (Heath, 1999). Furthermore, fish are captured from many of the water bodies in South Africa by recreational and subsistence fisherman, while commercial fishing and cage culture are undertaken at selected systems. Therefore, certain sections of the South African population that consume fish may be at risk from the possible exposure to contaminants accumulated by fish captured from freshwater systems. Information regarding the possible health risk due to the consumption of fish from the freshwater systems in South Africa is therefore urgently required.

The general objective of this paper is to provide a generic methodology that would give guidance in the undertaking of fish contaminant surveys in South Africa to provide information regarding the possible health risk if the fish are consumed by recreational and subsistence fishermen (Heath et al., 2004a,b). It must, however, be stressed that developing and implementing methodologies to manage and reduce the human health risk associated with the consumption of freshwater fish will also benefit the aquatic ecosystem at large. The ecosystem will benefit, as the ultimate goal of the management strategy would be to protect the freshwater aquatic environment and to put remedial actions in place that would ensure that the fish populations of the system are fit for present and future human consumption.

**METHODOLOGY DEVELOPED USING FISH TO DETERMINE POLLUTION AND HUMAN HEALTH RISK**

Although the United States of America has issued fish contaminant advisories since the mid-1970s the various Agencies have employed different methods to estimate the risks to human health from the consumption of chemically contaminated fish. Subsequently the United States of America Environmental Protection Agency (USEPA) has developed a series of four documents to provide guidance to Agencies issuing fish consumption advisories for non-commercial fishing (USEPA, 1995a,b, 1996, 1997). From these documents it is evident that a fish consumption advisory programme should consist of:

- fish sampling and analysis, therefore the collection of contaminant data,
- risk assessment,
- risk management and
- a risk communication and associated health advisory programme.

However, much of the information and guidance provided in these documents has a wider application and could assist in the development of any investigation related to the assessment of contaminant levels in fish and shellfish.

In order to promote the use of fish to determine both ecological contamination and human health risk, two publications were developed in South Africa entitled: ‘An Overview Guide to Determine the Human Health Risks of Eating South African Freshwater Fish’ (Heath et al., 2004a) and a ‘Reference Guide to Determine the Human Health Risks of Eating South African Freshwater Fish’ (Heath et al., 2004b).

The fundamentals of the methodology are based on catchment information (possible anthropogenic activities that can result in chemical pollution), socio-demographic
information of consumers of freshwater fish in the catchment, bioaccumulation potential and health risks of analytes, sound sampling design, risk assessment procedures and performing monitoring at different scales and depth (Figure 1). It is important to note the method developed is closely linked to the protocols proposed by the USEPA (1995a,b, 1996, 1997) for issuing fish consumption advisories for non-commercial fish and by Heath (1999) for the monitoring of pesticides and metals in South African rivers. The approach by Heath (1999) and the current approach are catchment-based, making it possible to use much the data and information when undertaking any of the proposed levels of investigation. Therefore, if projects are carefully planned using the same methodology and principles, the data and information can be exchanged, which would ensure the optimal utilisation of resources.

**South African Methodology**

The methodology developed for South African human health risks associated with eating freshwater fish identifies ten major steps, namely:

(i) selection of scale and depth of survey,
(ii) assessment of the water-body catchment,
(iii) monitoring system design,
(iv) field collection,
(v) laboratory sample processing and analysis,
(vi) analysis of and reporting of results,
(vii) risk assessment,
(viii) risk management,
(ix) risk communication, and
(x) evaluation and review of the programme.

The application of the protocol developed, even though it has several sequential steps, must be seen as an iterative process. For example, during the assessment of the water body (Step 2) it is important understand the needs of the risk assessment (Step 7) so that the relevant information is collected at the appropriate time. The same applies to the collection of sample in the field (Step 4), which need to be collected and preserved according to the analytical Procedures (Step 6).

The basic requirements of each step is highlighted as limited resources (financial, infrastructure and skilled personnel) in South Africa would limit the possibility of undertaking detailed assessments as undertaken by the United States of America Environmental Protection Agency (USEPA). Nevertheless, by applying the proposed protocol, sound comparable assessments, based on risk assessment methodology, can be made regarding the human health risk associated with the consumption of freshwater fish in South Africa.

Three monitoring levels are identified for the investigation of the chemical contaminant concentrations in freshwater fish tissue (Figure 1). The following levels of monitoring are considered for the South African surveys, namely Screening surveys, Intensive surveys (Phase I) and Intensive surveys (Phase II). The intensity of the level of
monitoring is determined up front by governmental authorities at a national or provincial level as well as project managers of specific surveys who are responsible for designing fish chemical contaminant surveys (Figure 1). It is also important to note that these surveys are interlinked and naturally flow from the one to the next. To be cost-effective these levels should be applied in a hierarchical manner.

**STEP 7 RISK ASSESSMENT FOLLOWED**

**Hazard Identification**

The likelihood that the exposure to a chemical under specific exposure conditions poses a threat to human health is assessed. General information such as the physical and chemical properties of the chemical, routes and patterns of exposure, structure-activity relationships, metabolic and pharmacokinetic properties, toxicological effects, acute and chronic animal exposure studies, human studies, bioaccumulation potential, persistence and prevalence in the environment, and the biochemical fate of the contaminant are reviewed in hazard identification.

The databases such as HEALTH EFFECTS SUMMARY TABLES (HEAST, 1998) Agency Toxic Substances and Disease Registry (ATSDR, 1999), Integrated Risk Information System (IRIS, 1999) and Toxicology Excellence for Risk Assessment (TERA, 1999) should be used to evaluate the toxicity and carcinogenicity of the various chemical contaminants. The software packages Risk*AssistantTM (Risk*AssistantTM, 1995) and the USEPA (1997) make this information readily available.

**Dose–response Assessment**

The relationship between the dose of a hazardous chemical (i.e. the amount of the chemical taken into the body through skin contact, breathing and ingestion) and the incidence of an adverse health effect in the exposed population is characterised. Hazardous chemicals can be broadly grouped as those with non-threshold effects (causing carcinogenic and genotoxic health effects) and those with threshold effects (causing acute, chronic or developmental effects). A distinction is therefore made in describing the dose-response variables for carcinogenic and non-carcinogenic chemicals. The above-mentioned toxicity databases and software packages are used to evaluate the dose-response relationships of the various chemical contaminants. The publications by the USEPA (1991, 1997) and Tchounwou et al., (1996) would also provide ready access to this information.

**Exposure Assessment**

The intensity, frequency and duration of human exposure to a chemical in potentially exposed populations is measured or estimated. Information and data on chemical residues in the fish and human consumption patterns are used to identify and describe potentially exposed populations. The risk-based consumption limits calculated for the determined analytes in can be used for the consumption of South African freshwater fish if information on the input values and risk values is not available for South African situations.
Risk Characterisation

All the information concerning the hazard identification, dose-response assessment, and exposure assessment are used to characterise and describe the extent of the overall individual or population risk. The most significant quantitative and qualitative aspects of these assessments, the assumptions used and the identified uncertainties are assessed, summarised and discussed to provide an overall estimate of individual risk. If information and data are available this can also be expanded to estimate overall population risk. Guidance on the risk characterisation process and examples of how to compile these documents can be found in the USEPA publications (USEPA 1997; du Preez et al., 2003a). To perform these risk calculations for the chemical contaminants found in freshwater fish from South African systems and for different scenarios the Risk*Assistant™ software package can be used.

DISCUSSION

The method suggested in this paper has been tested on rivers and impoundments in South Africa (Heath 1999, du Preez et al., 2003b) and the data indicate that there are potential metal and biocide health risks associated with the daily consumption of fish. The method has been further developed and tested by the regulatory authority in South Africa and published as a series of reports by the Water Research Commission (Heath et al., 2004a,b).

It is important to note that due to the differences in fish species, fish sizes and cultural beliefs freshwater fish are prepared and eaten in different manners in South Africa, ranging from filleting of larger fish to smaller fish being cooked whole. The consumers of these fish must be aware that certain tissues (such as liver, skin, testes and eggs) can have high levels of contamination, which could result in a human health risk (Heath et al., 2004b).

The different levels of assessment proposed in this method will also enable an accurate assessment of diffuse pollution run-off from mines dumps, mine water decants (heavy metals) and agricultural run-off (mainly biocides). There is an international move, with the Stockholm Convention on Persistent Organic Pesticides (POP’s) as one of the driving forces, for aquatic biota to be monitored for levels of POP’s (DWAF, 2006). The protocol proposed in this paper will be ideal for such surveys. There is trend in America and Europe towards Total Maximum Daily Loads (TMDL), or the amount of pollution that a water body can receive and still meet water quality objective. The water quality objective includes the aquatic life requirements. The methodology developed can be used as a check to determine if the fish are in fact healthy and able to reproduce in a sustainable manner as well as to ascertain if humans can in fact consume the fish without ill effects.

The methodology should provide guidance to governmental authorities at national or provincial level and project managers for the collection of data and information as well as for the assessment, management and communication of the health risks associated with the consumption of freshwater fish. The basic requirements are highlighted, as limited resources (financial, infrastructure and skilled personnel) in South Africa would curtail the possibility of undertaking detailed assessments as undertaken by the United States of America Environmental Protection Agency
(USEPA). Nevertheless, by applying the proposed methodology, sound comparable assessments, based on risk assessment methodology, can be made regarding the human health risk associated with the consumption of freshwater fish in South Africa. People responsible for these assessments would also be able to compare their data and information with other studies in the world, especially that of the United States of America.

These surveys will also identify areas in the aquatic system where aquatic life and especially fish have unacceptably high chemical contaminant levels due to anthropogenic activities in the catchment. This information can be used in catchment management programmes and thereby contribute to the general management of the catchments. Thus, by following and implementing the proposed methodology a major contribution would be made to the protection of the consumers of freshwater fish as well as the freshwater aquatic environment. This would in turn contribute to the ultimate goal of ensuring that the fish populations are fit for present and future human consumption.

All the data and results that are generated are documented and organised in a way that will facilitate their review and assessment. The risk assessment project leader or designated person should design specific forms to ensure proper documentation. The long-term goal must be to reduce the impacts on the water-body to such a level that the contaminant levels in the fish pose no health risk to consumers (see Heath et al., 2004a,b for more details).

The programme must be reviewed considering the objectives, activities and remedial actions that have been taken by other programmes, especially those related to catchment management and the River Health programmes of the Department of Water Affairs and Forestry. This review will enable the risk manager to adapt the programme as required thereby achieving the goals of reducing the health risk to the consumers of freshwater fish and contributing to the effective management of catchments.
**SELECTION OF SCALE AND DEPTH OF SURVEY**

- **Level 1:** Screening surveys: national or regional surveys
  Sites where contaminant levels in fish could pose a health risk to consumers?

- **Level 2:** Intensive surveys, Phase 1: surveys at potential risks sites as identified during Level 1 surveys.
  Magnitude of contamination in consumed fish species?

- **Level 3:** Intensive surveys, Phase II: surveys at the sites investigated during Level 2 surveys.
  Size class contamination level and geographical extent of contamination?

**ASSESSMENT OF THE WATER BODY CATCHMENT**

- What are the main features of the catchment?
- What chemical contaminants are expected?
  - What fish are expected?
  - What are the socio-demographics of the catchment?

- Geology, soil type, slopes
- Freshwater system: rivers, dam, weirs, natural barriers, seasonal hydrology
- Type of land use: agriculture, forestry, settlements, industries, mining
- Waste dumps, industrial sites, waste water treatment works effluent points

**Contaminants expected and site selection?**

- Fish population
- Species used by recreational and subsistence fishermen
- Seasonal availability
- Size, age and gender

**Fish expected and utilized?**

- Socio-demographics of recreational and subsistence fishermen: gender, age, household numbers, etc
- Fishing activities: number and species, frequency, etc
- Preparation and consumption patterns: cooking methods, amounts consumed, etc.

**Exposure population and amount of fish consumed?**

**MONITORING SYSTEM DESIGN**

- Where to Monitor?
- What to Monitor?
- When to Monitor?

- Sampling site selection
- Selection of analytes
- Selection of analyte concentrations

- Species selection
- Tissue type and mass selection
- Number of samples

- Sample time
- Sampling frequency

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*Figure 1: Methodology for freshwater fish chemical contaminant surveys for assessing the human health risks to consumers*
<table>
<thead>
<tr>
<th>Field Collection</th>
<th>Laboratory Sample Processing and Analysis</th>
<th>Analysis and Reporting of Results</th>
<th>Risk Assessment</th>
<th>Risk Management</th>
<th>Risk Communication</th>
</tr>
</thead>
<tbody>
<tr>
<td>- How to Collect Samples?</td>
<td>- What is the condition of the samples?</td>
<td>- Recording of results</td>
<td>- Hazard identification</td>
<td>- Evaluation of risk assessment data and information</td>
<td>- Risk Communication</td>
</tr>
<tr>
<td>- Associated sample and data to collect?</td>
<td>- How to remove and prepare tissue?</td>
<td>- Statistical analysis of results</td>
<td>- Dose-response assessment</td>
<td>- Assessment of risk management options</td>
<td>- Evaluation and Review of Programme</td>
</tr>
<tr>
<td>- How to pack and transport samples?</td>
<td>- What analysis methods to follow?</td>
<td>- Data storage</td>
<td>- Exposure assessment</td>
<td>- Assessment of positive and negative impacts</td>
<td>- Risk Assessment</td>
</tr>
<tr>
<td>- What documents are required?</td>
<td>- What quality assurances are followed?</td>
<td>- Reporting of data</td>
<td>- Risk characterisation</td>
<td>- Selection of most appropriate action and recommendations</td>
<td>- Risk Management</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Sample</th>
<th>Species identification and sorting</th>
<th>Sample package and preservation</th>
<th>Documentation</th>
</tr>
</thead>
<tbody>
<tr>
<td>- Sampling equipment</td>
<td>- Size measurements</td>
<td>- Health observations</td>
<td>- Laboratory and instrumentation requirements</td>
</tr>
<tr>
<td>- Species identification and sorting</td>
<td>- Scaling and skinning of fish</td>
<td>- Preparation of individual and composite samples</td>
<td>- Laboratory and instrumentation requirements</td>
</tr>
<tr>
<td>- Size measurements</td>
<td>- Fish health observations</td>
<td>- Distribution of samples</td>
<td>- Sample storage requirements</td>
</tr>
</tbody>
</table>

**Figure 1 (continued)**
ACKNOWLEDGEMENTS

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REFERENCES


DElVERING ENVIRONMENTAL BENEFITS THROUGH LAND MANAGEMENT CONTRACTS

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SUMMARY

From 2007 Land Management Contracts (LMCs) will be the main vehicle for delivering rural development support to rural land managers. This novel approach is designed to provide an integrated approach to rural development in order to achieve the Executive’s priority objectives as set out in the Rural Development Strategy for Scotland. The Strategy is structured around 3 proposed themes which address priorities and challenges facing rural Scotland. Some of these have already been identified elsewhere – for example the need to minimise diffuse pollution was highlighted in the Custodians of Change report. The Scottish Climate Change Programme (SCCP) is due to be published early in 2006 and the Water Framework Directive (WFD) will place pressure on the land use sector to meet water quality objectives.

The three themes are:

Theme 1 Underpinning performance and quality in the agriculture, food processing and forestry sectors.

Theme 2 Enhancing rural landscapes and the natural heritage.

Theme 3 Promoting a more diverse rural economy and thriving rural communities.

The actions we propose under Theme 2 will contribute to achieving a range of environmental benefits, from halting biodiversity loss to implementing the WFD and contributing to UK commitments under the Kyoto climate change agreement and associated commitments under the SCCP.

Current agri-environment schemes are designed to encourage beneficial management of a very wide range of habitats which, in turn provides benefits for wildlife, for water quality and for landscape.

The LMC concept allows for environmental public goods to be incentivized at different levels.

Tier 1 – the Single Farm Payment – is subject to cross-compliance requirements, which provide a basic environmental standard which all recipients must meet.

Tier 2 – the LMC Menu Scheme – was introduced in 2005 and includes a range of “light touch” agri-environment measures which are generally applicable across the countryside, do not require a high degree of expertise, and which all farmers and crofters can access.
Tier 3 – Tier 3 of LMCs is planned to come on stream in 2007, following consultation in March 2006 on the priorities for funding in a new Scottish Rural Development Programme for the period 2007-2013. The specific measures to be funded will be those which contribute to the Executive’s priorities for enhancing rural landscapes and the natural heritage.
CARROTS, STICKS, SERMONS OR HUGS? DESIGNING CO-ORDINATED POLICY MEASURES FOR THE UPTAKE OF ENVIRONMENTAL MANAGEMENT OPTIONS

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SUMMARY

Environmental policy instruments that encourage the uptake of environmental management measures have traditionally been focused at the level of the individual farm, and have aimed to re-direct the principal farm decision-maker towards adoption of alternative management options using a combination of voluntary and mandatory measures. These have combined incentives, regulations and advice delivered through a variety of channels – or in Kenneth Boulding’s terminology: ‘carrots, sticks and hugs’. There is now widespread recognition that emphasis on policy designed for the single farm is not a sufficient condition for achieving desired environmental quality targets, in particular where scale and spatial co-ordination are significant factors in effective uptake. Drawing on research on environmental collective action in Scotland, we draw out some issues with a more collective approach to policy design, and identify some key research challenges that need to be met to make these approaches more viable in the future.

INTRODUCTION

The range of policy options available to government for rural land management spans a wide range in theory, but the options available to specific agencies in practice are much narrower, constrained most clearly by both pragmatic considerations (most obviously financial and personnel resources) and regulatory jurisdiction. The underlying options in providing agri-environmental delivery are essentially fourfold:

- Changing how things are produced to ensure environmental goods and services are part of the process (e.g. Best Management Practices).
- Changing where things are produced, to reduce pressure on sensitive areas.
- Changing what is produced, to provide a better mix of goods and services, at a range of scales.
- Changing who produces things, to enable specialization in areas of management expertise.

Government and its responsible agencies have a range of policy instruments which can be deployed to bring about these changes, and these can be referenced to Kenneth Boulding’s (1989) analysis of power relations. The available ways of bringing about change can be broadly pursued through three strategies – the power of coercion (the stick), the power of exchange (the carrot) and the power of integration (the hug in Boulding’s own terminology – or the sermon to emphasise its persuasive rather than supportive element). These are more often described as economic instruments, regulation (or Command and Control) and moral suasion in the classic
policy literature, although this is clearly a more narrow definition than that developed by Boulding. A summary list of the range of instruments available, broadly delineated by these categories, is shown in Table 1.

It is important to note that though these elements are often separated, even quite specific instruments seldom exercise exclusively one kind of power (Frey, 1997). Thus pricing mechanisms for conservation goods not only offer the power of exchange, but send clear signals about the value from the public perspective of the goods that are being offered for exchange. Information provision can help to identify cost savings or profit opportunities that in turn bring their own rewards. Regulatory instruments backed up with the threat of prosecution also send a signal about what is ethically valued, as do market-based instruments aimed at delivering similar quality targets but through more flexible mechanisms (Winter and May, 2001). Single instruments are not therefore typically limited to the exercise of only one kind of power, even though this is a convenient way of characterizing them.

As Boulding notes, the hug is by far the most prevalent form of exercise of power, as it strikes to the heart of human relationships and is inherent in all efforts to engage humans in some form of activity. In Boulding’s terminology, the hug is founded on ‘love’, though reservations about the impact of such romantic language on political sensibilities have led to its more common characterization (suggested by Boulding himself) as ‘respect’. Thus although the power of exchange can be observed in recruiting farmers into various forms of agri-environmental schemes through payments, the success of these initiatives is still heavily reliant on mutual respect (essentially good will and honesty) on the part of participants, particularly when monitoring efforts are by necessity limited (Colman, 1994; Lowe et al., 1997).

TWO HOLY GRAILS OF AGRI-ENVIRONMENTAL POLICY

Many of the measures listed in Table 1 have been, and continue to be, employed with varying degrees of support and success. There is substantial experience of successful (in enrollment terms) initiatives and schemes, both broad and deep, led by agency project officers, extension staff, NGOs and government departments themselves (e.g. CRER, 2002).

However, two key objectives for rural land management have been fairly consistently elusive – integrated delivery mechanisms and co-ordinated uptake. Co-ordination is used here to refer to the appropriate scale of uptake, whether defined in relation to land area or numbers of participating farmers. Integrated delivery refers to multiple objectives that the state and its agencies hold in relation to the management of rural land, while acknowledging that land managers themselves are simultaneously involved in delivering multiple objectives for themselves and their families. These two key priorities add an additional complexity to policy delivery, but one where the challenges are significantly different to the past round of agri-environmental scheme development:

(1) A level of co-ordination in actions that brings synergy across multiple holdings.
(2) A level of co-ordination in objectives that brings synergy in multiple outputs.

At a strategic level, the problem is not the number and range of possible individual instruments, but the inability to co-ordinate these sufficiently at a comprehensive
enough scale and for a wide enough scope of objectives. The challenges here are numerous, including budgetary restrictions, jurisdictional boundaries, changing agency remits, competing stakeholder interests and pressures of new European legislation.

In this context, the introduction of Land Management Contracts (LMCs) offers a significant opportunity for far greater integration of objectives in delivery. However, the other challenge – co-ordination – has remained relatively underdeveloped in both UK research and policy terms. Undoubtedly this reflects the underlying land ownership structure where examples of collective land management are now largely confined to the management of common grazings. However, the emphasis on collective action is gaining prominence and reflects a new frontier for environmental working.

**COLLECTIVE INITIATIVES – PULLING MULTIPLE LEVERS**

Engagement in collaborative activities can be defined along a spectrum from Individual to Collective (Figure 1). At the individualistic end of the spectrum, farm actions are focused within a single farm boundary and without reference to wider objectives. At the other, collective, end of the spectrum lies full community land ownership, under which the entire decision-making process involves collective action. Collaborative or collective activities involving multiple land managers occur along this spectrum, and their advantages in delivering on both agri-environmental and rural policy objectives have received some – though quite limited – attention in past research (e.g. Slangen 1994; Hagedorn, 2002; Franks, 2003).

From the individual land manager’s perspective, there are some clear advantages to both modes of operation (Table 2). Although the benefits of co-operation are easily listed, the benefits of individual operation are far more familiar to most land managers. Currently, it is apparent that incentives to move towards more collective forms of action for environmental outcomes are notably weak (Davies *et al.*, 2004).

Various types of zoning regulation are the clearest examples aimed at delivering a broad degree of spatial co-ordination, with specific catchment and habitat initiatives targeting specific objectives more closely.

A principal advantage of collective action lies in its ability to address multiple objectives simultaneously, and to harness Boulding’s power of the ‘hug’ alongside more structured policy instruments. It can create a group which has the capacity to challenge traditional practice, and in which practices can be redefined, while simultaneously delivering – with the right supporting structures – direct benefits to participants either in efficiency savings or opening up new sources of revenue (either directly or indirectly) (Hagedorn, 2002). Four key elements of collective action can be characterised as co-learning, co-planning, co-acting and co-funding. Some activities may be limited to only one of these, but the more intangible elements – co-planning and co-learning – offer the greatest potential to secure attitudinal change over time.
However, the impetus required to move land managers to a more collaborative approach in environmental management is very substantial. Collaboration is costly in terms of time, is potentially risky, may take time to develop, can have uncertain objectives and threaten to constrain flexibility (Franks, 2003). Faced with this prospect, the rewards from collaboration need to be clear, significant and timely. Commercial co-operative ventures struggle with all these factors and their limited penetration in UK agriculture has long been evidence of the difficulty of turning theoretical advantages into reality.

**Table 2: Benefits to land managers of Individual and Collective management approaches**

<table>
<thead>
<tr>
<th>Individual benefits</th>
<th>Collective benefits</th>
</tr>
</thead>
<tbody>
<tr>
<td>Speed of decision making</td>
<td>Access to wider expertise</td>
</tr>
<tr>
<td>Flexibility in management</td>
<td>More powerful resource base</td>
</tr>
<tr>
<td>Personal pride in outcomes</td>
<td>Supportive learning environment</td>
</tr>
<tr>
<td>Simplicity in control</td>
<td>Sharing of burdens</td>
</tr>
<tr>
<td>Clear lines of responsibility</td>
<td>Potential capacity to specialize</td>
</tr>
<tr>
<td>Direct capture of benefits</td>
<td>Realise economies of scale and scope</td>
</tr>
<tr>
<td>Enjoyment of autonomy</td>
<td>Stronger voice in negotiation</td>
</tr>
</tbody>
</table>

The driving forces for collaboration in the past have been essentially twofold – problem solving and income generation (see Davies *et al.*, 2004) – and these principal motives are unlikely to change quickly. But securing benefits in response to either of these two motives is not inevitable even when the potential benefits of collaboration are clear. In the past, environmental collective initiatives have almost universally relied heavily on the driving force of project officers with time, expertise and access to funding to generate activity. The characteristics evident in generating enduring and productive collaborative action are complex and varied:

- Critical mass to realize collective benefits at appropriate scale.
- Clarity of purpose.
- Rapid and self-evident initial results.
- Outcomes exceed those available through individual effort alone.
• Individual rewards accrue from collective organisation.
• There are opportunities for continuing innovation and development.
• Individuals feel in control of the process.
• Sense of common goals and ownership.
• Appropriate and available training.
• Some secure funding streams to sustain co-operative engagement.
• Voluntary participation due to shared opportunities for gain.
• Strong and inspirational leadership.
• Flexibility in response to new opportunities.
• Continuing positive feedback and information flow to participants.

The extent of collective actions has scale and scope limitations, and not all will meet all these conditions. If collective approaches are to yield more of their potential, significant innovation is needed in mechanisms to deliver on the problems raised by these factors; without such innovation, the potential for moving towards more collaborative environmental initiatives is likely to remain very weak.

CHALLENGES FOR INNOVATION IN SUPPORTING MORE COLLABORATIVE ACTION

It should be noted that not everything worth doing is worth doing collaboratively. What is important is to provide mechanisms to move to the appropriate point on the Individual–Collective spectrum – appropriately defined in relation to the needs of both farmers and agencies, and the circumstances within which they are operating.

Collective approaches need therefore to be constructed from, and responsive to, complex combinations of incentives, regulation and social pressures that can effect these transitions. At present there are many good examples of past collaborative initiatives built on project officers’ expertise (Wondolleck and Yaffee, 2000); but to build on this experience in the UK context, developments are needed in several key fields:

(1) In monitoring systems that can enable greater clarity in benefits delivery:
• audit systems that send more precise and appropriate signals to managers; and
• strengthening returns to individual efforts within collective ventures.

(2) In understanding the full benefits of co-ordinated environmental actions:
• provision of evidence on scale and scope benefits from co-implementation; and
• recognition of the widest possible set of values delivered at the landscape scale.

(3) In leveraging additional rewards from other sources to reward multi-farm, multi-objective initiatives:
• circumventing the audit controls constraining government expenditure; and
• tailoring activities to local circumstances and opportunities.

(4) In mechanisms that enable farmers to tap into location-specific environmental expertise:
• stimulating farmer engagement in defining, rather than just delivering, on local priorities; and
• monitor and demonstration style farms, connected to environmental support groups.

(5) In creating farm-level incentives to increase recruitment to collective initiatives:
• by increasing co-operative rewards linked to larger scale initiatives.

(6) In using collective initiatives as gateways to other desired services:
• specialist training and software (e.g. in nutrient budgeting) via group membership; and
• compulsory elements of training in combination with farmer-defined priorities.

CONCLUDING COMMENT

Land Management Contracts should provide for a level of integration in agri-environmental policy delivery exceeding past approaches; however, delivering co-ordination across holdings remains exceptionally challenging under prevailing institutional structures. In seeking to re-orientate land managers to greater prioritization of environmental outcomes, Boulding’s observation on the power of the hug in particular may be particularly timely. To quote Senator Robert Byrd (speaking in 2003 on a very different topic, the invasion of Iraq):

‘the real power of America lies not in its will to intimidate, but in its ability to inspire’.

The step from coercion to inspiration is the necessary internalization of values that creates a community of shared interest from a disparate collection of individuals, a point observed two millennia ago by Aristotle (2000). Many, if not most farmers continue to fail to be inspired by the opportunities offered by environmental land management as a core activity, not only because it is a significant change of focus, but because it fails to offer dynamic opportunities for continuing positive innovation, rewards structured to monitorable outputs, flexibility in delivery and accessible benchmarks for these processes. Group structures are not a panacea for addressing these concerns, but they do offer a channel through which innovative responses to these challenges might be more effectively nurtured.
### Table 1: A typology of agri-environmental policy instruments

<table>
<thead>
<tr>
<th>Economic</th>
<th>Emphasis on use in UK*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Transferable property resource rights</td>
<td>–</td>
</tr>
<tr>
<td>Land covenants and trusts</td>
<td>+</td>
</tr>
<tr>
<td>Negotiated conservation management agreements</td>
<td>+++</td>
</tr>
<tr>
<td>Competitive public goods contracts</td>
<td>+++</td>
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<tr>
<td>Posted performance bonds</td>
<td>–</td>
</tr>
<tr>
<td>Crop insurance services</td>
<td>+</td>
</tr>
<tr>
<td>Hypothecated tax instruments</td>
<td>–</td>
</tr>
<tr>
<td>Tax relief and exemptions</td>
<td>+</td>
</tr>
<tr>
<td>Input substitution subsidies</td>
<td>–</td>
</tr>
<tr>
<td>Equipment upgrade grants</td>
<td>+</td>
</tr>
<tr>
<td>Output tariffs</td>
<td>–</td>
</tr>
<tr>
<td>Graduated user fees and charges</td>
<td>–</td>
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</table>

<table>
<thead>
<tr>
<th>Regulatory</th>
<th>Emphasis on use in UK*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Offset arrangements/planning concessions</td>
<td>–</td>
</tr>
<tr>
<td>Leasing and licensing schemes</td>
<td>+</td>
</tr>
<tr>
<td>Producer accreditation schemes</td>
<td>++</td>
</tr>
<tr>
<td>Zoning and development controls</td>
<td>++</td>
</tr>
<tr>
<td>Statutory procedural regulations</td>
<td>+++</td>
</tr>
<tr>
<td>Output/input controls and quotas</td>
<td>++</td>
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</tbody>
</table>

<table>
<thead>
<tr>
<th>Persuasive</th>
<th>Emphasis on use in UK*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Electronic and print media information services</td>
<td>++</td>
</tr>
<tr>
<td>Advisory and extension services</td>
<td>++</td>
</tr>
<tr>
<td>Process and product research and development</td>
<td>+</td>
</tr>
<tr>
<td>Education and training entitlements</td>
<td>–</td>
</tr>
<tr>
<td>Prizes and award Schemes</td>
<td>+</td>
</tr>
<tr>
<td>Voluntary audits and monitoring</td>
<td>++</td>
</tr>
<tr>
<td>Monitor and demonstration farms</td>
<td>+</td>
</tr>
<tr>
<td>Producer clubs</td>
<td>–</td>
</tr>
<tr>
<td>Discussion forums</td>
<td>+</td>
</tr>
<tr>
<td>‘Naming and shamming’ initiatives</td>
<td>–</td>
</tr>
<tr>
<td>Green benchmarking</td>
<td>+</td>
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</table>

<table>
<thead>
<tr>
<th>Mixed</th>
<th>Emphasis on use in UK*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Product certification schemes/green marketing</td>
<td>++</td>
</tr>
<tr>
<td>Industry sponsored producer clubs</td>
<td>+</td>
</tr>
<tr>
<td>Cross compliance conditions</td>
<td>+++</td>
</tr>
<tr>
<td>Local food initiatives</td>
<td>+</td>
</tr>
<tr>
<td>Regional development/diversification initiatives</td>
<td>+</td>
</tr>
<tr>
<td>Environmental clubs and cooperatives</td>
<td>–</td>
</tr>
</tbody>
</table>

* very little or not used; +, low emphasis; ++, medium emphasis; +++., high emphasis.
ACKNOWLEDGEMENTS

The support of the Scottish Executive Environment and Rural Affairs Department is gratefully acknowledged for the research leading to this paper. The views expressed here are entirely those of the author and not attributable in any way to the Department.

REFERENCES


AGRICULTURE POLICY REFORM: OPPORTUNITIES TO REDUCE DIFFUSE POLLUTION

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SUMMARY

Many of Scotland’s rivers, lochs, coastal and groundwaters are affected by diffuse pollution from agricultural sources. SEPA considers this to be the most significant cause of poor river quality, and expects it to be the largest cause of water pollution in Scotland by 2010. Water bodies affected by diffuse pollution from silts and excess nutrients become eutrophic with devastating consequences for wildlife. Many water-dependent species and habitats are of national and international importance. Existing policy measures which aim to tackle the problem are inadequate but comprehensive changes in agriculture and water policy are underway in Europe and Scotland. The Water Framework Directive is changing the way we manage our water environment, while reform of the Common Agriculture Policy gives greater emphasis to environmental objectives. Opportunities exist to align water and agricultural policy through these mechanisms, reducing diffuse pollution while enhancing biodiversity and delivering other public goods.

INTRODUCTION

Scotland is renowned for, and in many ways defined by, its water. A plentiful supply shapes its characteristic landscapes, enables production of world class goods, and supports highly popular recreational activities – all of which are vital to the Scottish economy. Many of our most valuable wildlife habitats are water dependant and often of national and international significance. Rivers and lochs, coastal salt-marshes and lowland grasslands, reedbeds and vast tracts of upland blanket bogs support a diverse array of Scottish biodiversity including aquatic species such as Atlantic salmon and freshwater pearl mussel, and huge numbers of wading birds and wildfowl.

There are many threats to the aquatic environment, but there is little doubt that diffuse pollution from agricultural sources, primarily excess nutrients, pesticides and silt, is causing major problems for both water quality and the vulnerable water habitats and species which depend on it. Diffuse pollution is a priority water policy issue for the UK Government, with the main driver for current action being the EU Water Framework Directive which requires a co-ordinated approach to protect and restore aquatic ecology – both water quality and quantity – at the river basin scale.

A range of practical measures can be used to reduce the cause of the problems derived from farmland. Various policy mechanisms are currently employed to address the issue: advice, awareness programmes, incentives and regulation in some areas. However, these appear to be having limited effect. SEPA considers that the problem is growing and that pollution from agricultural sources is likely to be the largest cause of water pollution in Scotland by 2010. The issue is politically and practically complex and there is currently no overall solution to the problem of diffuse pollution.
THE EFFECT OF DIFFUSE POLLUTION FROM AGRICULTURAL SOURCES ON BIODIVERSITY

Diffuse pollution arises from the cumulative effect of many small discharges. The total effect of a number of individually minor sources of contamination becomes increasingly significant over an entire catchment area and causes considerable harm to the environment. In particular, nitrogen (N) from inorganic fertilisers, slurries and manure is transported in surface and groundwater in the form of highly soluble nitrate. Nitrate is a major polluter of many saline and freshwater systems, where it increases the growth of certain aquatic plants and algae at the expense of less competitive species.

Phosphorus (P) from inorganic fertilisers, slurries and manure is a major plant nutrient and generally the limiting factor for eutrophication in freshwater systems. While nitrate is highly soluble and generally transported by leaching, phosphorus binds strongly to soil particles, and is often associated with pollution by soil erosion or run-off. However, as soils become saturated with phosphorus, more P becomes dissolved in water and reaches watercourses in a soluble form.

Silts are particles of soil that are washed into rivers and lochs from the surrounding land. Poor soil management combined with heavy rainfall greatly increases the natural level of erosion, changing the structure of river habitats by clogging up substrates and making them unsuitable for some fish and insect species. Silts can also make water turbid and prevent submerged plants from growing. Declines in mayfly populations are considered to be the result of a combination of silt and nutrient pollution, while silt deposition in rivers has serious impacts on salmonid fish which lay eggs in gravel. Excessive silt reduces oxygen flow to the eggs and provides a physical barrier to the hatching fry, resulting in serious reductions in breeding success. Other species vulnerable to silt pollution include freshwater pearl mussel and native white-clawed crayfish.

Pesticides can be toxic to aquatic life, and have potentially serious consequences for wildlife, including fish. These can include plant, insect or fish kills, hormonal disruptions, impacts on food chains as a result of changes or reductions in numbers of plants, or invertebrates providing food to mammals and birds. Assessing impacts of individual pesticides on the ecology of water environment is very difficult, but there is sufficient evidence to suggest that action is needed to provide better control of pesticide input into water.

Modern farming practices have tended to increase the risk of nutrient and silt loss. Ploughing, seedbed preparation, crop spraying, fertiliser spreading and application of slurry to land can all contribute to diffuse pollution. SEERAD (2005) stated that since the 1950s the amount of inorganic nitrogen applied to agricultural land has increased sevenfold (+600%), and phosphorus application has almost doubled (+70%). SEPA estimates that some 45,000 tonnes of nitrate and 2,800 tonnes of phosphate are lost from agriculture to surface waters and groundwater every year. Soil losses from agricultural activities are estimated at 780,000 tonnes a year (SEERAD, 2005).

The relatively low cost and ready availability of inorganic fertilisers, together with the geographical separation of crop and livestock production systems, has limited the re-cycling of N and P from manures and slurries as fertilisers. The cost of storing
and transporting manures and slurries has also reduced the likelihood of these
being spread on the farm at times and in places where crop uptake will reduce the
risk of nutrient loss. Where an excess nutrient is available, and not taken up by
a crop, it inevitably reaches surrounding watercourses. Current estimates (Defra,
2003) suggest that 40%–50% of P and 60%–70% of N in waters are derived from
agricultural activity, specifically the application of fertilisers and manure. While
some nutrient loss is the inevitable result of tilling the soil and keeping livestock, a
substantial proportion could be avoided by better soil husbandry, improved on-farm
nutrient management and more efficient use of existing sources of N and P to reduce
reliance on inorganic fertilisers.

The loss of hundreds of kilometres of riverine systems and associated wetlands over
recent decades exacerbates the problem. Naturally functioning wetland systems
can soak up pollutants and clean water in the process, as well as acting as natural
sponges capable of absorbing floodwater. However, development on floodplains,
extensive land drainage and the construction of embankments along rivers to protect
agricultural land from flooding all limit the capacity of natural habitats to cope with
flood events and pollution.

There can be little doubt that diffuse pollution is widespread and having a major
detrimental impact on the health of our water environment. SEPA (2005) identified for
the Scotland River Basin District (RBD) a total of 488 rivers, 57 lochs, 18 transitional,
59 coastal and 21 groundwater bodies currently affected by diffuse source pollution
caused mainly by agricultural activities. A further 198 rivers, 13 lakes, 3 transitional,
2 coastal and 16 groundwater bodies are affected by or at risk from diffuse pollution

Many of these are of international importance for wildlife. The Scotland RBD includes
235 water-dependant Special Areas of Conservation (SAC) and Special Protection
Areas (SPA). The Solway Tweed RBD includes 36 water-dependant SACs and SPAs.
Of these, many may not meet the Water Framework Directives (WFD’s) environmental
objectives in their current state. The intense pressure of diffuse pollution on some
of our most important aquatic habitats and species mean that these sites may
fail to meet the statutory favourable condition required by EU Habitats and Birds
Directives.

SPAs affected by or at risk from diffuse pollution include the Caithness Lochs, Loch
of Strathbeg, the Ythan Estuary, Insh Marshes, Loch of Skene, Loch Lomond, Loch
Ken and Dee Marshes, Loch of Kinnordy, Flanders Moss and Lake of Menteith, and
Loch Leven. SACs affected by or at risk from diffuse pollution include the Dunkeld–
Blaigowrie Lochs, and the Rivers Spey, Tay, Dee, South Esk and Tweed.

The UK Biodiversity Action Plan (BAP) includes a number of priority habitats and
species which depend on good quality water. Priority habitats include mesotrophic
lakes, rivers, fens, reedbeds and grazing marsh. Priority species include the otter,
water vole, medicinal leech, northern blue damselfly, great crested newt, freshwater
pearl mussel, vendace, yellow marsh saxifrage, slender naiad, river jelly lichen, marsh
clubmoss and Shetland pondweed. Action Plans for each of these habitats and
species highlight the threat from nutrient enrichment. Many of these are of principle
conservation importance in Scotland; the new Nature Conservation Act (Scotland)
requires the Minister to designate a list of such species and habitats in order to further their conservation.

In addition to the environmental cost, damage to aquatic ecosystems reduces the value of water bodies for amenity, recreation and drinking water supply with significant health and economic implications. The total costs of agricultural pollution in the UK have been estimated to be in the order of £250 million pounds per year, with significant clean-up costs borne by the taxpayer.

NEW OPPORTUNITIES IN WATER AND AGRICULTURE POLICY

Best Management Practices for reducing the problem of diffuse pollution are well documented (e.g. Vinten et al., 2005). However, policy mechanisms have so far failed to ensure widespread adoption of the measures needed. The implementation of the WFD through the Water Environment and Water Services (Scotland) Act 2003 offers the opportunity for effective policy change, particularly if carefully aligned with Common Agriculture Policy (CAP) reform measures and the Scottish Rural Development Plan. The delivery of CAP Pillar 2 funding through the proposed new system of Land Management Contracts (LMCs) must be a key element of any strategy to address diffuse pollution from agricultural sources.

DIFFUSE POLLUTION STRATEGY

A Diffuse Pollution Strategy for Scotland should be developed, with a key role for SEPA, in order to ensure the level of policy integration and implementation necessary on a widespread and highly targeted basis. Priority catchments need to be identified and targeted, with other measures being applied more generally for maximum effect. Promotion of organic farming techniques would help reduce the reliance on, and impacts of, inorganic fertilisers. A combination of incentives, advice and regulation designed to support, engage and ultimately deter farmers and landowners from carrying out damaging farming practices is required, including the following key elements.

Regulation

The new regulatory system required by the WFD should provide the necessary legislative basis for change; proposals by the Scottish Executive to introduce national and targeted General Binding Rules (GBRs) have the potential to ensure the application of good practice and should be properly integrated with the new LMC system of agricultural incentives.

Incentives

Agricultural incentives offered through the proposed new LMCs should promote good practice nationally, but particularly in priority catchments. There is considerable scope for addressing diffuse pollution through LMC Tiers 1, 2 and 3. Measures that are designed to meet multiple objectives including biodiversity, sustainable flood management and improvements to public water supplies provide the best value for funding:

- Widespread gains could be made in Tier 1 by introducing a requirement to prepare and implement a compulsory Nutrient/Soil Management Whole Farm Plan as part
of Good Agricultural and Environmental Condition (GAEC) in order to receive the Single Farm Payment.

- Tier 2 provides non-competitive incentives to land managers and should promote measures that deliver multiple benefits, such as the widespread use of buffer strips at riparian field margins, management of farm wetlands, provision of farm ponds, conversion from winter to spring sown cereals, provision of vegetative over-winter cover and strip cropping. Fencing off watercourses and provision of alternative drinking points for livestock are measures also suited to Tier 2 though capital costs may necessitate their inclusion in Tier 3.

- Tier 3 aims to enhance the environment and should provide significant competitive incentives for priority areas (identified from SEPA's ‘characterisation’ process) linked to targeted GBRs. There is great scope for major environmental gains by encouraging partnership working at the catchment scale. Enhanced buffer strips and provision of adequate manure storage facilities should be encouraged. The restoration of naturally functioning wetland areas including riverine and coastal floodplains is of particular importance. Wetlands deliver the multiple benefits sought in policy development and should be a major element of any strategy to combat the cause and effects of diffuse pollution.

Advice

The existing farm advisory service is under review and a new scheme needs to be introduced as a requirement of the Farm Advisory Service provisions of the CAP mid-term review. This should be considered in the context of the 3 LMC tiers with widespread easy access to free advice on nutrient and soil management planning and the provision of public benefits. Catchment advisors should be appointed in priority areas.

Planning

The development of River Basin Management Plans (RBMP) and the establishment of Area Advisory Groups in spring 2006 will provide a major opportunity to address diffuse pollution in priority catchments. Programmes of Measures developed to implement the RBMPs should promote and encourage a package of measures to reduce diffuse pollution, including direct regulation, the restoration of wetlands, sustainable flood management, and changes in land use where necessary to achieve the ecological objectives of the WFD.

CONCLUSION

The current shift in emphasis of agricultural and water policy means that significant ecological improvements in the water environment at a catchment scale are possible, and required. This is an unprecedented opportunity to address the issue of diffuse pollution from agricultural sources which is currently causing such great environmental damage at significant cost. Policy integration is the key to effective implementation – agriculture policy reform must be fully integrated with measures to implement the WFD and WEWS Act. Development of a strategy to tackle diffuse pollution is a priority – a combination of regulation, incentives and advice should be applied on a widespread basis and targeted in key areas. Measures designed to address the problem must also deliver multiple objectives including habitat
protection and restoration, sustainable flood management, cleaner water supplies and recreational waters, and public health improvements.

ACKNOWLEDGEMENTS

We are grateful for the input of policy officers from SEPA and SNH to this work.

REFERENCES


FARMER UPTAKE OF NUTRIENT MANAGEMENT BEST PRACTICE

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SUMMARY
Encouraging farmers to make changes in their farming practices is a major challenge for all involved in minimising the risk of diffuse pollution from agricultural activity. Effecting a change involves good communication to improve understanding; one-to-one advice illustrating the impact to the farmer's own situation appears to be most effective. Recent Defra campaigns have concentrated on ‘training the trainers’ with a view to maximising the dissemination of informed advice. Some best practice measures are easy for farmers to implement, while others may involve a major capital outlay or a complete change of management style. Incentives (either financial or through improved market opportunities) can encourage changes where these extra costs would otherwise be an insurmountable obstacle. However, changes in fertiliser use in designated NVZ areas suggest that regulation and non-compliance penalties may have the biggest impact.

INTRODUCTION
European legislation to control the quality of water has led to a wide range of measures being introduced across Europe to minimise the risk of diffuse pollution from agricultural activity. While we know much about best practices and mitigation methods for decreasing diffuse pollution, the challenge is bringing about change in farming practices. Some best practice measures are easy for farmers to implement, while others may involve a major capital outlay or a complete change of management style. The methods used to encourage uptake of best practice are often a combination of the ‘carrot and stick’ approach, where the ‘carrot’ usually takes the form of government support, and the ‘stick’ is regulation and non-compliance penalties. Because of the need to change farming practice, there have been several recent projects on these ‘softer’ issues and Defra has introduced a ‘Knowledge Transfer’ research programme. The aim of this paper is to review methods bringing about change in farmer practices using recent case studies, literature review and practical experience.

METHODOLOGIES
Communication
There are many ways to communicate best practice measures to farmers ranging from one-to-one through to larger study groups and conferences as well as media coverage in the press or on TV/radio (Dampney et al., 2001). These can be summarised as follows:

• One-to-One: most organisations regard direct one-to-one advice as the most effective method of achieving change on farms though recognise that it is
expensive. One-to-one contact becomes more important as issues become more difficult or controversial, and advice becomes more complex.

- **One to Group:** farm walks, discussion groups, technical seminars and demonstration farms are examples of ‘one to group’ activities that are being increasingly used, as they are more cost-effective than one-to-one sessions.

- **Farmer clubs:** the permanent group or ‘Farmer Club’ approach can encourage belonging, trust, bonding, ‘enjoyment’ and the likelihood of a closer interchange of farm experiences. A recent ‘Integrated Advice Project’ (Anon, 2004) found group spirit an important means of getting farmers to attend meetings on topics that they perceived to be less relevant to their businesses. This demonstrates a problem commonly seen, that participants have firmly held views on the relevance of topics to their business and are unwilling to readily explore wider issues, even though they might be of benefit. In the Integrated Advice Project this was particularly the case with the Environment module, although those that did attend these meetings found that discussions went well beyond their expectations.

- **Telephone:** telephone contact is widely used as a supporting measure. Many organisations have a telephone help-line that is considered to be cost-effective as a first point of contact but often requiring use of other more direct methods to adequately meet the needs of the enquirer.

- **Press:** the farming press is generally regarded as an important method to raise awareness of existing issues and new developments, technical, regulatory or economic. It is low cost, gets to a large audience and allows scope to repeat the same message in different ways.

- **Shows/events:** presence at a range of national and/or regional shows and other events attended by farmers and their advisers is considered useful to raise awareness and interest, and as a means of encouraging farmers to take the next step towards implementing change on their own farm.

- **Demo farms:** ‘one-off’ or ‘semi-permanent’ demonstration farms and farm walks allow farmers to see for themselves how practices perform at a whole farm scale. Seeing something work in practice under similar practical conditions can often give a farmer the confidence and final push to make a change on his own farm.

- **Conferences and ‘train the influencers’:** small interactive seminars and larger conferences are regarded as a good, cost-effective training mechanisms by targeting professional advisers, the trade and some leading farmers. They are not attended by the vast majority of farmers who will usually only benefit indirectly via their adviser, press coverage or informal contact with farmers who do attend.

- **Web/email:** websites are usually regarded as providing static reference material (e.g. publications lists, publication downloads, contact points), and links to centralised and standard information, which might support the implementation of change once a decision in principle, has been taken by the farmer. E-mail is a quick, cheap and effective means of responding to queries from professional advisers or for relaying new information. It is also a convenient method for sending out ‘alerts’ to notify a group of people of some change or development.

- **TV/Radio:** these media are not commonly used. Where used, it is to raise awareness and interest in the wider rural community as well as for farmers.
Regional TV and local radio can be used to highlight local issues that interest or affect farmers.

The Farm Adviser

The value of the adviser in face-to-face communication has been noted in a number of studies of information provision in agriculture (Jones et al., 1987; Cox et al., 1990; Fearne 1990; Angell et al., 1997). Farm visits and face-to-face advice was the preferred method of advice delivery stressed by farmers in farmer panels and agency interviewees, in an appraisal of conservation advice undertaken for English Nature (Winter et al., 2000). Farmers emphasise the need for a trusted individual to act as a mentor or interpreter and, for the most part, farmers put greatest value on information that has been interpreted and given in a specific form. Thus, Winter et al. (2000) found that farmers prefer face-to-face contact with someone who has interpreted the information within the context of their own business, reflecting the farmer’s lack of time, the increasingly technical and legislative nature of the required information and the perceived need for location specific advice. Where organisations only provide written material, they recognise that its value is greatest if it is read by professionals who then go on to provide advice to farmers (Angell et al., 1997). The success of agri-environment schemes, such as ESAs and CSS, has been shown to be critically dependent on the work of project officers or other advisers promoting the scheme (Cooper, 1999). The crucial role of advisers/consultants in supporting farmers in their use of complex nutrient management systems was also noted in a study of European Input-Output accounting systems (Goodlass et al., 2003). Morris et al. (2000), examining the uptake of arable options in the CSS, also found that face-to-face interaction with advisors on the farm was an important channel of communication. The research showed that, although the mass media appeared to have been the chief vehicle through which farmers became aware of the CSS, personal communication with advisors played a significant role in persuading farmers to take up the schemes that they had heard of through the farming press.

In the context of environmental pollution, the FACTS scheme of adviser registration and continuing professional development (CPD) has an important role to play in ensuring that advisers are up to date and well informed about the risks associated with poor fertiliser and manure management practices.

Voluntary Versus Compulsory Measures

Best practice measures may be voluntary, as for example in the Codes of Good Agricultural Practice. The adoption of such measures is likely to depend on how easy they are for the farmer to implement. Financial penalties are a significant disincentive to uptake whilst marketing advantages of compliance such as in some Quality Assurance schemes increase uptake (Goodlass et al., 2003). Overall the methods used to encourage uptake of best practice are often a combination of the ‘carrot and stick’ approach, where the ‘carrot’ usually takes the form of government support, and the ‘stick’ is regulation and non-compliance penalties.

In the absence of the legislative ‘stick’, a Danish study on what influences farmers to change their practices suggests that a combination of farmer study groups with associated one-to-one advice is very effective (Halberg et al., 2005). However, this is expensive in terms of advisory input time and only reaches a small number of
farmers. In England, recent policy within Defra has been to use the ‘train the trainers’ approach where workshops are used to train advisers and influencers who then pass the message on to their clients. Such workshops invariably provide FACTS cpd points for the advisers who attend. This approach has been used successfully in several recent campaigns such as raising awareness of organic manure management, promoting the latest fertiliser recommendations and explaining the English Action Programme for NVZ legislation.

Where measures impact badly on farm profitability there are clearly implications for financial sustainability. In England, the recent implementation of the stewardship schemes as part of the England Rural Development Programme (ERDP) provides farmers with an opportunity to receive payment for following best practice measures. Those most relevant to pollution from nutrients in the Entry Level Scheme are nutrient, manure and soil management plans. Advice on each of these topics is currently being made available in regionally organised workshops under the Environment Sensitive Farming campaign (Carter, 2006).

RESULTS

Information Sources

Several surveys have been carried out in recent years on farmers preferred source of information and advice. The British Survey of Fertiliser Practice (BSFP) (Goodlass et al., 2002) found that 74% of farmers used their own judgement, based on their knowledge and experience of their own farm when making decisions about fertiliser application rates. Information from advisers and consultants was the next most common method (41%). A survey of farmers in NVZs (Scott et al., 2003) also found that the farmers own past experience was regarded as the most important basis for fertiliser decisions, but the use of agronomic consultancy and farm waste management plans had increased when compared with the start of designation. Agronomic consultancy, in particular, was the source most frequently described as most useful in 2003 (in a 1996 survey, ‘past experience’ was chosen by more farmers).

In a 2004 survey, Chambers (2004) reported that RB209, MANNER and farmer meetings/conferences were the most commonly used sources of information and advice. Farmer workshops and on-farm demonstrations were also highly rated. Arable farmers (76%) made far more use of RB209 than grassland farmers (24%). Farmers, and in particular arable farmers (56%), made frequent use of independent consultants. Consultants used a wide range of information sources, including trade representatives. The farming press, although frequently used, was not well regarded as a source of credible technical information by either farmers or consultants. They rated one-to-one advice as the most effective means of communicating nutrient management information. Workshops/courses, local farming events/conferences and booklets/leaflets were all highly regarded communication vehicles. Around 80% of arable farmers and 70% of grassland farmers had made use of IT systems – although the survey database was skewed towards software users. IT systems were highly regarded, with c. 80% of arable farmers and consultants, and c. 75% of grassland farmers regarding them as either effective or very effective.
**Effecting a Change in Practices**

Farmers in the original (1996) NVZ designation were surveyed at the start of designation and again in 2003. This re-survey (Scott *et al.*, 2003) allowed an analysis of the factors (e.g. previous cropping, fertiliser and manure use) taken into account by farmers within the NVZs when making decisions about fertilisers. For all of the main factors, farmers were asked to record the frequency with which they took that factor into account. Responses were ranked from 1–5 (1 = always, 5 = never), so that an average score could be calculated for each factor. All of these mean scores were lower in 2003 than in 1996, for both fertiliser and manure planning. The biggest change was in ‘recent application of organic manures’. This suggests that farmers in the NVZs are taking the major factors into account more often, before spreading manure/fertiliser on their fields. In 2001, the BSFP indicated that, although few farmers used its tables of recommendations, 42% were aware of the RB209 fertiliser recommendations book (Goodlass *et al.*, 2002). In a later survey (2004) of farmer use of decision support tools such as the RB209 and software such as MANNER and ‘your farm and NVZs’, farmers were less likely to have used the software than would advisers/consultants (Chambers, 2004). In this survey, around 30% of arable farmers and 20% of grassland farmers indicated that RB209 had a major influence on their fertiliser planning.

Recent BSFP results (Goodlass and Welch, 2005) suggest that, for several crops, N fertiliser inputs are being reduced in the presence of organic manures (Table 1). Except for potatoes the amount of reduction is greater in farms within NVZ designated areas.

**Table 1: Effect of NVZ designation on reduction in applied fertilizer N in presence of manure**

<table>
<thead>
<tr>
<th>Crop</th>
<th>NVZ</th>
<th>N reduction (kg/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Winter wheat</td>
<td>No</td>
<td>-4</td>
</tr>
<tr>
<td>Winter wheat</td>
<td>Yes</td>
<td>-16</td>
</tr>
<tr>
<td>Winter oilseed rape</td>
<td>No</td>
<td>-31</td>
</tr>
<tr>
<td>Winter oilseed rape</td>
<td>Yes</td>
<td>-45</td>
</tr>
<tr>
<td>Potatoes</td>
<td>No</td>
<td>+27</td>
</tr>
<tr>
<td>Potatoes</td>
<td>Yes</td>
<td>+22</td>
</tr>
<tr>
<td>Sugar beet</td>
<td>No</td>
<td>+27</td>
</tr>
<tr>
<td>Sugar beet</td>
<td>Yes</td>
<td>-19</td>
</tr>
<tr>
<td>Forage maize</td>
<td>No</td>
<td>-33</td>
</tr>
<tr>
<td>Forage maize</td>
<td>Yes</td>
<td>-74</td>
</tr>
</tbody>
</table>

**Impact of Legislation**

As indicated above, the NVZ legislation has had an effect on the factors farmers take into account when making decisions and there is evidence (Scott *et al.*, 2003) that livestock farmers (especially those in the pig/poultry sector) have found this legislation has had a big impact on their farming practices. It is too early to say what impact Cross Compliance and the Stewardship Schemes will have on farmer
practices but, hopefully, the changes seen as a result of NVZs will gain momentum under the Environment Sensitive Farming campaign (Carter, 2006).

CONCLUSIONS

Given the advertising endeavour put into making us change our buying habits by large multinationals it is not surprising that making changes in farming practices takes time and effort. Mass media and generic literature is relevant to awareness creation, but personal contact and demonstration are critical if persuasion is to lead to action. The best advocates of agri-environmental schemes are participant farmers themselves. Surveys in England suggest that fertiliser and organic manure management are changing as marketing and legislative pressures encourage awareness of best practice. The current Environment Sensitive Farming campaign has major role to play in taking this forward.

ACKNOWLEDGEMENTS

This paper has drawn on the results from several projects, the work of other colleagues involved in these projects is acknowledged together with the funding received.

REFERENCES


A CASE STUDY: ADOPTION OF BEST MANAGEMENT PRACTICE IN BRITTANY (FRANCE) USING ECONOMIC INSTRUMENTS AND REGULATION

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SUMMARY

French intensive livestock production has become increasingly concentrated in Brittany over the past 40 years. This has resulted in a serious diffuse pollution problem. Increasing nitrate concentrations in waters and eutrophication have impacted on water treatment, the environment and tourism. To deal with this situation, environmental regulations were introduced in 2001 at both a regional and departmental level. Funding is now provided via the Agence de l’Eau and the regional and departmental councils to help farmers improve water quality. This paper uses the approach taken in Brittany as an example of how one European region has attempted to reduce diffuse pollution through a combination of education, economic instruments and regulation.

INTRODUCTION

After the introduction of the Nitrates Directive, EU member states were required to implement a programme of measures to reverse the trend of increasing nitrate concentrations in sensitive waters. The methods used to improve water quality vary between countries, according to the specific nature of the problem and the degree to which regulation and economic instruments are seen as acceptable means of attempting to address this issue. This paper provides an overview of the approach taken in Brittany (France) where regulation and economic instruments have been used since 1994 to encourage farmers to adopt what are considered to be the best management practices for the region.

CONTEXT

Only 5% of French agricultural land is in Brittany, but the region contains 55% of French pig production and a large proportion of dairy and poultry production. This concentration of livestock supports a food production and processing economy that is vital to the prosperity and identity of the region (SCEES, 1995). However, such intensification gives rise to a number of environmental and logistical problems. Large quantities of animal feed are imported into the region but the livestock themselves only retain a small proportion of the nutrients. As a result, in many instances the amount of nutrients excreted by livestock is far greater than the amount required to support crop growth at a farm or district level.

The intensification of livestock production in the Brittany region has also led to some conflict with tourism (Merceron, 1999). For example, the increase in eutrophication of surface waters and estuaries has been attributed to nutrient surplus (Figure 1). This has been an area of particular concern, given the importance of bathing waters and shellfisheries to the economy of the region.
Figure 1: The impact of algal blooms on the Brittany coast since 1997 – the quantity of algae collected and the number of communes affected (source: DIREN Brittany)

Figure 2 illustrates that in 2004 mean nitrate concentrations for a number of rivers in Brittany were well over the 50 mg/L limit. This is despite efforts made to reduce the amount of nitrate (and phosphorus) lost from the surrounding agricultural land.

Figure 2: Mean and maximum nitrate concentrations in the river water of Brittany. The dark columns indicate where values are greater than 50 mg/L (Source: Brittany Direction Régionale de l'Environnement)
The approaches taken by Brittany to reduce nitrate and phosphorus pollution have been influenced by a long history of legislation, starting in 1810, when a regulation for registered holdings (la réglementation des installations classées pour la protection de l’environnement – ICPE) was first introduced. This regulation was designed to control activities that are detrimental to environmental and public health.

The ICPE regulation has been modified a number of times and is applied to livestock enterprises holding numbers of animals above set thresholds. It imposes certain obligations on the farmer, such as the need for manure management plans and the active management of the nitrogen (N) surplus. Holdings are subject to inspections by the regional administration. In 2005, Brittany contained 12,000 authorised registered holdings (including 9782 livestock enterprises): representing 17% of the ICPE in France (Daumer et al., 2005).

**Nitrates Directive in Brittany**

Brittany is particularly sensitive to nitrate pollution as 80% of drinking water is derived from surface water sources. In 1994, the whole of Brittany was designated a Nitrate Vulnerable Zone (NVZ). Table 1 summarises the regulatory process and actions related to implementation of the Nitrates Directive. In the areas of highest livestock density, more stringent measures have since been introduced.

**Table 1: The regulatory framework associated with the implementation of the Nitrates Directive in Brittany (source: Daumer et al., 2005)**

<table>
<thead>
<tr>
<th>Date</th>
<th>Event</th>
</tr>
</thead>
<tbody>
<tr>
<td>December 1991</td>
<td>Nitrates Directive</td>
</tr>
<tr>
<td>December 1994</td>
<td>Brittany is designated a Nitrate Vulnerable Zone (NVZ)</td>
</tr>
<tr>
<td>1994–1995</td>
<td>Delineation of 71 ZES and 1st nutrient removal program</td>
</tr>
<tr>
<td></td>
<td>Nutrient Management Plans are mandatory</td>
</tr>
<tr>
<td>January 1998</td>
<td>Thresholds introduced for manure treatment and limits put on spreading areas</td>
</tr>
<tr>
<td></td>
<td>Delineation of complementary action zones (ZACs) including:</td>
</tr>
<tr>
<td></td>
<td>• Mandatory cover crops over winter</td>
</tr>
<tr>
<td></td>
<td>• Nitrogen application limit of 210 kg/ha/year</td>
</tr>
<tr>
<td>March 2002</td>
<td>Manure management plans mandatory for new young farmers and all registered holdings under the IPCE regulations</td>
</tr>
<tr>
<td>July 2002</td>
<td>Revision of ZES (104 cantons) and 2nd nutrient removal programme</td>
</tr>
</tbody>
</table>

An assessment was made of the manure N loading in each canton (local administrative district). Those cantons where the amount of manure N exceeded 170 kg/ha of agricultural land were designated as zones d’excedents structurels (ZES) (structural surplus zones). There are now 104 ZES and these cover 70% of the region. In these ZES cantons, actions have to be taken by the largest livestock producers to reduce the amount of nutrients spread on the land through a combination of two or more of the following measures:

- Reduce the number of livestock.
- Reduce the amount of nutrients excreted through changes in animal diet.
• Treatment of manures and slurries to remove P and N.
• The transfer of treated manures out of the region.

The requirement to treat depends on the size of the herd on farm and the extent of the manure N overloading in the canton. The largest pig producers (e.g. producing > 25,000 kg manure N) in sensitive cantons are obliged to both separate slurries for P removal (80% of the manure P must be removed) and treat the liquid to reduce the amount of N in the resulting effluent. However, the livestock threshold for mandatory treatment varies according to the overall manure N loading within each ZES.

Many of the treatment technologies are costly to set up and operate, and so funding has been made available to help with investment (Le Louarn, personal communication). For example, in the Department of Côte d’Armor the following funding is available:

• The Agence de l’Eau (Water Agency) provides up to 30% of the investment cost.
• The General Council (Departmental level) will provide up to 15% of investment costs for farm holdings producing 20,000 kg/a of manure N or less, a maximum of 110,000 euros per holding.
• The Regional Council of Brittany also provides up to 15% of investment costs for farm holdings producing 20,000 kg/year of manure N or less, but up to a maximum of 140,000 euros per holding.

It is therefore possible for up to 60% of the treatment costs to be paid by regional funding. However in reality, only those farm holdings that have met the IPCE and PMPOA (Programme de Maîtrise des Pollutions d’origine Agricole) regulations since 1994 are eligible. In the majority of cases, the level of funding obtained per holding ranges from 0% to 30% (Le Bris, personal communication).

Farms that are obliged to take action can obtain advice from the General Council, the Chamber of Agriculture and the various companies that market treatment technologies. Where treatment is an economically viable option, farms can group together to form a GIE (Groupement d’Intérêt Economique). However, the majority (88%) of treatment plants are operated by individual farms (Le Bris et al., 2005). On those farms for whom the treatment option is not economic, the only alternatives are to:

• transfer wastes to cantons where the manure N loading is less than 140 kg/ha;
• reduce the amount of N produced by introducing bi-phase or multi-phase feeding;
• incinerate the waste (feasible for poultry) with transfer of ash products; and
• reduce the number of livestock.
MANURE TREATMENT

By January 2005, there were 266 farm waste treatment plants in Brittany (Le Bris et al., 2005). Of these, about 70% use aerobic biological treatment, with or without separation. Most of the remainder use composting technologies involving straw and green waste combined with slurry. There are also a few (<10) physico-chemical, phase separation plants.

The most popular treatment technology on many farms is aerobic biological treatment using nitrification/denitrification. This approach can eliminate up to 70% of the total N in pig slurry (Burton et al., 1993, Melse et al., 2002, Béline et al., 2004) and, if combined with mechanical separation, up to 80% of the slurry P can also be removed (Béline et al., 2004).

Within such treatment plants the two processes of nitrification and denitrification are carried out via separate aerobic and anoxic phases. Under controlled commercial conditions, the denitrification of nitrate to di-nitrogen gas is carried out efficiently. However, when poor management leads to nitrite accumulation in the reactor, this can give rise to emissions of nitrous oxide. This is a potent greenhouse gas that contributes 260 times the effect of carbon dioxide on a weight basis (IPCC, 1990). Béline et al. (1999) found that up to 30% of total N can be lost as nitrous oxide during an aeration phase of between 4 days and 1 week. However, subsequent research (Béline and Martinez, 2002) has shown that a short aeration phase followed by an extended anoxic phase (aerobic to anoxic ratio = 0.375) can result in complete denitrification without N2O emissions, provided there is sufficient available carbon in the slurry. These recommendations have been incorporated in the management practices of commercial plants and no nitrous oxide emissions were found during recent measurements carried out on site.

Transformations can also occur in storage and there is a higher risk of nitrous oxide emissions when carbon availability in the slurry is low (Béline et al., 1999). It is therefore very important that aerobic treatment is well managed and that changes in livestock structure are taken into account to avoid nitrous oxide or ammonia emissions. One problem that has arisen in recent years is that, when changes occur within a GIE, aerobic biological treatment plants have to cope with changes in the quantity and type of slurry to be treated (Poilvet, 2005). Such incremental changes can force adaptations to the treatment system, such as residence time and sequence of events, which do not take account of the potential environmental risks. If these changes are not monitored, they could give rise to an increase in the amount of pollution swapping.

CONCLUSIONS

Aerobic treatment of surplus slurry is seen as a useful part of the armoury to help prevent nitrate pollution of water. Such systems are effective in reducing the amount of N entering soils and ultimately water. However, if they are not closely monitored and controlled they can give rise to the loss of N in the form of ammonia and nitrous oxide, both of which are potentially harmful pollutants. In order to support the creation and maintenance of efficient treatment systems, funding is available to farmers for up to 60% of investment. This and further research funding will be necessary to ensure that farmers are given sufficient scientific and technical support to carry out manure treatment while minimising the risks to the environment.
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REGULATORY OPTIONS FOR THE MANAGEMENT OF RURAL DIFFUSE POLLUTION

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SUMMARY

On 1 April, a new regulatory regime in Scotland came into force, bringing in regulations to implement much of the requirements of the Water Framework Directive (WFD). The WFD requires that diffuse sources of pollution are regulated, and this paper examines how far that can be achieved under the new Controlled Activities Regulations (CAR) regime, as well as considering opportunities for further control under additional regulations, and perhaps beyond them.

INTRODUCTION

Diffuse pollution is a statutory area of core business throughout the EU under the Water Framework Directive (WFD). Of course, even before WFD, diffuse pollution was statutory business in Scotland and the rest of the UK. For example, in relation to nitrates in Nitrate Vulnerable Zones (NVZs), faecal pathogens in bathing waters, nutrients in lochs, and for diffuse pollution hotspots anywhere. Indirect or advisory measures have also been used for many years, for example, in relation to forestry practices and many of the measures in the Prevention of Environmental Pollution From Agricultural Activity (PEPFAA) Code for agriculture. But the Water Framework Directive specifically requires that diffuse sources of pollution are controlled, not just a particular class of pollutants and not leaving the regulators with the soft option of merely controlling the pollutants in major point sources. This paper focuses on the use of the Water Environment and Water Services (Scotland) Act 2003, (the WEWS Act) and subsequent regulations, notably The Water Environment (Controlled Activities) (Scotland) Regulations 2005, (referred to as CAR below) for managing diffuse pollution from rural sources. These provisions came into force on 1 April 2006. This paper also considers the possible form and content of further regulations targeted more closely at the problem and possible regulatory solutions.

The effectiveness and appropriateness of regulation in the UK has been the object of serious consideration after the publication of the report by Philip Hampton for HM Treasury on regulatory inspections and enforcement. While there is no specific equivalent in Scotland, the report has been influential. One of the recommendations of that review was that consideration should be given to alternatives to regulation and to striking the correct balance for regulation; proportionate to environmental risks. The Water Framework Directive allows the use of what it calls supplementary measures (economic support schemes, voluntary initiatives, policy considerations with regard to pesticide or fertiliser taxes, etc.). An integrated approach to supplementary measures and regulation is discussed in Davies (2006) and Campbell et al. (2004), and is inherent in SEPA strategy (e.g. SEPA 2004a).

Although the requirement for regulation is explicit in WFD, there is scope for a light touch or for tighter regulation. Figure 1 presents a representation of the three key...
factors influencing a polluter's willingness to change behaviour and practices. There was no doubt that prior to the recent reform of the Common Agricultural Policy, when the farm payments system drove farmers to ever more intensive practices, it would have taken a hugely expensive regulatory and educational effort to counter that economic driver and prevent the environmental impacts of agriculture.

A. Economic drivers in opposite direction to pollution minimisation

B. Economic drivers in alignment with regulation

Figure 1: Minimizing regulation (Campbell et al., 2004)

It is hoped that the scenario presented in Figure 1B is now more applicable, with economic support now more aligned with the environmental drivers, allowing the adoption of a lighter touch for regulation.

Managing diffuse pollution under the provisions of the WEWS Act 2003 and regulations in CAR 2005

There is considerable scope for controlling pollution from diffuse sources under these statutes. Enforcement is predicated on the identified activity having an individually significant impact, at the scale of individuals undertaking the activity. At first sight that seems to rule out most of the significant diffuse source loads in a rural catchment, but it does allow for intervention in relation to diffuse pollution hotspots. Thus in circumstances where evidence of environmental impacts upstream and downstream of an activity can be collected, then there is an evidential basis for enforcement action, just as there would have been under COPA (1974 Control of Pollution Act). SEPA will also be able to serve notices to require corrective measures in such circumstances. This sort of regulatory control would apply to incidents where pollution is caused and a particular farm and farmer can be held responsible. The CAR 2005 provisions allow for the prevention of further such pollution, by embodying the necessary prevention conditions in a registration or licence. Example scenarios are given below.

The CAR offence provisions are set out in regulation 40 and, for example in relation to pollution incidents (diffuse pollution hotspots), provisions covering powers to serve notices and undertake work and recover costs, are set out in regulations 28, 29 and 31.
There are also measures that control pollution risks (as well as other environmental impacts) from a number of specific activities that are often undertaken in rural areas, established in General Binding Rules (GBR) (see next section and regulation 7 of CAR 2005). These are set out in CAR in Schedule 3, Part 1, and brief indications of scope and content are given below.

The relevant GBRs cover the following activities:

- Dredging a river, burn or ditch (GBR 5).
- Bridge construction (GBR 6).
- Laying a pipeline or cable beneath the bed or banks of a river, burn or ditch (GBR 7).
- Revetment work to control bank erosion (GBR 8).
- Operating any vehicle, plant or equipment for the purposes of activity 5, 6, 7 or 8 (GBR 9).
- Discharge of surface run-off from a surface water drainage system to the water environment from construction sites, buildings, roads, yards and any other built developments (GBR 10).
- Discharge into a surface water drainage system (GBR 11).

The pollution prevention requirements in GBRs 5–8 include (paraphrased for this paper):

- Vegetation shall not be disposed of into the channel.
- All reasonable steps must be taken to prevent increased erosion of bed or banks, and prevent transport of disturbed sediments beyond the reach where work is being done.
- Works shall not be undertaken during periods when fish are likely to be spawning, or in the period between any such spawning and the emergence of the juvenile fish.
- Revetments shall be constructed of non-polluting materials.

The pollution prevention requirements in GBR 9 include:

- Refuelling at least 10 metres from any surface water.
- Drip tray required for static plant, with 110% of capacity of fuel tank.
- Plant or other equipment with oil leaks shall not be used.
- Washing plant, etc. shall be at least 10 metres from surface water and wash waters shall not be allowed to enter surface water.

In contrast to the earlier GBRs, numbers 10 and 11 are primarily to prevent pollution, or damage to treatment systems. The pollution prevention requirements in GBR 10–11 refer to surface water drainage from the built environment and include:

- The discharge shall not contain any trade effluent or sewage, shall not cause visual impairment, destabilisation of stream bed or banks, or cause pollution.
- SUDS technology is to be used for new developments.
• The discharge shall not contain drainage from chemical, oil or other polluting matter loading/unloading/handling areas.

• Do not dispose of oil, paint or paint thinners, pesticides, detergents, disinfectants or other pollutants into a surface drainage system, or allow any material that might impair its performance and function to enter a surface water drainage system.

**GBRs, Registration or Licences?**

The CAR regulatory regime allows for three levels of authorisation (Part II, regulations 7–10), and SEPA may serve a notice on a person responsible for an activity, to change the level of authorisation (regulation 11). The most basic level is the General Binding Rule (GBR); essentially this way of controlling activity simply establishes the requirements to prevent environmental harm. It is an offence to carry out the activity in breach of those requirements. A precedent is the way the oil storage regulations have been established in England and Wales, and the successive regulations across the UK for controlling pollution risks associated with the storage of silage, slurry and agricultural fuel oil. For existing establishments there is no need to register with the regulatory agency, but enforcement can be undertaken by notices on inspection. The GBR is therefore the least bureaucratic regulatory option. To quote the recent rural diffuse pollution consultation by the Scottish Executive (2005):

‘...as far as is practicable, the basic level of authorisation, national GBRs, should be used as the basis of control. This minimises the regulatory burden on the industry, while requiring a basic minimum of action to be taken to prevent pollution...’

Registration is the next level of authorisation, requiring the farmer to register the farm with SEPA, and to comply with a simple set of pollution control measures. A fee will be payable to cover administrative costs; there could also be a subsistence charge to cover the costs for SEPA of inspections and meetings on the farms and associated office work. For the rare situations where a particular farm or other rural enterprise is individually significant in a catchment, then SEPA may require an authorisation in the form of a licence for the farm. That would allow for site-specific conditions, rechargeable costs for monitoring and inspection.

Licences have previously been held by many farmers, for example for septic tank discharges under COPA and earlier legislation, and more recently under the groundwater regulations. Under the CAR, these will be transferred to become either registrations or licences.

**What is Missing in CAR for Managing Diffuse Pollution from Rural Sources?**

The gaps are how to control the individually minor but collectively significant impacts; background contamination that is a function of land-use – as identified in the Scottish Executive consultation (2005) Diffuse Water Pollution from Rural Land Use. The SEPA strategy document for managing diffuse pollution (SEPA, 2004) highlighted the importance of the Best Management Practices (BMPs) approach developed in the USA (see chapter 2 in Campbell *et al.*, 2004, definitions and discussions in Novotny 2003, and D’Arcy and Frost, 2001). Many of the BMPs in those references are addressing aspects of farming and forestry practice that have not hitherto been
widely recognised as needing much attention in Scotland or indeed much of the
UK; contamination associated with livestock grazing in fields, with growing crops
in areas prone to soil erosion, harvesting timber, movements of livestock, access
to watercourses. There are also strategic decisions to be taken in catchment
management, for example recognition that with or without BMPs, there will be more
P and N loss from a field of potatoes than from a grass field or cereal crop: the land-
use decision is the first level for managing pollution risks. A general set of rules could
bring in national measures, but targeting specific high-risk areas of fields for soil
erosion control, lengths of watercourses to be fenced off, etc., would be site-specific
control and likely to be licence territory. It may prove best regulated by an alternative
approach, perhaps using supplementary measures for site-specific circumstances?

Can Regulation be Focused on the Target Areas within a Farm?

The following sections consider how a regulatory approach could ensure use of
BMPs to address rural diffuse pollution problems, and follows the BMP system
used in Campbell et al. (2004) and in the new BMPs Manual produced for SEPA and
stakeholder partners by CEH (Centre for Ecology and Hydrology). Four target areas
are considered: steading, in-field, riparian and planning tools.

Steading controls

Source apportionment studies have shown that steadings (farm buildings, yards
and any other associated roof and hard standing areas) are a significant source of
contamination (Edwards et al. in prep.) The GBRs 10 and 11 in CAR 2005 effectively
provide a statutory basis for managing pollution risks from steading drainage, and also
for surface run-off from other rural developments such as forestry offices, housing
and depots, golf course buildings and car parks, and poultry units, piggeries etc. An
additional requirement in a farm GBR could be to require the use of a biobed where
yard run-off at risk from pesticide handling does not drain naturally onto grassland.

A broad range of controls can be envisaged for managing pollution risks at steadings;
from control at source for high strength, low volume pollution risks (storage and
handling controls) to high volume, relatively low contamination where drainage
infrastructure measures are needed to attenuate the pollutants, such as SUDS and
other BMP drainage features (see Figure 2). Bhamidimarri et al. (in preparation) are
developing design guidance for constructed farm wetlands. The difficulty for steading
drainage from livestock farms in the UK, is at what point on the schematic (what level
of contamination) can dirty water be allowed to drain to a constructed wetland or
onto grass where no drainage system is used, without risking a contravention of the
Silage Slurry and Agricultural Fuel Oil regulations (SSAFO regulations).
Modifications to the SSAFO (Scotland) regulations are currently being considered by the Scottish Executive. The aim of these changes will be to permit more sustainable management of wider environmental issues and more cost-effective pollution prevention.

**In-field measures**

Land use decisions, which may change on an annual basis for any given farm, can drastically alter the diffuse pollution potential of a given area (see D’Arcy and Frost, 2001). As such, this is perhaps the most difficult area to control by specific regulations. It is hoped that the cross-compliance elements of the reform of the Common Agricultural Policy (CAP) of the European Community such as GAEC (Good Agricultural & Environmental Condition), and perhaps land management contracts, will go a long way to delivering better environmental farming practice. It may be possible to include GBR measures that require conservation tillage techniques for example in nutrient sensitive catchments, or prohibit autumn sowing in erosion prone soils in nutrient sensitive catchments. Some in-field hotspots can result in pollution where drainage enters a watercourse at a particular location; this is considered in the next section.

**Riparian measures**

Buffer zones for arable land and for livestock densities in excess of a threshold could be specified in a national GBR, but would probably have to be so small as to be in need of significant extension on a site-specific basis in particular problem catchments or designated areas. The measures are already widespread in forestry planting, and in agriculture a buffer zone at least 2 m wide measured from the stream bank top is required under GAEC. Design criteria for effectiveness in relation to different issues need to be established.

A farm licence would allow for site specific measures, and the use of maps appended to the licence showing the lay-out of steep or erosion prone slopes that would not be
ploughed, of rivers and streams that would be fenced to exclude livestock, etc. Where livestock are regularly standing in numbers in a watercourse, perhaps sheltering where a stream crosses the bottom corner of a field, the impact on water quality can be severe; a diffuse pollution hotspot. Under the regulations in CAR 2005, a notice can be served requiring the farmer to carry out measures such as stream fencing and protection from contaminated field run-off. Similarly pollution can be prevented from a poorly sited hay-feeder for livestock in a field, to require its relocation to a position where puddled ground around it will not result in contaminated drainage entering a watercourse. Use of such provisions to control diffuse pollution hotspots in arable land might be more difficult to achieve successfully, and it may be better to seek control by in-field measures specified in a GBR, backed by catchment initiatives and GAEC measures?

**Planning and other management techniques**

Nutrient budgets, farm manure and slurry management planning are all recognised good practice. In NVZs, nutrient budgets and farm waste management plans are already statutory requirements. Such commonsense approaches to use of resources and avoidance of unnecessary expense, as recommended in the PEPFAA code, should be more widespread. There are often significant savings to be made for the farmers, alongside the environmental opportunities. The smaller the farm units, the simpler the estimations required. These measures are obvious candidates for the national statutory measures to meet WFD requirements for controlling diffuse pollution.

**Even-handedness Across Rural Sectors**

It is important that an even-handed approach is taken, and seen to be taken, in regulating pollution risks. That has been achieved in CAR for the regulation of surface water discharges from the built environment, which includes steadings, forestry offices and depots, golf courses and everything else that involves buildings, yards, roads roof areas, etc. Surface water discharges are authorised by GBRs 10 and 11, as noted above and that applies across all sectors in Scotland.

Under the infield measures discussed above, a simple GBR approach for farming was also outlined, and reference made to the recent consultation by the Scottish Executive.

A simple GBR approach for forestry would be easy to devise, in direct consultation with forestry interests and at-risk environmental sectors such as fisheries and nature conservation interests. Scotland is soon to be a net exporter of timber, with potential risks to the trophic status of oligotrophic waters, as well as the spawning grounds of migratory fish.

**Cost Recovery Challenges: GBRs, Registrations or Licences?**

A subsistence charge could be attached to registrations, and certainly to licences, to enable SEPA to recover the cost of regulation. For non-registered GBRs, what are those costs likely to be? Direct funding for SEPA by central government for activity to manage diffuse pollution is likely to continue to be essential. In Canada, on the spot fines can be imposed by regulators for minor water pollution offences;
recourse to the law courts would only be for non-payment, at least for initial offences that are subsequently adequately resolved. How might this work in Scotland? First inspection visit could be at no charge; if anything is found to be in breach of GBR, then a letter would be sent to the farmer or forester or whoever, warning that if not corrected within a reasonable time (to be determined), an on-the-spot fine may be imposed. An enforcement procedure would be set out and farmers would be sent bills for payment, with risk of court action for non-payment. The size of the fine could be set at the cost to SEPA for the average time for two farm visits? N.B. this is not currently envisaged in regulatory options for Scotland. Precedents exist in Scotland, for example in relation to control of litter by local authorities, but not for work by SEPA. The idea is to have smaller fines in line with the environmental risk of the individual misdemeanour, yet still have some regulatory option to encourage best practice.

However, there is a wider issue of funding the inspection and promotion of good practice in the rural sector, including monitoring associated environmental quality. SEPA recovers as much of the costs of regulation as possible from the major point source discharges such as municipal sewage and process effluents from industry. SEPA is fully stretched servicing these few thousand such discharges; how will it recover inspection and enforcement action for the hundreds of thousands of individually minor sources that together can seriously influence water quality? The collection of subsistence charges from some rural businesses, or on the spot fines to cover necessary enforcement, will not be sufficient to fund pro-active campaigns and educational partnership initiatives as part of catchment management planning. Diffuse pollution is already a major driver for catchment planning projects in Scotland (SEPA, 2004b).

For agriculture, it is sensible for all the government-funded agencies whose field officers visit farms to collaborate:

a) to ensure consistent messages are conveyed to farmers, and
b) with its farm visits, SEPA can fulfil its regulatory duties by undertaking an audit role of the uptake of good practice and risk to the environment.

While delivery of the message to the sector may be primarily by the sector-based partners in such a scenario, SEPA would have to continue to play an ever more pro-active role in joint training for agricultural advisors if such a light touch for regulatory action is to be considered and likely to be effective.

**CONCLUSIONS**

Regulating rural diffuse pollution presents SEPA with many challenges and opportunities. With the introduction of WFD legislation and CAR there is an opportunity to take a fresh look at the means by which this is undertaken. One of the greatest challenges will come in relation to effectively enforcing pollution-sensitive land-use decision-making. Promotion of best practice will continue to be the standard means of diffuse pollution control from rural areas. However, the expense of educational (e.g. catchment based) initiatives is the hidden cost of light touch regulation: the regulated sectors need to be told about the GBRs for example, as well as about how to comply with them. At present central government funding to the agency
is the only mechanism identified for SEPA’s costs. Working closely with others in government and the rural sector offers cost effective ways to minimise costs as well as regulatory burdens.

ACKNOWLEDGEMENTS

Thanks to Dave Gorman, Jannette MacDonald and Lynda Gairns in SEPA and Dave Merrilees of SAC, for looking at drafts of the paper and offering constructive comments. This is a discussion paper, so opinions expressed herein are those of the individual authors and do not necessarily represent corporate views of their employers.

REFERENCES


THE FARM SOILS PLAN

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SUMMARY

The Farm Soils Plan (FSP) is a guidance booklet targeted at all farmers across Scotland. It aims to highlight various soil issues that could benefit both the farm business and surrounding environment. The FSP will help farmers comply with GAEC (Good Agricultural and Environmental Condition) under Cross Compliance and outlines soil management practices that will help to reduce the risk of diffuse pollution from farmland. The FSP follows on from the successful ‘4 Point Plan’ (www.sac.ac.uk/4pp) that helped livestock farmers identify and reduce the risk of diffuse pollution from agricultural activities.

The FSP contains basic guidance on a range of topics. It also lists contact addresses and websites that could act as a starting point for farmers looking for further information about alternative practices or new technologies. The main sections in the pack include:

- Recognising and rectifying poor soil conditions – highlights some common problems seen on Scottish soils and suggests possible solutions.
- Reducing soil erosion – minimising soil loss could avoid expensive re-seeds and remediation work while also protecting surrounding water quality.
- Targeting nutrient application – matching nutrient application to crop demand and taking account of nutrients from slurry and manure applications can help to reduce nutrient loss and make better use of bagged fertiliser applications.
- Preventing soil loss and protecting water quality – benefits both the farm business and the surrounding environment.
- GAEC checklist – non-compliance with GAEC could result in financial penalties under the Single Farm Payment Scheme.
- Field Notes – a reminder for farmers on how to hand texture soils to indicate the texture class and suggests ways to identify any problems visible in the soil profile.

During development of the plan, soil specialists and farm advisers were invited to comment on the document. Once near to completion, a group of farmers was also asked to give comments and suggestions prior to publication. The feedback from the farmer group suggested that the FSP was a useful guide to identifying soil problems that may have previously gone unnoticed such as over-compaction or the formation of a plough pan and also provided a useful reminder of general soil issues, confirming that they are doing the right things.

The FSP is a joint initiative supported by SAC, SEERAD (Scottish Executive Environment and Rural Affairs Department), SNH (Scottish Natural Heritage), WWF (Scotland) (funded through HSBC), SEPA (Scottish Environment Protection Agency), NFU Scotland (National Farmers Union) and FWAG Scotland (Farming and Wildlife Advisory Group).
PHOSPHORUS STORAGE IN FINE CHANNEL BED SEDIMENTS

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SUMMARY

Deposition and storage of fine sediment on channel beds represent an important component of a catchment’s sediment budget and can have implications for sediment-associated phosphorus fluxes and concentrations through water-sediment interactions. Spatial and temporal variations in the phosphorus content and storage in fine bed sediment have been studied in two UK lowland catchments, the Rivers Frome and Piddle in Dorset. Fine bed sediment was sampled in representative reaches on a bi-monthly basis using a re-suspension cylinder, and the resulting samples were analysed for total phosphorus, a range of phosphorus fractions and particle size. The results demonstrate the spatial and temporal variability of sediment-associated phosphorus concentrations and storage, with maximum and minimum phosphorus concentrations and storage occurring in late summer and winter respectively. Temporal variations in concentrations reflect residence times of the sediment and ambient phosphorus concentrations, while variations in storage are mainly due to hydrological regimes. Spatial variations reflect catchment characteristics and the location of inputs. Overall, the study highlights the need to consider both spatial and temporal controls on phosphorus storage associated with fine channel bed sediments.
SUMMARY

A revised PEPFAA (Prevention of Environmental Pollution from Agricultural Activity) Code was launched in March 2005 (Scottish Executive, 2005). A shortened version called the PEPFAA Do’s and Don’ts Guide (Scottish Executive, 2004) has been sent to all agricultural holdings in Scotland in the hope that it will be referred to on a regular basis in the workplace. The purpose of the Code is to provide practical guidance on minimising the risks of environmental pollution from farming operations. A new feature is the Red, Amber and Green highlighted text. The points highlighted in Red are mandatory for all farm businesses affected by the relevant legislation, Amber are a requirement for receipt of the Single Farm Payment, and Green are voluntary but if implemented will help minimise the risk of environmental pollution. There is considerable emphasis placed on planning, as well as keeping records to show that the planning has been implemented.

DIFFUSE AGRICULTURAL POLLUTION

A revised PEPFAA Code was launched in March 2005 (Scottish Executive, 2005). The section on Diffuse Agricultural Pollution has increased in importance. Although water quality in Scotland is generally good, the Scottish Environmental Protection Agency (SEPA) considers that diffuse agricultural pollution is now the most significant cause of poor river quality in certain parts of Scotland and that it will continue to be so unless appropriate action is taken at individual farm and catchment level to turn the situation around. Activities such as ploughing, seedbed preparation, crop spraying, fertiliser spreading and applying slurry may all contribute to diffuse pollution if inappropriately carried out. Run-off after rainfall from farm roads and yards, the surface of fields, and dirty roofs are all potential sources of diffuse pollution. Such pollution tends to arise over wide areas of land making it difficult to control. The key to preventing diffuse pollution by nitrate is to ensure that all inputs are carefully accounted for and that applications are not made in excess of the requirements of the growing crop. Preparation and implementation of a nitrogen management plan are mandatory (a Red highlight) on farms within designated Nitrate Vulnerable Zones (NVZs). This N plan is recommended as best practice from a cost-efficient and environmental point of view for all farms spreading organic or inorganic inputs onto land, and involves a Manure Management Plan (also known as a Farm Waste Management Plan or FWMP) on livestock farms.

SOIL PROTECTION AND SUSTAINABILITY

One aspect of Cross Compliance, and a condition of receipt of the Single Farm Payment, is a requirement to maintain the land in ‘Good Agricultural and Environmental Condition’ (GAEC). GAEC standards relate to soil erosion, soil organic matter, soil
structure and minimum level of maintenance. Good soil management practices will help ensure that the requirements of GAEC are met. Amber requirements include ‘All cropped land over the following winter must, where soil conditions after harvest allow, have either: crop cover, grass cover, stubble cover, ploughed surface or a roughly cultivated surface’ and ‘Fine seedbeds must only be created very close to sowing’. Good soil management also plays a significant role in minimising diffuse pollution.

**LIVESTOCK SLURRIES AND MANURES**

There are important, new, good practices highlighted in the section on The Collection, Storage and Application to Land of Livestock Slurries and Manures. An Amber requirement is to ‘incorporate livestock manures within 2 weeks after spreading on stubbles’. All livestock farms should have a FWMP.

‘The 4 Point Plan’ (SEERAD et al., 2004) is highlighted for the first time in a main PEPFAA Code. This Plan is aimed at livestock farmers and provides guidance on:

- minimising dirty water around the steading,
- better nutrient use,
- risk assessment for manures, and
- managing water margins.

4 Point Plans assess manure and dirty water volumes, storage conditions, steading discharges and clean water contamination and can form the basis of a full FWMP. A full Risk Assessment for Manure and Slurry (RAMS) is part of the FWMP. This involves an examination and soil survey of each field in order to identify their suitability for receiving organic manures and slurries. Risk based on soil, land, gradient, farming and climatic factors will be identified and mapped.

**ACKNOWLEDGEMENTS**

The PEPFAA Code is produced by the Scottish Agricultural Pollution Group comprising SEERAD, SEPA, NFUS and SAC. SAC receives financial support from Scottish Executive Environment and Rural Affairs Department.

**REFERENCES**


ESTIMATING DIFFUSE PHOSPHORUS LOADS TO LAKES: IMPLICATIONS FOR THE CALCULATION OF TOTAL MAXIMUM DAILY LOADS

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SUMMARY

Nutrients and the concentrations in which they are present within a water body are important factors regulating the growth of phytoplankton and other plants. Problematic overgrowth of these is generally recognised as the biological manifestation of eutrophication and poor water quality, often resulting in hypoxia, loss of biodiversity, fish kills, and release of algal toxins in the water body. As many freshwater environments are considered to be phosphorus limited, controlling and regulating the input of this particular nutrient is often one of the targets set under the total maximum daily load (TMDL) management approach.

A TMDL is the sum of the allowable loads of a single pollutant from all contributing point and non-point (diffuse) sources, including natural background levels. The major steps in the American TMDL initiative are: (1) to develop load calculations for non-point sources of pollution; (2) to calculate natural loads; (3) to calculate maximum total loads from all sources; and (4) to determine how to reduce the maximum load in order to meet water-quality standards. A key requirement of this approach is the determination of accurate and reliable pollutant loads to the receiving water from non-point sources.

Many of the methods that are currently being developed in the UK to assess nutrient inputs to lakes from non-point sources are based on data collected at weekly, monthly or even quarterly intervals. Previous studies have shown that such infrequent data collection results in pollutant load estimates that are of low accuracy and precision (Webb et al., 1997; Soerens and Nelson, 2001). There is now mounting evidence (e.g. May et al., 2005) that calculations based on such data may underestimate the pollutant load to water bodies from diffuse sources by up to 80%. This is because they fail to take account of the influence of storm events on the mobilisation and transport of pollutants. The method used to estimate the nutrient loading to a lake will have important implications to the final TMDL that is calculated. To be reliable, TMDLs must take into account the impact of high intensity, low frequency rainfall events on nutrient loadings to water bodies.
REFERENCES


SEDIMENT LOADS AND SOURCES IN THE BUSH CATCHMENT: A MOVE TOWARDS INFORMED MANAGEMENT STRATEGIES

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SUMMARY

A decline in the survival of salmon from ova to smolt had been reported in the River Bush, Northern Ireland, due partly to habitat degradation. A monitoring programme was initiated at four study sites on the river in order to promote improved sediment management at the catchment scale. Bed and suspended sediment loads were quantified leading to the identification of specific grain types and transport events contributing to the sedimentation of salmon spawning redds. Temporal and spatial variations were recorded in fine sediments transported through the channel in suspension (median value range 26.4–448 kg per week) and along the bed (median value range 0.214–9.55 kg per week). The resulting data indicated that lower parts of the river suffered from a relatively high sediment load. This poster also reports on supplementary work aimed at classifying the sources of these sediments in the catchment. A combination of visual observations, GIS erosion potential maps and bank erosion monitoring were used to assess large-scale sediment processes and guide more detailed study. For instance, bank erosion was highest in regions of the catchment with the least cohesive bank materials during high flow conditions (e.g. mean of 38.1 mm per storm at Magherahoney). These data targeted specific sites where sediment fingerprinting techniques were applied to elucidate the link between soil erosion in the Bush catchment and downstream sediment delivery to the four instream study sites. Livestock poaching and peak flows exacerbated damage to banks at a localised scale and led to selective patches of bare land being susceptible to further erosion, augmenting the sediment load (approximately 2% of the suspended sediment load and 60% of the bed load). Drainage maintenance work (60% and 30%, respectively), forest clearfell (1% and 2%, respectively) and ploughed land (37% and 8%, respectively) were also shown to influence the quantity of sediment transported through the study channels. Information generated from the sediment source ascription process was transferred into a scientifically justified plan aimed at reducing fine sediment transport in the Bush catchment. Nine key actions were suggested in order to improve habitat quality in the River Bush. These included wetland restoration, prohibiting drainage maintenance work, strategies to control conifer plantation felling, reducing bare ground, livestock access restrictions, construction site management, systematic macrophyte clearance, employment of a river warden and systematic dissemination of project recommendations to the general public to generate community involvement. Above all, this work has showed that effective catchment management has to be steered by detailed sediment budget information.
LOUD CATCHMENT PROJECT: WORKING WITH STAKEHOLDERS TO DELIVER BENEFITS FOR WATER AND WILDLIFE

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SUMMARY

Engaging the local community is at the heart of the Lound Project. The Project was started in October 2004 as a partnership between the Broads National Park Authority and Essex and Suffolk Water (ESW) to try to tackle diffuse pollution at source. It is focussed on the relatively small, rural catchment (30 km²) of shallow lakes on the Norfolk-Suffolk border, from which ESW abstracts water for public supply. It has four aims:

- To increase understanding of catchment hydrology and improve water quality;
- To manage the land owned by ESW to increase biodiversity;
- To build closer working relationships with farmers and landowners in the catchment, in order to encourage environmentally sensitive land management;
- To involve and include the site users and the local community through access, education and volunteering opportunities.

Perhaps the most challenging aspect has been building relationships with farmers in the catchment surrounding the Lound lakes. Visits to some of these farmers suggested a lack of understanding about how farming practices can affect water quality. Diffuse pollution, in particular nitrate, phosphate and sediment, is a major problem all over the Norfolk and Suffolk Broads. The National Park Authority is working to raise awareness of the issue with farmers and provide solutions that are practicable. The Lound Project is doing this through a variety of methods, including farm visits, advisory leaflets and workshops for farmers. These tools attempt to advise farmers directly on issues such as soil management and efficient fertiliser use, and to encourage farmers to opt into new government Environmental Stewardship schemes.

The Lound Project Officer undertakes regular monitoring of water quality, in particular nutrients such as nitrate and phosphate that largely come from agriculture in the Lound catchment. Although any change in farm practices in the surrounding catchment will take many years to have an impact on water quality, the sooner these changes are started, the sooner the benefits will be seen. This year the Project is working closely with the Somerleyton estate who own Fritton Lake, a significant visitor attraction located in the Lound catchment, to find ways of improving water quality to reduce algal blooms - something which will benefit tourism, public water supply, and aquatic wildlife.
On a smaller scale, the Project is managing the land adjacent to the reservoirs, owned by ESW, to promote biodiversity. The meadows and fen are managed through a DEFRA Countryside Stewardship Scheme. Woodland areas are being managed for wildlife with the help of a volunteer group established this year. This includes coppicing, creating glades and hedge laying. Local residents make good use of the public footpaths across the site, and the Project is keen to promote responsible access and associated infrastructure.

All of these measures, and more, feature in the Catchment Management Plan, written in 2005, which sets out the vision for the Lound Project and how it will be achieved.
MINIMISING THE PRESSURES AND IMPACTS ON FRESHWATER FROM AGRICULTURE IN THE UPPER BALLINDERRY RIVER SAC

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SUMMARY

With funding from HSBC and EHS, WWF has established the Ballinderry River Enhancement Project in partnership with Ballinderry Fish Hatchery. It aims to identify and reduce the causes of deteriorating water quality within one of the river’s tributaries.

The tributary displays a variety of pollution pressure points with detectable impacts on water quality. Agriculture is one of the main sources, and measures to reduce its impact have been considered.

Pollution reduction is being achieved through encouraging better uptake of agri-environment schemes and water protection measures and providing knowledge and incentives to farm to higher environmental standards.

Low-cost solutions to reduce the risk of pollution are also being developed and tested on a demonstration farm in the catchment. Management practices being adopted include the separation of clean and dirty water, secure storage and handling of dirty water and farmyard waste, water conservation and efficient application of organic and inorganic fertilisers.
A FRAMEWORK FOR VALUING THE HEALTH BENEFITS OF IMPROVED BATHING WATER QUALITY IN THE RIVER IRVINE CATCHMENT

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SUMMARY

A model predicting bathing water concentrations of Escherichia coli from livestock in the Irvine catchment has been adapted for faecal streptococci (FS). This has been used to predict risk of bather illness by extrapolation of published data on bather FS exposure vs. incidence of gastro-enteritis. Simulated reduction in the risk of illness by reduced faecal loading was multiplied by a willingness to pay for risk reduction to estimate the annual benefits of mitigation. Benefits of reducing loading by 95% at Irvine Beach were about £600 k pa. Estimated annualised costs of best management practises (BMPs) across the catchment were higher (> £1 m), and it is very unlikely that 95% mitigation is achievable with current stocking rates. However, more work on calibration and validation of this framework is needed.

INTRODUCTION

Current research is exploring applications of BMPs which reduce agriculture’s contribution of faecal bacteria to coastal bathing waters. An economic incentive is required for implementation of these measures. This paper presents a draft framework for valuing the health benefits to society of reducing faecal loading to bathing waters which may be achieved through BMP implementation. By relating a BMP-induced reduction in faecal loading to a decrease in risk of illness, benefits can be calculated using an estimate of society’s willingness to pay for reduction in swimming-associated illness.

MODEL FRAMEWORK

Three components were involved in evaluating reduced faecal loading in the Irvine catchment in Ayrshire (Figure 1). These will subsequently be linked to cost/effectiveness of mitigation.

Figure 1: Steps involved in determining the value of faecal loading reductions in the Irvine catchment
Predicting Bathing Water Quality

Vinten et al. (2004) devised a spreadsheet model to predict E. coli in bathing water by accounting for inactivation, sedimentation, transport and mixing in the Irvine Beach/River Irvine catchment system for various loading in the catchment. We have modified this model so FS levels in bathing waters could be predicted rather than E. coli, using changes to key parameter values, summarised in Table 1. The relative levels in animal faeces and inactivation factors show considerable variation in the literature, so these figures should only be considered indicative at this stage.

Table 1: Correction factors used with the model of Vinten et al. (2004) to predict FS concentrations in bathing water instead of E. coli

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Correction factor for FS/E. coli</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Levels in animal faeces</td>
<td>5.6</td>
<td>Sinton et al., 1993 Reddy et al., 1981</td>
</tr>
<tr>
<td>Half lives in soil and water</td>
<td>0.7</td>
<td>Sherer et al., 1992 in Merrilees, 2004</td>
</tr>
</tbody>
</table>

Bathing water quality predictions were made for current and 25–95% reduction in livestock loadings. Summer discharge percentiles were calculated using 14 years (1989-2002) of River Irvine data.

Bathing Water Quality and Risk of Illness

A dose–response function between the concentration of FS in bathing water and the probability of contracting gastro-enteritis was determined by Kay et al. (1994) and Wyer et al. (1999), based on risk of contracting illness from bathing water quality determined by the World Health Organization using a probability density function of bathing water quality throughout Europe (WHO, 2001). This relationship has data for up to 12% probability of infection, and these data were fitted to an exponential function (eq. 1) to predict risk of illness, including extrapolation where necessary beyond the data range.

\[
\text{Risk of illness} = 1 - e^{-0.000218c}
\]

Where \( c \) = concentration of FS in bathing water (cfu/100mL).

This equation was linked to model simulations of FS in bathing water to give the risk of contracting gastro-enteritis as a function of faecal loading and river discharge (Step 2 in Figure 1).

Economic Valuation of Reduced Risk of Illness

Consumer willingness to pay (WTP) for a reduction in risk of illness resulting from swimming in contaminated waters was determined using a benefit transfer of a contingent valuation used in a previous study in England and Wales (EFTEC, 2002). The EFTEC study determined consumers' WTP for a reduction in risk of illness from swimming in contaminated bathing water. Valuations from the study indicate that respondents are willing to pay between £1.10 and £2.00 per household per year.
for a 1% reduction in the risk of suffering a stomach upset (EFTEC, 2002) and we assumed a WTP of £1.10 per 1% reduction in risk. For comparison with Scottish bathing waters, a direct unadjusted mean value was used to determine the benefits of increased bathing water quality resulting from the reduction in faecal loading in the Irvine catchment. The EFTEC study is a suitable comparative site for Irvine Beach because of similarities below listed in Table 2. The annual benefit accruing to Irvine Beach from reducing faecal loading was calculated by accounting for the likelihood of high discharge events. Benefits were calculated at each percentile flow (above the 72nd) for a range of reductions in loading from 25–95%.

Table 2: Comparison of site characteristics between proposed Scottish benefit transfer and EFTEC WTP study in England and Wales

<table>
<thead>
<tr>
<th>Environmental quality parameter</th>
<th>England and Wales</th>
<th>Scotland</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bathing water quality</td>
<td>Bathing water quality</td>
<td></td>
</tr>
<tr>
<td>Households</td>
<td>24,000,000</td>
<td>2,200,000</td>
</tr>
<tr>
<td>Population</td>
<td>52,900,000</td>
<td>5,062,000</td>
</tr>
<tr>
<td>Average household size</td>
<td>2.204</td>
<td>2.3</td>
</tr>
<tr>
<td>Mean weekly household income</td>
<td>£302</td>
<td>£296</td>
</tr>
</tbody>
</table>

A spreadsheet model was developed to determine the benefits accruing to society from increased bathing water quality. Table 3 gives the equations used in the development of this model.

Table 3: Parameters used to model social benefits from increased bathing water quality

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Equation</th>
<th>Components</th>
</tr>
</thead>
<tbody>
<tr>
<td>Risk of illness</td>
<td></td>
<td>(1)</td>
</tr>
<tr>
<td></td>
<td>$r = 1 - e^{-0.000218c}$</td>
<td>$C = \text{FS concentration in bathing water}$</td>
</tr>
<tr>
<td>Risk of illness differences</td>
<td></td>
<td>(2)</td>
</tr>
<tr>
<td></td>
<td>$\Delta r = r_{\text{full}} - r_{%\text{reduction}}$</td>
<td>$r_{\text{full}} = \text{risk of illness at full FS loading}$, $r_{%\text{reduction}} = \text{risk of illness with reduction in loading}$</td>
</tr>
<tr>
<td>Social benefits at each discharge percentile</td>
<td>$B_f = \Delta r(WTP)h$</td>
<td>(3)</td>
</tr>
<tr>
<td>Total annual benefits ($TB$)</td>
<td></td>
<td>(4)</td>
</tr>
<tr>
<td></td>
<td>$TB = \frac{\sum_{f=0}^{100} \Delta r(WTP)h}{100}$</td>
<td>$F = \text{percentile flow}$</td>
</tr>
<tr>
<td>Lifetime benefits ($LB$)</td>
<td></td>
<td>(5)</td>
</tr>
<tr>
<td></td>
<td>$LB = \sum_{t=0}^{25} TB(1 + i)^{-t}$</td>
<td>$TB = \text{future value (in year } i)$, $i = \text{interest rate}$</td>
</tr>
</tbody>
</table>
Risk of illness percentages were calculated at various discharge percentiles for each loading reduction. The differences between risk of illness at full FS loading and the other loadings were determined. This accounts for the risk reduction expected by reducing FS loading.

Benefits accruing to the population were calculated for each discharge percentile. These values were aggregated across all percentile flows and corrected for the frequency of occurrence to determine total annual benefits (TB). Because the entire Scottish population was used, the total benefit value had to be corrected to elicit benefits occurring only at Irvine Beach. A 2004 study indicated that 3% of all Scottish beach visits were made to Irvine Beach (Scottish Executive Social Research, 2004). The total benefits to Scotland were therefore multiplied by 3% to calculate the benefits at Irvine Beach. To calculate future benefits, discounting was done to give the present value. This was done on a 25-year timescale with a 6% interest rate in this study, as they are similar to values used in evaluating costs of agricultural infrastructure and machinery.

Using the above framework, benefits from reducing faecal loading were calculated for the River Irvine catchment. Irvine Beach is located in Ayrshire in the west of Scotland (Grid Reference: NS305375). Discharge data from SEPA for the River Irvine for 1989–2003 were used to determine summer flow percentiles.

RESULTS AND DISCUSSION

FS concentrations at Irvine beach simulated with the modifications of the Vinten et al. (2004) model are shown in Figure 2. Risk of illness was determined for all loadings at various levels of discharge (Figure 3). Table 4 shows the annual benefit, and aggregated benefit over 25 years of improved bathing water quality at Irvine Beach.

![Figure 2: Predictions of faecal streptococci concentrations in bathing water at the Irvine Beach at various river Irvine scaled discharges](image-url)

Table 4

<table>
<thead>
<tr>
<th>Percentile Flow</th>
<th>Current loading</th>
<th>25% reduction</th>
<th>50% reduction</th>
<th>75% reduction</th>
<th>80% reduction</th>
<th>85% reduction</th>
<th>90% reduction</th>
<th>95% reduction</th>
</tr>
</thead>
<tbody>
<tr>
<td>75&lt;sup&gt;th&lt;/sup&gt;</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>80&lt;sup&gt;th&lt;/sup&gt;</td>
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<tr>
<td>85&lt;sup&gt;th&lt;/sup&gt;</td>
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<td>90&lt;sup&gt;th&lt;/sup&gt;</td>
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<td>95&lt;sup&gt;th&lt;/sup&gt;</td>
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</tbody>
</table>

BW Faecal Streptococc (cfu/100 mL)

Daily Catchment Discharge (mm/day)
Figure 3 shows that risk of illness is insignificant at all FS input rates below the 75 percentile flow. During 95 percentile flow, faecal loading needs to be reduced by 95% to give a risk of illness < 20%. These figures depend on the reliability of the extrapolation used in eq. (1) for risk of illness, and improvement of the epidemiological database for such modelling should be a research priority. There is also little information on the levels of FS in animals faeces, and literature data on inactivation suggest longer survival of FS than E. coli in soil (Cools et al., 2001) but higher sensitivity to sunlight (Sinton et al., 2002).

**Figure 3: Discharge and related risk of illness**

The WTP for avoidance of illness of BW improvement can now be compared with the costs of implementing BMPs to attempt to control diffuse pollution. Recent work has estimated the annual cost of installing BMPs on two farms in SW Scotland to be £3 k per farm plus £50 k capital cost (Merrilees, 2004). If we assume an annualised capital cost of £10 k, and that across the River Irvine catchment there are about 70 farms, total annualised mitigation costs are $(3 + 10) \times 70 = £910 k$. From Table 4 the required reduction of FS loading achieved needs to be > 95% to obtain a benefit/cost ratio which exceeds 1. BMPs to assist in bathing water quality improvements include measures such as stock reduction, fencing, buffer strips, constructed wetlands, slurry storage, cattle access routes, and separating clean and dirty water. In practice achieving > 50% efficacy may be very difficult, particularly considering that BMP performance during high intensity flow events is critical. Dickson et al. (2005) found a 40% reduction in high flow faecal indicator loadings where BMPs had been installed. If the entire Irvine catchment installed these practices, bathing waters may still pose a high risk of illness after storm events.
Table 4: Expected lifetime benefits of reducing faecal loading in the Irvine catchment

<table>
<thead>
<tr>
<th>Reduction in faecal loading in the Irvine catchment</th>
<th>Annual benefit (in £k)</th>
<th>Aggregated benefits over 25 years at a 6% discount rate (in £k)</th>
</tr>
</thead>
<tbody>
<tr>
<td>25% reduction</td>
<td>£63</td>
<td>£868</td>
</tr>
<tr>
<td>50% reduction</td>
<td>£153</td>
<td>£2,103</td>
</tr>
<tr>
<td>75% reduction</td>
<td>£312</td>
<td>£4,300</td>
</tr>
<tr>
<td>85% reduction</td>
<td>£428</td>
<td>£5,897</td>
</tr>
<tr>
<td>95% reduction</td>
<td>£617</td>
<td>£8,509</td>
</tr>
</tbody>
</table>

REFERENCES


DETERMINATION OF THE VETERINARY ANTIBIOTICS POLLUTION IN SOIL FROM AGRICULTURAL SOURCES

M Kaštelan-Macan, S Babić, D Ašperger, D Mutavdžić and AJM Horvat

Faculty of Chemical Engineering and Technology, Marulicev trg 20, 10000 Zagreb, Croatia, E-mail: mmacan@fkit.hr

SUMMARY

Emission of veterinary antibiotics has emerged as an environmental problem and a lack of knowledge concerning this kind of pollution requires further investigation. The antibiotics primarily used in animal medicine are tetracyclines, sulfonamides, ß-lactames and fluoroquinolones. Antibiotics reach agricultural soils directly through livestock or indirectly through the use of manure and sewage sludge as fertilisers. The aim of this work was to optimise the conditions for ultrasonic solvent extraction (USE) of the most commonly used antibiotics in Croatian veterinary practice: sulfamethazine, sulfadiazine, sulfaguanidine, norfl oxacine, oxytetracyclin, penicillinG/procaine and trimethoprim from soil samples. Sonication provides a more efficient contact between the solid and solvent than classic extraction procedures, resulting in a greater recovery of analyte. The extracted pesticides were identified using a thin-layer chromatography (TLC) system: HPTLC CN F254 chromatography plates as a stationary phase, and 0.05 M oxalic acid/methanol=0.80:0.20 as a mobile phase. Quantification was performed using a 3CCD colour video camera.

INTRODUCTION

In the past, the emission of so-called ‘emerging’ or ‘new’ contaminants has emerged as an environmental problem and a lack of knowledge concerning this kind of pollution requires further investigation. Emerging contaminants are the unregulated ones, mainly composed of products used in everyday life, such as pharmaceuticals, personal care products, surfactants, plasticisers, industrial additives, etc. Many believe that of all the emerging contaminants, antibiotics are the biggest concern, as their emission in the environment can result in an increased occurrence of resistant bacteria in the environment, which may lead to unforeseen consequences.

In veterinary medicine, a wide range of antibiotics are frequently used, and as a result of their consumption, excretion, and persistence, they are disseminated mostly via excrements and thus enter the soils. According to FEDESA, one-third of all antibiotics are used for veterinary purposes (FEDESA, 2001). No regulations exist for concentration limits of antibiotics in soils or water, but some EU directives prescribe the risk assessment of veterinary antibiotics (EMEA, 1997).

The antibiotics in primary use in animal medicine are tetracyclines, sulfonamides,ß-lactames and fluoroquinolones. The main sources of their release into the environment are animal farms and aquaculture for fish production.

Antibiotics reach agricultural soils directly through livestock or indirectly through the use of manure and sewage sludge as fertilizers. Manure samples from pigs contained up to 3.5 mg/kg of sulfonamides and up to 4 mg/kg of tetracyclines (Hamscher et al., 2002; Höper et al., 2002). Antibiotic residues and resistant micro-organisms can
affect the natural soil community and also disturb soil function. In addition, it was found that wastewater and run-off from agricultural land are mainly responsible for the contamination of aquatic systems (Alder, 2001).

For this purpose, it was necessary to establish a simple and reliable method for extraction, separation and determination of veterinary pharmaceuticals in soil. The drugs investigated included three sulfonamides (sulfaguanidine, sulfadiazine and sulfamethazine), sulfonamide synergist (trimethoprim), tetracycline (oxytetracycline), fluoroquinolone (norfloxacine) and b-lactam (penicillinG/procaïne). In this contribution, we worked out the procedure using ultrasound extraction (USE) of investigated compounds from soil, and their subsequent thin-layer chromatographic (TLC) determination.

MATERIALS AND METHODS

Materials

All used chromatographic solvents were p.a. grade (Kemika, Zagreb, Croatia). In this work, chromatographic plates HPTLC CN F254 10 x 20 cm, (Merck, Darmstadt, Germany) were used. Soil was collected at Medvednica, a hill near Zagreb. At least 10 years before collecting, the soil had not been treated with any antibiotics. Its composition was: sand 61.45%, silt 20.75%, clay 11.80% and organic matter 6.44%.

Standards

Trimethoprim (TMP), oxytetracycline (OTC), norfloxacine (NOR), sulfaguanidine (SGUA), sulfamethazine (SMETH), sulfadiazine (SDIAZ) and penicillinG/procaïne (PGP) (VETERINA Animal Health Ltd., Kalinovica, Croatia) were min. 99 % pure.

Stock solution of pharmaceutical mixture was prepared by dissolving accurate quantities of the powdered standards in methanol. Mass concentration of each pharmaceutical in the mixture was: 5 mg/L for NOR, 100 mg/L for SMETH, SDIAZ, OTC, TMP and PGP, and 200 mg/L for SGUA. Calibration standards were made by serial dilution of a stock standard solution with methanol in the working range from 0.15 to 1.5 mg/L for NOR, from 10 to 60 mg/L for OTC, TMP, SDAIZ, SMETH, and PGP, and from 30 to 110 mg/L for SGUA.

Soil Spiking and Extraction Procedure

Ten millilitres of standard solution was added to 100 g of dried and sieved soil. An additional 100 mL of methanol was added to cover the soil particles; the slurry was mixed for 1 h and then left at room temperature for 24 hours to allow solvent evaporation.

Accurately weighted spiked soil (10 g) was added to 20 mL of methanol and sonicated for 15 min in an ultrasonic bath (frequency 25–40 Hz, UZ-20R, Iskra, Kranj, Slovenia). The extracts were filtered through Whatman No.40 filter paper. The filtrate was evaporated on a rotary vacuum evaporator (R-114/A, Büchi, Switzerland) at 40° to dryness. The residue was dissolved in 1 mL of methanol and this solution was directly spotted on a TLC plate.
Thin Layer Chromatography

The samples were applied to the plates as bands by means of an automated sample applicator (Linomat IV, Camag, Muttenz, Switzerland): volume 5 μL, bandwidth 7 mm, distance between the middle of the bands 15 mm, delivery rate 1 μL/10 s. Ascending development was performed at room temperature in a CAMAG double-trough chamber without previous saturation. Plates were developed to a distance of 70 mm with 0.05 M oxalic acid/methanol = 0.80:0.20 as a mobile phase. After development, the plates were air dried and the chromatograms were visualised under UV light at λ = 254 nm and λ = 366 nm.

Video densitometry was performed with a highly sensitive 3CCD colour video camera HV-C20 (Hitachi Denshi, Japan). The parameters of video densitometry were as follows: close-up lens +2 Dpt, zoom lens 9.5 mm, integration period-exposure time 1 frame (= 20 ms) at 254 nm and 15 frames at 366 nm, frame accumulation-off mode, aperture (F-stop number) 2.0. Imaging, processing and archiving were controlled via VideoStore2 2.30 documentation software. The extended version of Camag VideoScan 1.01 software was used for quantitative evaluation of the stored TLC chromatograms. For quantification purposes, detection was carried out at 366 nm, and TMP was measured at 254 nm. Method was validated for selectivity, linearity, limit of detection (LOD), limit of quantification (LOQ), precision and recovery.

RESULTS AND DISCUSSION

Optimization of the mobile phase was performed by systematic variation of the composition of the basic mobile phase (methanol and 0.05 M oxalic acid). The quality of chromatographic separation was evaluated by chromatographic response function on the basis of the predicted R<sub>f</sub> values. In this work, the MRF criteria (De Spiegeleer and De Moerloose, 1987) was used as the separation criterion, and maximum value of MRF function was found using the genetic algorithm approach (Babic et al., 2005).

Optimum mobile phase composition obtained with these parameter settings was 0.05 M oxalic acid/methanol = 0.80:0.20. The obtained results indicate excellent agreement between experimental and calculated retention factors (R<sub>f</sub>-values), as well as very good overall resolution (Table 1 and Figure 1).

Table 1: Experimental and calculated R<sub>f</sub> values of pharmaceuticals at optimal mobile phase composition

<table>
<thead>
<tr>
<th>Pharmaceutical</th>
<th>Calculated</th>
<th>Experimental</th>
</tr>
</thead>
<tbody>
<tr>
<td>NOR</td>
<td>0.16</td>
<td>0.15</td>
</tr>
<tr>
<td>OTC</td>
<td>0.31</td>
<td>0.31</td>
</tr>
<tr>
<td>TMP</td>
<td>0.38</td>
<td>0.37</td>
</tr>
<tr>
<td>SDIAZ</td>
<td>0.45</td>
<td>0.44</td>
</tr>
<tr>
<td>SMETH</td>
<td>0.56</td>
<td>0.54</td>
</tr>
<tr>
<td>PGP</td>
<td>0.62</td>
<td>0.65</td>
</tr>
<tr>
<td>SGUA</td>
<td>0.71</td>
<td>0.71</td>
</tr>
</tbody>
</table>

For qualitative purposes, the method was evaluated by taking into account the precision in the R<sub>f</sub>-value and selectivity. Comparing the chromatogram of extracts with the chromatogram of standard solution (Figure 1a and b) a coincidence of
RF-values could be observed. A high repeatability in the $R_F$-values was obtained with relative standard deviation (RSD) values lower than 1.5% for standards and 1.7% for extracts. Optimization of the chromatographic method enabled an efficient separation of pharmaceuticals.

![Chromatograms](image)

**Figure 1:** Chromatograms of: a) standard pharmaceutical mixture and b) extracted samples from spiked soil, 1-NOR, 2-OTC, 3-TMP, 4-SDIAZ, 5-SMETH, 6-PGP and 7-SGUA

Linearity, limit of detection (LOD), limit of quantification (LOQ), precision and recovery were evaluated for quantitative purposes. The regression analysis was performed and the resulting parameters are shown in Table 2. $R^2$ values of the pharmaceuticals were higher than 0.99 thus confirming the linearity of the method.

Limits of detection (LOD) and quantification (LOQ) for each pharmaceutical in the mixture were determined from signal-to-noise ratio by comparing measured signals from the samples with known low concentrations of analyte with those of the blank samples (Table 2).
Table 2: Regression functions and correlation coefficients of the studied pharmaceuticals

<table>
<thead>
<tr>
<th>Pharmaceutical</th>
<th>Regression function</th>
<th>$R^2$</th>
<th>Linearity range (ng/spot)</th>
<th>LOD (ng/spot)</th>
<th>LOQ (ng/spot)</th>
</tr>
</thead>
<tbody>
<tr>
<td>NOR</td>
<td>$A = 2.286 \times 10^6 \cdot y + 930.8$</td>
<td>0.994</td>
<td>0.75–7.5</td>
<td>0.5</td>
<td>0.75</td>
</tr>
<tr>
<td>OTC</td>
<td>$A = 29.64 \cdot y - 429.3$</td>
<td>0.994</td>
<td>50–250</td>
<td>25</td>
<td>50</td>
</tr>
<tr>
<td>TMP</td>
<td>$A = 6.347 \cdot y - 6.821$</td>
<td>0.997</td>
<td>75–300</td>
<td>50</td>
<td>75</td>
</tr>
<tr>
<td>SDIAZ</td>
<td>$A = 38.84 \cdot y + 1445.0$</td>
<td>0.998</td>
<td>50–200</td>
<td>25</td>
<td>50</td>
</tr>
<tr>
<td>SMETH</td>
<td>$A = 26.95 \cdot y + 1496.0$</td>
<td>0.999</td>
<td>50–200</td>
<td>25</td>
<td>50</td>
</tr>
<tr>
<td>PGP</td>
<td>$A = 18.30 \cdot y + 1090.8$</td>
<td>0.993</td>
<td>50–200</td>
<td>25</td>
<td>50</td>
</tr>
<tr>
<td>SGUA</td>
<td>$A = 15.75 \cdot y + 3265.0$</td>
<td>0.996</td>
<td>150–550</td>
<td>100</td>
<td>150</td>
</tr>
</tbody>
</table>

A-peak area.
g-mass concentration (ng/spot) (spot = 5 μL).

Precision expressed in terms of RSD indicated high measurement repeatability (Table 3). In general, recoveries were dependent on the analyte concentration and values between 36.5% and 100.5% were obtained for all pharmaceuticals.

Table 3: Recoveries and RSDs obtained for studied pharmaceuticals

<table>
<thead>
<tr>
<th>Pharmaceutical</th>
<th>Recovery (%)</th>
<th>RSD (n = 3)</th>
</tr>
</thead>
<tbody>
<tr>
<td>NOR</td>
<td>36.5</td>
<td>4.05</td>
</tr>
<tr>
<td>OTC</td>
<td>n.d.</td>
<td>2.86</td>
</tr>
<tr>
<td>TMP</td>
<td>52.6</td>
<td>9.83</td>
</tr>
<tr>
<td>SDIAZ</td>
<td>83.2</td>
<td>6.00</td>
</tr>
<tr>
<td>SMETH</td>
<td>100.5</td>
<td>6.50</td>
</tr>
<tr>
<td>PGP</td>
<td>44.7</td>
<td>3.12</td>
</tr>
<tr>
<td>SGUA</td>
<td>78.5</td>
<td>4.88</td>
</tr>
</tbody>
</table>

ND: not detected.

According to validation parameters, it can be concluded that USE-TLC determination of veterinary antibiotics in soil is comparable with other commonly used extraction and chromatographic techniques, and convenient for determination of veterinary pollutants in soil (Thiele-Bruhn, 2003).

ACKNOWLEDGEMENTS

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REFERENCES


OPPORTUNITIES AND CONSTRAINTS FOR USING BEST MANAGEMENT PRACTICES: SOME LESSONS FROM THE TARLAND CATCHMENT INITIATIVE

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SUMMARY

The Tarland Catchment Initiative (TCI) is a partnership venture set up between researchers, land managers, regulators and the local community. The aims of the TCI are to improve water quality and increase awareness of catchment management. The implementation of these improvements has been guided by an objective assessment of existing water quality and selected ecological components across the catchment and by consideration of the differing needs and perceptions of the issues from a range of stakeholders. In order to engage the community, there has been a need to undertake capacity building exercises to introduce the concept of catchment management. This was also used to highlight the pressures and possible intervention through improved riparian land management to lessen the detrimental impacts on water quality and ecology. From this starting point, the TCI has initiated a plan of action based on best practice guidelines. The approach adopted allows for the implementation of measures on a systematic basis in order to evaluate their efficacy; to develop the work as a show piece and demonstration catchment from which others can learn. Results after intervention for water quality, ecology and acceptance by the community are largely a success. The approach adopted, the issues tackled and outcomes expected are central to many of the requirements set out in the guiding principles for the European Water Framework Directive.

INTRODUCTION

Increasingly attention is being given to how catchments can be managed in a holistic manner. One of the principle policy drivers for this in Europe is the Water Framework Directive (EC, 2000). The central features of the WFD and its implementation are: the focus on ecological status and stakeholder engagement (EC, 2000). Within Scotland one of the largest pressures on surface water bodies failing to meet good ecological status is due to the impact of diffuse pollution. As part of the characterisation and impacts report in developing a River Basin management Plan, SEPA identify 25% of river water bodies at risk from diffuse pollution largely linked to land use management activities (SEPA, 2005).

In order to reduce the impact of diffuse pollution on the environment, a range of Best Management Practices (BMPs) for agricultural land management have been included as parts of farm agri-environment schemes and good codes of practice, such as the PEPPFA code (SEERAD, 2005). A comprehensive report of many of the BMPs available for reducing diffuse pollution to surface waters by Hilton (2002) shows that many of these practices have not been evaluated in the context of the UK or in relation to impacts on aquatic and riparian ecology.
The implementation of BMPs requires an appreciation of the issues involved by practicing land mangers. To this end there is a need to connect the competent authorities developing catchment management plans, with land managers and research on the reduction of diffuse pollution impacts.

The aim of this paper is to summarise the development of a catchment initiative in terms of: (i) increasing awareness and stakeholder participation in relation to best management practice and the need for catchment management, (ii) improving land management practices in relation to streams and rivers and (iii) the establishment and use of the work as a demonstration site illustrating practical examples of best practice.

**MATERIALS AND METHODS**

The Tarland Burn is the most westerly tributary of the River Dee (Aberdeenshire, Scotland) that is dominated by intensive land use. These land uses, as characterised by the Land Cover of Scotland 1988 dataset, comprise: Heather moorland 7.9%, unimproved grassland 9.7 %, plantation forestry 19.1 %, mixed/broadleaved woodland 1.8%, improved grassland 35.7%, built-up area 0.7% and arable 25.1%. Farming in the catchment is typically mixed cereal and livestock, dominated by the fattening of cattle and malting barley production and some sheep on improved pasture on the higher ground. Across the catchment area that is used for agriculture, the practice is to utilise the maximum possible area, including up to the stream margins. When the riparian fields are used for cattle grazing the streams provide the cattle waterings.

Earlier work on the River Dee (Langan et al., 1997; SEPA, 2000) and an initial examination of the temporal variation in water quality at the bottom of the Tarland catchment indicated there were significant issues related to diffuse pollution, particularly faecal indicator organisms (FIO), N and P losses. On the basis of this information, an invitation to the major stakeholders in the catchment was sent out to discuss these results. As a result of this meeting, various issues were identified (see results), one of which was the need for more information on water quality variability. To assess the spatial variability in nitrogen, phosphorus and suspended solids across the catchment, monthly dip samples have been collected at 16 points since 2001. At the sites, quarterly samples for FIOs were collected and benthic macro invertebrates sampled using the standard SEPA 3-min kick sample. The invertebrate data are reported as a BMWP (Biological Monitoring Working Party) score (Hawkes, 1997) used to classify waters according to the tolerance of different families to pollution. These sites and parameters were selected to give a series of data in which the generation of water quality within each major tributary across the catchment could be examined. Chemical samples were returned to the laboratory and analysed for nitrate, ammoniacal nitrogen and phosphate within 3 days using automated colorimetric methods.

This monitoring and survey data provides a set of objective information against which together with discussions with stakeholders it has been possible to identify priority areas in which intervention management should be initiated. The intervention measures trialled are BMPs largely based on the requirements of agri-environment funding and the PEPPFA Code (Scottish Executive, 2005) with a view to assessing there efficacy for both water quality and invertebrate ecology and to demonstrate
to land managers and agency staff the issues involved. The range of environmental improvement measures suggested centred on the provision of off stream waterings for cattle, fencing off streams to provide a buffer strips, the creation of wetland areas and soft engineering to improve the diversity of the stream morphology and habitat.

In order to meet the objectives of an inclusive approach to catchment management, efforts aimed at increasing awareness and building capacity have been undertaken at the same time as the monitoring and used to guide intervention. This has been approached by providing a number of different forums for both presenting and gathering relevant information. A commentary on this process is provided in the discussion.

RESULTS

The results are presented in terms of water quality; stakeholder involvement and outcomes from a trial intervention.

Water Quality

Table 1 shows the range and mean concentrations of the major water quality determinands from the sites across the catchment. Each site represents a sampling point at the bottom of each tributary of the Tarland Burn. Table 1 also gives the BMWP score for each site. The data suggests major differences in water quality across the catchment. In the absence of any single large discharge source within the catchment, this variation is the result of the combined inputs of a number of smaller point and non-point sources of pollution. In overall terms, site 13 has the poorest water quality as noted in the high concentrations of FIOs and elevated concentrations of ammonia, nitrate and phosphate, relative to the other sites. The invertebrate analysis reflects this poor water quality.

Table 1: Variation in concentrations from monthly samples for water quality for selected chemical determinands and quarterly invertebrate samples in the Tarland catchment tributaries 2000 to 2002

<table>
<thead>
<tr>
<th>Site Number</th>
<th>PO₄-P mg/L</th>
<th>NO₃-N mg/L</th>
<th>NH₄-N mg/L</th>
<th>Faecal coliforms /100 mL</th>
<th>BMWP score</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0.02</td>
<td>3.49</td>
<td>0.03</td>
<td>7031</td>
<td>96</td>
</tr>
<tr>
<td>10</td>
<td>0.01</td>
<td>3.42</td>
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<td>586</td>
<td>103</td>
</tr>
<tr>
<td>13</td>
<td>0.11</td>
<td>4.72</td>
<td>0.11</td>
<td>127207</td>
<td>83</td>
</tr>
<tr>
<td>14</td>
<td>0.06</td>
<td>2.61</td>
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<td>1043</td>
<td>97</td>
</tr>
<tr>
<td>15</td>
<td>0.01</td>
<td>3.20</td>
<td>0.03</td>
<td>297</td>
<td>96</td>
</tr>
<tr>
<td>16</td>
<td>0.01</td>
<td>5.06</td>
<td>0.03</td>
<td>1885</td>
<td>92</td>
</tr>
<tr>
<td>20</td>
<td>0.01</td>
<td>3.97</td>
<td>0.02</td>
<td>253</td>
<td>104</td>
</tr>
</tbody>
</table>
On the basis of discussions with the land managers and other stakeholders it was agreed that any interventions to improve water quality should be undertaken in a systematic basis by concentrating efforts on individual tributaries. In the light of this and the data presented in Table 1 initial intervention work concentrated on the tributary represented by site 13. After further investigation of the tributary and in conjunction with the riparian land managers, three specific intervention measures were identified and implemented. These were: (a) the overhaul of the operation of a septic tank, (b) the installation of buffer strips and water troughs to reduce the need for cattle using the stream waterings and (c) the establishment of a small wetland where field drainage collector pipes were known to issue directly to the burn. These measures were implemented during the winter and spring of 2002/03. From the programme of samples taken comparison of the water quality and invertebrates for the year prior to and after the interventions were made is presented in Table 2. The data indicate significant reductions in the concentrations of phosphate, ammonia, nitrate and faecal coliforms. Furthermore a significant increase is seen in the BMWP score.

Table 2: Water quality and invertebrate data prior to and post-intervention above tributary site 13

<table>
<thead>
<tr>
<th>Determinand</th>
<th>Pre-intervention</th>
<th>Post-intervention</th>
</tr>
</thead>
<tbody>
<tr>
<td>PO₄-P (mg/L)</td>
<td>0.032 (&lt; DL–0.11)</td>
<td>0.018 (&lt; DL–0.050)</td>
</tr>
<tr>
<td>NH₄-N (mg/L)</td>
<td>0.043 (&lt; DL–0.166)</td>
<td>0.015 (&lt; DL–0.052)</td>
</tr>
<tr>
<td>NO₃-N (mg/L)</td>
<td>4.96 (3.48–5.89)</td>
<td>4.26 (3.33–4.89)</td>
</tr>
<tr>
<td>Faecal coliforms</td>
<td>252000 (640000–290000)</td>
<td>2080 (320–5000)</td>
</tr>
<tr>
<td>BMWP</td>
<td>72 (59–89)</td>
<td>90 (74–113)</td>
</tr>
</tbody>
</table>

Stakeholder Involvement

An important element of establishing, prioritising and agreeing to intervention works in the Tarland catchment has been to include and involve the catchment stakeholders. To fulfil this requirement stakeholders have been involved in a number of ways according to their interest and stake in the catchment. At the basic level, the community have been introduced to the issues, need for intervention and concept of catchment management through a mixture of informal and formal meetings, presentations, summary information and a web site (www.macaulay.ac.uk/tarland/). At the next level of engagement an annual seminar and discussion group with the farmers has taken place. Through this mechanism additional information from past experiences in the form of anecdotal evidence has been made available. Furthermore, the options for possible tributaries have been identified and the mechanisms for reducing pollution have been discussed together with possible advantages or constraints on sites have been identified. Finally the principal land managers (the MacRobert Trust estate manager, Scottish Water) together with Agency and research staff (SEPA, SNH, Macaulay Institute and Aberdeenshire Council) formed a steering group. This group has taken the available information and views expressed by the
stakeholders, together with their individual expertise and through consensus agreed the priority, scale and type of intervention possible. Each of these three groupings has had a different need and function in taking forward and improving the catchments management. For all the three groupings there has been a need to improve the level of understanding of how a catchment operates and highlight some of the pressures on it. In return this increased capacity to understand has given rise to a greater awareness and involvement. Examples of the additional outcomes arising from the close involvement of each of these groups are given in Table 3. The aspects of the high level of involvement of the stakeholders that were not fully appreciated are identified as common issues between the different groupings. In summary these are: (a) the high level of resource requirements, largely staff time, in organising, setting up and undertaking adequate consultation, (b) the strong role of an individuals personality and traits in influencing meetings, (c) the additional resources required to overcome conflict resolution which largely arise over mis-understandings between different stakeholder groups and (d) in overcoming issues a–c there is an inevitable slowed pace of implementation.

<table>
<thead>
<tr>
<th>Grouping</th>
<th>Examples of additional outcomes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Community</td>
<td>1. Identification of issues of habitat diversity for wader birds highlighted</td>
</tr>
<tr>
<td></td>
<td>2. Desire for active involvement</td>
</tr>
<tr>
<td>Land managers</td>
<td>3. Sharing of anecdotal evidence on fish numbers</td>
</tr>
<tr>
<td></td>
<td>4. Increased reporting of environmental issues and requests for assistance</td>
</tr>
<tr>
<td>Steering group</td>
<td>5. Implementation results in multi-purpose intervention to increase water quality, habitat and riparian access</td>
</tr>
<tr>
<td></td>
<td>6. Provide in kind and cash help to implement measures</td>
</tr>
<tr>
<td></td>
<td>7. Sharing of expertise</td>
</tr>
</tbody>
</table>

Table 3: Examples of additional outputs from stakeholder consultation and participation

DISCUSSION

The monitoring results have shown that there are a range of diffuse pollution issues within the Tarland catchment that are impacting on its ecological status. Through some simple interventions following best practice guidelines on one tributary in the catchment it has been possible to improve water quality and the biology (as measured by invertebrate BMWP). Added benefit to the intervention and the potential for including more wide ranging, multi-issue benefits have been brought about by the involvement and participation of a range of stakeholders. The visual establishment and demonstratable success of the first intervention has resulted in an increased willingness to expand the measures to other tributaries in the catchment. The provision of objective data through which the changes can be quantified and shown is an important element in discussions of future developments with the stakeholders. These positive outcomes need to be balanced by consideration of some of the more difficult aspects presented by the relatively slow pace of implementation and high resource requirements necessary to ensure adequate consultation and participation. Based on the experience to date in Tarland, possibly the most effective stakeholder involvement have been developed through largely unstructured informal approaches. A
major factor in the stakeholder involvement has been the necessity for communication between all parties at all stages and that this communication has been based on a common understanding of the stakeholder requirements and the technical issues. The success of the intervention for the stakeholders can be commented on through the continued interest and involvement in the work and apparent willingness to continue with the work and to roll it out across the catchment.

Through the approach taken, i.e. the monitoring of the impact of intervention of best management practices in the catchment, it is hoped that the site can act as a demonstration for others to learn. This will become increasingly important as the implementation of the WFD moves forward.

It is interesting to note that within the context of the WFD and the River Dee catchment that SEPA, through their characterisation report, have identified Tarland as at risk of failing good ecological status through pressures from diffuse pollution and morphological change. It is also likely that the catchment will appear as a priority in the Dee catchment management plan that is currently being drafted and consulted on. Further activity and work to reduce these pressures in Tarland is currently underway through the North Sea Commission Interreg funding (3 Dee Vision).

CONCLUSIONS

This paper has introduced some of the issues that arise in relation to water quality and describes stakeholder involvement and participation through which improvements have been achieved. The programme of action suggested has been designed in such a way that an evaluation of the benefits of the implementation of measures can be assessed through the monitoring and survey network already established. The pressures which generate the problems identified and the solutions with which to reduce their impact in the Tarland catchment are generic for many rural and agricultural catchments in Scotland and other parts of Northern Europe. It is hoped that the lessons learned in Tarland can be used in the development of other community and catchment based action plans elsewhere.

ACKNOWLEDGEMENTS

The author gratefully acknowledges the invaluable assistance and help given by the MacRobert Trust, Scottish Water, SEPA, SNH, Aberdeenshire Council and the farmers and community of Tarland and Howe of Cromar. Funding for the work has been from SEERAD, SNH and Aberdeenshire Council.
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CAN WE IMPROVE PREDICTION OF P CONCENTRATION IN LUNAN LOCHS BY CHANGING THE PLUS MODEL?

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SUMMARY

This project involves changing different factors of the PLUS model in order to test how sensitive the model is to these parameters. The first factor examined was whether modifying export coefficients to better reflect land use (spring–winter cereals) improved prediction of P loading. After checking the impact of crop choice, another scenario was set aimed at investigating how sensitive the model is to changes in the P export coefficient table used in previous PLUS application. In addition, an attempt was made to represent structural Best Management Practices (BMPs) in the GIS framework and predict potential reduction in phosphorus in-loch concentrations resulting from installing buffer strips.

INTRODUCTION

PLUS is a manageable model, constructed to predict the loss of phosphorus from a complex catchment to its drainage system. The model uses an export coefficient approach, calculating the total phosphorus (TP) load delivered annually to a water body as the sum of the individual loads exported from each nutrient source in its catchment.

The catchment of the Lunan Burn, in Scotland, was selected as the study site for the present project. The Lunan chain of Lochs in Perthshire is of high conservation status and requires informed and targeted environmental protection.

A study comparing predictions of PLUS with monthly observations of P status of the Lunan chain of Lochs (SEPA, 2004) found a marked discrepancy between observations and predictions. This discrepancy could be attributed to a number of factors, but the most likely issues identified were the inadequacy of the land cover data, the use of inappropriate loss coefficients and imperfect knowledge of septic tank outputs.

Comparison of the resulting predictions for in-loch concentrations of TP with observed values for the lochs showed reasonably good agreement for the upper three lochs in the Lunan chain. However, for Lochs Clunie and Marlee the predictions of the model were significantly higher than the observed values (SEPA, 2004).

SCENARIOS APPLIED

Two scenarios were applied to Lunan catchment, with the aim of determining which of the model parameters exert greatest influence over model output. For the purpose of these scenarios, the year of 1999 was chosen as the control year. This decision was taken because among the sets of years for which data are available, 1999 was closer to the initial PLUS application during which water chemistry data from January 2000 was used.
A third scenario was applied to Clunie subcatchment with the aim of investigating any potential reduction in phosphorus in-loch concentrations, resulting from installing buffer strips on both sides of Lunan Burn in the Clunie subcatchment.

**First Scenario’s Description and Assumptions**

The aim of this scenario was to check how sensitive the model is to the proportions of winter and spring cereals. Three sets of data were used: 1989, 1994 and 2004. The assumptions made for this scenario are listed below:

- First assumption: P loss coefficients for arable land, in the P export rates table provided by MLURI for the purpose of this project, are representative of the proportion of winter and spring cereals in Lunan catchment for 1999 (control year).

- Second assumption: The model works for all five subcatchments, so the final outputs are not to be questioned. The numbers used in order to draw conclusions are not the actual output numbers for P loads in the lochs but the difference in the values between 1999 and 1989, 1994, 2004.

- Third assumption: Relative P loss coefficient for winter cereals is equal to 1 and for spring cereals is equal to 0.39 based on literature. Winter cereals were considered to be the same as having no best management practice applied and spring cereals were considered the same as conservation tillage.

Applying this scenario involved modifying the export coefficients to better reflect land use (spring/winter cereals). In order for this to be accomplished the following steps were followed:

a. A formula that calculated correction factors for P export rates based on relative areas of winter/ spring crops was created. The relative areas were calculated using agricultural census data per parish.

b. P export table was transformed according to the correction factors and a different table for each application (1989, 1994, 2004) was produced.

**Formula used to calculate P export coefficients:**

\[ A_e = \left( \frac{S}{TA} \cdot S_e \right) + \left( \frac{W}{TA} \cdot W_e \right) \]  \hspace{1cm} (1)

Where:
- \( A_e \) is the P export coefficient for arable land.
- \( S \) and \( W \) are the areas of spring and winter cereals, respectively.
- \( TA \) is the area of total arable land.
- \( S_e \) and \( W_e \) are the P loss exports for spring and winter cereals, respectively.

Based on the third assumption:

\[ S_e - 0.39 \cdot W_e \]  \hspace{1cm} (2)
From (1) and (2): 

\[ A_e = 0.39 \cdot \left( \frac{S}{TA} + \frac{W}{TA} \right) \]

\[ W_e = \frac{0.39 \cdot \left( \frac{S}{TA} + \frac{W}{TA} \right)}{A_e} \]  

(3)

For 1999 (default year): \( \frac{S}{TA} = 0.954 \) and for flat slope the minimum \( A_e = 0.40 \).

So, from (3) we get \( W_e = 0.957 \) and from (2) \( S_e = 0.373 \).

Using \( W_e \) and \( S_e \), as they were calculated based on 1999’s data, and the relative areas of winter and spring cereals for 1989, 1994 and 2004 we get from (1) \( A_e^* \) for each of the previous years.

A correction factor \( \left( \frac{A_e^*}{A_e} \right) \) is used in order to reproduce the series of export coefficients according to land slope.

Where: \( A_e \) is the export coefficient for the default year, and \( A_e^* \) is the export coefficient for arable land representative of the proportion of winter–spring cereals of the year that data are used in the equation (1).

**Second Scenario’s Description and Assumptions**

This scenario investigated how sensitive the model is to changes in the P export coefficients table used in the PLUS application. Specifically, the P loss coefficients that correspond to arable land, which is one of the biggest contributors to TP in the lochs, were improved and afterwards the effect that this change had on the model’s TP calculation, was examined.

Table 1 summarizes export coefficients for arable land, as they were collected from literature research. Based on the figures listed in Table 1, P export coefficients for cereals are lower compared with other arable crops.

**Table 1: Summary of P export coefficients from literature review**

<table>
<thead>
<tr>
<th>Land cover type</th>
<th>P (kg/ha)</th>
<th>Location</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Min</td>
<td>max</td>
</tr>
<tr>
<td>PLUS Arable</td>
<td>0.4</td>
<td>2.0</td>
</tr>
<tr>
<td>Agriculture (general)</td>
<td>0.1</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>0.1</td>
<td>6.8</td>
</tr>
<tr>
<td>Arable (general)</td>
<td>0.8</td>
<td>2.5</td>
</tr>
<tr>
<td></td>
<td>0.5</td>
<td>5</td>
</tr>
<tr>
<td>Cereals</td>
<td>0.22</td>
<td>0.9</td>
</tr>
<tr>
<td></td>
<td>0.06</td>
<td>0.9</td>
</tr>
</tbody>
</table>
Based on a detailed distribution of the different land uses in Lunan catchment and according to agricultural census data per parish for 1999, 52.8% of the total arable land in Lunan catchment was under cereals crops. This fact combined with the conclusion that cereal crops have lower export coefficients compared with other arable crops allowed us to accept the assumption that P export coefficients for arable land used for the default application could be reduced by 40% and still be representative of the Lunan catchment.

So, applying this scenario involved reducing arable land P export rates by 40% and creating a new P export table for the application.

Third scenario’s description and assumptions

This scenario was based on two main assumptions:

- First assumption: Buffer strip efficacy concerning prevention of phosphorus losses is 40%, based on literature search (Vinten et al., 2004).
- Second assumption: In order for the buffer strip to be effective, the slope should be less than seven degrees (Vinten et al., 2004).

The only way to implement a BMP, like a buffer strip, in the PLUS model, is by trying to include the effect of such practice in the P loss coefficients. Figure 1 shows the fields that were assumed to be influenced by the installation of the buffer strip.

![Figure 1: Fields that were assumed to be influenced from the installation of the buffer strip (dotted area across both sides of the Lunan Burn depicts buffered fields)](image)

After identifying the buffered fields, P loss coefficients that corresponded with those fields were reduced by 40% from their initial value, according to the first assumption. This new P export table was then used in order to run the PLUS model.
RESULTS

Figure 2 shows the estimates of TP concentration for Clunie loch from all the three scenarios applied to Lunan catchment compared with the default application.

The outputs from the first scenario suggest that taking into account the change into winter/spring cereals distribution has some positive effect on estimating TP in loch concentration, but the difference is not statistically significant at 95% confidence.

The outcome of the second scenario applied, demonstrates that model calibration by inspecting and adjusting the export coefficients is required and has the potential to result in a more accurate estimation of the total in-loch concentrations.

The descriptive statistics support the conclusion that the residuals of the second scenario are better behaved compared with the default application. The mean of the residuals is closer to zero and furthermore the standard deviation of the residuals is smaller compared with the one of the control year. So, the notion that the scenario application is more efficient is verified, since the high values of the residuals in the case of the default application could mean that the independent variables used did not explain satisfactory the variance of the dependent variable. It should be stated though that the limited number of observations available diminishes the credibility of the results.

The results of the third scenario, compared with the results of the default application suggest that potential installation of buffer strip would have positive effect on holding of TP to Clunie Loch. Specifically, minimum in loch TP concentration was estimated to decrease from 22.9 to 13.2 µg/L, while the maximum was estimated to decrease from 39.3 to 20.3 µg/L.

DISCUSSION

The conclusion that when the different proportions of winter and spring crops were taken into consideration did not have any significant difference in the model outputs
could be misleading. The results of the first scenario can be attributed to the following factors:

- The proportion of winter crops in the Lunan catchment is very low, and specifically winter crops in the catchment are limited to winter cereals, which compose just 6.7% of the overall cereal crops, based on 1999 agricultural census data;
- The amount of potatoes could also be considered as winter crop, given that potatoes increase the risk of run-off;
- Finally, the third assumption that was made could be questioned.

The range of P export coefficient used by PLUS for arable land is 0.40–2.00 kg/ha/year, while the range of P export coefficient for arable land used for the purpose of the second scenario is 0.24–1.20 kg/ha/year. In order for this assumption to be justified, results of P export studies that were carried out on two main subcatchments/ farms in the Lunan Lochs chain are used:

- Craigie Burn/ Hillocks of Gourdie Farm
- Cattymill Burn/ Meadows of Ballied Farm

Based on the samples (Table 2), the exports are higher than 1 kg/ha in specific areas in the catchment. This figure weakens the argument that the PLUS exports (0.40–2.00 kg/ha) are too high. It has to be kept in mind though that the ditch that drains the field where repeated applications of poultry manure takes place drains afterwards into Black Loch. So, the high value of P loss could be attributed to the poultry manure application. However, the Cattymill catchment is mainly arable. More accurate conclusions regarding the range of P exports could be drawn if a series of samples well distributed across the whole Lunan catchment was available.

Table 2: Summary of P export from three catchment areas

<table>
<thead>
<tr>
<th></th>
<th>Catchment area (ha)</th>
<th>Sampling period</th>
<th>Mean [TP] (mg/L)</th>
<th>P loss (kg/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Craigie Burn</td>
<td>Ditch draining a field with poultry manure</td>
<td>18</td>
<td>13/10/04 to 13/3/05</td>
<td>0.27</td>
</tr>
<tr>
<td></td>
<td>Black Loch outlet</td>
<td>60</td>
<td>8/4/05 to 30/7/05</td>
<td>0.49</td>
</tr>
<tr>
<td>Cattymill Burn</td>
<td>Meadows of Ballied</td>
<td>13/10/04 to 12/05</td>
<td>0.34</td>
<td>0.99</td>
</tr>
<tr>
<td></td>
<td>Chapelton</td>
<td>2453</td>
<td>13/10/04 to 19/10/05</td>
<td>0.22</td>
</tr>
</tbody>
</table>

CONCLUSIONS – FUTURE WORK

Taking under consideration the change into winter/spring cereals distribution has some positive effect on estimating TP in loch concentration, but the difference is not statistically significant at 95% confidence. However, this is attributed to a number of factors related to Lunan catchment and to the assumptions of the scenario.

Changing P export coefficients for arable land, based on literature search, resulted in a decrease of the calculated TP in the lochs by up to 20%. This demonstrates that model calibration by inspecting and adjusting the export coefficients is required and has the potential to result in more accurate estimation of the total in-loch concentrations.
It is worth considering ways to represent BMP’s in GIS framework, as it can be a useful tool when it comes to catchment management.

Input data refinement is one of the areas for future improvement. Updated land cover theme is required as the land cover theme used at the present project depicts year 1988 (LCS88) and questions the comparison of the model’s output with 2000 water chemistry data.

ACKNOWLEDGEMENTS

I would like to acknowledge SEPA, SEERAD and MLURI for providing me with the data I needed for the present project.

REFERENCES


TACKLING DIFFUSE NITRATE POLLUTION: SWAPPING EUTROPHICATION FOR CLIMATE CHANGE?

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SUMMARY

Large areas of the UK have now been designated ‘Nitrate Vulnerable Zones’ (NVZs) as a result of the high levels of nitrate in surface and groundwaters. Where nitrate is lost from the fields through drainage there are two main remediation strategies available. The first involves the creation of strips of uncultivated land, known as ‘buffer strips’ to intercept the leached nitrate before it gets into drainage waters. The other is designed to intercept any nitrate already in drainage waters by diverting the water through natural or constructed denitrifying wetlands.

The use of both buffer strips, often in riparian zones, and denitrifying wetlands, has become increasingly widespread in the past few years (Mitch et al., 2001; Sabater et al., 2003), but doubts remain over their suitability in many areas (Dosskey, 2002). As well as the potential problem of leached nitrate by-passing the buffer strip and wetland soils, there is also a danger that denitrification in their soils - the primary process of nitrate removal – will lead to greatly increased nitrous oxide (N2O) emissions (Hefting et al., 2003), so swapping a water pollution problem for a climate change issue (Reay, 2004). Here we present a synthesis of N pollution swapping studies to date and introduce a wide-ranging project aimed at addressing this key issue.

REFERENCES


CATCHMENT SENSITIVE FARMING ON THE HAMPSHIRE AVON

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SUMMARY

The Hampshire Avon Multi-Agency Catchment Sensitive Farming (MACSF) pilot, started in 2005, takes much of its local lead from the ‘Landcare’ project, which has been running since 1997. However, the MACSF project is farther reaching than its local forerunner. It focuses not only on the local reduction of diffuse pollution from agriculture but also on a robust evaluation strategy, effective communication and providing key information, lessons learned and support for the, soon to be appointed, CSFOs.

Delivery of CSF on the Avon is in the form of soil management planning workshops and follow-ups surgery, cultivation trials on demonstration farms, 1:1 field-specific specialist advice, farm visits and agronomy newsletters. Monitoring and evaluation strategies vary over the four MACSF pilots, the Avon strategy focuses on changes in actual behaviour and land management, changes in soil structural degradation and storm inspection surveys. Less emphasis has been placed on water quality monitoring due the impact of weather patterns on the measured parameters and the short duration of the project. Issues being addressed include raising general and farmer awareness of diffuse pollution, stakeholder expectation management and designing an evaluation strategy that will produce high quality data independent of the weather.

If the pilot catchments succeed, they will not only reduce diffuse pollution locally, and inform other catchments of best practice, but importantly, they will be used in assessing whether the voluntary approach can reduce diffuse pollution from agriculture sufficiently for the demands of the Water Framework Directive. In an industry where the general perception is one of over-regulation, a successful voluntary approach to reducing diffuse pollution is an attractive idea.
AMMONIA VOLATILISATION FROM CATTLE SLURRY APPLIED TO GRASSLAND: EFFECTS OF APPLICATION TECHNIQUE AND RATE

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SUMMARY

The effects of slurry application rate (20–80 m³/ha) and method (broadcasting vs. bandspreading/shallow injection) on ammonia (NH₃) emissions were studied at a clay loam grassland site in Cheshire. Shallow injection reduced NH₃ emissions by c. 60% compared with surface broadcast application, because the slurry was retained within the shallow injection slot, which reduced the exposed surface area that was susceptible to NH₃ loss. However, the trailing shoe application did not reduce NH₃ emissions compared with surface broadcasting, which was due to a combination of the high hydraulic loading rate from the trailing shoe slurry application (c. five times greater than the surface broadcast) and the moist soil conditions, which delayed the rate of slurry infiltration into the soil. There were no relationships between broadcast, shallow injected or trailing shoe slurry application rates and NH₃ emissions. This work indicates that factors which affect the infiltration rate of slurry into the soil (i.e. hydraulic loading rate, soil moisture, slurry dry matter content) are important factors in controlling NH₃ losses from slurry applications.

INTRODUCTION

Ammonia (NH₃) emissions from UK agriculture were estimated at 218,000 tonnes NH₃-N per year in 2000, with losses after the land spreading of farm manures responsible for c. 36% of total emissions (Defra, 2002). In order to protect sensitive habitats, the UK has signed a number of international agreements to reduce NH₃ emissions, including the UNECE Gothenburg Protocol, the EC National Emission Ceilings Directive (NECD) and the EC Integrated Pollution Prevention and Control (IPPC) Directive.

Previous work in the UK has shown that at application rates of 30–35 m³/ha slurry bandspreading and shallow injection on grassland can reduce ammonia emissions compared with surface broadcasting by c. 40% and 70%, respectively (Smith et al., 2000). In addition, bandspreading techniques (i.e. trailing shoe and hose) allow slurry to be spread on growing crops in the spring, with much reduced slurry contamination. However, the effectiveness of band spreading and shallow injection techniques in reducing NH₃ emissions can be variable, with factors such as application rate (which can be controlled by the farmer) and the retention of slurry in a band (for bandspreading) or slot (for shallow injection), likely to be important in controlling NH₃ emissions.

The aim of this study was to evaluate the effects of slurry application rate and method on NH₃ emissions from cattle slurry applied to grassland.
MATERIALS AND METHODS

Cattle slurry was applied to a clay loam textured grassland soil in Cheshire using broadcast and shallow injection application techniques in November 2003, and broadcast and trailing shoe application techniques in March 2004. Slurry was applied at five application rates (20, 35, 50, 65 and 80 m$^3$/ha), with each treatment replicated three times. Each plot was 6 m wide x 12 m long, and was arranged in a randomised block design. Slurry was applied using a specialist plot applicator (Plate 1, Basford et al., 1996), which was representative of commercial practice, with the slots/bands 20 cm apart. Ammonia emissions were measured from each plot for 7 days after application, using wind tunnels (Plate 1) based on the design of Lockyer (1984). The depth of the injection slots (after slurry injection) and width of the slurry bands (after both the shallow injected and trailing shoe slurry application) were measured, and visual assessments were made of the time taken for the slurry to infiltrate into the soil.

RESULTS AND DISCUSSION

In November 2003, shallow injection reduced (P < 0.001) NH$_3$ emissions by an overall mean of 59% compared with surface broadcast application (Figure 1). The cattle slurry (5.2% dry matter, 3.1 kg/m$^3$ total N) was effectively retained in the 7–8 cm deep injection slots at the application rates between 20 and 50 m$^3$/ha (c. 20% soil surface occupied by slurry), although there was some ‘overspill’ of the injection slots at the two higher application rates (65 and 80 m$^3$/ha; 25–30% soil surface occupied by slurry). Ammonia emissions were equivalent to a mean of 14% of the total N applied (range 10–18%) from the broadcast application and a mean of 6% of the total N applied (range 3–7%) from the shallow injected application. The reduction in NH$_3$ emissions achieved by the shallow injection technique reflected the smaller emitting surface area compared with the surface broadcast applications. There was no relationship (P > 0.05) between broadcast or shallow injected slurry application rates and NH$_3$ emissions in November 2003.

In March 2004, the trailing shoe application did not reduce NH$_3$ emissions compared with the surface broadcast application, even though the high dry matter slurry (8.9% dry matter) remained in a band, covering 18% of the soil surface area at 20 m$^3$/ha.
and 27% of the soil surface at 80 m³/ha. Ammonia emissions were equivalent to a mean of 10% of the total N applied (range 7–14%) from the broadcast application and a mean of 12% of the total N applied (range 7–20%) from the trailing shoe application (Figure 2).

Figure 1: Ammonia losses after shallow injection and broadcast slurry applications to grassland in November 2003

The higher hydraulic loading from the trailing shoe slurry application (11–30 mm per unit of surface area occupied by the slurry) compared with the surface broadcast application (2–8 mm), meant that while the slurry stayed in a band, infiltration into the moist clay soil was restricted (Table 1). Hence, the NH₃ emission patterns and rates were similar from the two techniques, even though the emitting surface of the trailing shoe application was only 18–30% that of the surface broadcast application. Again, there was no relationship (P > 0.05) between broadcast or trailing shoe slurry application rates and NH₃ emissions in March 2004.

Figure 2: Ammonia losses after bandspread and broadcast slurry applications to grassland in March 2004
Table 1: Broadcast and trailing shoe slurry application details (March 2004)

<table>
<thead>
<tr>
<th>Application rate (m³/ha)</th>
<th>N applied (kg/ha)</th>
<th>Surface area occupied by band spread slurry (%)</th>
<th>Hydraulic loading rate (mm)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Broadcast</td>
</tr>
<tr>
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<td>363</td>
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<td>8.0</td>
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</tbody>
</table>

CONCLUSIONS

Shallow injection of slurry into the clay loam grassland soil (at application rates between 20 and 80 m³/ha) reduced NH₃ emissions by c. 60% compared with surface broadcast application, largely as a result of the smaller emitting surface area. Although there was some ‘overspill’ from the injection slots at the higher application rates, the majority of the slurry remained within the slot and this did not affect NH₃ emissions as a % of the total N applied.

Slurry application using a trailing shoe did not reduce NH₃ emissions compared with surface broadcast application. Although the slurry remained within a band (surface area occupied 18–30%), the combination of a high hydraulic loading rate (11–30 mm) and wet soil conditions restricted the rate of slurry infiltration into the soil, which resulted in similar NH₃ losses to the broadcast application.

These data indicate that for slurry shallow injection and bandspreading techniques to successfully reduce NH₃ emissions, slurry needs to be retained in a band/slot (to give a smaller emitting surface area) and to rapidly infiltrate into the soil. If both conditions are not satisfied, the NH₃ reduction benefit of bandspread/shallow injection spreading techniques will not be realised.

ACKNOWLEDGEMENTS

Funding of this work by Defra is gratefully acknowledged.

REFERENCES


FIELD TESTING OF MITIGATION OPTIONS FOR PHOSPHORUS AND SEDIMENT (MOPS)

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SUMMARY

Diffuse phosphorus (P) pollution contributes to the eutrophication of surface waters and is a serious problem in the UK. Losses of P associated with soil particles are often linked to soil erosion. There are already a wide range of options for reducing soil erosion and the subject has received considerable research effort. However, less is known about the effectiveness of these methods for reducing P losses. To address this gap, the Mitigation Options for Phosphorus and Sediment (MOPS) project focuses on a range of treatments with potential for mitigating P losses associated with combinable crops. Field experiments have been established at three contrasting sites in the UK. At each site a number of different treatments are being investigated including cultivation techniques, tramline management, cover crops and vegetative barriers. These treatments reflect different levels of intervention. The project will also consider the financial costs associated with adopting these different mitigation options.

INTRODUCTION

Diffuse P pollution, predominantly from agricultural sources (including fertiliser and animal waste), is a serious problem in the UK and contributes to the eutrophication of waterways and standing water bodies. Surface waters in the UK are strongly limited by P and even small additions can cause eutrophication. P occurs naturally within soils, but high inputs in the form of organic and inorganic fertilisers from agricultural crops have resulted in considerably higher levels of soil P than are utilised by the crop plants. Inputs in the form of fertiliser and manure are between 20 and 50 kg ha/year (Haygarth et al., 1998) and (Haygarth et al., 2005) have shown that total losses of P from soils are estimated to be approximately 1 kg/ha/year.

Phosphorus binds to soil particles and losses of P associated with soil particles are often linked to soil erosion. As a result of the research effort since the 1930s there are a wide range of effective mitigation options for reducing soil erosion. However, much less is known about how effective these mitigation options are for reducing the P losses associated with sediment.

The Mitigation Options for Phosphorus and Sediment (MOPS) project focuses on a range of treatments with potential for reducing P losses associated with combinable crops. The treatments being investigated reflect different levels of intervention and are suitable for inclusion under Cross Compliance and the Entry Level Stewardship scheme or Higher Tier options. All the treatments have the aim of reducing sediment losses by reducing overland flow velocity, protecting the soil surface or managing soil structure.
MATERIALS AND METHODS

Field Experiment

Field experiments have been established at three contrasting sites (Figure 1). The soils at Loddington in Leicestershire consist of Hanslope and Denchworth series clays on an erodible slope. The site is run by the Allerton Trust which seeks to demonstrate means of farming profitably with minimal environmental impact. The soil at the Moorfield site at ADAS Rosemaund consists of a silty clay loam of the Bromyard/Middleton series. The site at Old Hattons, near Wolverhampton is owned and operated by Seven Trent Water plc. The soils here are of a sandy loam texture, have a high P content and are prone to erosion. They are poorly structured sandy loam soils overlying compacted glacial till. A number of different treatments with the potential to reduce P losses will be investigated at each site.

Figure 1: The location of the Loddington, Moorfield and Rosemaund field sites

At each field site run-off from unbounded plots (12 m wide and the length of the hill slope) will be collected in troughs towards the base of the slope and will run through pipes to sample splitters. The sample splitters will enable the collection of between 50% and 12.5% of run-off depending on conditions. The collected run-off will be stored in tanks for subsequent sub-sampling and analysis.

TREATMENTS

Cultivation Techniques

Minimum tillage is currently being promoted as an erosion control tool and experiments in Bedfordshire have shown that cultivation across the slope and minimum tillage
techniques can significantly reduce soil erosion on sandy soils (Quinton and Catt, 2004). Unpublished data by the same authors has indicated that it can also reduce losses of total P. We will trial minimum tillage (at Loddington and Rosemaund) and across slope cultivation (at Loddington) to determine the effects on P losses.

**Tramline Management**

Tramlines are a key conduit for the transfer of sediment and P at the field scale and previous work (DEFRA project number PE0111 – ADAS, 2003) has shown that concentrations of total P in run-off from fields with no tramlines were half of those with tramlines. However, tramlines are an important component of modern agriculture so methods (e.g. tramline disruption and vegetative techniques) to reduce run-off on tramlines will be investigated.

**Cover Crops**

The use of cover crops is recommended in Nitrate Vulnerable Zones (NVZs) but there is little data available to demonstrate whether they are suitable for reducing P loses. Cover crops rely on establishing rapid canopy cover to prevent raindrop impact and reduce the velocity of surface run-off therefore reducing P losses in sediment. Crops to be investigated in this project include wheat and barley.

**Crop Residue Management**

Widely used in the United States to control soil erosion, crop residue management involves the use of residues of the previous year’s crops to protect the soil surface and maintain an open soil structure so water can infiltrate. This will potentially be a key technique to use on the weakly structured soils.

**Vegetative Barriers**

Long slopes were identified as important for the transport of P in a previous investigation (ADAS, 2003). In this project we will reduce slope length will be reduced by introducing live vegetative barriers across slopes in the form of beetle banks. These are already funded under the Environmental Stewardship scheme to enhance biodiversity and introduce natural predators.

**Laboratory Analysis**

The run-off samples collected will be analysed for total P and total P < 0.45 mm using acid molybdate/antinomy with ascorbic acid reduction (Department of Environment, 1992; USEPA, 1985) and determined spectrophotometrically (880 nm) using a Seal Analytical AQ2 analyser. Total nitrogen will be determined using high temperature catalytic oxidation and chemoluminescence. Total suspended solids in the run-off samples will be determined using a standard filtration and drying technique (Bartram and Balance, 1996).

**DISCUSSION**

This project will allow the development and testing of recommendations for mitigating P losses with consideration of pollution swapping. Measures taken to reduce the losses of one pollutant to the environment can cause other pollutants to increase. This project is primarily looking at the impact of the selected mitigation option on
P losses but will also consider losses of N and sediment in order to recommend management practices that can provide maximum environmental benefit.

The testing of the mitigation measures examined in this project will provide evidence to help farmers select the most suitable methods for mitigating losses of P from their land. It will also help bodies such as DEFRA to target funding for farmers to undertake mitigation measures. If mitigation measures are to be adopted by farmers they need to be at worst cost-neutral. Analyses will consider the costs associated with introducing and maintaining the mitigations practices and the effect of adopting mitigation options upon economic land management. These analyses will also consider whether any cost shortfall may be met from other funding sources.

ACKNOWLEDGEMENTS

The authors are grateful to DEFRA for funding this project.

REFERENCES


NITRATE CONTAMINATION OF GROUNDWATER FROM AGRICULTURE IN CANTERBURY, NEW ZEALAND: MEASUREMENT AND MANAGEMENT OF A HIDDEN PROBLEM

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SUMMARY

Nitrate contamination of groundwater from agriculture is an increasingly important issue for resource management in Canterbury. Most people in the region are dependent on untreated groundwater for drinking, and although communities place a high value on groundwater quality, over much of the plains the resource is unconfined and therefore at risk from nitrate contamination. While samples from only a small proportion of wells in the monitoring network (2-6%) have nitrate-N concentrations in excess of NZ Drinking Water Standards, the data show a trend of increasing concentrations (up to 0.7 mgL⁻¹ yr⁻¹) across large parts of the alluvial aquifer system. This is particularly marked in those areas where the primary source of recharge is rainfall, as opposed to seepage from rivers. Agricultural development on the Canterbury Plains has been ongoing for many years, but there has been a shift in production systems over the last decade in particular, from mixed cropping and dry-land sheep grazing to more intensive irrigated pastoral and arable farming, driven in part by developments in irrigation technology. Dairying has emerged as a major land use in formerly dry-land areas. This intensification has the potential to contribute to further increases in nitrate concentrations in groundwater. The management response to the nitrate trend has included: increased research effort to quantify N contributions from multiple sources over time and at a range of spatial scales, more stringent regulation of point-source discharges, and the promotion of best management practices. The most important new initiative is a proposed rule requiring landowners to estimate nitrate-leaching losses, and to undertake nutrient budgeting and other measures if leaching thresholds are exceeded.
RAPID INCORPORATION OF SOLID MANURE AS A BEST MANAGEMENT PRACTICE?

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SUMMARY

After solid manure spreading, rapid incorporation by ploughing has been recognised as a successful technique to reduce ammonia (NH₃) emissions. However, the reduced NH₃ loss conserves nitrogen which may subsequently be emitted from the soil as the greenhouse gas, nitrous oxide (N₂O). Emissions of N₂O were monitored at two UK sites (central and south west England) after application of cattle manure, pig manure, layer manure and broiler litter to arable land. At both sites, ploughing reduced NH₃ emissions compared with those from plots where manure remained on the soil surface (P < 0.001). However, the effect of incorporation on N₂O loss was inconsistent. In the warm and wet south west, incorporation had no effect on N₂O emissions (P > 0.05), but at the cooler and drier central site, N₂O losses increased (P < 0.001), suggesting that rapid incorporation may be considered as a best management practice (BMP) for both N₂O and NH₃ abatement only under site-specific conditions.

INTRODUCTION

Around 43 million tonnes of solid manure are handled annually in the UK (Williams et al., 2000), with 36% of agricultural ammonia (NH₃) emissions (82 kt N) estimated to arise from the management of solid manures (Misselbrook et al., 2000). After land spreading, ammonia emissions of manures represent both a large loss of potential crop available nitrogen and a considerable environmental risk. Deposition of NH₃ can lead to soil acidification and nitrogen (N) enrichment of sensitive habitats, consequently the UK has signed a number of international agreements [UNECE Gothenburg Protocol, EC Integrated Pollution Prevention and Control (IPPC) Directive, EC National Emission Ceilings Directive] to reduce NH₃ emissions. After the application of solid manures to arable land, rapid incorporation has been identified as an effective measure to abate NH₃ emissions (Webb et al., 2005). However, the reduced NH₃ loss conserves N increasing the mineral N pool in the soil, which may subsequently be available for microbial nitrification and denitrification and the production of the greenhouse gas, nitrous oxide (N₂O). Nitrous oxide is a potent greenhouse gas with a global warming potential 310 times that of carbon dioxide (IPCC, 1996). The current UK emissions inventory (2003) shows that 67% of N₂O is produced from agriculture with the majority emitted from agricultural soils (Baggott et al., 2005). As a result of the Kyoto protocol, the UK has agreed to a legally binding reduction of greenhouse gas emissions of 12.5% of 1990 levels by the period 2008–12. It is, therefore, important that measures implemented as a BMP to reduce NH₃ emissions do not result in an increased loss of N₂O.
MATERIALS AND METHODS

At two UK sites: site 1 at ADAS Gleadthorpe, Central England and site 2 at IGER North Wyke, south west England, \(\text{N}_2\text{O}\) and \(\text{NH}_3\) emissions were monitored from replicated (x 4) plots (6 x 10 m) after a spring (February/March) application of solid manure. The plots were established on cereal stubble on a loamy sand soil (site 1) and on bare arable ground on a coarse sandy loam soil (site 2). Cattle farm yard manure (FYM), pig FYM, layer manure or broiler litter were spread at a target application rate of 250 kg N/ha and either left on the surface or immediately incorporated (to 20–25 cm depth) by ploughing. Control treatments were included where no manure was added.

Measurements of \(\text{N}_2\text{O}\) were made from two static flux chambers (40 cm wide x 40 cm long x 25 cm high), that were placed in random positions on each plot after the incorporation treatment had been completed. Chambers were pushed into the soil up to a depth of 5 cm to ensure an airtight seal and headspace samples analysed as soon as possible after collection by gas chromatography. The \(\text{N}_2\text{O}\) flux was calculated based on the linear increase in \(\text{N}_2\text{O}\) concentration inside the chamber over a 40-minute enclosure period. Nitrous oxide emission measurements were carried out immediately after manure application and at regular intervals over a c. 60-day period.

Ammonia emissions were monitored for up to 2 weeks after manure application using a modified wind tunnel technique (one per plot) based on the design of Lockyer (1984). Each wind tunnel consisted of two parts; a transparent polycarbonate canopy (2.0 x 0.5 m) which covered the plot area, and a stainless-steel duct housing a fan which drew air through the canopy at a speed of 1 m/s. A sub-sample of the air entering and leaving the tunnel was drawn through flasks containing 0.02 M orthophosphoric acid, which absorbed \(\text{NH}_3\) present in the air stream. The acid was subsequently analysed for ammonium-N concentration by automated colorimetry. The \(\text{NH}_3\) emission was calculated as the product of air that flowed through the tunnel and the difference between the concentrations of \(\text{NH}_3\) in the air entering and leaving the tunnel.

RESULTS AND DISCUSSION

Ploughing reduced mean \(\text{NH}_3\) emissions (\(P < 0.001\)) by 97-98%, with losses from the ploughed treatments at –0.3% and 0.4% of total-N applied compared with losses of 15.5% and 19.0% of total-N applied from the plots where manure was left on the surface. However, at site 1 ploughing increased \(\text{N}_2\text{O}\) losses (\(P < 0.001\)) compared with those from surface application (Figure 1). Mean \(\text{N}_2\text{O}\) losses from the ploughed treatments were a factor of 4’ larger at 1.39% of total-N applied compared with losses of 0.36% of total-N applied from the plots where manure was left on the surface. In contrast, at site 2, there was no effect of incorporation on \(\text{N}_2\text{O}\) emissions (\(P > 0.05\)), although there was a suggestion that ploughing reduced \(\text{N}_2\text{O}\) losses compared with surface application. Losses were 0.54% of total-N applied and 1.44% of total-N applied from ploughed and surface applied treatments respectively. The discrepancy in the effect of ploughing on \(\text{N}_2\text{O}\) emissions between sites is probably related to differences in soil conditions.

The mean soil temperature (5-cm depth) over the monitoring period at site 2 was 10.7°C with 19% of the mean daily temperatures 8°C or less. At site 1, the mean
soil temperature was 9.3°C, but with 41% of the mean daily temperatures ≤ 8°C. Site 2 was wetter than site 1 with twice as much rain (174.4 mm) falling over the N₂O monitoring period compared with site 1 (85.8 mm). Numerous studies in the literature have shown that N₂O production increases with temperature and can be stimulated with a rise in soil moisture (Dobbie et al., 1999; Scott et al., 2000). Ploughing is likely to increase the length of the diffusion pathway from the site of N₂O production (the FYM) to the soil surface. Soil texture and structure would influence this diffusion rate. So at site 2, although conditions were more favourable for N₂O production than at site 1, the rate of N₂O diffusion through the soil was probably slower due to the heavier textured and wetter soil. This would provide a greater opportunity for N₂O reduction to N₂ and hence lower N₂O emissions.

Later work from this project at a site with a clay soil in central England has shown similar results to those at site 2. Despite no effect of ploughing (P > 0.05) on N₂O losses calculated over c. 60 days, surface application resulted in mean losses 4x larger than from ploughing. Evidence from the literature also indicates an uncertain effect of ploughing/simulated incorporation of solid/liquid manures such that, depending on site conditions, the impact on N₂O may be neutral, or result in enhanced or reduced emissions (Velthof et al., 2003; Thorman et al., 2005a, b).

![Figure 1: Total N₂O emission (% total-N applied) calculated over c. 60 days after application of solid manure to cereal stubble (site 1) and bare arable ground (site 2) in spring. CM, cattle manure; PM, pig manure; LM, layer manure; BL, broiler litter; surf, surface application; plo, plough. Error bars represent ± one standard error of the mean](image)

**CONCLUSIONS**

This study shows that the effect of rapid incorporation by ploughing on N₂O loss was inconsistent. In the wet, south west ploughing had no effect on N₂O emissions, but at the drier, central site N₂O losses increased, suggesting that rapid incorporation by ploughing as a best management practice for NH₃ abatement may only be used as a ‘win–win’ technique under site specific conditions.
ACKNOWLEDGEMENTS

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REFERENCES


METHODS TO IMPROVE POLLUTION CONTROL POTENTIAL OF WOODCHIP CORRALS

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SUMMARY

This project investigated the efficacy of different potential sublayers for removal of some key residual pollutants after passing slurry through woodchip corrals. Pollutants studied were: ammonium, nitrate, Escherichia coli, mineral phosphate and organic carbon (TOC). The sublayers investigated were loamy sand subsoil, sandy loam topsoil, iron ochre granules (to remove P by adsorption) and straw (to remove N by immobilisation). Good pollutant removal was obtained by all sublayer treatments, with the exception of the straw treatment. This suffered from problems with packing the straw between woodchips and subsoil, leading to uneven flow conditions. Nitrate concentrations in leachate rose to > 40 mg/L but results suggest significant denitrification was occurring. The value of the iron ochre and straw was demonstrated in a laboratory column experiment, in which removal of nutrients equivalent to 188 kg N/ha and 92 kg P/ha was achieved by straw and ochre layers.

INTRODUCTION

In Scotland, woodchip corrals are being widely used as a means of overwintering beef cattle. The leachate from corrals still has a considerable pollution potential (Vinten et al., 2006), some of which can be mitigated if it has the chance to pass through suitable sublayers before release to ground or surface waters. The idea of this project was to investigate the efficacy of different sublayers for removal of the key pollutants: ammonium, nitrate, E. coli, suspended solids, phosphate and dissolved organic carbon (DOC). The sublayers investigated were sandy subsoil, sandy loam topsoil, iron ochre granules (to remove P by adsorption) and straw (to remove N by immobilisation).

MATERIALS AND METHODS

Eight lysimeters (1.1 x 0.9 x 1 m) were set up at SAC Auchincruive (Figure 1). There were four sublayer treatments (each in duplicate): control (40 cm of sandy subsoil of Dreghorn series from the site area), topsoil (40 cm of sandy loam topsoil of Dreghorn series, from the site area), iron ochre (10 cm of dried granular iron ochre from waste lagoons at Polkemmet open cast coal mine, underlain by sandy subsoil) and straw (20 cm of barley straw coarsely chopped, underlain by sandy subsoil). Overlying these sublayers were 20-cm depth of medium-size woodchips (approx. 15 x 5 x 5 cm chips) and 20 cm of large woodchips (20 x 10 x 10 cm). Weekly inputs of slurry (14 mm equivalent depth) containing 1800 mg/L NH4-N, 33,000 mg/L total organic C (TOC), 3.4 x 107 colony forming units (cfu) E. coli/100 mL and with electrical conductivity of 14 mS, along with natural rainfall were applied from January to September 2004. Weekly accumulated samples of leachate were analysed for ammonium, nitrate, phosphate, dissolved organic (DOC), and electrical conductivity.
by standard methods and fresh, end of week samples were analysed for *E. coli* using the Colilert method (IDEXX, 2001). Volume of flow was also noted.

Two laboratory columns were also set up: control (washed sand) and straw/iron ochre overlaying sand to demonstrate the nutrient removal concept for these materials. Dilute slurry (50 mm/day) containing on average 90 mg/L NH$_4$-N, 107 mg/L DOC, 5.9 mg/L soluble reactive P (SRP) and with electrical conductivity of 343 mS was applied to these columns from 29 October to 12 December 2003.

RESULTS

Figure 2 shows the breakthrough curves of EC, DOC, *E. coli*, NH$_4$-N, NO$_3$-N and PO$_4$-P for control (40 cm subsoil) and ochre (10 cm ochre and 30 cm subsoil) lysimeters. In the interests of clarity, results for the topsoil and straw treatments are not shown in Figure 1, but the average leachate concentrations in the period from May to September 2004 are given in Table 1. Over the time course of the experiment, the results show that removal of all pollutants was good for all sublayers except straw. The straw treatment was the least effective, probably because the water did not move uniformly through this layer due to difficulties in packing of the straw layer between subsoil and chips.

![Figure 1: Summary of lysimeter layout and design](image-url)
Table 1: Concentrations of pollutants in leachate from lysimeters, compared with slurry input (mean and SD of samples taken from 3rd May [after cumulative input of rainfall (403 mm) + slurry (224 mm)] to 30 August [after further cumulative rainfall input of 344 mm])

<table>
<thead>
<tr>
<th></th>
<th>Material below woodchips</th>
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<tbody>
<tr>
<td></td>
<td></td>
<td>Control</td>
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<tr>
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<td>0.57</td>
</tr>
<tr>
<td>Log E. coli</td>
<td></td>
<td>7.5</td>
<td>2.3</td>
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</tbody>
</table>

*Mean input slurry dilution factor with rainfall = 2.8.

The EC relative to the average EC value of the rainfall + slurry inputs (EC/EC_{input}) until slurry application ceased (after 166 mm of drainage, 403 mm rainfall and 224 mm of slurry, 12 January to 3 May 2004) increased to a maximum of 0.62 in the control treatment. If we assume the EC is controlled primarily by transport of weakly sorbed ions such as Cl⁻, this line is indicative of the behaviour of a conservative, non-reactive, non-adsorbed tracer. The initial EC in leachate from both the ochre and straw lysimeters was significantly higher than from the other treatments, but it is not clear why at present. The DOC concentrations began to rise at the same time as the EC, but values were much lower, relative to input, showing that microbial degradation and/or sorption occurred. The NH₄⁻N showed a similar pattern to DOC, and the DOC/NH₄⁻N ratios were similar to the input values, suggesting that oxidation and sorption of organic C and NH₄⁻N follow similar patterns. Nitrate concentrations rose steadily in the drainage water with time, to a maximum of approximately 40 mg/L and there was no delay relative to EC, suggesting that rapid nitrification of NH₄⁻N occurred. However, NH₄⁻N concentrations relative to input values were much lower than EC/EC_{input}. This suggests either that for a large part of the NH₄⁻N added, nitrification is slow, and/or that denitrification is occurring. Given the time course of nitrate breakthrough, the latter explanation is more likely, but the two alternatives cannot be properly distinguished without further work, for example by comparison of the d15N content of input and output N (e.g. Krapac et al., 2002) and assessment of ammonium adsorption, which will delay the pollutant transport. The E. coli concentrations in the leachate rose before the EC showing that there was significant exclusion of bacterial cells from smaller soil pores. Except for the straw treatment, E. coli concentrations were very low relative to input values, showing that most of the E. coli was inactivated during transport through the soil. The concentrations were
comparable to those in water draining slurry treated or sheep-grazed drained plots (Vinten et al., 2004). The PO₄-P concentrations draining from all treatments were very low, comparable to those draining from arable fields, but the values were lowest from the ochre treatment. The ratio of P to NH₄-N in the drainage water was similar to that in the input slurry.

Figure 2: Lysimeter leachate concentrations of EC, DOC, E. coli, NH₄-N, NO₃-N and PO₄-P for control (40 cm of subsoil) and ochre (10 cm of ochre and 30 cm of subsoil) lysimeters
The results of the column experiment are summarised in Figure 3. These show that P removal by the ochre/straw was nearly complete (equivalent to 92 kg P/ha), and N removal equivalent to 188 kg N/ha during addition of 450 kg N/ha occurred, relative to the sand only column. This confirms that the ochre and straw treatments have the potential to remove substantial quantities of P and N respectively, if the hydraulic properties are suitable.

Figure 3: Mineral N and P in leachate of control column (sand) and ochre + straw column, receiving dilute slurry, showing removal of nutrients equivalent to 188 kg N/ha and 92 kg P/ha of slurry nutrients by the ochre + straw layers

DISCUSSION

The results show that the subsoil and ochre sublayers are both effective ways of attenuating pollutants (NH₄-N, TOC, faecal indicator organisms, P) from the leachate from corrals, at least in the short term. The mineralization and oxidation of N from the slurry leads to a rise in the nitrate concentration, which could create problems with Nitrate Vulnerable Zone compliance. However, the results also suggest that nitrate removal through denitrification is occurring; this will be greater than would occur where the faecal and urinary nitrogen was deposited at lower rates (e.g. during overwintering in fields).

The results interface closely with work on full-scale woodchip corrals (Vinten et al., 2006) which shows that freely drained sandy soil is working as an effective mitigator to prevent groundwater pollution by ammonium, organic carbon and faecal indicators, but that high nitrate concentrations below corrals occur. Further work to assess the amount of denitrification under corrals is ongoing. It should be borne in mind that the nitrate leaching per unit area from such an installation can be very high, although the nitrate leaching per animal will be lower than where animals are more spread out in fields over winter.
Results may be influential in determining whether attenuation through infiltration can be considered an acceptable method of pollution prevention under corrals on freely drained sites, as an alternative to lining. The leachate pollutant concentrations are at levels that would be considered indicative of very effective treatment, for example, from constructed wetlands.

ACKNOWLEDGEMENTS

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REFERENCES


RESPONSES TO WATER QUALITY PROBLEMS IN THE LEET CATCHMENT

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SUMMARY

The study focuses on the Leet Water catchment, a left-bank tributary of the River Tweed, Berwickshire, south-east Scotland. The Leet Water and its subcatchment, the Lambden Burn, cover an area of approximately 114 km² within the Lothian and Borders Nitrate Vulnerable Zone (designated in 2002). Pollution impacts were modelled using a modified export coefficient approach by integrating land cover with available chemical and fertiliser data. Results of modelling scenarios of simple land use changes found that reducing fertiliser use by 10% can reduce the number of fields in the ‘very high-risk’ group from 191 to 16, equivalent to reducing the high-risk area from ~3255 ha (29% of the catchment) to ~428 ha (3.3% of the catchment). This method of water quality modelling provides a means of integrating field research on water quality with the results of socio-economic surveys.

INTRODUCTION

The study area, the Leet Water catchment, is a left-bank tributary of the River Tweed having its confluence at Coldstream, Berwickshire (NGR 3844 6395). The Leet Water and its subcatchment, the Lambden Burn, together cover an area of approximately 114 km². The area is largely rural, of low population and mainly agricultural. In contrast to the faster flowing, upland streams in the Tweed Basin, the Leet and Lambden Burn are small and relatively slow flowing. These watercourses have been significantly altered in the past, in particular during land drainage schemes of the 1970s. This has contributed to problems of poor water quality in the catchment.

Previous water quality monitoring by SEPA has found high concentrations of nitrate and phosphate. Levels of N have often been above the EU Nitrate Directive permitted level (50 mg/L NO₃; 11.3 mg/L NO₃-n). Phosphate levels have been found to range from < 0.1 to 1 mg/L. Leet Water and the Lambden Burn have been classed as eutrophic (Institute of Hydrology, 1996).

The export coefficient approach has previously been applied to intensive lowland agricultural systems in the UK (Johnes, 1996; Johnes and Heathwaite, 1997). This model predicts the nutrient loading at any site in the drainage network of a catchment as a function of the export of nutrients from each source in the catchment above that site.

MODELLING LAND USE CHANGE IN THE LEET WATER CATCHMENT

In this study, the export coefficient model was constructed using readily available data including:

- spatial distribution of land use and fertilisers applied to each land use type;
• numbers and distribution of livestock in the catchment;
• input of nutrients to the catchments through nitrogen fixation and atmospheric deposition; and
• export coefficients derived from literature sources to determine the rate of loss of nutrients from each source to the surface drainage network.

This research had produced a land cover map for 2002 classified from multispectral remote sensed data and aerial photography (Widdison, 2005). This was used as the base map to calculate the extent of risk of N loss using the equation:

\[ L = \sum_{i=1}^{n} E_i[A_i(I_i)] + p \]  

(Johnes, 1996)

where: L is loss of nutrients; E is export coefficient for nutrient source I; A is the area occupied by land use type, number of livestock type i; I is the input of nutrients to source; p is the input of nutrients from precipitation.

ArcGIS software was used to calculate total N loss kg/ha/year on a per-field basis. Using the 2002 land cover as a base map provided a spatial dimension of predicted output relevant to the main stakeholder group (i.e. the farming community). Results of the preliminary modelling found the agricultural area within the Leet catchment to be 11,213 hectares. Calculated total nutrient input (organic and inorganic fertilisers) was a little over 2 million kg resulting in a total loss of nitrogen from the catchment of 285,540 kg, an average of 25.5 kg/ha/year. Table 1 summarises the results for land cover associated with the main farming activities. Cereal crops are the most significant land cover in terms of area, input and losses of total nitrogen, accounting for approximately 74% of the total land use and nitrogen input in the catchment but approximately 92% of the nutrient losses.

<table>
<thead>
<tr>
<th>Land cover source</th>
<th>~Area ha</th>
<th>% land cover</th>
<th>Fertiliser inputs (kg)</th>
<th>Total export of nitrogen (kg)</th>
<th>% of the total loss of nutrients</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pasture</td>
<td>1403</td>
<td>12.52</td>
<td>264017</td>
<td>17622</td>
<td>6.17</td>
</tr>
<tr>
<td>Rough grazing</td>
<td>121</td>
<td>1.08</td>
<td></td>
<td>381</td>
<td>0.13</td>
</tr>
<tr>
<td>Cereal crops*</td>
<td>8302</td>
<td>74</td>
<td>1743427</td>
<td>263168</td>
<td>92.2</td>
</tr>
<tr>
<td>Wood &amp; Hedge</td>
<td>530</td>
<td>4.73</td>
<td></td>
<td>1670</td>
<td>0.6</td>
</tr>
<tr>
<td>Fallow &amp; Set-aside</td>
<td>333</td>
<td>2.97</td>
<td></td>
<td>1050</td>
<td>0.4</td>
</tr>
</tbody>
</table>

*Includes three fields of potatoes (9.37 ha) which accounted for nutrient loss of 377kg.

Furthermore, when the impacts of winter sown wheat and oilseed rape are examined, these account for approximately 47% of the total area and ~59% of nutrient losses (Table 2).
### Table 2: Nutrient export from winter wheat and winter oilseed rape (2002)

<table>
<thead>
<tr>
<th>Land cover source</th>
<th>-Area Ha</th>
<th>% land cover</th>
<th>Fertiliser Inputs (kg)</th>
<th>Total export of nutrients (kg)</th>
<th>% of total loss of nutrients</th>
</tr>
</thead>
<tbody>
<tr>
<td>W. wheat</td>
<td>4763</td>
<td>42.5</td>
<td>1000261</td>
<td>135736</td>
<td>47.5</td>
</tr>
<tr>
<td>W. oilseed rape</td>
<td>549</td>
<td>4.9</td>
<td>115296</td>
<td>35091</td>
<td>12.3</td>
</tr>
</tbody>
</table>

These results illustrate the significance of cereal cropping, and in particular the significance of winter sown crops to the problems of nutrient export in the catchment as a whole.

In the monitored sub-catchments, the average annual loss of N ranged from 18.25 to 164 mg/L. The highest N losses (> 146 mg/L p.a.) occur in areas contributing to gauging stations in sub-catchments where arable land use is more than 80% so this can be seen to be one of the contributing factors to such high losses. Potential risk of N loss was attributed to each field plot by establishing which plots lie within 50 m of the watercourses and therefore pose the greatest threat to water quality. Using a combination of factors, the spatial distribution of N loss was presented in ArcGIS as risk maps to be used by stakeholders as part of a suite of land use decision-making tools.

### The Nutrient Export Risk Maps for the Leet Water Catchment

Relative risk was assigned using weighted values for a range of criteria; when these are combined with a weighting for proximity to a water-course, a combined risk score can be calculated. Weighting values ranging from 1–5 were given to the parameters shown in Table 3 below. For land use, nutrient input and export the full range of weighted values were available for each field. However, for distance from watercourse, the weight range was restricted to reflect the significant contribution of field drains to potential nutrient loss.

### Table 3: Weighted values for parameters in risk assessment

<table>
<thead>
<tr>
<th>Land use</th>
<th>Nutrient input kg/ha/year</th>
<th>Daily nutrient export N (mg/L)</th>
<th>Distance from water course (m)</th>
<th>Weighted value for each variable</th>
</tr>
</thead>
<tbody>
<tr>
<td>Woodland</td>
<td>0</td>
<td>0–0.09</td>
<td></td>
<td>1</td>
</tr>
<tr>
<td>Fallow/set-aside</td>
<td>1–99</td>
<td>0.1–0.19</td>
<td>&gt; 50</td>
<td>2</td>
</tr>
<tr>
<td>Grazing</td>
<td>100–199</td>
<td>0.2–0.29</td>
<td></td>
<td>3</td>
</tr>
<tr>
<td>Spring cereals</td>
<td>200–249</td>
<td>0.3–0.39</td>
<td>11–50</td>
<td>4</td>
</tr>
<tr>
<td>Winter cereals</td>
<td>&gt; 250</td>
<td>&gt; 0.4</td>
<td>0–10</td>
<td>5</td>
</tr>
</tbody>
</table>

This enabled a total risk assessment score of up to a maximum of 20 points to be assigned to each field plot. Risk was then classified into five categories based on the following intervals:

- Very low risk: 1 – 4 points
- Low risk: 5 – 8 points
- Medium risk: 9 – 12 points
- High risk: 13 – 16 points
- Very high risk: 17 – 20 points.
Using this method of classification allowed a much more realistic assessment of the level of risk that each field plot contributed to water quality in the catchment. These results, Figure 1 and summarised in Table 4 below, show there were no fields in the very low risk category. This is because the proximity to water course weighting forces a minimum possible score of 5 points.

**Table 4: Summary results of risk assessment of 2002 land use**

<table>
<thead>
<tr>
<th>Category</th>
<th>Number of field plots in category</th>
<th>Approximate area (ha)</th>
<th>Approximate N loss (mg/l)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Very low risk</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Low risk</td>
<td>922</td>
<td>3120</td>
<td>184</td>
</tr>
<tr>
<td>Medium risk</td>
<td>414</td>
<td>2219</td>
<td>101</td>
</tr>
<tr>
<td>High risk</td>
<td>569</td>
<td>3717</td>
<td>363</td>
</tr>
<tr>
<td>Very high risk</td>
<td>191</td>
<td>3256</td>
<td>345</td>
</tr>
</tbody>
</table>

**Modelling Land Use Change Scenarios**

Although it is not possible to predict a precise moment when nutrient loss will exceed the EU limit, the export coefficient model can be applied to predict the impact of changing land use on nutrient losses at the field scale. This modelling will be useful to the farming community as it can provide information to be used in their decision making processes. Currently the catchment includes 74% arable land use which is responsible for 87% of total fertiliser input and 92% of the total predicted nutrient loss. The four scenarios described below involve changing the way in which arable farming is practised.

The predicted impacts of the scenario modelling are shown in Table 5 below. The first land use change scenario involves the installation of fixed width grassland buffers at 5 or 10 m, to all water courses in the catchment. These buffers would remove approximately 150–320 ha of land from production, of which approximately 90–190 ha are currently used for arable production. As part of the management of these buffers, it is assumed that fencing is installed to prevent livestock accessing the stream; application of chemicals including fertilisers ceases; and most importantly, all field drains discharging to streams are blocked to prevent nutrient losses by-passing the buffer. In the short-term, vegetation would return to rough grassland, although further management of the buffer could include planting native species woodland which would increase nitrate removal in these zones. In these two scenarios nitrogen input is limited to that from atmospheric deposition (3.15 kg/ha/year) therefore the total N loss is recalculated using the export coefficient for land in set-aside/woodland. Installing fixed width buffers results in N losses being reduced by 0.16–0.25% for the whole catchment. In terms of benefits for the farming community, fertiliser usage is reduced by 1.17% to 2.47% of the existing use and this would reduce the cost. However, this would have to be balanced by the loss in income from grain sales on these buffers.

The buffer scenario was further investigated by changing the buffer to 50 m. In an intensive arable regime, the land within 50 m of the water course can make a significant contribution to nutrient loss. Under current farming practices, there are a
significant number of arable fields adjacent to the water course without any buffers. Therefore the majority of fields are classified as high or very high risk. With the implementation of a 50-m buffer, risk is reduced to low. When applied to the whole of the catchment there is a net reduction of total N losses of 1.86%. However, implementing such large buffers may not be acceptable to farmers with smaller farming units as this will remove a greater proportion of their land from economic production.

The second scenario investigated reducing fertiliser use (by 10% or 20%) on the existing land use regime. The results found that nutrient losses for the whole of the catchment were 8.8% and 17.5% respectively. The third scenario examined the impact of a radical change in land use. Table 5 shows that if all arable land use was changed to permanent pasture (i.e. only livestock farming is practiced), nutrient losses are predicted to be 55% less than they would be under the current intensive arable regime. However, a livestock-dominated farming system is not popular. This is because cereal production is seen to be more profitable than livestock farming. But more importantly to the farming community, the recent outbreak of foot and mouth disease and previous impacts of BSE have shown how vulnerable livestock can be to contagious or infectious diseases and farmers would be very reluctant to specialise.

The final scenario is the most radical model, requiring all agricultural land (including pasture land) to be taken out of production and converted to woodland. Although this would reduce the current annual nutrient loss from approximately 902 to 111 mg/L, the economic, social and environmental impacts are extreme. Assuming plantation woodland takes over, this takes upward of 30 years to mature before felling takes place and an economic return made, so this would not be economically viable for the current farming community. In terms of social change, the range of employment activities in the local area would alter as the specialised skills of forestry workers and the number of workers required change. In environmental terms, although nitrate losses would reduce, there would be increased acidification of water courses, leading to a decline of aquatic biodiversity. Terrestrial biodiversity would also change as existing wildlife habitats were destroyed.
Table 5: Results of land use scenario modelling (catchment scale)

<table>
<thead>
<tr>
<th>Land use scenario</th>
<th>Land removed from production (ha)</th>
<th>Arable land removed (ha)</th>
<th>Existing land use contributes annual N loss of kg/year</th>
<th>Scenario land use contributes N losses mg/L</th>
<th>% net reduction N loss</th>
</tr>
</thead>
<tbody>
<tr>
<td>5m buffer</td>
<td>152.7</td>
<td>3257.1</td>
<td>10.3</td>
<td>480.6</td>
<td>1.48</td>
</tr>
<tr>
<td>10m buffer</td>
<td>317.3</td>
<td>6896.9</td>
<td>21.3</td>
<td>9985.9</td>
<td>2.28</td>
</tr>
<tr>
<td>50m buffer</td>
<td>1685.9</td>
<td>39158.1</td>
<td>123.6</td>
<td>5310.4</td>
<td>16.24</td>
</tr>
<tr>
<td>10% reduction of fertiliser</td>
<td>0</td>
<td>285540</td>
<td>902.3</td>
<td>260518.6</td>
<td>823.22</td>
</tr>
<tr>
<td>20% reduction of fertiliser</td>
<td>0</td>
<td>285540</td>
<td>902.3</td>
<td>235496.9</td>
<td>744.15</td>
</tr>
<tr>
<td>Convert all arable land to permanent pasture</td>
<td>0</td>
<td>8635.9</td>
<td>285540</td>
<td>902.3</td>
<td>126567.7</td>
</tr>
<tr>
<td>Convert all agricultural land to wood</td>
<td>10160.9</td>
<td>8635.9</td>
<td>285540</td>
<td>902.3</td>
<td>35323.6</td>
</tr>
</tbody>
</table>

Figure 1: Extent of risk (2002 land use)
In terms of the risk assessment for each field, Table 6 shows that even a 10% reduction in fertiliser use can have a significant impact on each field. Under the current farming practice of using the maximum recommended fertiliser rates, 191 fields are classified as very high risk (Figure 1) but this number is reduced to 16 with a 10% reduction in fertiliser use (Figure 2) and to 13 fields with a 20% reduction in fertiliser. This also has a knock-on effect on fields classified as high or medium risk.

Table 6: Impact of fertiliser reduction to the extent of risk (number of field plots)

<table>
<thead>
<tr>
<th>Land use 2002</th>
<th>Very low risk</th>
<th>Low risk</th>
<th>Medium risk</th>
<th>High risk</th>
<th>Very high risk</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>922</td>
<td>414</td>
<td>569</td>
<td>191</td>
<td></td>
</tr>
<tr>
<td>10% reduction in fertiliser use</td>
<td>0</td>
<td>922</td>
<td>642</td>
<td>516</td>
<td>16</td>
</tr>
<tr>
<td>20% reduction in fertiliser use</td>
<td>0</td>
<td>922</td>
<td>646</td>
<td>515</td>
<td>13</td>
</tr>
</tbody>
</table>

However, farmers believe this scenario would affect the grain outputs from arable production and the number of livestock units per hectare and therefore farm income. On the other hand, farm expenditure on chemical fertilisers and grain seed would be less. For example, at 2002 figures, fertiliser costs were £79–£105 ha depending on the chemical mix required for the range of crops grown.
CONCLUSIONS

As a decision support tool, the export coefficient approach has many advantages. These include: simplicity in calculating nutrient loss; relatively few data requirements; use of a spreadsheet (database) system; and coupling with GIS mapping, to provide a visual interpretation. Most importantly, it provides a robust and relatively inexpensive means of evaluating the impact of land use and management on water quality. The export coefficient approach enables a risk assessment to be applied to each field plot in the catchment, and modelling land use change in a range of scenarios has shown that significant changes can be made to the risk associated with each field.

ACKNOWLEDGEMENTS

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REFERENCES


NITROGEN LOSSES AFTER CATTLE SLURRY APPLICATIONS TO A DRAINED CLAY SOIL

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SUMMARY

Nitrogen losses in drainage water were measured after autumn, winter and spring cattle slurry applications to a drained clay soil under arable and grassland farming over winter 2003/04. Mean nitrate losses were greatest (P<0.05) from the arable plots at 37 kg/ha N compared with 4 kg/ha N from the arable reversion grassland plots. On arable land, nitrate leaching losses after the autumn cattle slurry application were equivalent to 11% of total N applied compared with 5% after the winter application. The spring application led to ammonium-N concentrations of up to 4.5 mg/L NH₄-N (c. fivefold greater than the EC Freshwater Fish Directive limit of 0.78 mg/L NH₄-N) in drainage waters. These data indicate that changing slurry application timings from autumn to spring will reduce nitrate leaching losses from drained arable clay soils. However, if rainfall occurs soon after applying slurry to wet soils in spring then this can result in elevated drainflow ammonium concentrations. This is particularly likely where there is good connectivity between the soil surface and drainage system, via cracks/mole channels.

INTRODUCTION

An estimated 47 million tonnes of livestock slurry supplying c. 210,000 tonnes of nitrogen (N) are applied to agricultural land in the UK each year (Williams et al., 2000). These applications need to be carefully managed to minimise diffuse pollution as nitrate leaching losses after autumn/winter manure applications in the UK are estimated at 58,000 tonnes N/annum (Chambers and Smith, 1995). On sandy textured soils, over-winter drainage (and nutrient loss) occurs by piston displacement in the unsaturated phase (commonly called ‘matrix’ flow). Consequently autumn slurry applications on such soils represent a significant risk of nitrate contamination of ground waters because crop N uptake between application and the end of drainage in spring is generally low (Beckwith et al., 1998).

However, the impermeable nature of clay textured soils has led to the perception that they are nutrient retentive and that autumn slurry applications are likely to pose a low risk of nitrogen contamination of surface waters. However, on drained land (which accounts for an estimated 6.4 million ha of England and Wales; Withers et al., 2000) the rapid transfer of water from the soil surface to drains, via soil macropores (commonly called ‘by-pass’ flow) has the potential to lead to high nitrogen concentrations and losses in drainage waters after slurry application. It is also likely that the pattern of nitrogen losses from arable land will be different to that from grassland because factors that affect water movement (e.g. ground cover, cultivations, surface compaction, pore size distribution and continuity) differ between the two land use
types. This paper reports results from the first year of a 3-year study to quantify the effects of different timings of cattle slurry application on nitrogen losses in water from a drained clay soil under arable and grassland management.

MATERIALS AND METHODS

The experiment was carried out on a heavy clay textured (60% clay) soil of the Denchworth Association, at the Brimstone Farm experimental facility, near Faringdon in Oxfordshire. The site consists of 18 hydrologically isolated plots (each 40 m x 48 m), which had been in arable production for over 20 years until autumn 2001 when grassland was established on nine plots. The soils are drained with pipe drains at 1 m depth and 48 m spacing, and have gravel backfill to within 30 m of the surface, with secondary mole drains at 2 m spacing and 50 cm depth at right angles to the pipe drains. Cattle slurry (c. 40 m³/ha, 120 kg/ha total N) was applied to the arable and grassland plots in September and December 2003, and March 2004, using an 11-m³ Joskin tanker fitted with a 12-m trailing hose boom. There were three replicates of each application timing. Inorganic nitrogen fertiliser was applied all plots in early and late April 2004, using standard recommended rates, to ensure that subsequent crop growth was representative of commercial practice.

Drainage and surface run-off volumes were measured continuously using V-notch weirs. Drainage water samples were collected on a flow proportional basis using automatic water samplers and analysed for nitrate-N, ammonium-N and soluble organic N (SON).

RESULTS AND DISCUSSION

Drainage Volumes

Low rainfall volumes during autumn 2003 (70% of long-term average between 1st September and 30th November 2003) meant that drainage did not begin until the end of November 2003 on the arable plots, and mid-December on the grassland plots. A wet April, when rainfall was 60% greater than the long-term average, meant that drainage continued until mid-May 2004. Mean drainflow volumes during the whole drainage period (up to mid-May 2004) were 77 mm from the grassland compared with 130 mm from the arable plots (long-term arable mean drainage at Brimstone = 204 mm). The lower drainage volumes and later return to field capacity on the grassland plots reflected the greater soil moisture deficit that had developed as the grass grew during the summer and early autumn. Surface run-off volumes were less than 7 mm on all plots.

N Concentrations in Drainage Water

On the arable plots, nitrate concentrations in drainage water up to late March 2004 (until inorganic fertiliser N was applied) were greatest (P<0.05) after the autumn application and peaked at 130 mg/L NO₃-N in the first 10 mm of drainage. The winter slurry application had no effect (P>0.05) on nitrate concentrations in drainage waters, probably as a result of the cold and wet soil conditions delaying the nitrification of slurry ammonium-N to nitrate-N. Nitrate concentrations declined on all the treatments to between 30–50 mg/L NO₃-N after c. 30 mm of drainage (Figure 1a), but remained above the EC limit of 11.3 mg/L NO₃-N throughout the drainage period.
In contrast, on the arable reversion grassland plots nitrate concentrations in drainage waters were low and generally below the EC limit throughout the drainage season. On these plots, there was no effect (P>0.05) of slurry application timing (Figure 1b) on drainage water nitrate concentrations.

Figure 1: Mean nitrate-N concentrations in drainage water at Brimstone Farm (2003/04)

The autumn and winter slurry applications had no effect on ammonium-N concentrations in drainage waters from either the arable or grass plots. In both instances, these remained below the EC Freshwater Fish Directive limit of 0.78 mg/L NH₄-N throughout the drainage season (Figure 2a and b). However, after the spring slurry application timing, when c. 20 mm of rain fell 9-12 days after the slurry was applied, ammonium-N concentrations peaked at 4.5 and 3.9 mg/L NH₄-N on the grassland and arable plots, respectively. These measurements suggest that the rainfall soon after application had caused slurry nitrogen to move rapidly through the soil profile, via macropores (‘by-pass’ flow), to the drains.

Figure 2: Mean ammonium-N concentrations in drainage water at Brimstone Farm (2003/04)
N Losses in Drainage Water

Mean nitrate losses up to the end of March 2004 (until fertiliser N was applied) were greatest (P<0.05) from the arable plots at 37 kg/ha N compared with 4 kg/ha N from the arable reversion grassland plots (Figure 3). On the arable plots, nitrate losses after the autumn and winter slurry applications (up to the time slurry was applied in the spring) were equivalent to 11% and 5% of the total slurry N applied, respectively.

On the arable reversion grassland plots, slurry application timing had no effect (P>0.05) on nitrate leaching losses up to the end of March 2004. Losses from the autumn and winter applications were equivalent to 2% and 4% of the total N applied, respectively. The low nitrate leaching losses from the grassland system were probably a reflection of the recently established grass sward accumulating N within organic reserves and greater crop uptake of slurry N (c. 20 kg/ha N compared with less than 5 kg/ha N on the arable plots) in the period between application and the start of drainage.

![Figure 3: Nitrogen leaching losses Brimstone Farm (2003/04)](image)

Ammonium-N losses were less than 0.2 kg/ha N from all the treatments. Soluble organic N losses ranged from 2.5 to 3.6 kg/ha N on the arable plots and 0.6 to 0.8 kg/ha N on the grassland plots.

CONCLUSIONS

The data from the first year of this study indicate that drained arable clay soils may not be as retentive of slurry N as had previously been thought, although the arable reversion grassland plots were very nitrate retentive. Moving the timing of the slurry application to arable land from autumn to spring reduced nitrate leaching losses. However, applying slurry to ‘wet’ drained soils, where rainfall follows soon afterwards, can result in elevated drainflow ammonium-N concentrations (so called ‘pollution swapping’). This is particularly likely where there is good connectivity between the soil surface and drainage system via cracks/mole channels.
ACKNOWLEDGEMENT

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REFERENCES


SOIL AND CROP MANAGEMENT EFFECTS ON SEDIMENT AND PHOSPHORUS CONCENTRATIONS IN RUN-OFF FROM AGRICULTURAL LAND

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SUMMARY

To encourage the adoption of best management practices in a priority catchment (Hampshire Avon) in south-west England suffering diffuse pollution, surface run-off from field demonstration plots on different soil types and land management treatments was monitored over two winter seasons. Late drilling increased run-off up to fivefold, and mobilisation of sediment and phosphorus (P) by up to an order of magnitude, compared with early drilling using traditional cultivation techniques. Tramlines running up-and-down slope generated significantly greater run-off, and often increased run-off sediment and P concentrations, than where tramlines were absent, or running across-slope. Adopting reduced cultivations (minimum tillage) either decreased run-off, the mobilisation of sediment and P in run-off, or lessened the impact of tramlines. The results support the uptake of early drilling, timeliness of cultivations to avoid soil compaction, better tramline management and non-inversion cultivation techniques to help reduce agriculture’s impact on water quality.

INTRODUCTION

Sediment and phosphorus (P) concentrations in land run-off, and loads of these diffuse pollutants entering surface waters, are a major environmental concern and need to be controlled to acceptable levels in order to preserve or improve good water quality. Much of the agriculturally-derived P entering waterbodies via surface and sub-surface run-off is associated with fine topsoil particles that have been enriched with P from previous P fertilisation (Edwards and Withers, 1998). Combinations of erosion vulnerable soils, over-cultivation presence of tramlines and lack of crop cover during storm events significantly increase the risk of sediment and P transport, and off-site impacts (Chambers et al., 2000). Control over run-off initiation and mobilisation of sediment is therefore an important part of the integrated approach to catchment management required for diffuse pollution control.

In 1999, a catchment management initiative called Landcare was started in the Hampshire Avon river basin upstream of Salisbury, England to help reduce the agriculturally-derived loads of pollutants, particularly sediment and P entering the major tributaries (Huggins, 1999). As part of this initiative, farmer demonstration plots organised by the Environment Agency (EA) for farmers and their advisers were established at field sites on the three major lithologies that dominate the catchment: Upper Chalk (Wilton), Upper Greensand (Pewsey) and Kimmeridge Clay (East Knoyle). To supplement this demonstration activity, the plots were monitored over two winter periods (2002/03 and 2003/04) to provide supporting data that would encourage the adoption of more sensitive land management practices.
MATERIALS AND METHODS

The field sites contrasted strongly in their soil characteristics, hydrology and P fertility. At Pewsey, the moderately sloping (5°), fine sandy soil (67% sand, Ardington Association) had been in arable cropping for many years with low levels of organic matter (c. 2.3% OM) and good levels of soil P fertility (c. 30 mg/kg Olsen-extractable P). The arable soil at Wilton was a shallow, highly calcareous and free draining silty clay loam (Upton Association, 4.1% OM), but with steep slopes (8°) and very low levels of Olsen-extractable P (c.10 mg/kg). The heavy clay soil (28% clay, Whickham Association) at East Knoyle was underdrained and contained a higher level of OM (5.1–8.6% OM) and soil P fertility (46–67 mg/kg Olsen-extractable P) than the other two sites, reflecting a history of grass-based dairy farming including forage maize and frequent manure application.

The demonstration plots were cultivated and drilled either early (E) or late (L), and adopted either traditionally cultivations (TC) or reduced cultivations (RC), providing four treatment combinations: E-TC, E-RC, L-TC and L-RC. The plots were not replicated but were large in size (20 x 20 m) as required for demonstration purposes. Site management was under the control of the farmer using local cultivation practices. At Wilton in 2002/03, the farmer did not establish the late drilled treatments due to very wet weather, and at East Knoyle in 2003/04 the L-TC treatment was replaced with an E-RC headland plot (hereafter referred to as ‘headland’) due to field size restrictions. Early drilling was usually at the end of September (range 20 Sept. to 2 Oct.), but late drilling varied from late October to early January depending on the weather.

Traditional cultivation included ploughing (with or without a press), and either tine harrowing or power harrowing before drilling. For the reduced cultivation treatment, the farmers adopted either heavy discs (Pewsey and Wilton), or a heavy harrow (East Knoyle) instead of the plough (Table 1). The field at East Knoyle has also been regularly subsoiled. At Wilton and Pewsey, the plots were drilled up-and-down slope and tramlines established after drilling by one or more tractor passes. At East Knoyle, the plots were drilled across-slope, except for the headland plot which was drilled up-and-down slope. Residues from the previous crop were returned to the soil and all sites grew winter cereals in both years.

On each demonstration plot, three run-off traps each measuring 15 m long by 2 m wide were installed to monitor sediment and P in surface run-off generated by successive storm events. At East Knoyle in 2002/03, prolonged wet weather prevented the installation of any run-off traps. The traps were hydrologically isolated using 30 cm deep stainless-steel dividers driven into the ground, and each trap contained a tramline. A collecting tray was cemented in place at the bottom of each trap, and angled to direct the run-off to a 110-mm-diameter pipe, which fed directly into a 500 litre covered fibre glass tank. After each major rainfall event, the run-off that had collected in the tanks was measured and recorded, and a 250-mL sub-sample taken for determination of suspended sediment (SS), total P (TP) and dissolved P (DP, < 0.45 mm).

The traps were installed as soon as was practicable after drilling and removed in April to allow fertilizer and spraying operations on the field. For the early-drilled
treatments, typically 10 storm events were monitored, while c. five events were monitored for the late-drilled treatments. To investigate the effect of tramlines, three additional replicate traps without a tramline were installed on selected treatments at Pewsey, including an off-plot area where the tramlines were running across the slope instead of up and down slope. Rainfall amount and intensity were measured at each site with an automatic rain gauge, supplemented as necessary with data from the nearest meteorological station. As there was no replication of the four main treatment combinations, data from the individual run-off traps were taken as independent treatment replicates and treatment effects analysed using one-way ANOVA using Genstat.

RESULTS

Cultivation Effects

Prolonged heavy rain fell approximately 2 weeks after early drilling in 2002/03 causing surface compaction (capping) of the soil surface on the E-TC treatment at Pewsey, which generated twice as much run-off compared to the E-RC treatment where surface straw residues gave some protection from raindrop impact (Figure 1). The greater run-off mobilised significantly more sediment and P, with twofold greater flow-weighted SS and TP concentrations on the ploughed soil (Figure 1). Cumulative exports of SS and TP over the whole monitoring period were consequently increased fivefold (up to 0.5 t/ha) and fourfold (up to and 0.4 kg/ha), respectively on the E-TC treatment compared with the E-RC treatment. In contrast on the chalk soil, where run-off volumes were much lower, there was no effect of cultivation method on run-off, although sediment and P mobilisation was significantly lower on the reduced cultivated treatments due to the presence of crop cover (Figure 1). Hence, the exports of SS and TP were still fivefold and twofold, respectively greater on the E-TC treatment than on the E-RC treatment. The late-drilled treatments were installed only at Pewsey in 2002/03 and little rain fell afterwards, resulting in no significant treatment effects.

![Figure 1: Effect of cultivation method on run-off and flow-weighted TP concentrations in run-off at Pewsey and Wilton in 2002/03](image-url)
In 2003/04, heavy rain fell after the late-drilled treatments were established, again causing capping of the soil surface on the sandy soil at Pewsey with rill erosion down the tramlines. This rain also caused gushing of run-off down the tramlines on some traps on the L-RC treatment at Wilton, and increased run-off from the headland area at East Knoyle (Table 1). Consequently, run-off volumes were up to c. fivefold greater on the late-drilled treatments than on the early-drilled treatments. As in the first year, the increased run-off mobilised significantly more sediment and P on the ploughed soil compared with the reduced cultivated soil at Pewsey (Table 1). At Wilton, flow-weighted SS and TP concentrations were increased on the L-RC treatment but not significantly so. However at East Knoyle, flow-weighted SS and P concentrations were significantly lower in the run-off on the headland area, perhaps reflecting a more consolidated surface that was more resistant to sediment entrainment. Sufficient crop cover had developed on the early drilled treatments at all sites in 2003/04 to prevent accelerated sediment and P mobilisation, and there were no significant soil cultivation treatment effects. Mean cumulative sediment and TP export ranged up to 0.8 t/ha and 0.6 kg/ha in 2003/04, with the majority (60-97%) of the TP export in particulate (> 0.45 mm) form at all three sites (Table 1).

Table 1: Treatment effects on cumulative run-off volume, and loads of SS, TP and DP at each site in 2003/04. Flow-weighted concentrations (g/L for SS and mg/L for TP and DP) are given in brackets

<table>
<thead>
<tr>
<th>Site</th>
<th>Run-off Treatment (mm)</th>
<th>SS (kg/ha)</th>
<th>TP (g/ha)</th>
<th>DP (g/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pewsey</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>E-TC</td>
<td>3.2</td>
<td>77 (2.3)</td>
<td>48 (1.5)</td>
<td>3 (0.08)</td>
</tr>
<tr>
<td>E-RC</td>
<td>3.2</td>
<td>57 (1.8)</td>
<td>38 (1.1)</td>
<td>4 (0.12)</td>
</tr>
<tr>
<td>L-TC</td>
<td>15.8</td>
<td>650 (4.3)</td>
<td>548 (3.6)</td>
<td>31 (0.18)</td>
</tr>
<tr>
<td>L-RC</td>
<td>14.1</td>
<td>184 (1.3)</td>
<td>183 (1.3)</td>
<td>23 (0.16)</td>
</tr>
<tr>
<td>Significance</td>
<td>***  ***  ***</td>
<td>***</td>
<td>***</td>
<td>**</td>
</tr>
<tr>
<td>l.s.d.</td>
<td>129</td>
<td>107</td>
<td>17</td>
<td></td>
</tr>
<tr>
<td>East Knoyle</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>E-TC</td>
<td>1.5</td>
<td>49 (3.2)</td>
<td>47 (3.0)</td>
<td>5 (0.32)</td>
</tr>
<tr>
<td>E-RC</td>
<td>2.5</td>
<td>75 (3.3)</td>
<td>72 (3.2)</td>
<td>9 (0.39)</td>
</tr>
<tr>
<td>Headland</td>
<td>9.7</td>
<td>70 (0.7)</td>
<td>148 (1.4)</td>
<td>19 (0.22)</td>
</tr>
<tr>
<td>L-RC</td>
<td>1.8</td>
<td>32 (1.7)</td>
<td>30 (1.6)</td>
<td>5 (0.28)</td>
</tr>
<tr>
<td>Significance</td>
<td>NS  NS  *</td>
<td>NS</td>
<td></td>
<td>*</td>
</tr>
<tr>
<td>l.s.d.</td>
<td>9</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wilton</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>E-TC</td>
<td>2.0</td>
<td>63 (2.8)</td>
<td>51 (2.3)</td>
<td>3 (0.12)</td>
</tr>
<tr>
<td>E-RC</td>
<td>2.4</td>
<td>43 (1.8)</td>
<td>39 (1.6)</td>
<td>2 (0.07)</td>
</tr>
<tr>
<td>L-TC</td>
<td>5.0</td>
<td>150 (3.0)</td>
<td>97 (2.0)</td>
<td>12 (0.06)</td>
</tr>
<tr>
<td>L-RC</td>
<td>13.2</td>
<td>787 (4.6)</td>
<td>583 (3.4)</td>
<td>3 (0.10)</td>
</tr>
<tr>
<td>Significance</td>
<td>NS  NS  NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
</tbody>
</table>

*, ** and *** denotes significance at the 5%, 1% and 0.1% level, respectively. NS, not significant; l.s.d., least significant difference.
Tramline Effects

In both years, observations of flow rates from the traps during storm events indicated that the tramlines might be having a significant impact on run-off, sediment and P export. Run-off was being initiated sooner in the tramlines than in the surrounding soil, and measurements at Pewsey in both 2002/03 and 2003/04 showed significantly (P < 0.001) more run-off (on average +46%, or 1–2 mm in total over the monitoring period) from tramlined than from non-tramlined areas (Figure 2). The greater run-off down the tramlines also increased flow-weighted concentrations of SS and TP two- to threefold (up to 4 g SS/L and 4 mg TP/L) on the ploughed soils, but there was no significant effect of tramlines on sediment and P concentrations in run-off from the reduced-cultivated soils. Greatest entrainment of sediment and P was obtained where the tramline was more indented by the tractor tyres, and when the tyre lugs were pointing in a downward Ù rather than in an upward Ù direction. For example in 2002/03, tramlines were 10 mm deeper and more incised on the E-TC treatment than on the E-RC treatment.

![Figure 2: Rainfall-run-off relationships for traps on the early-drilled treatments with and without tramlines at Pewsey](image)

DISCUSSION

Treatment effects on run-off and subsequent entrainment of soil particles and associated P were site specific depending on soil and site characteristics, and were variable between years depending on weather patterns. At Pewsey, the inherent vulnerability of the fine sandy soil to capping when there was little crop cover was the main factor causing increased run-off and erosion. There was little the farmer could have done to prevent this in 2002/03 using traditional ploughing, whilst in 2003/04 it did not rain appreciably until after the late-drilled treatments were established, and under these circumstances, the beneficial effects of early drilling were considerable. These data provide the first quantitative evidence of the effects of capping and late drilling on run-off, and the resulting impacts on the mobilisation of sediment and P.
The soils at Wilton and East Knoyle were not as erosion vulnerable as at Pewsey probably due to the more stable soil structure resulting from their higher clay, organic matter and/or calcium carbonate contents. The key factors influencing run-off rates on the chalkland site were timeliness of cultivation and the presence of tramlines running up-and-down slope, which concentrated the flow. The comparatively low run-off volumes and export of SS and P at East Knoyle reflect the beneficial effects of subsoiling, and drilling across-slope, even though the site was relatively flat. The consistent beneficial effects of non-inversion cultivation methods in reducing run-off, and/or SS and P mobilisation under heavy rain at Pewsey and Wilton support earlier work (Carter, 1998). These benefits reflect both better crop cover and improved resilience to trafficking after drilling. The lack of any effect of reduced cultivation on the heavier clay soil at East Knoyle can be related to the lack of crop residue cover afforded by maize stubble.

In our study, run-off volumes were increased by up to 65% and SS and TP loads were increased up to five- and fourfold, respectively, where tramlines were present. Earlier initiation of run-off, the lack of any crop cover to protect the soil and the channelling effect created by the depth and pattern of indentation left by the lugs of the tractor tyre were key contributory factors. Variation in the impact of tramlines will depend on the extent to which these key factors are represented. These data strongly suggest that the timing of tramline establishment in relation to antecedent soil moisture conditions can have an important influence on the risk of increased run-off and erosion from these soils.

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REFERENCES


Notes