

BRITISH HYDROLOGICAL SOCIETY

Managing our aquatic environment in the 21st Century:

contemporary issues of water quality

BHS National Meeting
Institution of Civil Engineers, London
12th November 2003

Edited by Colin Neal and Ian Littlewood

BHS Occasional Paper No. 14

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British Hydrological Society 2004
ISBN 1 903741 09 2

BHS Occasional Paper No. 14
Produced by the Centre for Ecology and Hydrology
on behalf of the British Hydrological Society

April 2004

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Preface

There is an increasing need for water quality and quantity aspects of the hydrological cycle to be considered together for water resources planning and environmental management, particularly in the context of changes in legislation such as the EU Water Framework Directive.

On the 12th November 2003, a one day meeting hosted by the British Hydrological Society at the Institution of Civil Engineers, London, considered wide ranging issues tied to this subject. The meeting comprised a series of eleven presentations of work by research institutes, consultants, universities and environment agencies. There were three sessions to the meeting, Chaired consecutively by Colin Neal, Paul Withers and Bruce Webb.

The papers covered a diverse range of subjects. They ranged between the current status of water quality law in the UK from a Water Framework Directive perspective (Howsam, 2003), to the impacts of EU legislation on nutrient river loadings (Daldorph & Wheater, 2003), national risk assessment strategies (Heathwaite & Dils, 2003) and the management of canals (Swanson *et al*., 2003). The water quality issues of concern ranged from upland, wetland and lowland UK ecosystems (Reynolds *et al*., 2003; O'Brien *et al*., 2003; Stratford *et al*., 2003; Naden *et al*., 2003) to highly polluted mine drainage in Romania (Aston, 2003). Methodologies of data handling (Natschke *et al*., 2003) and modelling (Cox *et al*., 2003) were also well covered.

We thank the authors for their valuable presentations and for the lively participation of the Chairs and members of the audience.

The papers in this volume are the result of allowing the authors time to reflect and, if they wished, to make changes to their draft papers distributed to delegates at the meeting on the 12th November 2003.

Colin Neal & Ian Littlewood
Centre for Ecology and Hydrology
Wallingford

April 2004

The current status and impact of water quality law in Britain with particular reference to the Water Framework Directive

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ABSTRACT

Water quality law in Britain can be traced back to the 14th Century. One may think that after over 600 years of regulation we would not have any water quality problems – but we do. We need innovative new legislation to deal with the conditions we face in the 21st Century.

The Water Framework Directive (WFD) is a big step in the right direction. It pulls together a number of *ad hoc* pieces of legislation, it applies to all waters, and for the first time will tackle diffuse source pollution. While having a strong focus on ecological water quality – it will address the physical/hydromorphological and as well as the chemical and biological condition of water bodies. Requiring an integrated River Basin based approach not only to water, but also land, management, it could have far reaching socio-political and socio-economic implications.

The devil lies both in the science (data and understanding leading to establishment of reference conditions for water bodies) and the context (the legal and institutional framework) of its requirements. The Directive which came into force in December 2000, has to be transposed into UK law by the end of next month (i.e. December 2003). The approach that the Government have proposed, and implications for implementation and impact will be discussed.

BACKGROUND

Howarth & McGillivray (2001) identify the first piece of water quality protection / pollution control related legislation as the 1388 Act for Punishing Nuisances in towns, which talked in terms of:

"*... so much dung and other filth of the garbage and entrails as well of beasts killed, as of other corruptions, be cast and put in ditches, rivers and other waters, ... that ... is greatly corrupt and infect, and many maladies and other intolerable diseases do daily happen, as*

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well as to the inhabitants and those that are conversant in the said cities, boroughs, towns, and suburbs, as to others repairing and travelling thither, ……."

"….that all they which have cast and laid such annoyances, dung, garbage,…….in ditches, rivers, waters and other places aforesaid, shall cause them utterly to be removed, avoided and carried away……., everyone upon pain to lose and to forfeit to our Lord the King, twenty pounds…"

One can imagine the picture: it was accepted practice to use rivers as waste disposal systems and up to a point the rivers could cope by processes of dilution and biodegradation; but then the point had to occur when settlements had so increased in size that the amount of waste produced exceeded the rivers natural capacity to deal with it. The driver behind this legislation was probably however the dislike by the wealthy and articulate of the nuisances associated with dirty, smelly rivers rather than any concern for the water environment itself. This law despite significant sanctions, achieved very little. Nothing really happened, in fact it got a lot worse, until nearly 550 years later, with the introduction of what might be described as more genuinely socially concerned public health legislation. Indeed concern over public health has continued to be the prime driver for pollution control, as can be seen from the evolution of water pollution and water quality legislation (Howarth & McGillivray, 2001). Only in the 21st century have circumstances dictated that legislation (i.e. the WFD) be introduced to deal with ecological water quality and protection of the water environment for its own sake.

Another important feature of the WFD is its requirement for a more integrated approach to water management. But we have known about the 'water cycle' for hundreds of years. It should be obvious that when dealing with water we need to manage it as a continuum. Yet it is only relatively recently that we have realised the need to do so. In the UK the 1948 Rivers Board Act introduce the idea that water was better managed by River Boards working within hydrologically defined boundaries rather than those under which the previous custodians, Local Government, operated. The 1963 Water Resources Act introduced River Authorities and then these were replaced by ten Water Authorities by the 1973 Water Resources Act. This consolidated a process of catchment (or river basin) based water management which could deal with the whole water cycle within the major hydrologically determined units. Several decades later the European Community as a whole realises that River Basin management is the most appropriate approach. The Directive 2000/60/EC, establishing a framework for Community action in the field of water policy (commonly known as the Water Framework Directive - WFD) formally came into being on 22 December 2000 (Official Journal of the European Communities, L327/1-72, 22.12.2000).

As with the WFD these earlier approaches were only a step in the right direction. Whilst providing an institutional and legal framework for integrated catchment management a fully integrated approach to water management did not materialise. Surface water and groundwater were dealt with separately, as were clean water and dirty water operations, with perhaps the most significant weaknesses being the non-integration of water quantity and water quality.

The legal framework for water quality currently includes a variety of different, not always well coordinated, legal options. Primarily these are:

- the imposition of controls on discharges to water (e.g. s.85 Water Resources Act 1991 makes it an offence to do so without a consent, which limits what, how much, when and where, discharges are permitted);
- the setting of statutory water quality objectives for different waters; and
- control of specific activities/substances with the potential to adversely affect water quality (e.g. control on nitrate application in sensitive areas).

One would have thought that regulations putting limits on inputs, would have been coordinated with those setting receptor quality standards. After all they should both have the same objective - but this has not always been the case.

Of course it is easy in hindsight to criticise what went before. In reality in many ways what happened then was appropriate at that time. Our current dissatisfaction stems from the increasing pressures arising from human activities on water which now make circumstances worse. The availability and degradation of water resources is becoming critical in some areas, both in terms of water quantity (e.g. regional/seasonal variations in rainfall; increased level of demands by an increasing population – one bath a week to two showers a day) and water quality (increased levels of polluting waste and greater demands on water based recreation by an increasing population).

WATER FRAMEWORK DIRECTIVE

A précis of key legal requirements under the Directive is provided in Box 1, below.

However before outlining the key implications of its implementation, it is worth recalling that this integrative Directive started life as a yet another single issue water Directive (i.e. on ecological water quality), following those on surface water for drinking quality, bathing water quality, shellfish water quality, freshwater fish water quality, groundwater quality, urban wastewater (treatment of), etc. Ecological water quality remains as a core feature of the WFD. *"Red hot revolutionary, the most significant piece of legislation affecting ecology for decades"* Moss (2001).

In due course the WFD will replace seven of the current water Directives (e.g. Surface Water, Groundwater, Freshwater Fish, Shellfish). However the WFD goes only part of the way towards a fully integrated legal framework for water. A number of significant water quality related Directives will remain outside the framework (e.g. Bathing Waters, Drinking Waters, and Urban Wastewaters)

It should be noted however that a greater degree of integration will be achieved under the WFD requirement (Annex V) that the programme of measures, necessary to achieve and maintain 'good water status', must include as a minimum, all measures currently required under all current relevant Directives (which along with those mentioned above will also include: Nitrates, Habitats, Environmental Impact Assessment, Integrated Prevention & Pollution Control, Control of Major Accidents & Hazards).

Box 1 Summary of main articles of the Water Framework Directive
(Adapted from The Water Framework Directive 2003a)

ARTICLE 1 – Purpose = **to establish a framework:**

- **to prevent deterioration of, and where necessary enhance, the status of aquatic and related ecosystems;**
  NB = the requirement to "prevent" is a strong / strict obligation**.**
- **to promote sustainable water use;**
  NB the requirement to only "promote" is a weak obligation
- **to aim to progressively reduce and for priority substances eliminate, pollution from hazardous substances;**
  NB the requirement to "aim to" is a moderate obligation
- **to ensure reduction/prevention of groundwater pollution;**
  NB the requirement to "ensure" is a strong obligation
- **to contribute to the mitigation of floods & droughts;**
  NB the requirement to only "contribute to" is a weak obligation

ARTICLE 3 – Co-ordination of administrative arrangements within river basin districts

Requires the establishment of River Basin Districts and for each to appoint a competent authority **with responsibility to achieve environmental objectives via a co-ordinated programme of measures. The latter will include control of land-based as well as water based activities.**

ARTICLE 4 – Environmental objectives

For surface waters there is a requirement to take any necessary measures to prevent deterioration and to achieve good ecological and chemical status; and for groundwaters, to prevent deterioration and to achieve good volumetric and chemical status, by 2015. These requirements apply to all waters except those identified and designated as artificial or heavily modified bodies, in which certain temporary (until 2027) derogations can apply. There is an overriding requirement to reduce / prevent pollution especially by priority hazardous substances.
The Directive requires that all existing water Directives are fully implemented and enforced

ARTICLE 5 – Characteristics of the river basin district, environmental impact of human activity and economics of water use.

Requires an analysis to be undertaken of these three features for each river basin district

ARTICLE 6 – Register of protected areas

Requires the establishment and maintenance of a register of all areas within each river basin district, designated by legislation as requiring special protection

ARTICLE 8 – Monitoring of surface water / groundwater status and protected areas

Requires the establishment of programmes to monitor, using standard methods, the status of water bodies

ARTICLE 11 – Programme of measures

Requires the establishment and regular updating of a programme of measures to achieve the obligatory environmental objectives; Includes basic measures (= those within existing legislation + new controls on abstraction and pollution; and any necessary supplementary measures);
(NB introduces for the first time, legal requirement to control diffuse source pollution)

ARTICLE 13 – River basin management plans

Requires the production of plans, as prescribed, for each River Basin District – the first by December 2009, reviewed/updated versions by December 2015 and thereafter every 6 years

The importance of the WFD is such that the House of Commons EFRA Select Committee deemed it necessary to undertake its first water related inquiry. The inquiry addressed the process and implications / impact of WFD implementation. The WFD is recognised as imposing very ambitious objectives in a very ambitious timetable. The Select Committee concluded:- "*...our most urgent plea is that the Government adopt a more positive and more active approach to the Directive.*" (The Water Framework Directive, 2003b)

The new and challenging water quality obligations under the WFD are that it:

- introduces ecological water quality standards for the first time;
- applies to <u>all</u> waters and not just those previously separately designated for specific purposes;
- aims to deal with surface water and groundwater in an integrated manner;
- provides for the first time a legal framework to tackle diffuse pollution; and
- requires integrated catchment management and planning in which there must be increased stakeholder/public participation.

Key steps in implementation and their implications are represented in Figure 1 below

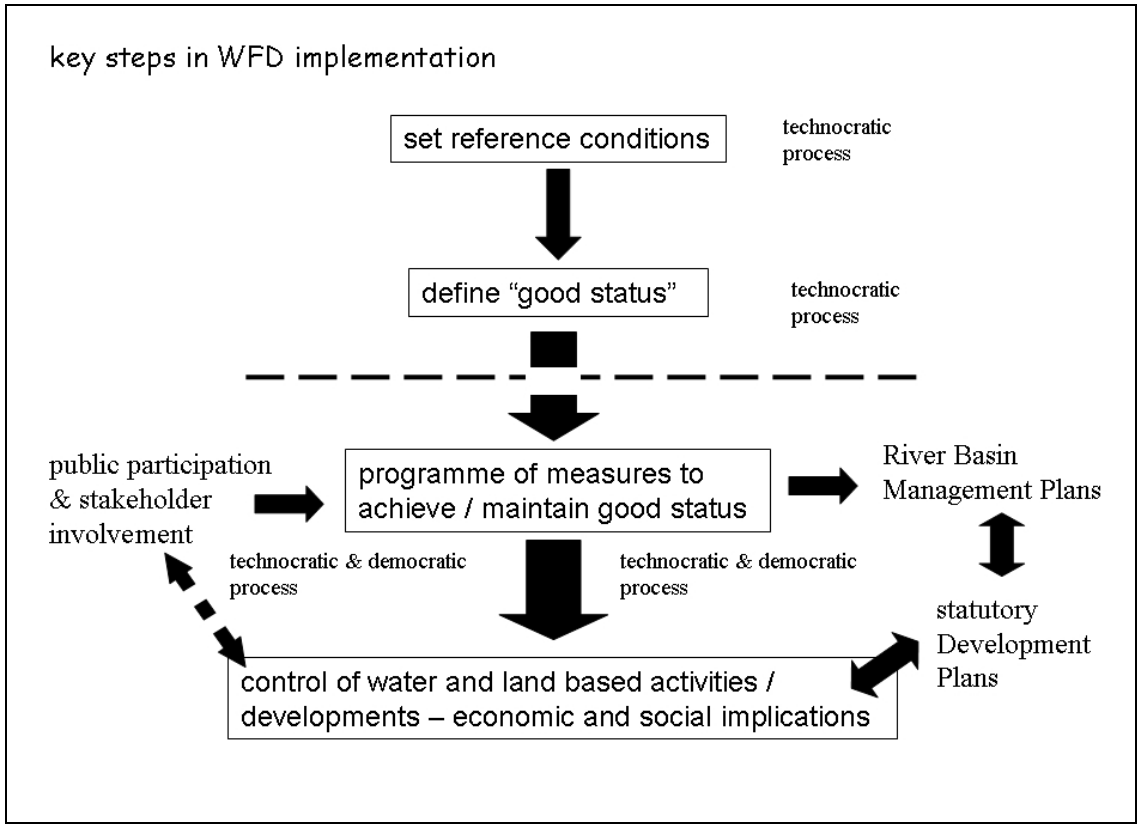


Figure 1 Key steps in WFD implementation

WATER QUALITY ISSUES

While significant improvement in river water quality has been achieved under previous regimes (which used limited physical, chemical and biological parameters – primarily indicating oxygen and organic pollution status), it would be misleading to assume that the situation in the UK is OK. It is expected that many water bodies considered of good quality under the current QGA / RE classification will not be consider so under the new WFD regime – not least because of the new parameters which will now be included; i.e. monitored and assessed, and for which levels of good status will be set (e.g. ecological - aquatic flora, benthic invertebrate and fish fauna; hydromorphological – quantity and dynamics of water flow, interconnection with groundwater, river continuity/depth/width, and river bed / riparian zone characteristics). The latter has been widely impacted by over-abstraction, drainage and flood defence operations. One aspect related to both point and diffuse source pollution is that of nutrient (N & P) loading – with evidence of significant increases over the last sixty years attributed to population growth and changing land-use (mainly agriculture related) practices.

Perhaps the greatest initial challenge will be to identify reference conditions for specific bodies of water, from which 'good status' will be derived as prescribed by the WFD. Reference conditions should be identified on the basis of sound scientific information but if this is not available then 'expert judgement' will be used – e.g. in cases where a water body has been impacted over a long period of time, and no, or inadequate, data exists on the original/natural conditions and/or on chemical parameters not routinely monitored. Reference conditions proposed by Member States will be subject to an intercalibration exercise by the EC. This is an attempt to standardise levels set for similar water body types in similar environments, across Member States.

IMPLEMENTATION ISSUES

Regulations to transpose the WFD into UK law will be in place by the end of December 2003. The Regulations as proposed (Defra, 2003), are very general and place a lot of power in the hands of the Secretary of State to establish the necessary details without further Parliamentary scrutiny. Such a transposition reflects the fact that we still do not have a clear picture of what implementation of the WFD will mean.

The characterisation of River Basins and the water bodies therein, the identification of reference conditions and the subsequent definition of good status for all water bodies, will be largely a technocratic processes. This is being undertaken by the Environment Agency who the Government are proposing to be the 'competent authority' responsible for implementing the WFD.

How we deal with the technocratic - democratic interface process which is at the heart of WFD implementation, is crucial. The legal obligation under Art.14 of the WFD (and in line with obligations under the Aarhus Convention), for stakeholder and public involvement in decision making, goes beyond the normal mode of consultation as currently practiced. Technocrats (scientist and engineers) do not generally feel comfortable with this technocratic – democratic interface, and therefore there is a lot to learn and change. One option is to fully engage Local Authorities, who through the land-use planning / development control regime, have much experience operating at the technocratic - democratic interface.

If the 'authorities' do not get it right then other legal liabilities will arise, as private citizens take action in the courts. Recent notable Common law actions relating to water include the cases of Marcic *v.* Thames Water, Hart *v.* Anglian Water, and that brought by villagers in Bangladesh against BGS/NERC.

CONCLUSIONS

The obligations under the directive are strict – good water status once set, must be achieved and no deterioration of status will be allowed. Three years into implementation there is still not a clear idea of the implications and impact of the WFD – which are potentially significant and even worse if we get it wrong.

Many professionals whose work interfaces directly or indirectly with water and its environs are getting involved and making a contribution – but more need to do so. Hydrologists who have yet to integrate quantity with quality must do so in order to enhance what is a necessary and important contribution to optimising the outcome of WFD implementation.

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A risk assessment methodology for determining nutrient pressures on surface water bodies at the river basin scale

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The Water Framework Directive 2000/60/EC (WFD) is the most significant piece of European water legislation to be produced for over twenty years. It rationalises and updates existing water legislation, and introduces an integrated and co-ordinated approach to water management in Europe, based on the concept of river basin planning.

Nutrients, in particular phosphorus (P), in freshwaters, are considered to be a pressure which may result in an increased risk of eutrophication. The current regulatory drivers for nutrient control tend to be focused on controlling point or diffuse nutrient sources, rather than a combined approach. The Directive, however, requires each Member State to undertake an initial assessment of risks and impacts from all human activity, which combined with information on the sensitivity to the pressures, will identify those waterbodies at risk of failing to meet the Directive's environmental objectives. This initial risk assessment must be completed by the end of 2004.

The risk assessment needs to be structured so that the level of detail needed is in proportion to the difficulty in judging whether or not a water body is at risk. The process consists of three components:

- characterisation of water bodies - differentiation into types and identification of reference conditions;
- identification of type and magnitude of the pressure (nutrients in this case); and
- assessment of the sensitivity/susceptibility of the receiving waterbody to the pressure (nutrients in this case). This will depend on a water body's natural characteristics and on the other pressures acting upon it.

This paper describes work being undertaken jointly by the Environment Agency and a research consortium towards developing a risk assessment methodology to identify nutrient pressures on biological assemblages in lakes (EA project P2-260/9 *Risk Assessment Methodology for Determining Nutrient Impacts in Surface Freshwater Bodies* (NUPHAR)). Here we focus on the approach being developed within the NUPHAR project for predicting diffuse agricultural nutrient losses as this is one of the most difficult parts of the risk assessment.

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The NUPHAR project is seeking to develop a tiered modelling framework which incorporates different approaches, ranging from coarse pressure screening tools to meet the immediate WFD 2004 deadline, to simple semi-distributed models. Three approaches for assessing diffuse agricultural phosphorus pressures were evaluated: land use export coefficients using the GB lakes inventory; pressure-delivery risk screening matrix; and the Phosphorus Indictors Tool (PIT).

The simplest approach uses export coefficients based on land use information held within the CEH GB lakes inventory. Nutrient export models such as this are 'black-box' models that make no attempt to explain the processes involved in generating nutrient outputs from a set of input parameters but allow some evaluation of the impact of changing inputs or managing outputs within agricultural systems. Catchment polygons were created for a series of 50 test lakes to allow an estimation of the diffuse nutrient pressure within each lake catchment based on simple export coefficients.

A more sophisticated approach, the Phosphorus Indicators Tool (PIT), is also being evaluated (Heathwaite *et al*., 2003). PIT predicts phosphorus delivery from agricultural land to receiving waters using the most appropriate indicators of phosphorus loss. The indicators are organised in three interlinked modules which feed from one layer to the next: (layer 1) P loss-potential indicators, (layer 2) P transfer indicators and (layer 3) P delivery indicators. Currently the model runs on an annual time step and at a 1 km^2 grid scale using nationally available data. The PIT model is being tested on six of the fifty test lakes (Barton Broad, Slapton Ley, Loch Leven, Windermere, Esthwaite Water and Blelham Tarn).

As an intermediary between the export coefficient approach and PIT, a simple phosphorus pressure-delivery risk screening approach was developed by the Environment Agency and NUPHAR research consortium based on catchment phosphorus inputs (pressures) and assuming phosphorus delivery via two principal transport pathways (particulate transport via surface flow and soluble transport via groundwater movement). To determine phosphorus inputs, agricultural census data was used to calculate the areas of crops and numbers of livestock in each 1 km^2 grid. Standard numbers for the amount of P applied to crops or produced by animals was developed using the coefficients in layer 1 of the PIT model. From the gridded agricultural census data and the standard application/production rates, an estimated phosphorus input/source rate per km^2 was calculated using ArcView 3.1 standard routines. The input/source rate was then ranked into high-medium-low categories. Since the main route of phosphorus to surface water is via attachment to sediment particles, the likelihood of sediment-bound phosphorus reaching watercourses from any area is related to the soil type, slope of the land, rainfall and the distance of the land from a watercourse. These factors have been combined to produce a map giving the estimated sediment load which could erode based on a 1 km grid square (Environment Agency, 2002). This map was re-classified into high-medium-low risk of sediment delivery. The two high-medium-low grids (source and delivery) are then multiplied together to give a final ranked pressure map. Similarly, the Base Flow Index (BFI) was chosen to represent areas where there is a high degree of connectivity between the surface and groundwater, and where surface waterbodies are highly influenced by the quality of the groundwater. A 1 km^2 grid of BFI values was then reclassified into high, medium or low groundwater to surface water connectivity. The reclassified P pressure grid and the BFI grid were then combined to give a pressure map for P losses to groundwater. Validation of this matrix approach is being undertaken by overlaying catchment polygons for the fifty test lakes on the two pressure matrices to calculate the

proportion of the catchment occupied by each of the bands, and comparing this with water quality data. Some of the initial findings using this approach will be presented.

Analysis of the three models to date has demonstrated a tiered risk assessment approach is appropriate. On this basis, where the coarser export coefficient and the pressure matrix approach showed a catchment to be at high risk, it was justifiable to undertake a more detailed assessment using the PIT approach. Recent findings will be presented to develop these points.

This research is also exploring the potential for linking modelled nutrient loads (using models such as PIT) to metabolic models of phytoplankton response (Reynolds & Maberly, 2002) and to within lake P and chlorophyll measurements based on the GB lakes database (Hughes *et al*., in press). The overall objective of the work is to produce a robust risk-based prioritization protocol for lakes and slow-flowing rivers in relation to nutrient pressures and phytoplankton response.

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Fine sediment in the aquatic environment: issues for successful management

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INTRODUCTION

Fine sediment is an often neglected component of water quality. Nonetheless, both the quantity and quality of fine sediment can be critical for the aquatic environment. Suspended solids concentrations and the deposition of fine material, either on or within the channel bed, affect the habitat of a range of fish and invertebrate species (Theurer *et al*., 1998; Hastie *et al*., 2000; Gayraud *et al*., 2002; Newson, 2002). Suspended solids are also an important transport vector for phosphorus (Owens & Walling, 2002), metals (Neal *et al*., 1999) and organic contaminants (Long *et al*., 1998). Moreover, the exchange of nutrients and contaminants between sediments and the water column, as well as their bioavailability within deposited sediments, are fundamental to both water quality and ecological status (Wade *et al*., 2001; House & Denison, 1998; den Besten *et al*., 2003). Other causes of concern arise from the loss of reservoir capacity (White *et al*., 1996) and the need to dredge sediment for navigation or flood defence purposes (Bates & Hooper, 1997).

DRIVERS FOR SEDIMENT MANAGEMENT

The necessity for sediment management in the UK, arising from this plethora of interests, is embodied within a number of EU Directives with a disparate set of objectives. The Freshwater Fish Directive (78/659/EEC) is concerned with rivers and canals designated as salmonid or cyprinid waters, while the Habitat and Species Directive (92/43/EEC) is focused on Special Areas of Conservation which are designated for specific habitats or species. In both these cases, relevant objectives are concerned with sediment quantity and relatively low levels of suspended solids concentrations (e.g. mean value < 25 mgl^{-1}) and siltation (e.g. $< 10\%$ fines in top 30 cm of bed sediments). By contrast, the Water Framework Directive (WFD 2000/60/EC) is directed towards all water bodies and whole river basins, but is far less prescriptive in terms of suspended solids. The transport of solids is quoted as an optional classification element of water bodies, while the structure and condition of the substrate are included in the hydromorphology. Sediment quality is specifically mentioned in the WFD with relation to good ecological status for phytoplankton, macrophytes and phytobenthos. Sediment quality is also important with regard to recent changes in

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the regulations for the disposal of dredgings (Landfill Directive 1999/31/EC) and the potential requirement for remediation measures (Köthe, 2003).

CURRENT MANAGEMENT APPROACHES

In line with these disparate drivers, management responses have tended to focus either on what is "delivered" to the river system or on deposits associated with specific sites or specific issues of contamination. The response to environmental objectives relating to fine sediments, or to diffuse pollution associated with eutrophication, has largely been aimed at reducing the amount of fine sediment reaching watercourses through changes at the farm scale (Defra, 2002). Measures which address the management of sources (e.g. reducing erosion vulnerability) and transport (e.g. buffer strips and sedimentation ponds) prior to delivery to the river system have been assessed and costed in a number of documents (Hilton, 2002; RPA, 2003). However, their effectiveness in meeting objectives at a basin scale has still to be assessed. At specific sites, the problem of siltation has been recognised as a river restoration issue and either gravel cleaning advocated (Shackle *et al*., 1999) or changes put in place to increase the sediment conveyance of the local river reach (Environment Agency, 2000).

However, what has yet to be generally appreciated is the spatial coherence of some of these issues and the temporal dynamics of the fluvial sediment system as a whole. Questions which still need to be addressed are: How do changes in the delivery of sediment to the river affect downstream sites? Where is it most cost effective to focus schemes for reducing sediment inputs? Will a reduction in fine sediment delivery lead to channel erosion elsewhere? What are the consequences of fencing off channel banks? The purpose of this paper is to provide two examples – one at the catchment scale and one at the river reach scale – which illustrate the need for a better appreciation of the fluvial sediment system as a whole.

UNDERSTANDING THE SEDIMENT SYSTEM AT THE SCALE OF THE RIVER NETWORK

Mapping of erosion vulnerability i.e. potential sediment export from the land (McHugh *et al*., 2002) and sediment delivery to the river (Naden & Cooper, 1999; Watts *et al*., 2003), has provided some capability in estimating suspended solids inputs to river systems over large catchment scales. However, in order to look at downstream impacts, the role of in-stream processes must be assessed. Even in cases where the suspended solids inputs to the river system appear to be balanced by exports on an annual time scale, the routing of the water and sediment waves during a flood event may have a significant influence on the shorter-term sediment budget in individual river reaches.

As part of an investigation into fluxes, continuous monitoring of suspended solids concentrations was carried out, within the NERC-funded LOIS programme, on the River Swale, North Yorkshire (Figure 1). The catchment area is 1363 km^2 and the focus of interest in this paper is the 55 km reach between Catterick and Crakehill. Upstream of Catterick, the land use is predominantly upland pasture, some of which has been subject to moorland gripping. More intensive agriculture is practised in the Lower Swale which includes part of the Vale of York with grazed turf and tilled land becoming the dominant land uses.

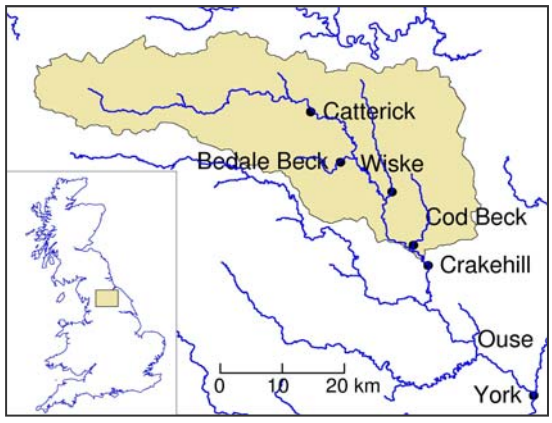


Figure 1 Study area showing river monitoring stations and tributary inputs

Suspended solids concentrations measured at Catterick and Leckby Grange, near to the gauging station at Crakehill, were based on turbidity monitoring at 15 minute intervals coupled with triggered automatic sampling followed by gravimetric analysis for suspended solids. The turbidity measurements were converted into suspended solids concentrations using event-specific relationships between suspended solids concentrations and turbidity. The data were analysed on an event basis (Smith *et al*., 2003a, 2003b), and over 50% of the annual load was estimated to have been transported during a single event lasting less than 10% of the year. Figure 2 shows results from two contrasting events. In the first example (Figure 2a), the dominant source of the water and sediment is from the upper part of the catchment. As shown the water wave is attenuated as it moves downstream. This reduces the carrying capacity of the flow during the event with the net result that the sediment load conveyed out of the reach is much less than that entering the reach. During the second event (Figure 2b), the dominant source of both water and sediment is from the lower part of the catchment and the flood wave builds between the two measurement points as does the sediment flux. In this case, the sediment budget for the reach is approximately balanced.

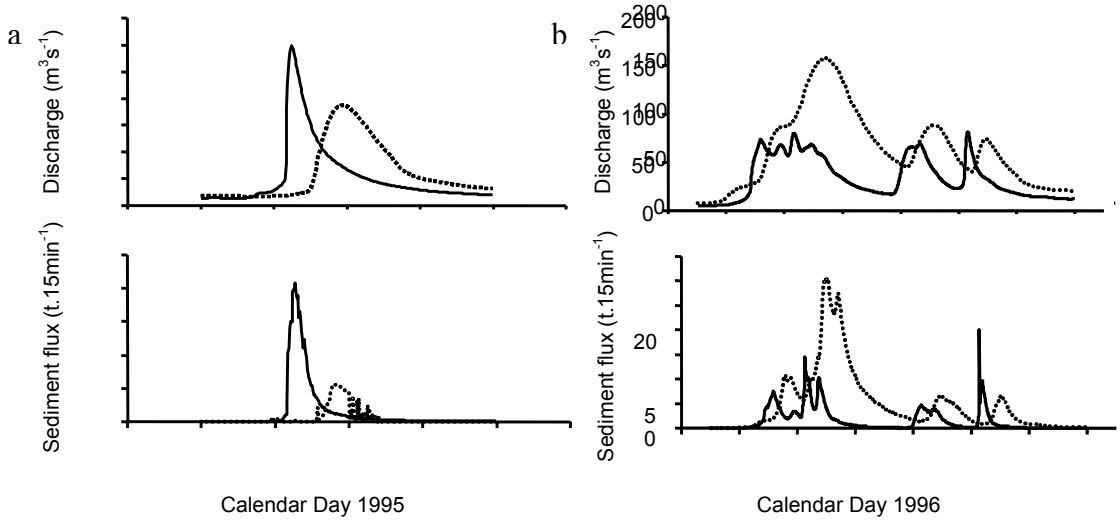


Figure 2 Example flow (m^3s^{-1}) and sediment load ($t.15min^{-1}$) during two contrasting events: Catterick shown by solid line; Crakehill/Leckby Grange shown by dashed line.

The analysis of 17 events during the period October 1994 to February 1996 reveals that the water and sediment waves show a range of different behaviour, depending on the dominant sources and timing of inputs during events. This is further complicated by sediment availability. For example, clockwise hysteresis in the relationship between suspended solids concentration and flow at Catterick implies the occurrence of sediment exhaustion in the upstream part of the catchment and, under these conditions, the channel within the Catterick to Crakehill reach can become an important source of sediment. Table 1 gives input and output totals for the reach during five of the events analysed. The export from the downstream end of the reach lies between 40% (±3%) and 159% (±13%) of the material input from the upstream part of the catchment and the tributaries within the reach. The net effect is therefore inferred periods of deposition and erosion within the reach of interest.

Table 1 Sediment flux during example events for a 55km reach of the River Swale

Event	Flux at Catterick (tonnes)	Flux from tributaries (tonnes)	Flux at Leckby Grange (tonnes)	Output as % of inputs – range based on assessment of errors shown in brackets	Inferred channel erosion (-) or deposition (+) (tonnes)
Example in-bank events					
11-26 Nov 1994	778	750	2281	149 (99 to 236)	-753
15-18 Nov 1995	2230	21	901	40 (39 to 40)	+1350
8-21 Feb 1996	1942	4137	6029	99 (76 to 138)	+50
Example overbank events					
24 Dec 94-4 Jan 95	3284	5281	6188	72 (59 to 91)	+2377
19-26 Feb 1995	4534	3095	12110	159 (140 to 180)	-4481

The implications of this work for sediment management are threefold. First, it is not straight forward to predict the impact of changing inputs through land management practices on downstream suspended solids concentrations or siltation. Secondly, a quantitative understanding of the sediment system may help determine where measures to reduce suspended solids levels are best targeted and areas where they will either have no effect or perhaps even a detrimental effect downstream. Thirdly, the interaction of sediment with the overlying water may be mediated by periods of erosion or deposition. Newly deposited sediment may be high in nutrients or contaminants or may bury the existing sediment with less contaminated material. Similarly, erosion may cause the exposure of a different quality of sediment. In cases where erosion or deposition is a dominant process, it will not be cost effective to consider *in situ* remediation.

UNDERSTANDING PROCESS INTERACTIONS AT THE RIVER REACH SCALE

While a quantitative understanding of the sediment budget at the catchment scale is arguably a necessary pre-requisite for sediment management, process interactions within the river reach may also play an important role in determining the impact of any control measures. For example, one option for reducing sediment inputs is to fence off riparian areas. This has additional benefits of reducing nutrient inputs direct to the river system, stabilising channel banks and enhancing habitat by re-vegetation of banks. However, the re-growth of bankside vegetation and the growth of macrophytes within the channel may also mean that flow velocities are reduced and silt deposition may occur.

Recent work on the River Blackwater in Farnborough has highlighted the association of sediment deposition with macrophyte growth and begun to quantify this in relation to the reduction in flow velocity. Figure 3 shows results from a transect across a lateral bar colonised by a stand of *Sparganium erectum*. A seasonal pattern in the deposition and erosion of fine sediment is observed with build up of sediment during the summer months for the period of macrophyte growth. Erosion of the bar during high flows in the winter months involves both surface removal and trimming back of the bar from the centre of the main channel.

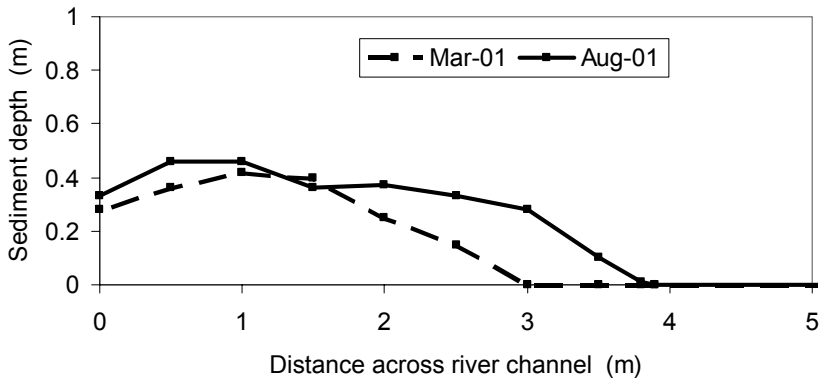


Figure 3 Seasonal variation in a single transect of River Blackwater near Farnborough

Within the Blackwater as a whole, macrophyte growth affects some 27% of the low-flow channel and comprises three elements: (i) emergent forms such as *Sparganium erectum* and *Typha latifolia*; (ii) submerged forms such as *Sparganium emersum* and *Potamogetan natans*; (iii) bankside vegetation comprising nettles, grasses and osiers. The interaction of these plant elements with the flowing water is a fully three-dimensional flow pattern providing a patchy mosaic of velocities and shear stresses. Work has begun on modelling this interaction. Some preliminary results for a single plant of *Ranunculus* spp. within a rectangular channel are shown in Figure 4. The recirculation zone of much slower velocity in the lee of the plant is clearly demonstrated and can be an efficient trap for fine sediment. Plant die back or removal during the autumn may then result in the release of fine sediment, suggesting that macrophyte growth is an important mediator in the behaviour of fine sediment and associated nutrients and contaminants.

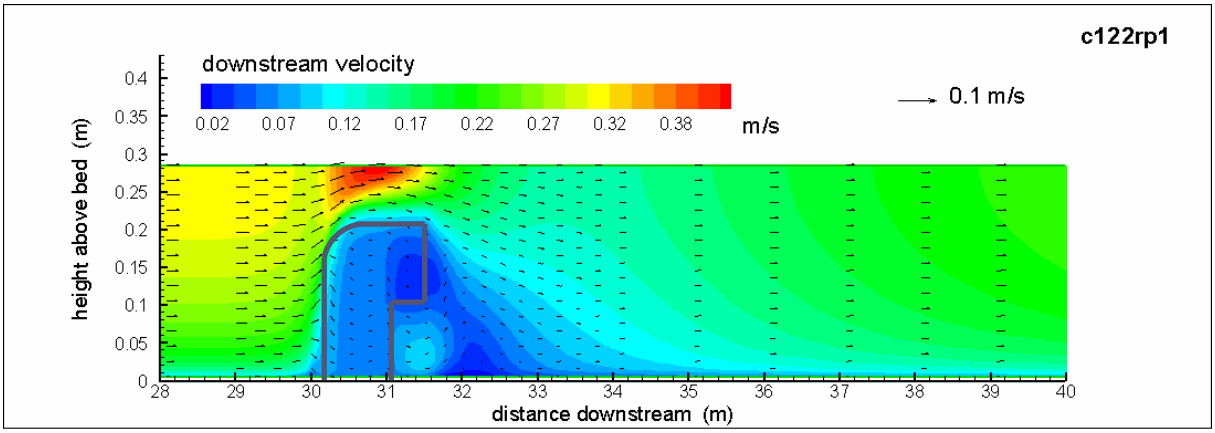


Figure 4 Simulated velocity field for single plant of *Ranunculus* spp. in straight rectangular flume. Generalised outline of plant shown. Vertical exaggeration 10x.

THE WAY FORWARD

The examples presented here highlight some of the space-time process interactions which need to be appreciated in order to understand the effect of sediment management practices. For successful management, sediment objectives need to be defined in the context of the whole river basin. At a qualitative level, a Fluvial Audit (Newson & Sear, 1997) will provide a general understanding of how the river system works and what historical legacies there may be within the system. An initial assessment against existing data and against this qualitative understanding can provide evidence as to whether sediment objectives are realistic. However, to answer the sorts of management questions posed earlier, a quantitative appreciation of amounts, sources, rates and process interactions will also be needed. This will require adequate monitoring strategies with a high temporal resolution and a spatial coverage designed to address the key issues for the particular catchment concerned.

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Aquatic environment management: a case study of the Rosia Montana gold mine project, Romania

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ABSTRACT

The Rosia Montana gold and silver ore deposit has been mined for over 2000 years. This was a jewel in the crown of both the Roman and the Austrian-Hungarian Empires. More recently the site has gone through over 40 years of state-owned underground and open-pit mining, with few environmental controls. Together these activities have created a network of underground workings totalling more than 140 kilometres of tunnels, which now form a complex conduit for Acid Rock Drainage (ARD) in the region. The name "Rosia Montana" meaning "Red Mountain" is due to the colour of the exposed leached rocks of the mountain and the red streams flowing down its valleys.

Now the mine is being privatised and redeveloped. The management of the new mine is faced with the dual task of remediating 2000 years of aquatic pollution and complying with current Romanian, EU and World Bank environmental standards.

An extensive mine-related aquatic monitoring system has been in operation since the year 2000 measuring both the quantity and quality of all waters impacted covering an area of more than 170 km^2. This includes six flow measurement sites and more than 70 hydrochemistry sampling sites.

This paper presents the findings of the baseline study to date, identifies the major sources of pollution and shows how the project will manage this environment in such a way as to meet or exceed the requirements of all Romanian and EU environmental regulations.

INTRODUCTION

The Rosia Montana gold and silver low sulphidation epithermal ore deposit is situated towards the top of Rosia Montana valley in the Apuseni Mountains in west-central Romania. This deposit has been mined for over two thousand years, being a jewel in the crown of both the Roman and the Austro-Hungarian Empires and the most important gold-producing region of Europe throughout these ages (Figures 1 and 2).

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More recently the mining of this deposit has encountered a period of economically and environmentally non-sustainable operation. Today the Rosia Montana mine is being privatised and redeveloped. The new mine is faced with the dual environmental challenge of:

- remediating two thousand years of aquatic pollution; and
- complying with Romanian, EU and International environmental standards.

HISTORICAL SETTING

Gold has been mined in Rosia Montana since before the Romans invaded Romania. Starting off with low intensity methods using hammer, chisel and grinding blocks, mining has gradually intensified. In the 1970s mining went from underground to open pit. To date, mining in Rosia Montana has been conducted using few environmental controls.



Figure 1 The Rosia Montana gold deposit has been mined for over 2000 years



Figure 2 Ore processing in the past in Rosia Montana

CURRENT SITUATION

The open cast mining currently being conducted in Rosia Montana started on a hill known as "Cetate" (Figure 3) in the 1970s. Ore is extracted from here, crushed, and then dropped down a vertical shaft to an underground railway (Figure 4), from where it is transported seven kilometres down valley to a processing plant. Tailings are deposited in the Saliste Valley tailings dam.

In addition to the current open cast mining, the extensive historic mining activities have resulted in a network of underground workings totalling more than 140 km of tunnels. These form a complex source and conduit for Acid Rock Drainage (ARD) (Figure 5). Due to the manner that the existing and previous mining operations have been conducted, the site has been listed as an environmentally high-risk industrial hot spot by the International Commission for the Protection of the Danube River (ICPDR).

With the recent period of economically and environmentally non-sustainable operation drawing to a close, the Rosia Montana mine is being privatised and redeveloped by Rosia Montana Gold Corporation (RMGC), a joint venture between the Romanian Government and a Canadian mining company. The proposed project has become known as the Rosia Montana Project (see www.RosiaMontanaProject.net)



Figure 3 The existing environmentally unfriendly Rosia Montana open pit



Figure 4 Lowest exit of Rosia Montana's underground galleries



Figure 5 ARD from the historic mining galleries

THE ROSIA MONTANA PROJECT

The Rosia Montana Project is founded upon the commercial exploitation of the world class ore deposit in Rosia Montana. Physically, this interpretates into the environmentally responsible and sustainable development of a mine including:

- four open pits;
- a processing plant;
- a cyanide detoxification unit;
- waste rock stockpiles;
- a tailings dam; and
- associated road infrastructure and water and waste pipelines.

The Rosia Montana Project's proposed development, however, is more than just a mine as it will also include:

- assistance for the closure of the current government subsidised mining operation;
- community resettlement activities;
- cultural property activities (archaeological surveys, assessments, cataloguing and preservation of significant artefacts);
- ongoing exploration throughout the local and regional area;
- mitigation of historical negative environmental impacts; and
- support for development of the local communities.

The Project began in 1997 with planning and exploration activities. Construction is expected to commence in late 2004, leading to an initial capital investment of approximately 450 million US dollars in Rosia Montana. Mining operations are scheduled to begin in 2006. Based on the known ore reserves, the Project will have a mine life of at least 17 years, to be followed by a mine closure phase of several years with associated additional monitoring and follow-up work. In addition, there is significant potential to extend this life should additional resources become proven.

The objective of the Rosia Montana Project is to use best proven technology to develop a modern gold and silver mine consistent with the social, economic, and environmental objectives of the local community, the Government of Romania, the European Union (EU) Directives and World Bank Group policies.

The activities of the Rosia Montana Project are interdependent. In addition to local, regional and national benefits, the Project will provide investors with a reasonable rate of return on investment, hence securing the financial backing for the Project. For the environment, it is through the finance raised through the development of the new mine that the environmental clean-up becomes financially possible. This clean-up involves three steps. The first is to identify the current pollution sources. The second step is to identify the pathways through which this pollution enters the downstream environmental system. The third and final step is the isolation and/or removal and treatment of the pollution source.

The Rosia Montana Gold Corporation (RMGC) has been monitoring the existing environment since 2000. The resulting aquatic monitoring system covers more than 170 km^2 upstream,

downstream and throughout the existing Rosia Montana Brownfield site. Biodiversity, also, has been monitored throughout the Project area and the surrounding countryside.

BIODIVERSITY IN THE PROJECT AREA

Biodiversity is intrinsically linked to the aquatic environment in Rosia Montana. To address this correctly is to address it in a holistic manner.

Landscape and habitat structure in the Rosia Montana Project Area has been significantly influenced by human activities over the past two thousand years. The deterioration of the area falls into two broad categories, namely deterioration through landscape structural changes and deterioration through changes at the ecosystem level. These changes are attributed to (a) historic and current mining activities and related pollution, including acid rock drainage (ARD), (b) transformation of natural systems to meadow, human settlements and planted forests, (c) development of semi-natural systems (man-made lakes) and (d) the exploitation of renewable resources.

Such activities have resulted in significant changes to the local flora and fauna, and as a consequence, the number of habitats of interest in the Project Area from a conservation point of view is relatively low.

In the long-term, the Rosia Montana Project hopes to improve the overall quality of biodiversity in the area through an integrated program that includes:

- establishing Environmental Protection Areas (EPAs) to protect key sites within the Project Area;
- re-vegetation of wildlife corridors between the EPAs and along riparian habitats;
- provision of micro-habitats for small animals (such as nest boxes for birds);
- establishment of habitat plots that will serve as seed banks and nurseries for rehabilitating derelict lands; and
- rehabilitation of degraded streams to support healthy aquatic communities.

To date, the Rosia Montana Project has conducted comprehensive baseline studies on terrestrial and aquatic flora and fauna, and has identified several areas of conservation concern in the Project Area. Over the course of the project, the success of initiatives aimed at protecting and hopefully improving the quality of biodiversity in the area will be measured through annual monitoring programs. Community education will play a key role in the success of these initiatives, encouraging more and more members of the local community to play a positive and active role in biodiversity management and protection.

ENVIRONMENTAL MANAGEMENT PHILOSOPHY

The existing state of the aquatic environmental degradation in the region has been documented. The challenge is now to clean it up and create a healthy environment through sustainable development. The Rosia Montana Project's environmental management philosophy can be summed up as follows:

- contain and remove existing pollution; and
- conduct the new mining operation using internationally recognised modern environmental management.

This philosophy is made practical through the natural geology and layout of the Rosia Montana and Corna valleys. From a hydrological point of view both valleys are dominated by surface water flow. This is due to groundwater systems being predominantly only present as flow through the top 10 to 15 metres of weathered rock on the sides of the valleys or alluvial material on the valley bottoms. Below this zone is a low permeability shale bedrock. The near-surface groundwater flow responds rapidly to rainfall and pollutants are flushed rapidly into the streams at the base of the valleys. No evidence of groundwater flow parallel to the surface water streams has been found.

The Rosia Montana Project will exploit this situation by capturing the historic acid rock drainage (ARD) through the construction of water retention dams downstream of the ore body in both valleys. This captured water will then be pumped for direct use in ore processing or to an ARD treatment plant near to the processing plant. Only treated water will be released into the streams downstream of the dams. Clean water from upstream of the ore deposit, which is currently flowing into the ARD rich streams, will be diverted around the mine site to join the clean treated water coming from the treatment plant.

As the Rosia Montana Project is being conducted in Romania (an accession country to the EU) and being financed by the international financial community, all the Project's components will conform with current or foreseen regulations of the Romanian State, EU Directives and International Guidelines (as defined, for example, by the Equator Principles).

To ensure that the Project has the administrative structure to deliver on its environmental and related commitments, associated management plans are being developed following international best practice on the following topics:

- Environmental and Social Management System;
- Public Consultation and Disclosure Plan;
- Resettlement and Relocation Action Plan;
- Water Management and Erosion Control Plan;
- Tailings Facility Management Plan;
- Cyanide Management Plan;
- Waste Management Plan;
- Air Quality and Noise Management Plan;
- Emergency Preparedness and Spill Contingency Plan;
- Biodiversity Conservation Plan;
- Cultural Heritage Management Plan;
- Mine Reclamation and Closure Plan;
- Environmental and Social Performance Improvement Plan;
- Environmental and Social Monitoring Plan;
- Community Development Plan; and
- Outline of Regional Development Plan.

Cyanide management will follow the UNEP facilitated International Cyanide Management Code (www.cyanidecode.org). Cyanide will be transported to site in its solid form in a container with triple containment.

The dams will be designed and built to meet all Romanian and International safety standards and regulations. The tailings management dam will not be constructed of tailings, unlike many other tailings dams. It will be constructed as a centreline, rock-fill dam with the starter dam including a low permeability core.

RESULTS

The major sources of pollution are identified and classified in line with Romanian and EU legislation. Streams in the region are highly contaminated with pH as low as 2.5, and water chemistry pollution existing over 70 times over the Romanian legal limits (Figure 6). The Rosia Montana Project will change this situation by collecting the historic acid rock drainage using dams and treating the water to applicable Romanian and EU standards prior to release. Further, by the very fact of developing the new mine, the source of this pollution will be either partly removed or stabilised.

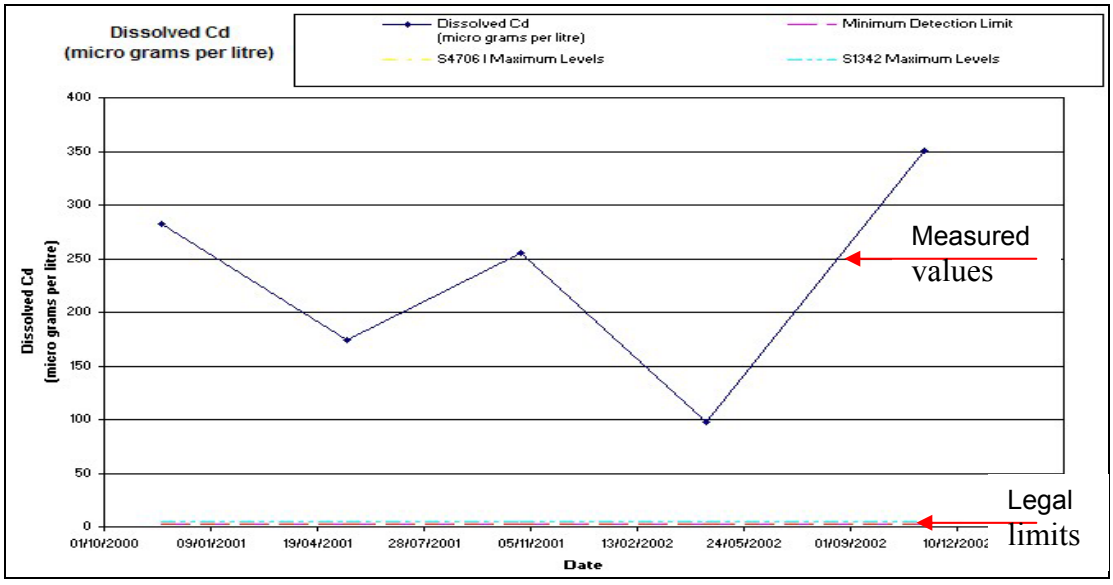


Figure 6 Existing dissolved cadmium concentration being discharged from the mine galleries.

CONCLUSION

The new Rosia Montana Project will develop the existing mine site to create a financially viable modern gold mining project within a sustainable and healthy environment. Through the resources available from the responsible development of the geological richness of this area, Rosia Montana will once again become a world-class mining centre. The difference this time will be that the project will be developed and conducted in respect of the laws of a 21st century Romania incorporating 21st century environmental management.

The challenge remains for the developers of the Rosia Montana Project, the Romanian authorities, the European Union, the international financial community, and all concerned individuals and institutions to work together to ensure that the Rosia Montana Project becomes a success story. The development of the Rosia Montana Project should provide an example to the international community of how private capital and enterprise can work with the local authorities and other stakeholders to resolve the legacy of years of environmentally unfriendly and non-sustainable mining and create a platform for future sustainable development.

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A scoping study for a Welsh acid waters strategy

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INTRODUCTION

Soil and surface water acidification due to the deposition of acidifying pollutants in Wales remains a significant environmental issue. This is because of the extent to which upland soils and streams are acidified, and the particular impacts on salmonid fisheries. However, considerable progress has been made over the last two or three decades both nationally and within Europe in terms of reducing the emissions of acidifying pollutants. Against this background, the Welsh Assembly Government commissioned a scoping study to inform the development of a national strategy for tackling surface water acidification in Wales (Reynolds *et al.*, 2002). The principal objectives of the scoping study as defined by Welsh Assembly Government were:

- To assess the benefits of these emission reductions for upland surface waters in Wales.
- To assess the potential benefits to a suite of Welsh freshwater sites of the emissions reductions recently negotiated in the United Nations Economic Committee for Europe Multi-Pollutant, Multi-Effect (Gothenburg) Protocol. This protocol is designed to achieve emissions reductions, relative to a base year of 1990, of 75% for sulphur dioxide, 50% for nitrogen oxides and 12% for ammonia by the year 2010.
- To assess the potential benefits of reduction scenarios additional to those recently negotiated in the Gothenburg protocol. These amounted to a further: i) 30% reduction in UK NO_x emissions; ii) 30% reduction in NO_x emissions from Wales alone; iii) 16% reduction in UK NH_3 emissions and iv) 16% reduction in NH_3 emissions from Wales alone.
- From the baseline of the predicted benefits of the Gothenburg protocol, to assess the options for local remediation measures to improve Welsh freshwaters based on:
  - Liming (direct dosing);
  - Agri-environment schemes (AES);
  - Re-introduction of aquatic invertebrates at specific sites.

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The study used models to predict, under current international agreements, the effects of reducing polluting emissions of sulphur and nitrogen on acid deposition in Wales. Effects of lowered deposition on the acidification of streams, rivers and lakes was investigated using steady-state and dynamic models applied to data from the 1995 acid waters survey in Wales (Stevens *et al*., 1997). The study also reviewed local management options for remediating the effects of acidification on freshwaters and made recommendations for future action. The main findings of the scoping study are summarised in this paper to illustrate the approach taken and to highlight some of the uncertainties which still need to be addressed.

MODELLING CHANGES IN ACID DEPOSITION IN WALES

The Hull Acid Rain Model (HARM) model (Metcalfe *et al.*, 1995; 2001) was used to predict annual excess sulphur (i.e. that proportion of sulphur deposition not derived from sea salts) and NO_x deposition across Wales for 1990, 1995/97 (assumed "present day") and 2010. Deposition of reduced nitrogen compounds (NH_x) was modelled using the Fine Resolution Atmospheric Multi-pollutant Exchange (FRAME) model (Singles, 1996). Both models have been validated against data from the UK deposition monitoring networks and have been found to perform satisfactorily. In summary, the principal conclusions concerning acid deposition to Wales and the predicted effects of the Gothenburg protocol were that:

- Wales is currently a net exporter of sulphur and NO_x but receives a small net import of NH_x. This remained the case under the Gothenburg scenario in 2010.
- Total inorganic nitrogen deposition is dominated by NH_x which currently accounts for 69% of the total, increasing to 74% in 2010.
- The Gothenburg protocol is predicted to cut sulphur deposition to Wales in 2010 by 52%, NO_x-N deposition by 25% and NH_x-N deposition by 10% with respect to 1995-97 values, which are assumed to represent present day conditions.
- Achieving future deposition reductions of sulphur and NO_x-N particularly in north and mid-Wales will be highly dependent upon emission reductions outside Wales but within Europe. These will contribute 60% of sulphur and 54% of NO_x-N deposition to Wales in 2010.
- Power stations and industrial sources, which are concentrated in South Wales will contribute 93% of the Welsh SO_2 emission in 2010 making them a possible target to achieve detectable local deposition benefits.
- The additional N emission reduction scenarios of 30% for NO_x and 16% for ammonia beyond Gothenburg for Welsh only and UK sources made only a minor impact on predicted N deposition.

MODELLING THE ACIDIFICATION OF FRESHWATERS IN WALES USING STEADY-STATE CRITICAL LOADS

The acidification status of upland freshwaters in Wales was assessed by applying the First-order Acidity Balance (FAB) steady state critical loads model (Posch, 1995) to 123 survey and monitoring sites comprising a mixture of streams and lakes. This relatively simple model uses a mass balance approach to consider acidification by sulphur and nitrogen deposition simultaneously. Essentially it calculates the amount of sulphur and nitrogen deposition that can be sustained by a catchment without the Acid Neutralising Capacity (ANC) of the surface water falling below a specified threshold. The ANC threshold is

defined in relation to the requirements of key aquatic biota. Within Europe the key biological target is usually brown trout (*Salmo trutta*). A relationship between brown trout survival and ANC in Norwegian lakes (Lien *et al.,* 1992) has been used to define a critical ANC value of zero for the application of critical loads in Europe, the UK and hence this study. Using spatial data mounted on a Geographical Information System, the acid loading to a catchment can be determined from data such as that provided by HARM and the amount by which acid deposition exceeds the critical load can be calculated. The FAB model assumes that steady state conditions pertain; it can give no indication as to the timescale over which exceedance of critical loads might occur although the number of exceeded sites has been used to give an indication of acidification impact. The steady-state modelling exercise showed that:

- The number of stream and lake sites receiving acid deposition in excess of the critical load is predicted to decrease substantially from 80 (65%) in 1995/97 to 34 (28%) in 2010, provided the Gothenburg protocol is implemented on schedule.
- None of the stream or lake sites exceeded in 2010 will require reductions in excess sulphur deposition to reach the critical load.
- Nine (26%) of the sites exceeded in 2010 will require reductions in N deposition to reach the critical load.
- All other sites exceeded in 2010 can be protected by reductions in either S or N deposition.
- Overall, the future deposition reductions required to protect sites still exceeded in 2010 are less than they were in 1990 and 1995/97, as all the exceeded sites will receive deposition which is less than 1 keq-H^+ ha^{-1} yr^{-1} above the critical load.
- Thirty percent additional reductions in UK and Welsh (only) NO_x emissions beyond Gothenburg make virtually no difference to the predicted status of the sites in 2010.
- An additional 16% reduction in UK NH_x emissions beyond Gothenburg is predicted to increase the number of protected sites by 12% (11 sites) in 2010.
- The number of protected sites in 2010 is predicted to increase by 8% (7 sites) for an additional 16% reduction in Welsh (only) NH_x emissions.
- If N deposition continues at 2010 levels, the FAB model predicts that substantial nitrate leaching will occur at exceeded sites at an unspecified time in the future, once steady-state conditions have been attained. This results in significant acidification at many of the exceeded sites.
- Given the large uncertainty in the timing and probability of N breakthrough in current critical load models, these predictions should be treated with caution. Until the dynamics of the deposited nitrogen are understood, a precautionary approach would be prudent.

DYNAMIC MODELLING OF SURFACE WATER ACIDITY AT SELECTED WELSH SITES

A regional application of the MAGIC (Modelling Acidification of Groundwater in Catchments; Cosby *et al*., 1985a) dynamic acidification model was used at a subset of 45 stream sites to predict the timescale and extent of changes in acidity in response to the sulphur and nitrogen deposition scenarios predicted by the HARM/FRAME models. MAGIC is a process-oriented, intermediate-complexity dynamic model for the long term reconstruction and future prediction of soil and surface water acidification at the catchment scale (Cosby *et al.*, 1985a,b). The model consists of two sets of equations. Firstly, soil-soil solution

equilibria equations in which the chemical composition of the soil solution is assumed to be governed by simultaneous reactions involving sulphate adsorption, cation exchange and dissolution and speciation of inorganic and organic carbon. Secondly, mass balance equations in which the fluxes of major ions to and from soil and surface water are assumed to be governed by atmospheric inputs, mineral weathering, net uptake by biomass, and loss to streamwater. MAGIC uses a lumped parameter approach to aggregate the complex chemical and biological processes active at the catchment scale into a few readily described processes, and to represent the spatial heterogeneity of soil properties throughout the catchment. Dynamic simulation of soil and surface water chemistry is achieved by coupling the equilibria equations with the dynamic mass balance equations for each of the major cations (Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}) and major anions (Cl^{-}, SO_4^{2-}, and NO_3^{-}). The MAGIC model has been extensively tested on catchments in the USA (Cosby *et al.*, 2001), Scandinavia (Wright *et al.*, 1994) and the UK (Jenkins *et al.*, 1997). The MAGIC model currently includes N dynamics based conceptually on an empirical model described by Gundersen *et al*. (1998) whereby the soil C/N ratio controls the amount of incoming inorganic N which is leached.

The findings from the MAGIC model simulations were:

- Surface water annual mean ANC is predicted to improve significantly towards its pre-acidification level under the Gothenburg Protocol by 2010, assuming that the emission controls are implemented on schedule.
- The improvement in annual mean ANC is predicted to continue beyond 2010, with deposition maintained constant at 2010 levels, although at a slower rate.
- By 2010 only 2% of sites (1 out of 45 sites) will have an annual mean ANC < 0 $\mu eq\ l^{-1}$. In contrast, FAB predicts an ANC < 0 $\mu eq\ l^{-1}$ for 18 out of the 45 sites.
- By 2045, all sites are predicted to have annual mean ANC concentrations greater than 20 $\mu eq\ l^{-1}$.
- The recovery in annual mean ANC predicted by the model results almost entirely from reductions in S deposition.
- Nitrogen deposition does not contribute significantly to predicted surface water acidity even in the longer term.
- The variations on the Gothenburg scenario involving additional reductions in NO_x and NH_x deposition have no affect on the long-term surface water acidity predictions based on Gothenburg alone.
- The discrepancies between these predictions and those of FAB reflect differences in the representation of N dynamics in the two models and are a significant uncertainty that needs to be addressed by further research. Until the dynamics of the deposited nitrogen are understood, a precautionary approach would be prudent, based on the prediction of the FAB model.

MODELLING THE BIOLOGICAL RESPONSE OF SURFACE WATERS TO CHANGES IN ACIDITY

MEDUSA (Model of Ecosystem Dynamics Under Stream Acidification; Ormerod *et al.,* 1990a,b) has been used to simulate the likely impact on biological populations of the improvements in water chemistry predicted by MAGIC for the subsample of 45 sites. Aimed in particular at simulating trends among a range of acid sensitive organisms, MEDUSA predicts trout densities; trout survival time in acid waters; the probability of the occurrence of

a river bird, the dipper *Cinclus cinclus;* and the composition of invertebrate communities. Rather than being a complex process-based model, the essence of the approach is empirical and involves

- Relating patterns in the distribution and/or abundance of organisms to key features of water chemistry (i.e. acid base status as pH, Al^{3+}, Ca^{2+}, ANC etc) across large numbers of sites.
- Quantifying the relationships from a) using a range of statistical algorithms such as multiple regression, logistic regression and discriminant analysis.
- Predicting change in the target species using the algorithm derived from b) under any new conditions of water chemistry.

The main findings from the biological modelling are:

- Approximately 50% of sites should have trout densities greater than 10 per 100 m^2 by 2010 representing an improvement of about 10% on observed densities for 1995.
- The emission controls under the Gothenburg protocol are insufficient to return trout densities to pre-industrial (1845) levels when over 90% of sites are predicted to have had trout densities greater than 10 per 100 m^2. However, it is debatable whether pre-industrial conditions are an appropriate target for recovery given other changes, unrelated to acidification, that have occurred in the Welsh uplands since that time.
- For macro-invertebrates, approximately 30% of sites are characterised by acid-sensitive taxa by 2010, representing an improvement of about 10% on observed and modelled data for 1995.
- Approximately 55% of sites should have dippers present by 2010 reflecting little improvement on observations for 1995.
- The modelling approach contains a number of uncertainties including those present within the predictions from MAGIC used as input to MEDUSA and the poor correspondence between observed and modelled data for trout and dippers. These factors must be recognised in the interpretation of the results.

REVIEW OF REMEDIAL MEASURES

It was recognised at the outset of the project that whilst the key to solving the problem of acid waters in Wales is the reduction of emissions of sulphur and nitrogen compounds, local remediation measures could provide potential solutions to aid recovery in the short term. Three options were considered and assessed following literature reviews.

In relation to liming, it was concluded that targeted direct dose liming of streams could provide an effective option for the symptomatic treatment of acid waters by restoring chemical conditions to those suitable for acid sensitive biota. However, tributaries upstream of a doser would not be treated and this represents a limitation to the technique. There is evidence that targeted liming confers positive biological effects, at least where there are nearby sources for recolonisation, probably by downstream drift (invertebrates) or unimpeded upstream movement (fish). Limestone 'sand' treatments warrant further research as this approach can be targeted to protect the sensitive early life stages of fish which occur in river sediments. In order to achieve maximum cost-effectiveness from liming with least conservation impact, a scheme is required which prioritises Welsh rivers in terms of suitability for new liming projects. One such scheme which uses a series of chemical and

biological criteria to assess the suitability of a catchment for liming is proposed. The selection criteria include: fisheries and conservation status, stream chemistry, and nearness to recovery.

A review of current Agri-Environment Schemes (AES) operating in Wales showed that there is considerable potential to improve the habitats of rivers and streams in Wales through riparian management under AES. AES might benefit biological recovery of acidified streams where chemical conditions are becoming more favourable for acid sensitive aquatic biota. At present, however, quantitative evidence of their effectiveness is lacking. Of the three Welsh AES, Tir Gofal is currently the only one with specific riparian management prescriptions, but uptake is very small. Wetland management in the Organic Farming Scheme is poorly defined and the Woodland Grant Scheme is currently not used near rivers. A new focus is needed to combine the strengths of these three AES for the benefit of the riparian environment, within the context of integrated catchment management.

Using artificial means of re-establishing and re-enforcing aquatic macro-invertebrate populations in Welsh rivers is a possibility but quantitative evidence for its effectiveness is lacking. This approach would represent a real advance in conservation research and has rarely been attempted in freshwaters. There are two significant barriers to such an option which, for the short-term at least, limit its viability as a practical management tool. Firstly, there is a lack of information on many of the processes and logistics involved. Secondly, the cause of the limited persistence of macro-invertebrates in streams which have been limed or which are recovering from acidification is unknown, and until identified and eliminated, new populations should not be added.

CONCLUSIONS

The main overall findings of the scoping study were:

- Emission control policies, particularly for SO_2 are delivering improvements in the acid deposition climate for Wales.
- Implementation of the proposed emission reductions under the Gothenburg protocol will provide further substantial improvements in the acidification status of the 123 sites used in this study as measured using a steady-state critical loads model. A key question remains as to the extent to which these sites are representative more generally of acid waters in Wales.
- Efforts to ensure that the Gothenburg protocol emission targets are delivered internationally as well as within Wales should be an important policy priority which is likely to directly benefit Welsh acid waters.
- Targeting future SO_2 emission controls on power stations and industrial sources in south Wales will probably achieve detectable local benefits.
- There is very large uncertainty surrounding the prediction of the future importance of nitrogen deposition to recovery of acid waters in Wales. From an acidification perspective, specific recommendations concerning the need to reduce emissions of NH_x (the dominant form of N in deposition) cannot be made, although in view of the uncertainty surrounding model predictions, a precautionary approach would seem prudent. Reductions in NH_x emissions will benefit sensitive terrestrial ecosystems subject to eutrophication by excess nitrogen deposition. In the light of the

uncertainties surrounding predictions for freshwaters, terrestrial impacts may well drive future policy on ammonia emissions reductions.

- Despite the most optimistic predictions for recovery of surface water chemistry following implementation of the Gothenburg protocol, biological recovery is predicted to be slow and incomplete. Thus, some form of intervention should be considered to aid biological recovery in selected Welsh upland surface waters.
- Of the three local remediation measures reviewed in the scoping study, targeted direct liming is an option to include within an acidification recovery strategy provided it is used in support of continued emissions reductions and within the context of fisheries management objectives. Benefits should be justified economically and in terms of conservation impact. AES have value within integrated catchment management but at present there is no quantitative evidence for their direct benefit in ameliorating acidification impacts. Artificial re-introduction of aquatic invertebrates is not a viable management tool at the present time, and requires further research.
- The need for an adequate and co-ordinated monitoring strategy to assess the effectiveness of the Gothenburg protocol for the recovery of Welsh acid waters was identified by the study. Three elements are required: i) continued support of existing long-term monitoring sites, ii) the consideration of new sites to cover critical areas for recovery and iii) consideration of a third acid waters survey. Further work is required to assess the extent to which the existing long-term monitoring and acid waters survey sites adequately represent the population of acid waters in Wales and to identify the areas of critical concern.

- The following research needs were also identified which focus on: i) uncertainties in modelling nitrogen dynamics; ii) the importance of acid episodes; iii) quantitative assessment of the benefits of existing AES to freshwater habitats; iv) assisted colonisation by aquatic invertebrates and v) the environmental consequences of a change in forest management from patch clear felling to continuous cover forestry as proposed by the Welsh woodlands strategy (National Assembly for Wales, 2001).

ACKNOWLEDGEMENTS

We wish to thank the following for their various contributions to the study: Havard Prosser, Julia Salter, Michael Eklund (National Assembly of Wales), Wendy Merritt, Liz Roblin, Julian Bray, Steve Barnard (Environment Agency, Wales), Simon Bareham, Brian Pawson, Tristan Hatton-Ellis (Countryside Council for Wales), Andrew Pullin (University of Birmingham), Tony Prater (Royal Society for the Protection of Birds), Tom Nisbet and Steve Hunt (Forestry Authority). The study was funded by the Welsh Assembly Government.

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Upland catchment management and discolouration of water supplies

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ABSTRACT

This paper presents results of the first 18 months of a five-year experiment relating water quality to catchment management in upland blanket peats. Water draining from these catchments is a major source of water supply in the UK, potentially providing a cheap source of high quality water via gravity, but it is often highly coloured due to release of humic and fulvic acids. Records indicate that discolouration of water used for direct supply is increasing in many upland areas, along with associated costs of water treatment and resources. Traditionally, water colour has been managed by the addition of chemicals at the water treatment works, but catchment management may offer a preventative strategy for reduction of the problem in order to meet required standards.

The Nottingham Trent University is involved in an innovative partnership with Severn Trent Water, the National Trust and English Nature to conduct research on the management of peat erosion and associated water discolouration in the High Peak, Derbyshire. The UK and Peak District National Park Biodiversity Action Plans recognise blanket bog as a target habitat in need of protection and conservation with much of the South Pennines designated as a SSSI and nominated in the South Pennines Moors candidate SAC.

This five-year research project reflects the need for a long-term catchment investigation into the feasibility of controlling water colour into Ladybower Reservoir by means of sustainable land management in the Ashop catchment, Derbyshire. A paired catchment study involving monitoring of six sub-catchments of the River Ashop has allowed patterns of discolouration and hydrological responses to be identified, together with analysis of topography and vegetation prior to intervention in land management by means of changing burning, grazing and drainage regimes.

INTRODUCTION

Upland catchments, such as the Southern Pennines, are a major source of water supply in the UK, potentially providing a cheap source of high quality water via gravity (Naden & McDonald, 1989; Mitchell, 1991). However, water draining from blanket peat upland catchments is often highly coloured due to release of humic and fulvic acids. Excess colour

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builds up in peat catchments during the summer, being released in autumn and winter, producing an 'autumn flush' as the peat becomes re-hydrated (Naden & McDonald,1989; Butcher *et al.,* 1992; Pattinson *et al.*, 1995).

Discolouration of surface waters has escalated particularly since the severe droughts in the 1970s and mid 1990s, along with associated costs of water treatment and resources. Such discolouration is a major concern for Severn Trent Water in the Peak District and high water colour levels have been recorded at Bamford Treatment Works, Derbyshire (see Figure 1). Discolouration of these supplies and the associated costs have increased significantly in the last ten years.

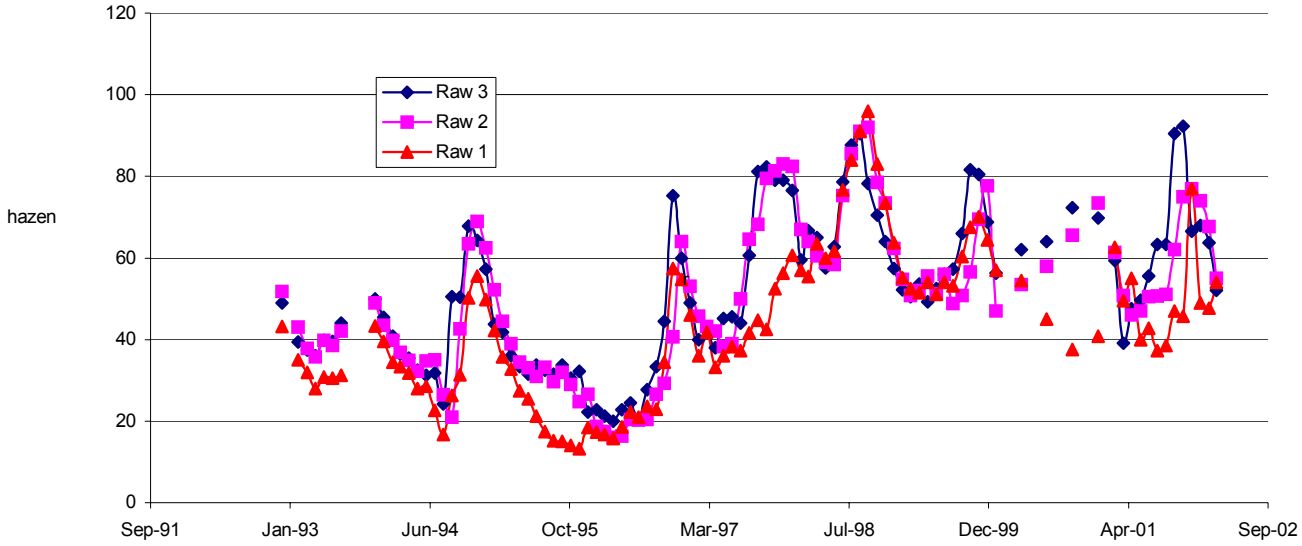


Figure 1 Long-term trends in water colour at Bamford Treatment Works (data from Severn Trent Water plc)

The problem of water colour is exacerbated by current management practices. Moorland management practices such as burning and grazing are known to cause blanket bog erosion and degradation (Anderson & Radford, 1994; Bragg & Tallis, 2001). Anderson (1986) and Tallis (1987) found burning regimes caused the onset of widespread erosion and exposure of bare peat, whilst intense grazing reduces the competitive vigour or kill plants through a process of feeding, trampling and poaching (MacDonald, 1990; RSPB, 1995). Ditch cutting, together with the naturally occurring intense drainage network has also been associated with peat desiccation (Burt, 2001). These practices and natural drainage patterns are present on the catchment feeding Bamford Water Treatment Works, and are thought to be closely linked to water discolouration by increasing the rate of humification of the peat.

Attitudes to UK upland management have, in recent times, changed emphasis from land utilisation for grazing and game management towards environmental protection. The UK and Peak District National Park Biodiversity Action Plan recognises blanket bog as a target habitat in need of protection and conservation, with much of the south Pennines designated as a SSSI and nominated in the South Pennines Moors candidate SAC. Organisations such as English Nature are a major instigator in its protection and conservation. Landowners and farmers also play an important role in the management of blanket peat moorlands, with

individual management practices and views having a significant impact on these upland catchments. The National Trust is a major landowner in the Southern Pennines, and therefore in a strong position to influence tenant farmers and game keepers managing their estates. Much of the National Trust land in this area, lies within the water gathering upland catchments which eventually drain into the Howden, Derwent and Ladybower reservoirs. These are managed by Severn Trent Water, and via Bamford Water Treatment Works provide a major potable water resource to the industrial towns and cities of the Midlands.

Although water colour has traditionally been managed by addition of chemicals at the water treatment works, catchment management may offer a preventative and thus more attractive strategy for reduction of the problem. It may be more effective and environmentally more sustainable to manage the moorland to reduce long term colour generation. This innovative project unites some of the most influential land managers in the UK, namely Severn Trent Water, the National Trust, English Nature and Environment Agency, whose objectives may previously have been divergent. Combining business, conservation and research, the land managers are working jointly with the Nottingham Trent University on a catchment-scale project to examine the roles of management practices (burning, grazing and drainage), in the generation and amelioration of drought-induced discolouration of surface water supplies.

The principal aim of the research project is to examine the role of blanket peat moorland management to mitigate or reduce water colour. The project focuses on water supplies to the Bamford Water Treatment Works. A control monitoring period of 18 months (recently completed) is now being followed by implementation of changes to land management at a catchment scale. This comprises changes on selected catchments to remove grazing, cease burning, and block eroded peat gullies. This project therefore reflects the need for a long-term catchment-scale investigation into the feasibility of controlling water colour into Ladybower Reservoir by means of sustainable land management in the Ashop catchment, Derbyshire.

THE STUDY AREA

The River Ashop is approximately 11 km in length and is one of the major rivers in the Dark Peak area of the Peak District in Derbyshire. The river drains a catchment area of 45 km^2, exhibiting a high drainage density. It supplies one of the major inflows to Ladybower Reservoir, a major source of potable water to large populations in the Midlands.

The study area, shown in Figure 2, comprises a set of six sub-catchments varying in altitude from 475 – 530 m. The area is composed of Namurian rocks, including the relatively soft Shale Grits and the more resistant Kinder Scout Grits. These are overlain with an almost continuous cover of Winter Hill Association (1011b) peat, which is extensive and varying in depth, producing a generally smooth, undulating surface on the summit plateaux. The soft, almost permanently water-logged peat is vulnerable to erosion, forming radial systems of gullies related to the surface topography. The main drainage channels are deeply incised into underlying rock, exposing the mineral substrate.

Cotton grass moorland dominates the highest plateaux, and heather, (*Calluna vulgaris*), together with other dwarf-shrub species such as bilberry and crowberry (*Vaccinium mytillus* and *Empetrum nigrum*) which dominate the drier bog. Current land management varies between catchments, but is primarily for sheep grazing and rearing of grouse. Heather

burning is used as the main management tool to maintain a diverse age structure of heather from pioneer to mature, in order to provide a nutritional food source and cover for wildlife (English Nature, 2001).

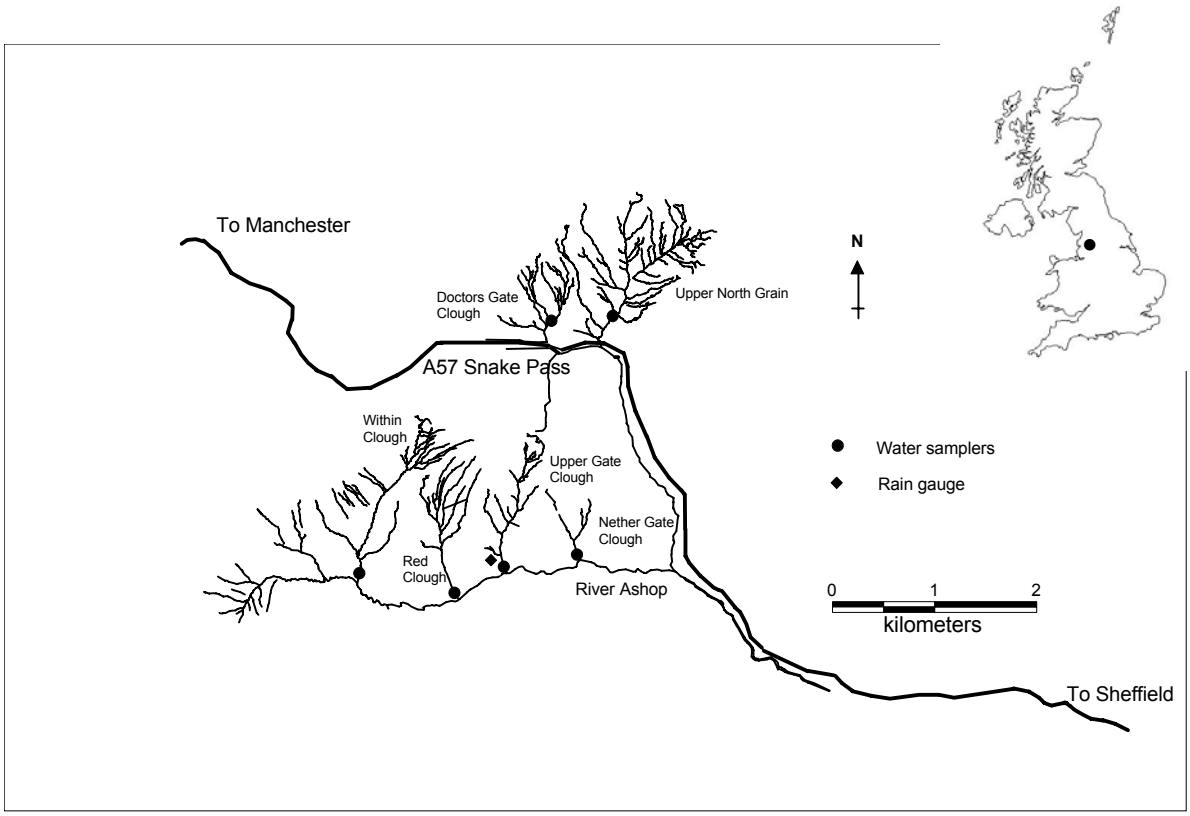


Figure 2 Map of Study Area (River Ashop, Derbyshire, UK)

EXPERIMENTAL METHODOLOGY

Preliminary studies of the streams in the Derwent, Westend and Ashop catchments by Blackmore (2001) and Walker (2001) identified those sub-catchments of Ladybower reservoir contributing most colour in terms of both true colour and "true colour load" (a conceptual quantity of colour, derived as the product of discharge in the stream and true colour of the water that day, and important in terms of costs of water treatment). On commencement of the current project, further spatial sampling confirmed those sub-catchments within the study area to have disproportionately high true colour in the streams.

Six sub-catchments of the River Ashop (Doctors Gate Clough, Upper North Grain, Nether Gate Clough, Upper Gate Clough, Red Clough and Within Clough) were thus instrumented to establish baseline relationships between meteorological inputs and hydrological responses. Water discolouration, morphological, pedological and land management characteristics were also examined during a one-year control period in order to identify those sub-catchments which could be paired together and responses predicted pre- and post-management intervention.

Six Xian 1000 automated water samplers were installed on each sub-catchment near to the confluence with the River Ashop. These were set to sample hourly and to provide a composite daily sample over a 24-hour period. Six Ott Thalimedes shaft encoders were installed next to the samplers and set to record stage at 5 minute intervals. Stream discharge was measured regularly in a natural rated section of the main channel on each catchment, using an electro-magnetic flow meter (Aqua Data Systems). Mean daily discharge at each site was then calculated from the stage record.

Apparent colour of the water samples and true colour (i.e. after filtering through cellulose acetate) were measured in Absorption units.m^{-1}, using a Cecil 1000 spectrophotometer at 400 nm. These values will be converted to Hazen units (see Watts *et al.*, 2001).

Transects of dipwells were installed on the plateaux of each catchment and ground water levels are being monitored monthly. Two Ott Thalimedes shaft encoders were also installed in dipwell tubes, at Upper Gate Clough and Within Clough, set to record ground water level at 15 minute intervals.

Precipitation is being monitored continuously with a 0.2 mm Casella tipping bucket rain gauge and is supported by a Snowdon manual gauge, together with regional daily rainfall obtained from the Environment Agency and MORECS evaporation estimates.

Habitat mapping and general land use was initially established through aerial photographic interpretation and supported by ground truthing. A Site Condition Assessment was conducted using the English Nature Upland Vegetation Condition Assessment for blanket mire (Jerram & Drewitt, 1998), noting abundance of sphagnum, dwarf shrub cover, graminoid cover, lichens and erosion. Impacts of grazing, burning and drainage were also noted.

Vegetation community boundaries were identified to general habitat and NVC level (Rodwell, 1991) using aerial photograph interpretation and 10 m × 10 m vegetation quadrats located by stratified random sampling. Permanent 1 m × 1 m quadrats were set up at each dipwell point across the catchments to identify species present and area of bare ground, vegetation cover, condition and sward height. A point-transect of vegetation monitoring was also established with vegetation species and height noted every 2 m across the open moor, and every 0.2 m in the gully channel.

RESULTS

Figure 3 shows variations in area-specific daily mean discharge and true colour for Within Clough over the control period October 2002 to the end of September 2003. The runoff rises and falls rapidly in response to rainfall, with very little baseflow component. True colour mirrors the short-term peaks in discharge but also demonstrates a seasonal pattern, with higher colour levels in autumn and winter.

Table 1 shows that Nether Gate Clough has the highest rates of runoff, as measured by the area-specific daily discharge, and this, combined with a mean true colour of 9.41 produces the highest area-specific true colour load. Upper North Grain, Doctor's Gate Clough and Upper Gate Clough have very similar colour loads, whilst Red Clough is somewhat lower and Within Clough much higher than average.

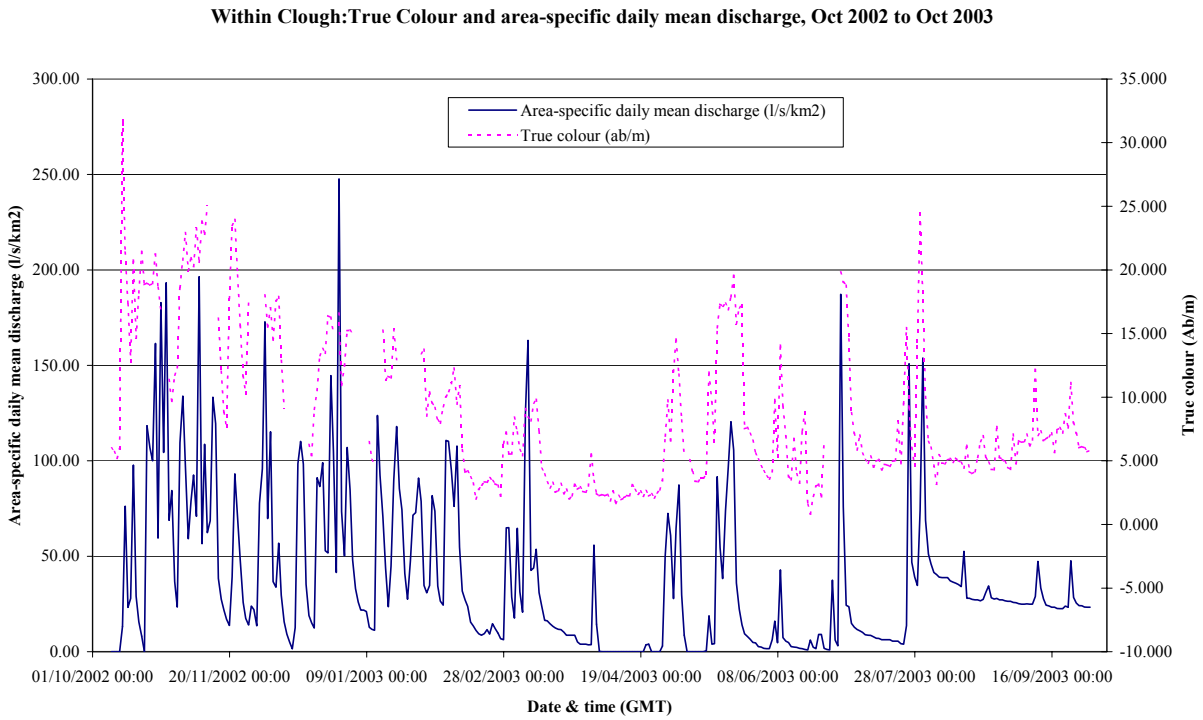


Figure 3 **Area-specific daily mean discharge and true colour for Within Clough over the control period October 2002 to the end of September 2003.**

Table 1 **Means of daily mean discharge and colour values for the six experimental catchments, October 2002 to end of September 2003**

	Area-specific daily discharge ($l.sec^{-1}km^{-2}$)	True colour ($Ab.m^{-1}$ at 400nm)	Area-specific true colour load ($Ab.m^{-1} * l.sec^{-1}.km^{-2}$)
Doctors Gate Clough	38.19	14.261	683.024
Upper North Grain	40.77	6.655	398.928
Nether Gate Clough	45.38	9.406	702.109
Upper Gate Clough	27.26	7.528	316.592
Red Clough	19.82	7.998	161.697
Within Clough	38.07	8.613	447.491

For each catchment, relationships have been established to predict true colour from discharge and antecedant rainfall over various time periods. Colour is shown to be positively correlated with rainfall on the day and within the last month; this reflects the flushing mechanism of flow through the upper layers of the peat. Colour is however, negatively correlated with rainfall over the preceding four months ($p = 0.01$ or better) for all six catchments. This can be attributed to the effects of drought periods in generating colour within the upper layers of the soil, as has been reported by a number of authors such as Naden & McDonald (1987). The R^2 value varies from 0.366 for Red Clough to 0.604 for Nether Gate Clough.

Pairings of catchments produced very strong relationships, as illustrated in Figure 4 for daily mean area-specific true colour load at Within Clough and Upper Gate Clough. Such correlations are one means of identifying the strongest pairings between catchments, and these have confirmed the similarities identified between catchments at the pilot stage of the project, by the use of cluster analysis on catchment characteristics (vegetation, topography, burning regime etc).

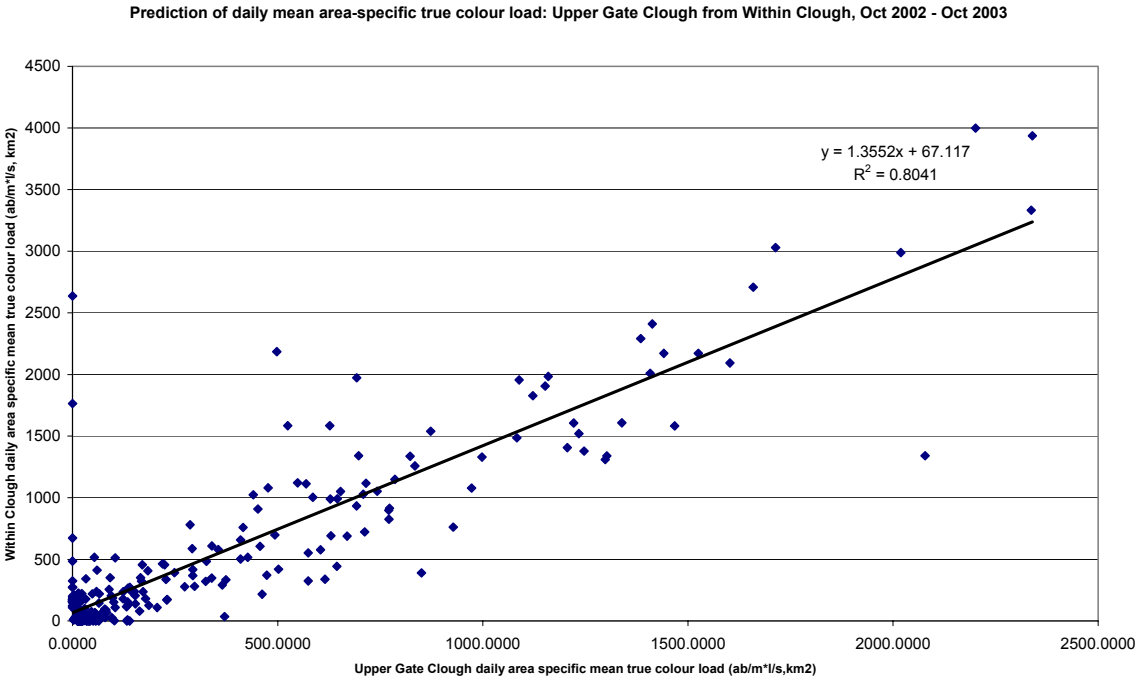


Figure 4 Relationship between Within Clough and Upper Gate Clough: Daily mean area-specific colour load for the control period, October 2002 to October 2003

DISCUSSION

Management changes have now been implemented on Doctors Gate Clough (removal of grazing), Nether Gate Clough (cessation of burning) and Within Clough (blockage of gully channels). The other three catchments remain as experimental controls. Monitoring will continue for a further three years, to determine whether any discernible changes in colour response occur in the treatment catchments.

Whilst many workers have predicted the effects of catchment management on water colour, and there have been experiments at laboratory and plot scales, there have been no experiments in which a high degree of experimental control has been incorporated at a catchment scale. The results of this project should allow water companies and land managers to predict the effects of specific management processes on water colour in the reservoir and at the treatment works.

There is an increasing convergence in the interests of bodies such as water companies and land managers. Where previously the needs of water companies and land managers may have differed, in areas for example of burning and stocking densities, the need to raise soil

moisture levels throughout upland areas is now accepted almost unanimously. This is now leading to a much greater degree of cooperation amongst land managers and a more proactive response on the part of water supply companies with regard to their catchment areas.

ACKNOWLEDGEMENTS

Severn Trent Water plc has provided funding for this project and The National Trust and English Nature have provided material support. The authors are grateful to all the people who have been involved in fieldwork, including Tom Davies, Tessa Levens, Garry Rushworth and John Corr.

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Managing our aquatic environment in the 21st century: wetlands and water quality

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ABSTRACT

In contrast to the past, contemporary European society places high value on wetlands for their biodiversity and important recreational and aesthetic qualities. In the UK, considerable resources are being expended in attempts to restore "near-natural" wetland ecosystems. Water chemistry is a key factor in determining the composition and maintenance of different wetland habitats and both acute and chronic pollution can undermine restoration efforts. Conversely, a pragmatic reason for wetland creation is as a low-cost solution to the treatment of polluted runoff from both agricultural and industrial sources. In this paper both aspects of wetlands, as natural systems threatened by human impacts on water quality and as tools to be used for water treatment, are reviewed. Results are presented from UK research into: i) the nutrient dynamics of a wet grassland; ii) the seasonal changes and trends in water quality within a recreated reed bed and iii) the use of a constructed wetland to treat polluted runoff. The findings illustrate that knowledge of the linkages in physio-chemical and biological functioning is a pre-requisite for creation, restoration and management of all wetland types.

INTRODUCTION

Wetlands have historically been viewed as unproductive wastelands and sources of disease. During the 17th century major drainage schemes converted thousands of square kilometres of British wetlands into what is today some of the most productive farmland in the country (Cook & Williamson, 1999). Throughout the twentieth century, as demand for food increased and technology advanced, the rate of conversion from wetland to agricultural land increased. As a result, throughout the UK there is now a range of modified and degraded wetland landscapes (Acreman & José, 2000).

In recent years, there has been a radical change in attitude towards wetlands. They are now widely perceived to be of value as highly productive ecosystems that maintain environmental quality and support biodiversity, and for their recreational and aesthetic qualities (Maltby, 1986). For these reasons, in the UK and throughout Europe, considerable efforts are being made to protect and restore wetland ecosystems and in places recreate wetland habitats (Klötzli & Grootjans, 2001).

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Although drainage, resulting in dewatering, remains the greatest hazard to wetland habitats in the UK, human-induced changes in hydrochemistry are a significant threat. If chemical loadings exceed the physiological tolerances of key microbial and plant species, environmental degradation is likely to occur. This is especially true where wetlands are small "remnants" of previously larger systems. In a recent study of 48 UK wetlands, 15% were threatened by water quality changes, largely as a consequence of agricultural activities on adjacent land (Gilman, 1994). Furthermore, hydro-chemical fluxes are often a key constraint to the restoration of degraded wetlands (Klötzli & Grootjans, 2001).

Despite this, wetlands can be very effective at treating polluted water by sedimentation, filtration (through the litter layer), physical and chemical immobilisation, microbial interactions and uptake by vegetation (Kadlec & Knight, 1996). Their capabilities are highly ephemeral though, due to the dynamic growth and metabolic processes within them (Prior & Johnes, 2002). It is often difficult to harness the benefits of natural wetlands as they are sometimes remote, of an inappropriate size for the required task, or have an unsuitable layout and plant assemblage. As a result, there are relatively few 'natural' treatment wetlands and the majority are specially constructed for particular situations.

Constructed wetlands provide effective, relatively low cost, treatment for many types of wastewater pollution, effectively removing or converting large quantities of organic and inorganic pollutants from point sources and non-point sources. These include urban highway runoff, agricultural pesticide runoff, petroleum industry wastewater, domestic sewage, paper mill waste, and dairy farm waste (Shutes *et al.*, 1999)

This paper summarises the findings of three scientific studies conducted in the UK to gain insight into the hydro-chemical dynamics of both natural and man-made wetlands.

NUTRIENT DYNAMICS OF WET GRASSLANDS: TADHAM MOOR CASE STUDY

The development of agri-environment schemes in the European Union toward the end of the 20th century arose from a growing concern as to the extent and quality of unimproved farmed habitats. By 1985, the extent of unimproved pasture in England and Wales had been reduced to just 3% of that present fifty years earlier (Fuller, 1987). Measures were therefore introduced to protect the extent of these communities and to take steps to restore them where they had become degraded through drainage and fertiliser application.

Preliminary research found a dearth of rigorous experimental evidence on the effects of fertiliser use on the botanical composition of grasslands of biodiversity importance, especially lowland wet grasslands. A study was therefore setup to answer the three following questions:

- What changes occurred in sward composition in response to different application rates of nitrogen fertiliser?
- How long would it take for swards to revert to a semi-natural composition following the end of fertiliser usage?
- What would be the interaction between cessation of fertiliser use and the raising of water-levels in grasslands to achieve benefits for breeding and wintering birds?

The greater part of this study took place at Tadham Moor in Somerset, where exceptionally rich meadows occurred on low-lying peaty soils. The grassland at Tadham had been mown for hay

for many years, with stock grazing the aftermath, but there was no history of fertiliser use. The experiment was conducted in three phases:

- ***Phase I*** (1986-89): Sub-plots were identified and treated with either 0, 25, 50, 100 or 200 kg N ha^{-1} $year^{-1}$. Untreated control plots were also identified. In each location, sward composition was monitored to ascertain whether there was a compromise level that could be applied to species-rich grassland without reducing floristic diversity.
- ***Phase II*** (1990-93): Fertiliser application ceased at half of the sub-plots and the change in sward composition was monitored. Monitoring continued in the sub-plots with ongoing fertiliser application.
- ***Phase III*** (1994-2002): All fertiliser application ceased, the water levels were raised in the ditches surrounding some of the sub-plots, and sward changes were monitored.

Fertilisation/Eutrophication

All levels of nitrogen fertilisation resulted in increased plant production but reduced plant diversity, and caused a significant increase in grass domination. When the site was studied three years after fertilisation had been stopped, the species that had become dominant under N treatment (such as the competitive grasses) remained dominant, and the species that had declined, such as the forbs, sedges, rushes and mosses, had not regained their previous abundance. It was calculated that it would take nine years for the vegetation that received the highest fertilisation loadings to return to pre-fertilisation status (Mountford *et al.*, 1996). As the length of time over which fertilisation occurred increased, three effects were observed: greater domination by grasses, greater suppression of sedges and forbs, and greater decline of species diversity.

Raised water levels

Raised water levels resulted in an initial reduction of vegetation coverage and plant species diversity in comparison with fields where water levels remained unaltered. Subsequently there was some recovery in species diversity but the raised water level fields remained less diverse than the fields with lower water levels. Higher water levels resulted in an increase in wetland grasses and sedges. Micro-topographic variations in elevation and drainage, and therefore the likelihood of plants experiencing either drought or aeration stress, were important and resulted in clear plant community variation. Such variation is one reason for the high species richness at Tadham Moor (Silvertown *et al.*, 1999).

Interaction between nutrients and raised water levels

Species whose abundance had been reduced by the nitrogen fertilisation increased to a greater extent in the raised water level fields than in the fields where the water level had remained unchanged e.g: *Carex spp* (sedges) and *Filipendula ulmaria* (meadowsweet). Where high N treatments had ceased, competitive-ruderal species (Grime *et al*., 1988: e.g. *Ranunculus repens* and *Agrostis stolonifera)* were able to recover faster in the raised water level fields than in the fields with lower water levels but did not recover to the same abundance in those fields that had never received fertiliser treatments.

The ability of such competitor-ruderal species to invade communities after perturbation is a significant issue in restoration ecology (Mountford *et al.*, 2001). Such species may come to dominate the sward where previously there were not so abundant and this can complicate the processes of restoration.

WETLAND RECREATION: HAM WALL CASE STUDY

In parts of the Somerset Levels and Moors, long-term removal of peat for horticulture has left a wasteland of disturbed peat and clay, which is unsuitable for the immediate re-establishment of the 'natural' acid bog habitat that existed in the past. One option for the restoration of these peat pits is to establish reed beds, similar to the ancient reed swamps that covered the area in prehistoric times.

The Royal Society for the Protection of Birds (RSPB) is presently investigating the efficacy of restoring reed beds (*Phragmites australis*), as suitable habitat for both rare and endangered wetland bird species (RSPB, 1998). At Ham Wall, located within the catchment of the River Brue, 140 ha of old peat workings are being converted to reed beds in a phased approach. Work commenced at the site in 1994 and since then hydro-chemical monitoring has been conducted to assist the management and restoration efforts by providing information on the temporal variation in water quality.

The research has shown that the wetland is a highly complex and dynamic system with considerable spatial and temporal variation in chemical and biological processes (McCartney *et al*., 2003). As submerged macrophytes and algae photosynthesise, dissolved oxygen and dissolved carbon dioxide levels increase and decrease respectively. In addition to this, the decomposition of organic matter by microbes consumes dissolved oxygen and releases carbon dioxide. As a consequence, there is a large diurnal variation in dissolved carbon dioxide and therefore in pH (Figure 1).

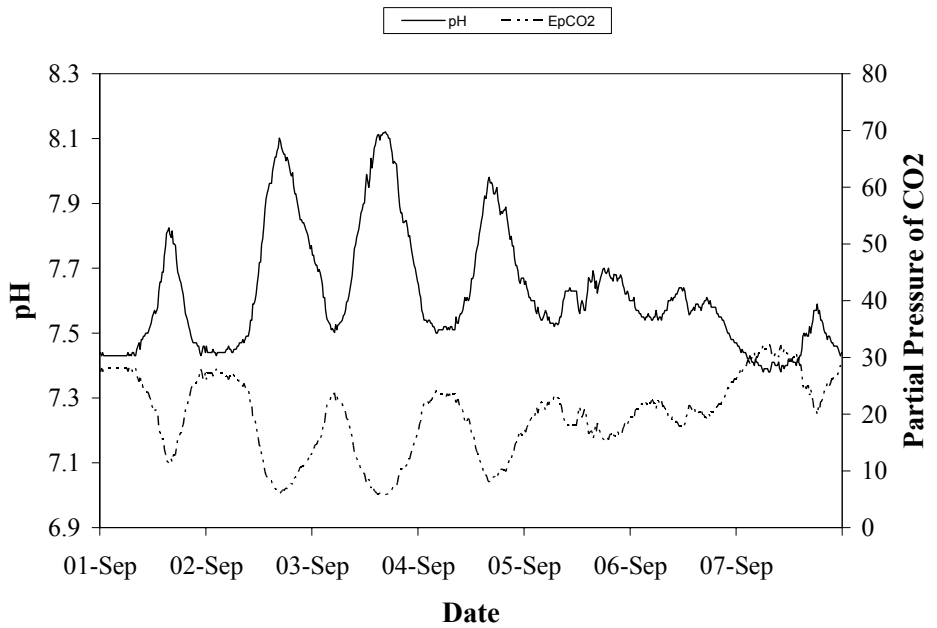


Figure 1 Example of the diurnal variation in dissolved carbon dioxide and pH. Period shown is 1st to 8th September 2000.

During the day, dissolved carbon dioxide is between 5 and 20 times atmospheric equilibrium and at night can be in excess of 40 times atmospheric equilibrium, an order of magnitude higher than observed in most unpolluted river systems (Neal *et al.*, 1998). The large diurnal variations in carbon dioxide pressure and dissolved oxygen are likely to be detrimental to some biota.

As the reed bed has matured, organic litter has built up due to both natural processes and management practices. This has resulted in an increase in respiration and consequently an upward trend in average carbon dioxide pressure. The increased respiration is also a likely cause of decreasing dissolved oxygen concentrations (Figure 2). It is anticipated that if the decomposition of organic matter continues to increase in the future, the associated decline in dissolved oxygen levels may make the reed bed an unsuitable habitat for some invertebrate and fish species.

Although the water of the reed beds is eutrophic, the nutrient levels (NO_3-N typically < 0.6 mg l^{-1} and P average 0.26 mgl^{-1}) are generally lower than those in a nearby canal polluted with agricultural runoff (NO_3-N average 5.8 mgl^{-1} and P average 1.7 mgl^{-1}) despite the fact that agricultural drainage is a major source of water in the wetland. However, since 1998, phosphate concentrations in the reed beds have increased significantly and in 1999 have at times exceeded 1 mg-Pl^{-1}. The increasing phosphate concentrations are attributed, at least in part, to avian inputs arising from the huge number of starlings (in excess of one million birds) that roost in the wetland during winter months.

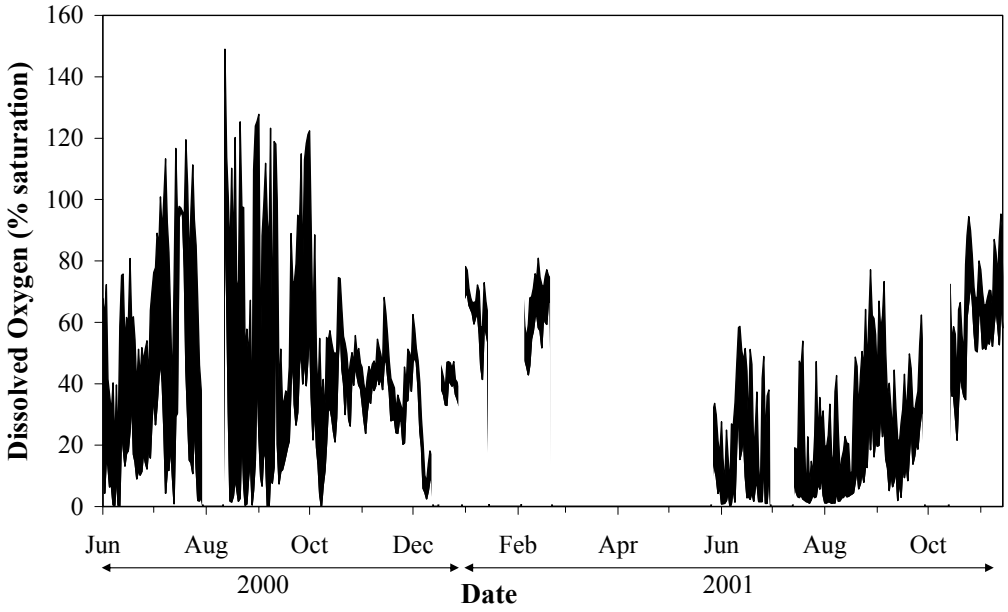


Figure 2 **Dissolved oxygen concentration over the sample period. The area between daily maximum and daily minimum is shaded black. Missing data are largely due to site restrictions during the foot and mouth disease outbreak.**

USE OF CONSTRUCTED WETLANDS TO IMPROVE WATER QUALITY: HEATHROW AIRPORT CASE STUDY

For pollution control, constructed wetlands offer several advantages over the use of natural wetlands such as the ability to determine the substrate and vegetation types, the wetland size and location. Three types of system are used for water quality treatment: surface flow, sub-surface flow, and rafted systems (Figure 3 shows the surface flow and sub-surface flow types). Surface flow wetlands emulate the hydrological regime of natural wetlands, where water flows over the soil surface, planted with wetland plants such as *Typha latifolia, Typha angustifolia, Phragmites australis, Schenoplectus* and *Iris pseudacorus*. These emergent macrophytes are extensively utilised because of their widespread availability and high level of pollution tolerance (Revitt *et al.*, 1997). In sub-surface flow systems, water flows through a gravel based substrate planted with macrophytes. This increases the contact time between the wastewater and the roots and rhizomes, enhancing treatment performance per unit area, and reducing the likelihood of exposure of humans and wildlife to pollutants. Rafted systems usually contain a buoyant plastic lattice into which the macrophytes are planted where their roots bind together to enhance the mat, and grow down into the water to create a 'treatment curtain'.

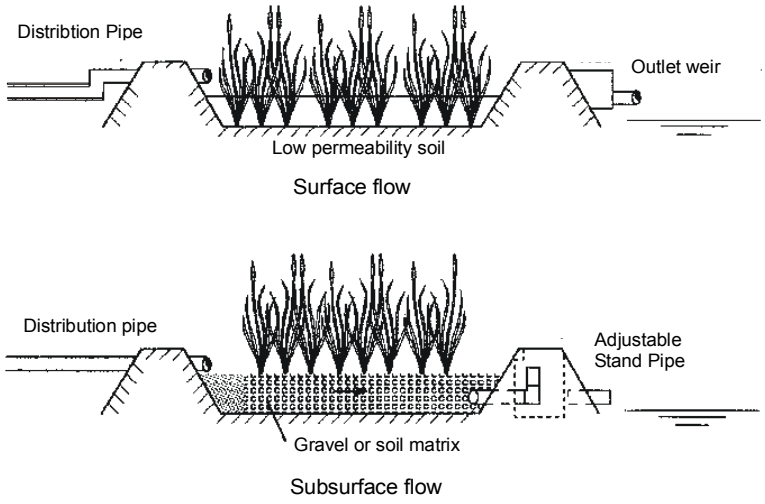


Figure 3 Schematic of the surface flow (top) and sub-surface flow (bottom) constructed reed bed systems. (Adapted from Kadlec & Knight, 1996)

London Heathrow (LHR) airport is one of the world's busiest international airports. In winter months glycol based de-icers and anti-icers are applied to runway areas and aircraft to prevent the formation and build-up of ice. As a result, the airport runoff can be heavily contaminated. Concentrations in runoff can be highly variable with levels as high as 4,500 mg/l for BOD and 6,000 mg/l for total glycols, which can create serious problems of water quality within the sites receiving runoff both in terms of ecosystem stress from low oxygen levels and direct toxicity to aquatic organisms (Ellis *et al.*, 1997).

In the early 1990s LHR set out to investigate the most appropriate technology for dealing with the glycol runoff problem. Techniques included reverse osmosis, carbon filtration, modified cellulose filtration, UV catalytic oxidation and constructed wetlands. The investigation on the use of constructed wetlands began with a two year trial using pilot plant reed bed systems based on the three designs described above (Revitt *et al.*, 1997). The results demonstrated that reed bed technology could effectively treat BOD and glycol contaminated airport runoff.

Over the two-year monitoring period, the average BOD removal efficiencies were 30.9% for the surface flow bed and 32.9% for the sub-surface flow bed. The average glycol removal efficiencies determined from three dosing experiments were 54.2% for the surface flow bed and 78.3% for the sub-surface flow bed. The better performance of the latter is explained by the higher surface area of the gravel substrate in conjunction with the greater amount of contact time between wastewater and rhizomes. Data from the pilot plant trial were used to design a scaled-up constructed wetland capable of 90% treatment efficiency for the BOD and glycol load from the entire airport. However, given the local constraints of land availability, the required surface area was too large, and therefore a hybrid system was designed using a modified existing receiving/aeration reservoir with its output treated to the required standard by passage through constructed wetlands (Figure 4). The wetlands were fronted by rafted systems, which simply rose with the flow and thus coped better with hydraulic shocks, while the main component of the system was constructed of sub-surface flow reed beds giving rise to high level microbiological treatment. The facility was constructed two kilometers south of the airport, minimizing the risk of aircraft striking birds attracted to the wetland (Revitt *et al.*, 2001).



Figure 4 Partial view of the full size constructed wetland at Heathrow Airport

The pilot plant results showed that the reed beds could treat BOD and glycol effectively and provided essential data for the design and costing of a scaled-up treatment system. When compared with the costs of other treatment technologies, the constructed wetland proved most effective in terms of treatment performance, cost and sustainability. Other technologies faced problems or were too expensive. Reed beds also demonstrated that as well as dealing with glycol, they could effectively treat hydrocarbons and metals coming from aircraft tyres and engine exhausts (Revitt *et al.*, 2001).

SUMMARY

In the UK the perceived value of wetlands to society has changed over time. They are now recognised as highly productive elements of the landscape, critical for the maintenance of biodiversity and landscape heterogeneity. It is also understood that they can make a positive

contribution to water resources through protection of water quality. However, the management of wetlands for conservation, alongside agriculture and other development, requires a thorough understanding of the complex interrelationships between hydrology, hydrochemistry and biological processes. Similar understanding is required to optimize the design of constructed wetlands used to treat polluted water. More research is needed to conserve and maximize the benefits to be gained from diverse wetland types under the impact of escalating human demands and climate change.

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WISKI - software for water quantity and water quality data management

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ABSTRACT

The recent adoption of the European Water Framework Directive (WFD) has led to the reforming of European water laws. The WFD places central emphasis on river basin management as well as water resource management into the future, particularly qualitative aspects such as the preservation of a good ecological and chemical condition of our inland waters. Furthermore there is also a need to consider the complicated relationships between water quantity and water quality. Possible measures to achieve the protection and improvement of water quality are justified by the need for sustainable use of our water resources. Because of the need to quantify and identify the effects of water resources management – and considering the wide range of relevant influencing variables - the application of environmental software are required.

This paper describes the WISKI-WQM software package for water quantity and water quality data management. WISKI is a Windows based client/server system based on relational databases, designed in cooperation with water agency authorities, engineers and hydrologists. It combines modern standards of data management with advanced tools to collect, edit, store and present time series data to intranet, internet and GIS users. WISKI consists of three parts; a remote data acquisition server, a central database server for storage and management, and a windows based hydrological workbench. The system allows the automatic and effortless flow of time series data from the measuring site into the database. The data is then reviewed using powerful graphical and tabular interface tools. Accepted water quantity and water quality data are then ready for further analysis either using internal or external statistical and modelling packages.

INTRODUCTION

The Water Framework Directive 2000/60/EC (WFD) is the most significant piece of European water legislation to be produced for over twenty years. The WFD will rationalise and update existing water legislation and introduce an integrated and co-ordinated approach to water management in Europe based on the concept of river basin planning. The consistent area-oriented and river-basin based approach and the objective of achieving 'good status' of all waters within 16 years will have a considerable impact on water management.

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To achieve the 'good status' the WFD requires:

- water protection of all waters;
- good status within 16 years;
- water management based on river basins;
- combined approach; and
- sustainable water resources management.

These requirements are defined particularly by ecological criteria. This will lead to the following tasks:

1. monitoring of all water bodies;
2. extended water quality data management;
3. combined water quality and quantity data analysis;
4. assessment of current water body status; and
5. presentation of results to the public.

To fulfil these tasks within the required period software solutions are required. For water quantity management the Water management Information System KISTERS WISKI is an established software package which is already applied in many present member states of the European community. To meet the requirements of the WFD in terms of water quality the WISKI package was enhanced by the water quality module WQM. This module was developed in cooperation with the Hessian environment agency within the scope of an R&D project.

In the following first the WISKI package is briefly described. Subsequently key features of the water quality module are pointed out.

WATER QUANTITY MANAGEMENT USING WISKI

Key features

WISKI is a client/server system based on relational databases, which was designed under close collaboration with water agencies and hydrologist. WISKI is made up of three parts, a data acquisition server for remote control and telemetry, a central database server and a graphical user interface termed the hydrological workbench.

The hydrologist uses the hydrological workbench (Figure 1) to perform all of the necessary work to transform raw data to final data. This process can consist of quality control of new data using the graphical and tabular visualization tools for error correction, gap filling, flow calculation, statistics and also summary reports for documentation in hydrological yearbooks or publication to the Internet. The workbench also has the ability to act as an alert system for flood management. Alarms can be configured for monitoring tasks such as the calculation of storage for retention basin management or early warning systems.

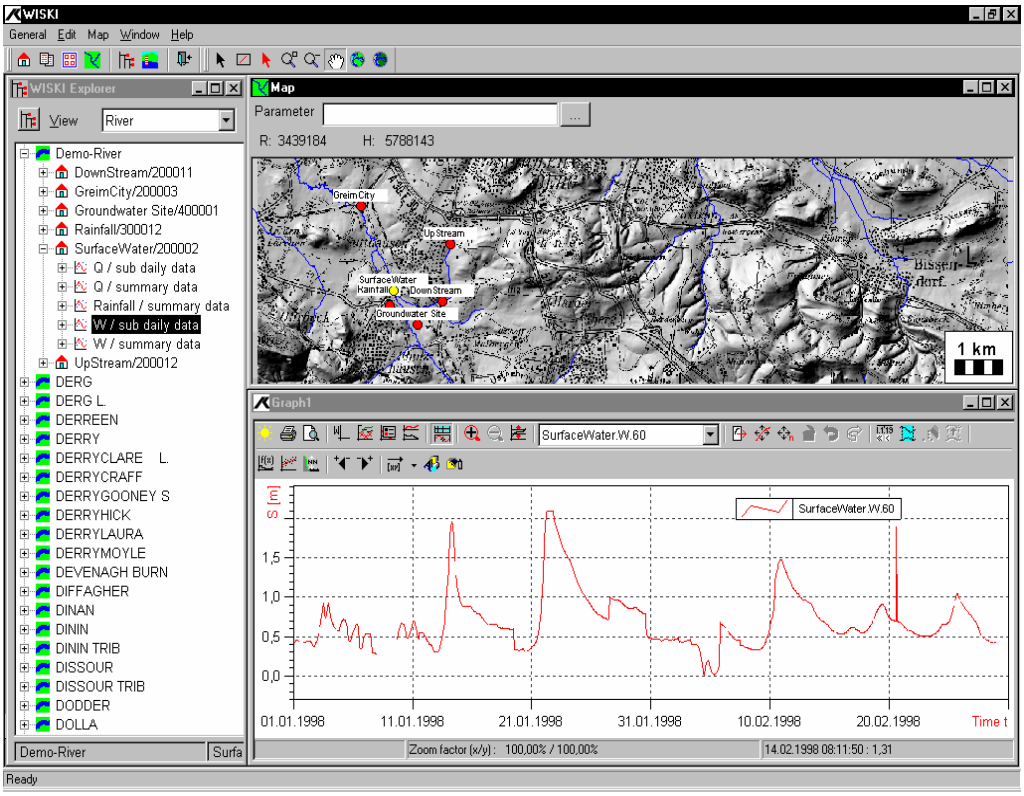


Figure 1 The hydrological workbench of WISKI, easy access to the data

Different tools are integrated into the hydrological workbench to support the hydrologist's work. There are configurable tabular and graphical data editors, automated procedures for standard tasks such as quality control checks, filling of data gaps, primary value calculations or transformation of measured water levels to discharge rates. One of the most important time savings tools of the workbench is the rating curve editor. This tool supports the hydrologist by assisting in the rapid construction of rating curves from discharge measurements as well as monitoring valid rating curve periods. Another feature of WISKI is the GIS interface. This module allows the user to display a map of the river basin as well as geographical information, which allows the user a graphical access to station information and associated time series data directly from the database server.

Data Monitoring

Monitoring water quality data refers on the one hand to water quality samples and on the other hand to automatic and continuously recorded data. For sample monitoring the new water quality module can be applied. This module is described in the next section. Monitoring of equidistant data is already applicable in the general WISKI package. Figure 2 depicts the system for remote call from water quality stations.

SODA is a communication server, which can perform data retrieval via telephone network and GSM network. It is the ideal partner for WISKI to add telemetry functionality to your hydrometric information system. It is based on industry standards. The equipment is vendor neutral (e.g. Ott, SEBA, Läufer, Rittmeyer, DDP, Sommer). Fetched data is directly imported into the WISKI Database and ties up to the workflow of plausibility checks and data correction provided by WISKI.

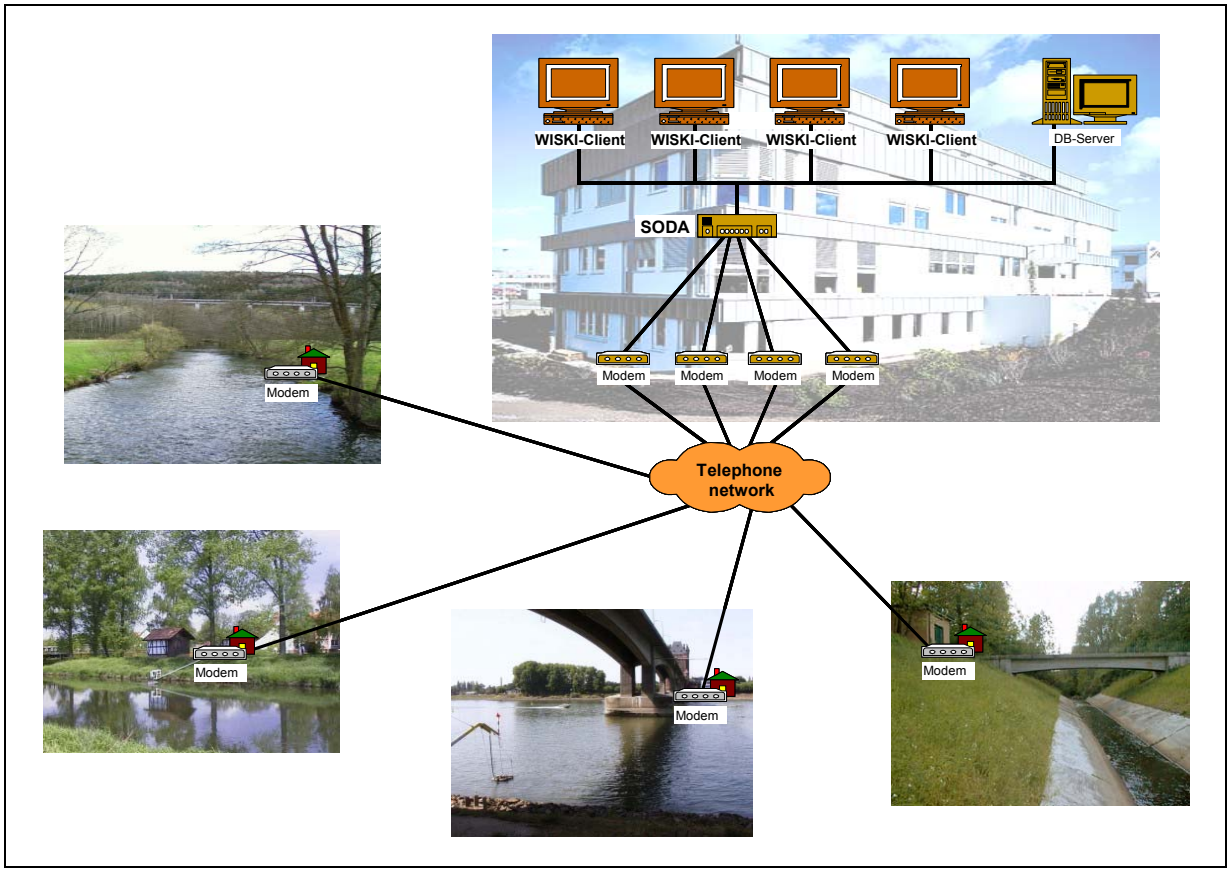


Figure 2 Water quality monitoring using the WISKI telemetry system

WATER QUALITY MODULE WQM

Water quality data management

Between water quantity data and water quality data an essential distinction exists. Whereas in water quantity generally only a few parameters are measured water quality comprises hundreds of different parameters. But in contrast to water quantity the measurement of these parameters is generally very infrequent. The measurements of water quality parameters are grouped in samplings and samplings again are grouped in measurement programs. Furthermore, to simplify access to water quality data different list types were introduced. These lists will enable the user to group different parameters in individually defined lists, e.g. a nutrient list.

To consider these aspects new attributes had to be integrated, which means the database structure had to be enhanced considerably. These new attributes comprise both general descriptions of samplings, the water quality parameters themselves, water quality values but also relationships between water quality sites, measuring programs, parameter lists and samplings.

The different types of data structure, which had to be integrated into the WISKI system will be clarified in the following by a description of new WISKI-explorer modes.

The first new explorer mode is the view mode "measuring programs". Combined with the new edit mode "sampling" access to WQ samples can be achieved. Different levels for quick search of samplings are available. These levels are:

Level 1 "measuring programs";
Level 2 "water quality sites";
Level 3 "sampled years"; and
Level 4 "samplings".

Figure 3 depicts these different levels.

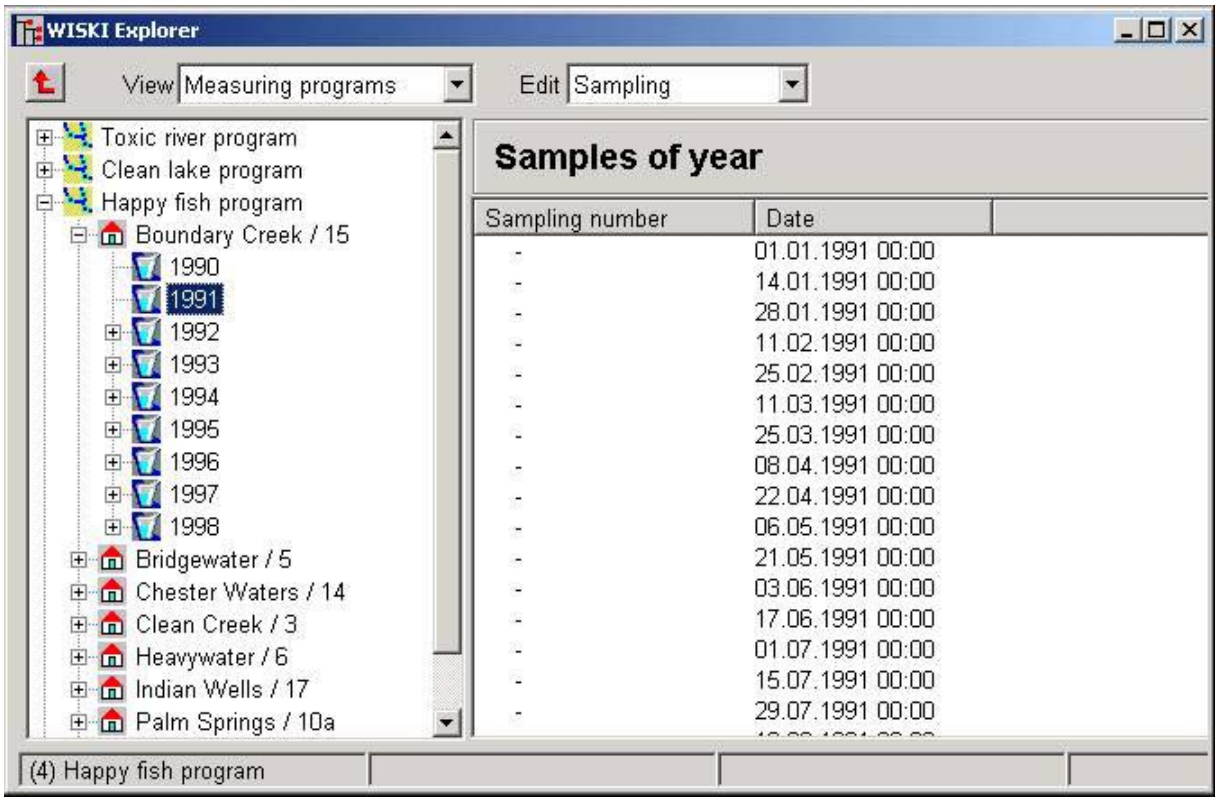


Figure 3 WISKI-Explorer – access to WQ samples

For each sampled parameter, which contains at least one single value, a time series is created. To get access to these water quality time series the edit mode "sample parameter" is available. This mode can be combined with different view modes, e.g. "station types". Different levels of this mode are:

Level 1 "Water quality sites";
Level 2 "Measuring programs";
Level 3 "WQ Parameter lists";
Level 4 "WQ Parameter"; and
Level 5 "WQ time series".

The following figure depicts these different levels.

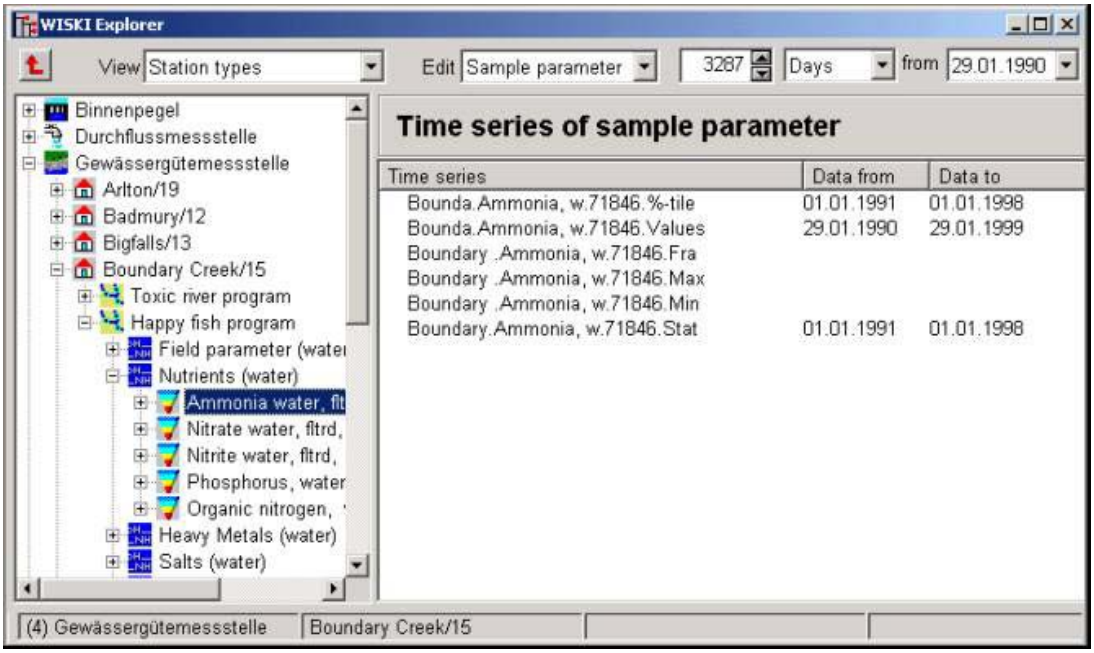


Figure 4 WISKI-Explorer – access to WQ time series

WQ data import

Import of water quality data, means sampling data, manual import as well as automatic import is available.

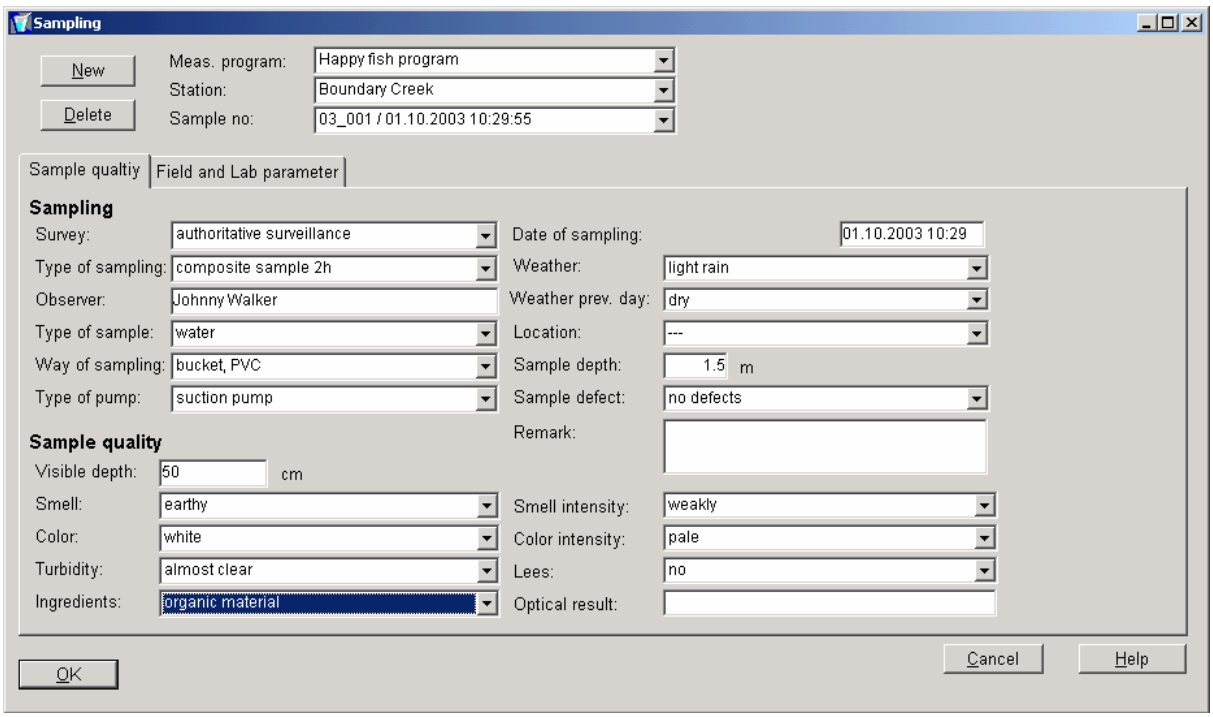


Figure 5 Sampling dialog for viewing, editing and importing WQ data - Sample quality

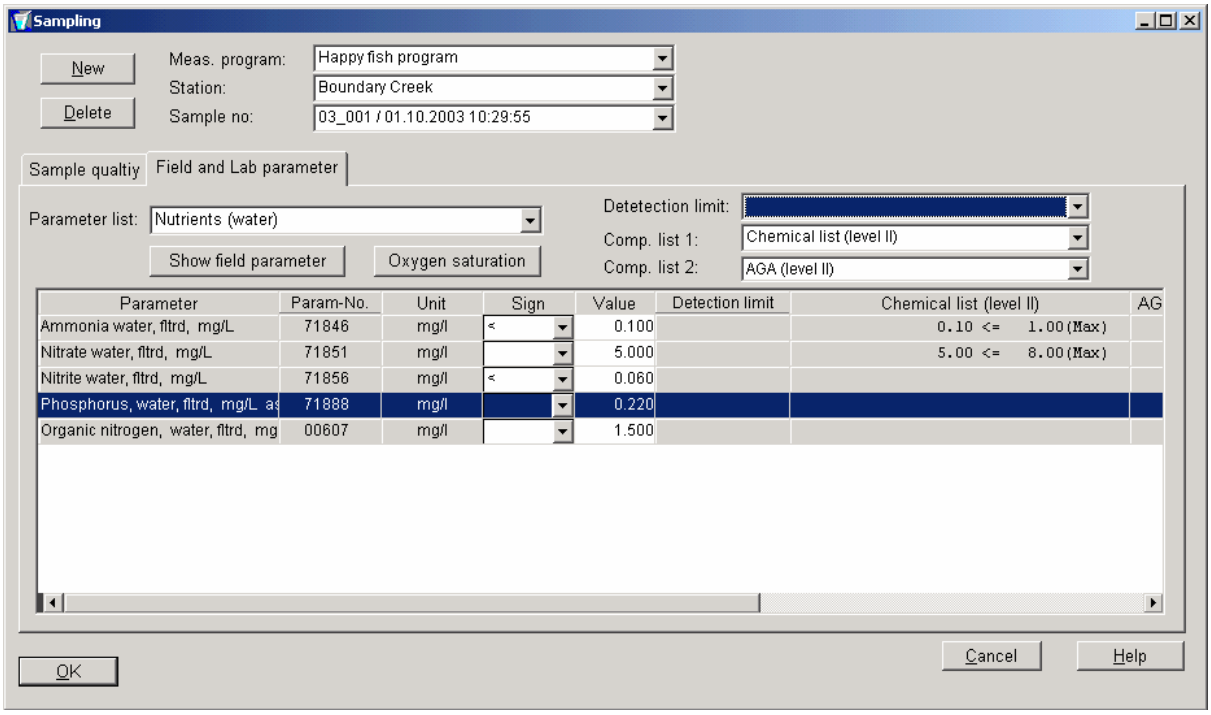


Figure 6 Sampling dialog for viewing, editing and importing WQ data – Field and laboratory parameters

Most of the WQ data are managed in Excel and are available in csv or simple ASCII formats. In order to simplify the import procedure and to make the import process as flexible as possible, KISTERS developed a configurable import interface. The configurable importer is fully integrated into the import framework of the WISKI core system.

To interface with laboratory information management systems (LIMS) WISKI offers an XML Interface to baseline communication and data exchange to standard data transfer.

Data analysis

After the import of data automatic plausibility check routines are carried out either automatically by the WISKI Calculation Server or when data is retrieved by table, graph, export or report. The most commonly used plausibility checks are Rate of change, Check of completeness, Boundary check, Distance in time and various calibration checks. Individual plausibility checks can be programmed by using the WISKI KBasic programming language.

The WISKI Core System - especially the time series engine - provides a standard set of summary data. For a water quality station the annual summary statistics are calculated as described in Table 1.

Table 1 Derived time series

Derived time series	Equations or Methods
Statistics	Mean: $\bar{x} = \frac{1}{n} \cdot \sum_{i=1}^{n} x_i$ Standard deviation: $s = \sqrt{\frac{1}{n} \cdot \sum_{i=1}^{n} (x_i - \bar{x})^2}$
Extrema	Min: $\mathrm{Min} = \min(x_i)$ Max: $\mathrm{Max} = \max(x_i)$
Percentiles	Methods: 1. HAZEN-method 2. SAS-Def. 4 3. SAS-Def. 5 4. EXCEL-method 5. VDI-Standard 2450
Loads	$F^{E}_{Jahr} = \frac{364 \cdot 86400}{1000 \cdot 1000} \cdot \frac{1}{n} \sum_{i=1}^{n} c(t_i) \cdot Q(t_i)$

The Water Quality Module is fully integrated into the WISKI Environment. All analysis tools of the WISKI Core System are available to perform data analysis, by eyeball control or using analysis tools such as regression or cross correlation.

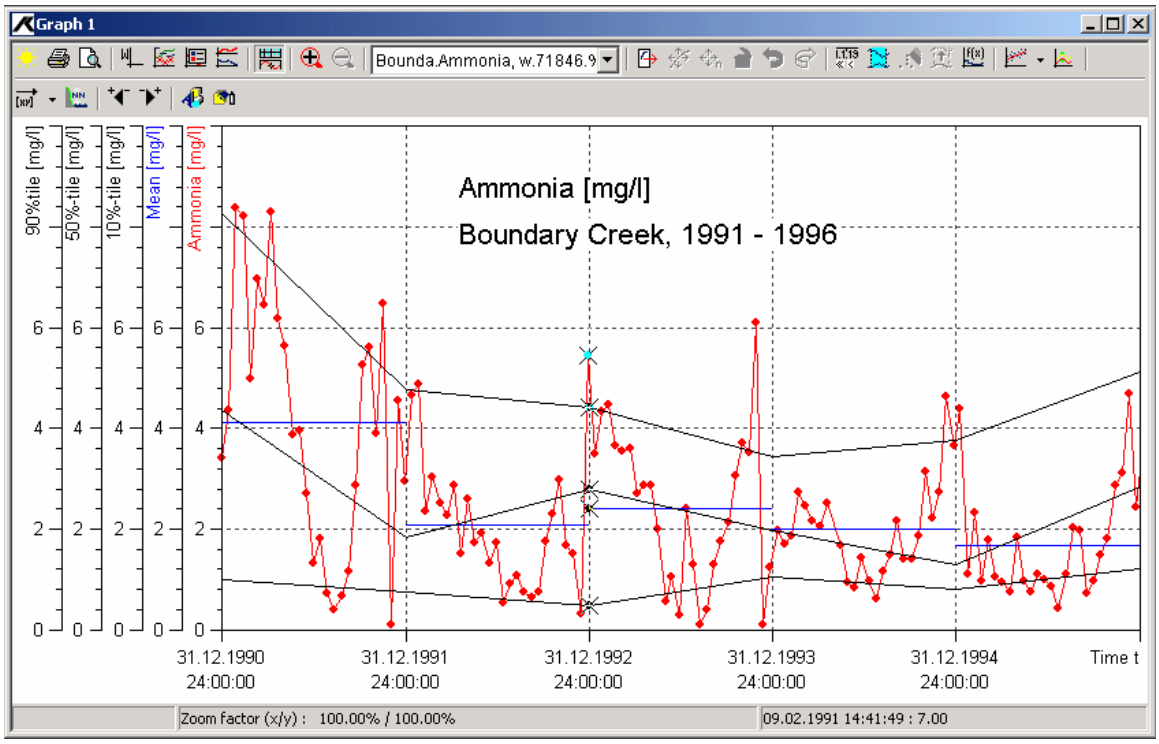


Figure 7 WISKI-graph – WQ time series

Presentation

GIS systems like ArcGIS focus on managing spatial information for various purposes. Common GIS datasets have a temporal validity of months, years, and several decades related to their adequate precision scale (topography, soil, geology). Time-series data traditionally arise at point locations (e.g. river gauge) and apply for various temporal resolutions (seconds to decades), parameters, and summarization techniques.

Both the systems are specialized to their purposes and their data worlds are very different. Thus the connection of ArcGIS and WISKI is established in a loose coupling. The coupling covers two areas: the links between the data worlds itself and the connection between both the specialist software. The point of contact between both the data worlds is actually the point of data measurement. WISKI uses this location as reference to the spatial data model (ArcHydro or others, raw data sets) to implicate time series data in a spatial context. The actual software coupling depends on the combination of topics and the procedures. For topics such as hydrology, meteorology, groundwater or water quality the ArcGIS Extension offers tools for data administration, validation and preparation of data for determining areal rainfall, and the visualization of quantitative and quality data in specialist's charts and graphs.

The WISKI ArcView Extension will be widened after February 2004 with the inclusion of a functionality to communicate with the WISKI Water Quality Module. With GIS technology and WISKI data management capabilities River Quality maps can be developed and analysed.

Quality and discharge data are used to derive map data (Shapefile) with where information can be shown as width and colour. Data are included as point layer into the GIS's layer list where all specific functions apply to this layer. GIS methodology allows to visualize various symbol maps (pattern, colours, sizes, label mechanisms, ...) and line visualization such as Line representation is derived directly from point data. Stream geometries are generalised, buffered and finalized with next river geometries. For layout purposes diagrams and width charts are derived from current layers and mapscale and integrated into map layout.

The following map types will be provided: Symbol maps, diagram maps and quality discharge maps (band visualisation)

SUMMARY

WISKI provides hydrologists with the latest in hydrological data management software interfaced to standard client/server database technology. The hydrological workbench provides the user with all necessary tools for data validation, correction, evaluation, and publication, thereby increasing the efficiency of calculations and quality control checks and enabling the user to manage his measurement network more effectively.

Since the requirements of the WFD affect in particular water quality data management, WISKI was enhanced by the water quality module WQM. This module allows extended management, analysis, assessment and presentation of water quality data.

The WISKI-WQM package is a complete solution for decision support and the management of resources or events and therefore a useful tool to meet the requirements of the WFD.

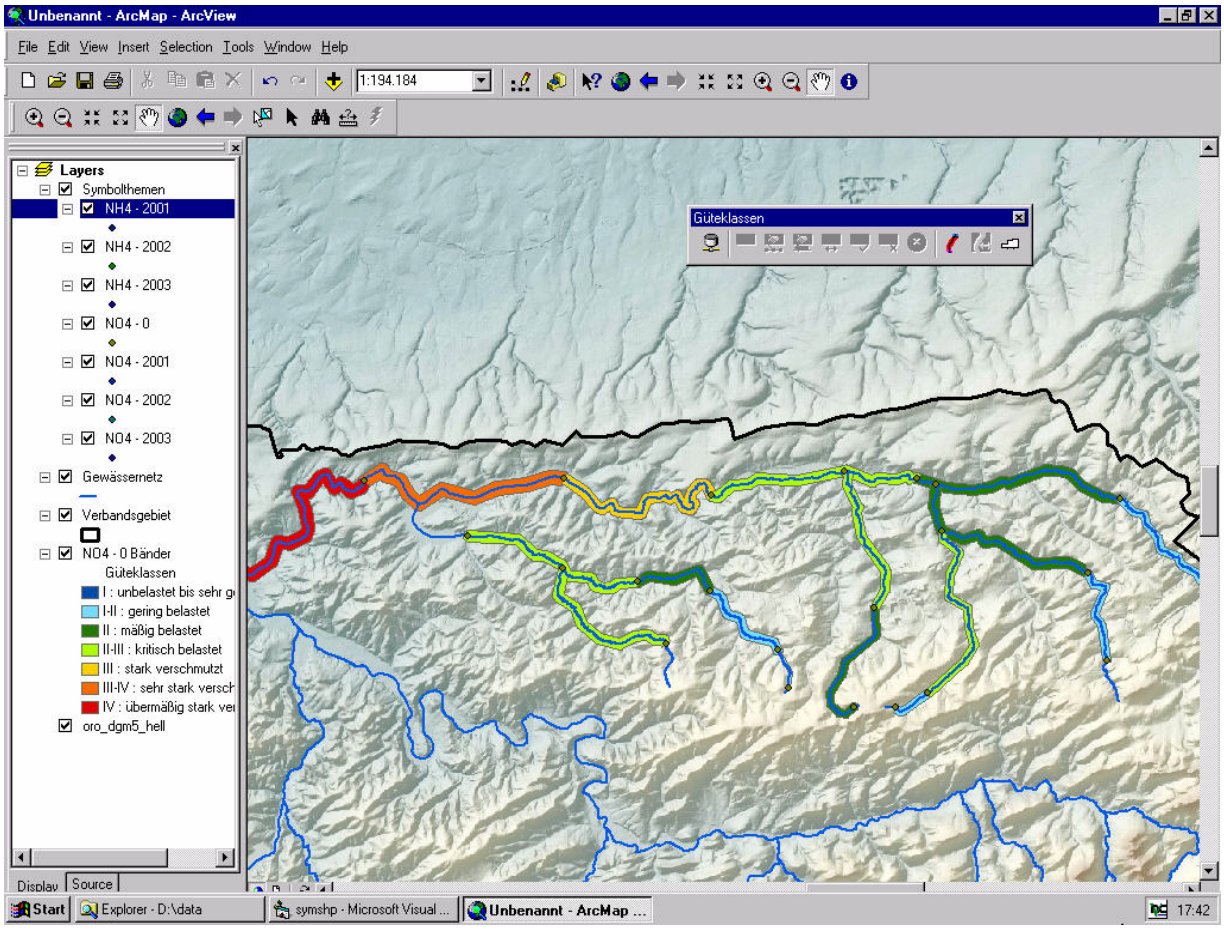


Figure 8 Derivation of water quality map

Toward innovative management of canal systems: determining fundamental properties governing water quality

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ABSTRACT

This research investigates current and future influences on the water quality of canal systems with the aim of designing innovative management strategies for their improvement. The implementation of the Water Framework Directive (WFD) will be particularly significant for canals as it is the first legislation to include artificial impoundments. The WFD requires that the quality of all water bodies is characterised and that ultimately they achieve "good ecological potential". For artificial water bodies such potential is as yet undefined.

The hydrology of canals is uniquely different from that of natural river systems. Influencing factors of particular importance include the extremely low flow rates (enabling flow directions to be reversed during rainfall events or high winds), the multiple overflow discharge points; the interaction with groundwater, and the regular channel dredging. These lead to unpredictable flow directions, long residence times for pollutants, a lack of rapid dispersion processes, and sudden mixing events of bed sediment into the water column. This research uses the Union Canal, central Scotland, as a case study for fundamental research of advection-dispersion processes. Detailed velocity profiles and tracer experiments have produced unusual results such as maximum velocity measurements near the bed of the canal as opposed to just beneath the surface. Research is also focused on understanding the mechanisms governing canal baseline water quality. Field data show low levels of dissolved oxygen (DO), coupled with ortho-phosphate concentrations up to 10 times greater than those normally found in rivers.

The results of this research will provide the first fundamental study of the processes governing water quality and solute transport in canals. Such information is vital for the design of water chemistry and biological monitoring programmes for assessing the ecological status of canals and for the development of appropriate management strategies for improving their water quality.

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INTRODUCTION AND BACKGROUND

The EU Water Framework Directive (2000/60/EC) is the first comprehensive piece of legislation to impact on water resource management at an international level. It extends, for the first time, environmental protection to artificial impoundments such as canals, dams, water abstractions and engineering works. The directive requires that canals reach "good ecological potential" and that no deterioration of water quality occurs. Deterioration in all water bodies will be assessed using benchmark biological and ecological parameters. However, whilst such benchmarks are available for rivers, canals are distinctly different both hydraulically and ecologically, and as yet such benchmark criteria for canals are completely undefined.

Historically, there has been an almost total lack of hydrological research carried out on canal networks. Hence, it is first useful to describe their main characteristics. In contrast to rivers, canals generally have an artificial trapezoidal shaped cross sectional bed profile, the contributing catchment area is narrow, longitudinal and may only contribute via one bank, and the lack of channel bedslope leads to an entirely different hydraulic regime. Canals are fed water through an artificial feeder system usually taken from a nearby river or reservoir; in addition, they also receive incoming water from direct rainfall, surface runoff and groundwater. Unlike river systems, canals do not lose water via channel flow. Instead, distributed losses occur via groundwater leakage, evapotranspiration and overflow weirs. Owing to their slow flowing nature, flow directional change can occur frequently in canals due to runoff from local rainfall events, activation of overflow weirs and wind effects at the canal surface.

As a consequence of the hydraulic nature of canals outlined above, the water quality and sediment accumulation also differs greatly from that of river systems. In general, the low flows and incoming land runoff produce a predominance of nutrient bound clay and silt sediments settling on the channel bed. There is a lack of rapid dispersion processes. Hence, residence times for pollutants are long and sudden mixing of sediments into the water column from navigational activity causes high turbidity and an influx of nutrients into the otherwise relatively still water column. The lack of flood events to remove sediment build up and clear channels worsens the sedimentation problems in canals producing a continuous rise in the channel bed profile. In general, banks become choked; vegetation encroaches into the main channel and sedimentation increases until channels become unnavigable (Haslam & Wolseley, 1981). Hence, dredging of channels to remove sediment loads from the channel bed every few years is essential (Hopkins & Brassely, 1992).

In recent years, legislation and canal restoration programmes have altered the management of canals, which now promotes the protection of canal water quality and their sustainable use. In the 2000 kilometres of UK network, canals not only have navigational, amenity and fishing interests but are also conservationally and economically important supporting over 1000 designated nature conservation sites in the UK alone (Sturt, 2001). They also provide over 265 billion litres of water for abstraction to industry annually (Lassiere, 2001). In order to comply with the requirements outlined in the WFD, however, further efforts are required to ensure canals meet the criteria of 'good ecological potential'. In particular, the collection of baseline hydraulic, water quality and ecological data is required to provide benchmarks for monitoring and assessment of current canal status. Once these data are available for canals, then the necessary improvements to management systems can be designed and put in place.

KEY RESEARCH OBJECTIVES

The key objectives of this research are to investigate the current influences controlling water quality within canal systems from both an ecological and hydrological perspective and to provide currently unavailable data to help meet the requirements of the WFD.

This research uses the Union Canal in central Scotland, UK, as a case study (Figure 1). Built in 1822, the Union Canal is a contour canal that uses aqueducts to maintain a constant bed height of 240 metres above ordinance datum and as a consequence, it has no locks. The canal fell into decline during the last century until by 1965; a number of sections were completely blocked. In April 2001 it was re-opened as part of the £78million funded Millennium Link project and since then, attention has focussed on its conservational value and sustainable use.



Figure 1 The Union Canal in Central Scotland

Figure 2 shows the Union Canal and its contributing catchment area for surface runoff. The canal is 31 miles long surrounded predominantly by agricultural land. The water source is a feeder from a reservoir in the south; the canal runs from Edinburgh to the Forth and Clyde canal in the west at Falkirk.

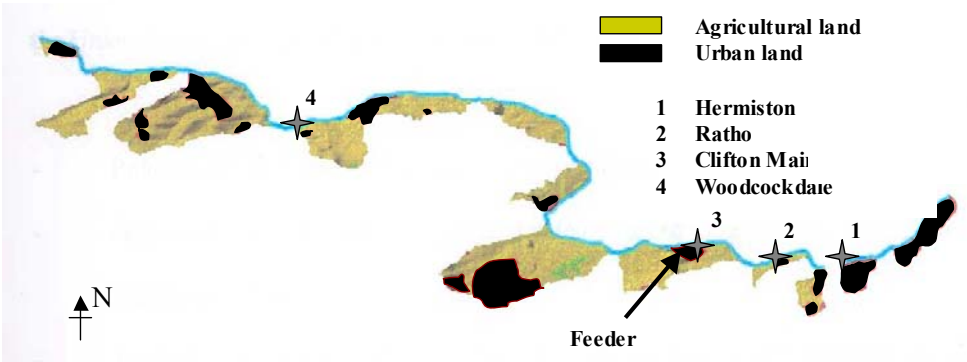


Figure 2 The Union Canal contributing catchment area, land use and sampling locations (Veldema, 2002)

WATER QUALITY OF THE UNION CANAL

Although no comprehensive water quality study of the Union Canal has been completed to date, water chemistry parameters such as dissolved oxygen (DO), phosphates and nitrates are collected every two months by the Scottish Environment Protection Agency (SEPA). Generally the water quality of the canal is good, although trends in the data suggest eutrophication problems in some stretches. The DO levels of the canal are frequently very low at several monitoring sites, with 105 samples below 5 mgl^{-1} during 1985 – 2001. Such low levels adversely affect the survival and functioning of biological communities, in particular 25 incidents with severely reduced oxygen concentrations below 2 mgl^{-1} resulted in extensive fish kills. Simultaneously, ortho-phosphate levels in the same stretch of the canal have reached up to 0.235 mgl^{-1}: 10 times what would normally be found in rivers (Chapman, 1996). Potentially, these high ortho-phosphate levels could be caused by phosphates bound on clay and silt sediments being washed in via run-off from surrounding agricultural fields. This in combination with the slow flow rates in the canal could be contributing to the low DO events. Questions that need addressing are: How long are the residence times of pollutants, such as sediment bound phosphate, in the canal? To what extent does the slow flowing nature of a canal inhibit mixing process and maintain 'hot-spots' of poor water quality? This paper investigates these issues by: presenting the preliminary results of detailed velocity profiling and tracer experiments; discussing the apparent flow regime and mixing processes derived from these experiments; and considering their influence on water quality control within the canal system.

DETERMINATION OF CANAL VELOCITY PROFILES

The 3D velocity distribution in a watercourse controls both the advective transport of pollutants and the mechanical mixing processes. Since velocity patterns in the canal are entirely unknown, it was necessary to collect some fundamental velocity data. To do this, detailed velocity profile measurements were taken at various cross sections within the Union Canal at a number of times throughout the year. The locations of the velocity profiles taken are marked on Figure 2. Figure 3, shows a typical profile of the canal bed, the steeply rising southern side of the bed is a consistent feature of all the locations visited. To determine a cross-sectional velocity profile at each location, velocity was measured at one metre chainage across the canal and at 10 cm intervals vertically using the Sensa RMX electromagnetic probe device as shown in Figure 4 (Aqua-data Services, 2003).

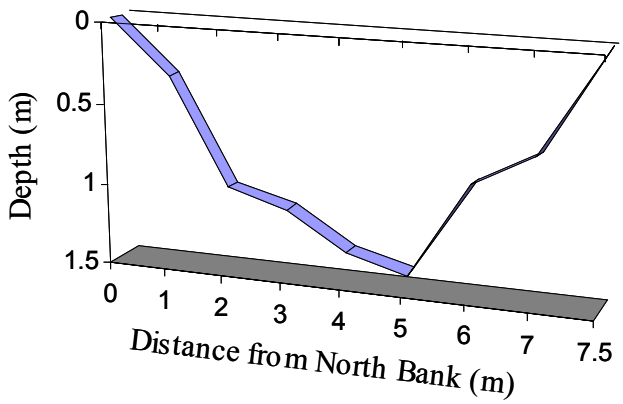


Figure 3 Cross-sectional velocity profile at Hermiston



Figure 4 Sensa RMX probes

Table 1 gives a summary of the main features derived from the velocity profiles taken at each location marked in Figure 2. The largest flow rate recorded is 0.198 m^3s^{-1} at Clifton Mains, corresponding to a mean recorded velocity of 30 $mm.s^{-1}$. This is indicative of the slow flow found in the canal compared to that of a river. Clifton Mains cross section shows the highest velocities due to its close proximity to the canal feeder which is the main source of incoming water. Table 1 shows that, as anticipated, the flow then diminishes with distance from the feeder. The Froude and Reynolds numbers calculated from these data for the Union Canal are 0.1 and 120,000 respectively, classifying the canal as sub-critical, turbulent flow.

Table 1 Summary data derived from the velocity profiles at the locations marked in Figure 2

Canal Site	Distance from feeder (km)	Max velocity (mm/s)	Min velocity (mm/s)	Average velocity (mm/s)	Cross sectional area (m^2)	Flow Rate (m^3/s)
Clifton Mains	1	75	0	30	6.7	0.198
Hermiston	7	51	0	27	6	0.163
Ratho	3	28	1	20	7.2	0.146
Woodcockdale	15	42	0	15	5.8	0.086

Within the cross sectional velocity profiles collected there is considerable spatial variability. In river channels, the velocity is normally expected to increase rapidly from the bed towards the free surface with an approximate logarithmic distribution (Graf, 1998). In the case of the canal, however, the highest velocities are found near the surface, as would be expected, and at the deepest part of the channel bed. For example, Figure 5 shows a velocity contour map from a cross section measured at the Hermiston site. Here, a clear increase in velocity

towards the channel bed is recorded in the deepest part of the section. This also is the case on cross sections taken at Woodcockdale, Ratho and Hermiston.

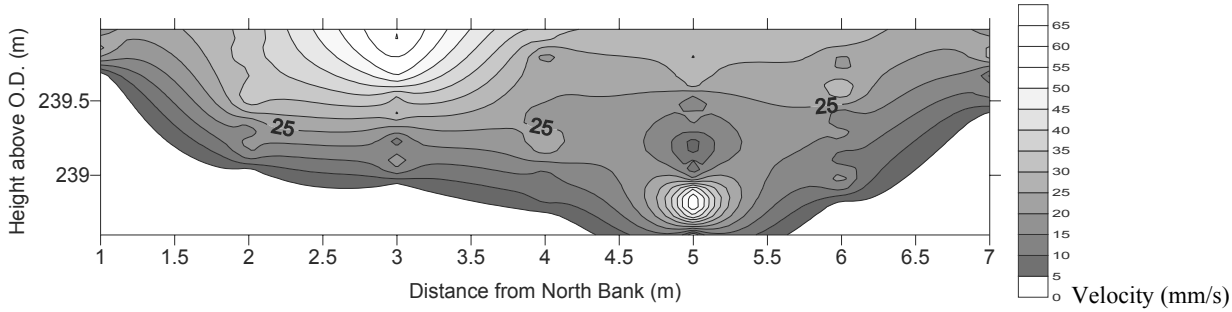


Figure 5 Cross sectional velocity profile at Hermiston

The contour map in Figure 5 also shows unstructured very low flows towards the southern canal bank. This side of the canal contains significant in-stream vegetation which may, in part, be due to the increased siltation from runoff from the adjacent farmland. Since the Union Canal is a contour canal, surface runoff only drains in from the topographically higher land to the South (see Figure 2). As discussed earlier, in all cross-sections, the shape of the canal bed is similar to that in Figure 3, with a steep rise toward the South bank. So, a combination of the unusual channel shape and the dense vegetation probably accounts for the variable low velocities that have been observed in all profiles at the southern bank.

PRELIMINARY TRACER EXPERIMENTS IN THE UNION CANAL

To determine the rate of mixing and hence pollutant residence times in the Union Canal, it is first necessary to investigate its mixing characteristics. Dye dispersion tests are widely used in hydrology for dilution gauging (Cobb & Bailey, 1965) and for dispersion experiments in rivers (Yotsukura *et al.*, 1970). By adding a chemical, usually a fluorescent dye of known concentration to the flow and measuring downstream where the chemical is completely mixed with stream water, one can measure time of travel and mixing characteristics. There are two basic techniques in tracer experiments: gulp injection and constant rate injection. The gulp injection method is normally the preferred option in rivers owing to its ease of execution. However, the aim in the canal was to undertake a qualitative examination of the nature of the cross-sectional mixing. This is most appropriately done by observing a steady-state plume formed downstream of a point or line source. Consequently, tracer test design for the canal focussed on a continuous injection test. Ideally in tracer testing: the tracer concentration must be at least 10 times the limit of detection in order to discriminate between signal and background interference; the type of dye used must be conservative; it must be injected at a constant rate with minimal physical impact on the water surface so as not to disturb the flow; and in the canal the test must not impede navigation. The tracer used in this study was the fluorescent dye rhodamine (WT) due to its conservative nature, it being moderately easy to handle, it being readily measurable to low concentrations (Rutherford, 1994; Smart & Laidlaw, 1977), and its low sorption tendency.



Figure 6 Constant rate injection method

The tracer test was designed such that the dye was injected at a slow constant rate using a Marriott vessel; where there is a constant pressure head at the outlet and hence a constant flow rate (Rutherford, 1994). The vessel was attached to a pipe and sprinkler system suspended from a bridge as illustrated in Figure 6. This sprinkler system was designed in the School's workshop to ensure as little disturbance as possible to the canal's mixing process. Due to the difficulties posed in finding the tracer once in the canal water, the sprinkler was designed to be one metre in length with sprinkle holes every cm. This ensured that a wide plume would develop. Sampling involved scooping surface water at various points downstream and within the canal cross-section. The field samples were analysed using a filter fluorometer: Turner designs Picofluor unit (Turner Designs, 2003) where excitation energy at 540 nm provided by a light source is passed through a filter and absorbed by the dye to be remitted with a longer wavelength as fluorescence. This light is measured and compared to the reference light path and produces a read out of discrete sample readings averaged over 5 second intervals enabling determination of dye concentration at sample points. The results of the initial tracer tests are outlined in Figure 7 below.

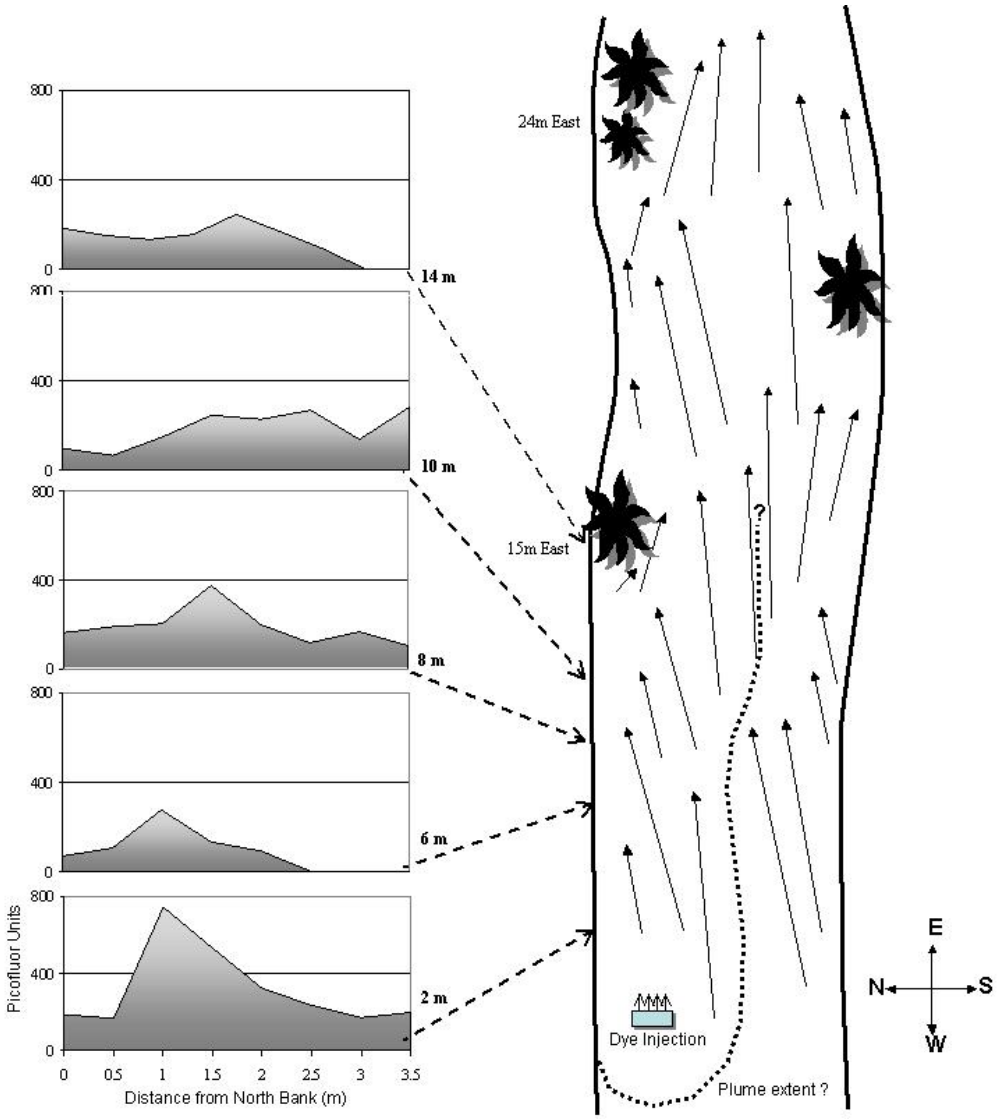


Figure 7 Results of tracer experiment with accompanying flow patterns

The right hand sketch in Figure 7 shows the flow patterns in the canal, as recorded during the tracer test, from visual inspection in the field. South-westerly winds were driving flow toward the northern canal bank. At 15 m downstream, in-stream vegetation and a small protrusion of the north bank were then redirecting the flow back towards the centre of the channel, producing an overall pattern of flow paths 'snaking' downstream. The location of the 1 m long injection sprinkler is marked on the figure. The rhodamine solution was injected for a duration of 100 minutes, with sampling beginning after 30 minutes to ensure a steady plume location. Samples were collected at approximately 2m intervals in the downstream direction and every 0.5 m across the width of the canal. No samples could be taken more than 3.5 m from the northern canal bank due to the restricted length of the sampling device. The results of this tracer test are shown on the left hand side of Figure 7. The meter reading taken by the picofluor unit is plotted for each downstream sampling location. The picofluor unit was calibrated such that a background reading for the canal water is approximately 10 and the injected dye concentration is 999. The results show that the peak dye concentration at 2 m distance is directly downstream of the point of injection. The plume then appears to widen with distance and the concentrations diminish. At 10 m and 14 m, the peak concentration has

migrated towards the centre of the channel, which is consistent with the observed flow pattern shown in the adjacent sketch.

A number of spot samples were also taken toward the end of the tracer test to investigate three effects: vertical mixing; upstream dispersion; and the presence of in-stream vegetation. First, to investigate vertical mixing, samples were collected at 0.5 metre intervals over depth using a calibrated rod and syringe, these showed the plume to be fully mixed in the vertical. Second, to investigate the extent of upstream dispersion, samples were collected upstream of the source, results showed significant upstream transport over a distance of at least a few metres. This suggests that diffusion processes may be important. Finally, samples were collected within and around in-stream vegetation on the northern bank. The samples collected within the vegetation contained much lower concentrations than the surrounding canal water, implying a lack of mixing in vegetated areas, hence, such areas may well form 'dead zones' to canal flow.

A significant number of difficulties were encountered in designing a tracer test for the Union Canal (or any other canal system). These difficulties were caused not only by the low flows, but also by the high turbidity of the canal water: the picofluor meter was extremely sensitive to the sediment content of the sample, making accurate measurements of rhodamine concentration difficult to obtain. Consequently, there still remain a number of unanswered questions regarding the shape and location of the tracer plume. Marked on the sketch in Figure 7 is what we believe to be the most likely plume extent in the canal. However, further field work is required to verify the dimensions of this plume and the testing procedure is currently being modified to achieve this.

CONCLUSIONS

The EU Water Framework Directive (2000/60/EC) will extend environmental protection for the first time to include artificial impoundments such as canals. The directive will require that, in the future, canals reach "good ecological potential" and that no deterioration of their water quality occurs. Canals are fundamentally different to rivers both hydraulically and ecologically. Hence, to be able to meet the first stage of the directive requirements of characterising existing water bodies by 2004, it is critical that researchers address the dearth of historical information and collect baseline data from which canal status can be evaluated. In addition, research is urgently required to determine the fundamental processes that control canal water quality, so that future management practices can be improved.

The research presented in this paper forms the first detailed data to be collected on the hydraulic factors affecting canal water quality in the UK. Here, the Union Canal has been selected as a case study, and velocity patterns and dispersion properties have been measured. Detailed surveys of the canal bed at a number of locations show a consistent profile of sediment accumulation toward the southern canal bank due to incoming surface runoff. Accompanying velocity profiles demonstrate some unusual features including relatively high velocities at the bed in the deepest part of the channel and extremely low velocities as the bed rises steeply toward the southern canal bank. These results differ substantially from river channels, where the velocity is normally expected to increase rapidly from the bed towards the free surface. Results from preliminary tracer testing in the Union Canal confirm the observations of low, spatially variable fluid velocities. In addition, the tracer tests have also indicated significant up-stream diffusion of the tracer from the injection point, full vertical

mixing and the existence of possible 'dead zones' caused by in-stream vegetation. Further research is ongoing to improve the tracer test design for canals to determine the full spatial extent of the plume.

Implementation of the WFD requires not only baseline data collection in canals, but also a much greater understanding of those processes that influence canal water quality. To achieve this understanding a predictive framework is required. Hence, the data presented here are now being combined with 3D numerical modelling studies of canal flow. The results of this research will be used to produce improved management strategies for canal systems. These will focus on interactions with groundwater, regulation of inflowing agricultural and urban runoff, and optimal design and location of inflowing drainage pipes.

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Impact of EU legislation on a UK catchment (The Wash): phosphorus control in catchments with high population density

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ABSTRACT

The Integrated Lake and Catchment Model (ILC) was used to assess the effectiveness of existing phosphorus control measures and possible future measures in achieving the objectives of the Water Framework Directive in Eastern England. Practical and economic constraints would make it extremely difficult to achieve target phosphorus concentrations in areas with high upstream urban density and low baseflow. In contrast, significant benefits are achievable in parts of the catchment with lower population density and high baseflow. Targeted investment in P removal would, therefore, be beneficial in maximising environmental benefits.

INTRODUCTION

In linking water quality objectives to ecological status, the Water Framework Directive provides an opportunity to optimise investment to meet environmental benefits and avoid the 'piece-meal' nature of environmental investment that has tended to occur in the past. However, efficient use of resources to improve the ecological status of rivers and lakes will only be possible if the strategic issues related to the targeting of investment are considered at an early stage.

The research presented in this paper, funded by the European Union Joint Research Centre, combines water quality modelling with economic analysis to assess the effectiveness of possible future environmental measures to achieve ecological targets for phosphorus in the rivers and coastal zone in the Wash catchment in Eastern England (European Union, 2003). For the purpose of this paper, two sites are compared with contrasting requirements to achieve phosphorus targets to illustrate issues of optimising investment in phosphorus control.

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THE CATCHMENT

Figure 1 shows the location of the two reference sites at Grafham Water and at Stoke Ferry on the River Wissey. The catchment of Grafham Water, a large pumped storage reservoir receiving water from the Bedford Ouse, includes several large towns including Milton Keynes and Bedford. In contrast, the urban population density upstream of Stoke Ferry is relatively low. Impermeable clays and mudstones dominate the catchment geology of the Bedford Ouse, whereas chalk dominates the Wissey catchment.

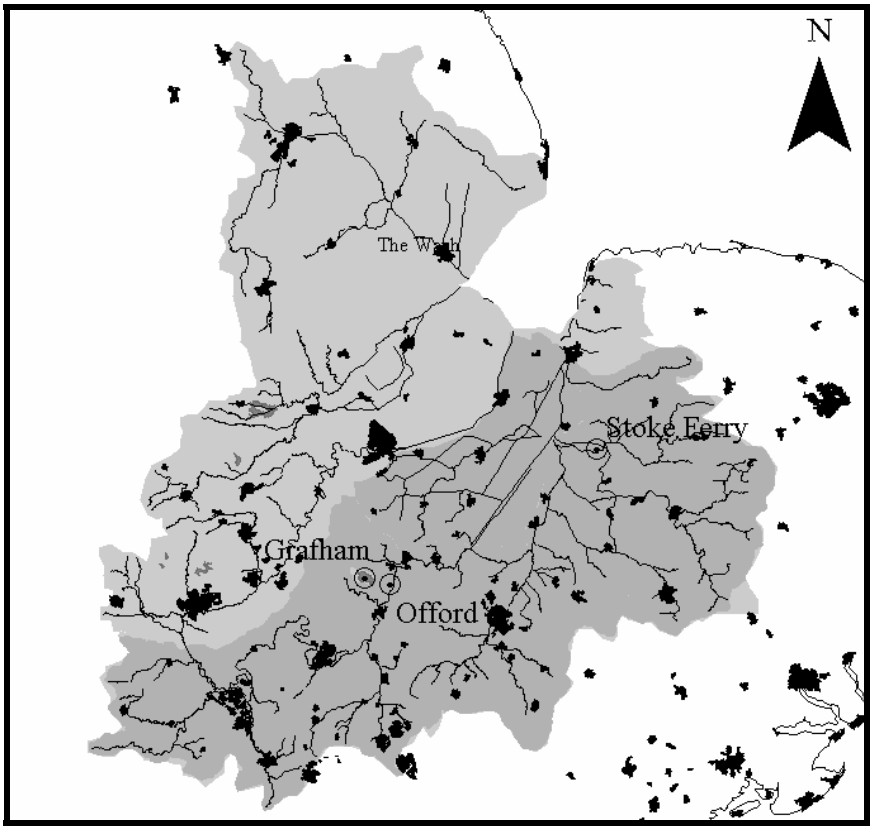


Figure 1 Location of reference sites

WATER QUALITY MODELLING

The *Integrated Lake and Catchment Model (ILC)* is a strategic management tool, incorporating the key hydrological and anthropogenic controls on nutrient transport in rivers and lakes (Daldorph *et al.,* 2001a,b; European Union, 2003). Phosphorus and nitrogen inputs to the soil are based on distributed land use data whilst releases from the soil are determined by a combination of export coefficients and temporally variable losses related to soil hydrology. Inputs of nutrients from point sources are calculated using geo-referenced effluent data. Dynamic inputs of nutrients are incorporated into a hydrologically based, multi-reach, model that also simulates abstractions and reservoirs/lakes. Routing of water velocity and flood wave velocity are modelled separately whilst solute dispersion is modelled using an aggregated dead zone approach. Complex operational and licensing rules are taken into account in the simulation of rivers and lakes to provide a tool to assess the influence of the environmental controls available to managers.

Diffuse phosphorus inputs, related to different land use were based on information from Defra (MAFF, 2000) and the Environment Agency (Environment Agency, 2000). Fertiliser application rates over the period were adjusted to take into account seasonal patterns and long-term trends using data from the Fertiliser Manufacturers Association (Defra, 2000). The models were calibrated for the period 1990-2000 and then validated for the period 1980-1989. The period 1990-2000 was chosen for calibration because information on land-use and operation of the wastewater treatment plants was more complete for this period. Figure 2 compares observed and simulated orthophosphate at Offord, Grafham Water and Stoke Ferry.

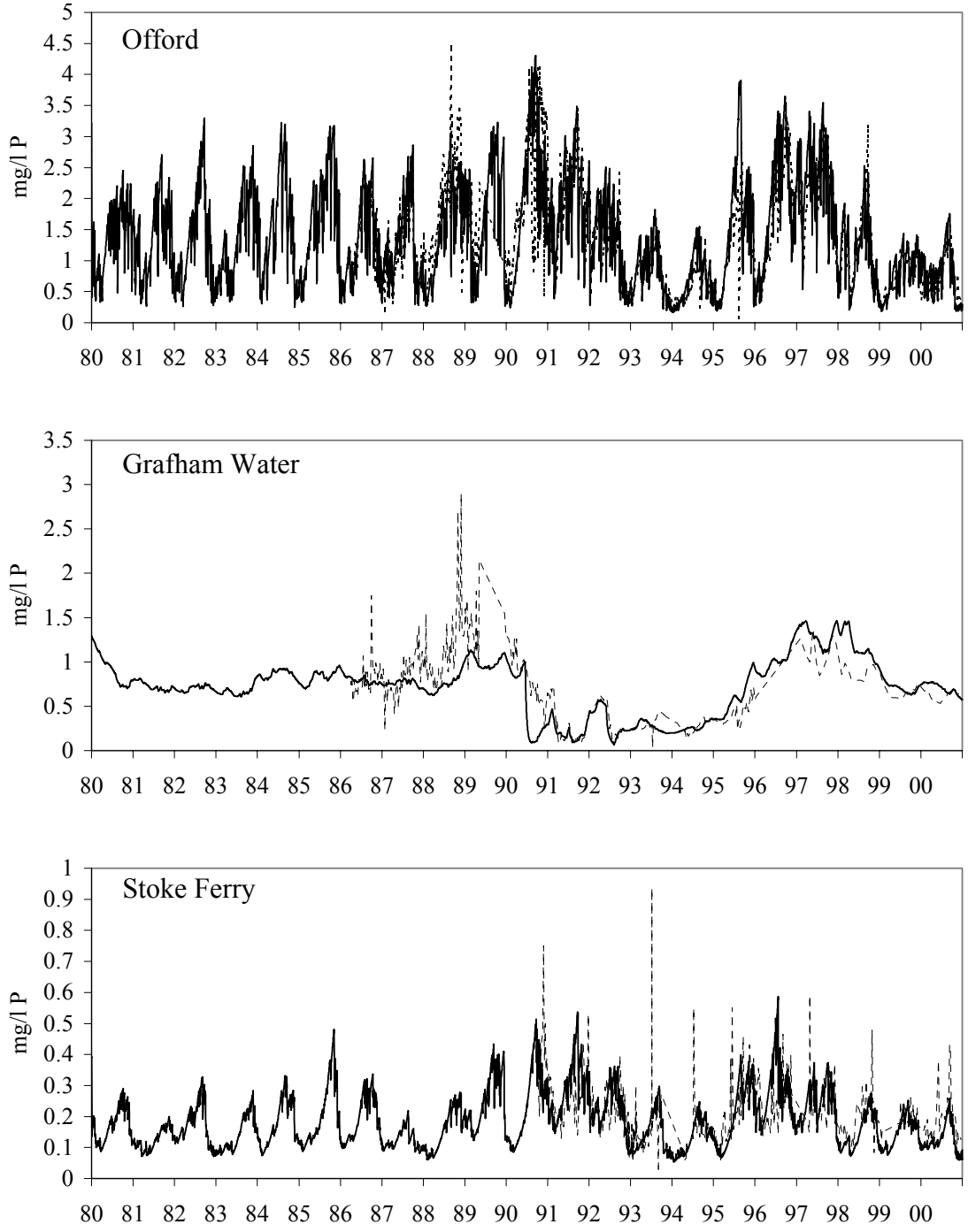


Figure 2 Comparison of observed and simulated orthophosphate concentrations at Offord, Grafham Water and Stoke Ferry

Table 1 lists simulated environmental measures. AMP measures are already in operation or have been planned, whereas Water Framework Directive Measures (WFD) are proposed as possible measures to achieve ecological targets in the rivers and coastal zone.

Table 1 Nutrient control measures analysed using the ILC model

Scenario	Type	Measures
1	None	No phosphorus or nitrogen control.
2	AMP2	Specified wastewater treatment works to meet Urban Wastewater Treatment Directive Targets (effluent target of < 2mg/l P at works > 10,000 p.e. and < 1mg/l P at works greater than 100,000 p.e.). Already operating).
3	AMP3	Further specified wastewater treatment works to meet Urban Wastewater Treatment Directive Targets (will be in operation at the beginning of 2004).
4	AMP4	Further specified wastewater treatment works to meet Urban Wastewater Treatment Directive Targets (will be in operation at the beginning of 2009).
5	WFD	All activated sludge sewage works greater than 10,000 p.e. to meet an effluent target phosphorus of 1mg/l P. All filter bed works greater than 10,000 p.e. to meet an effluent target phosphorus of 2 mg/l.
6	WFD	All activated sludge sewage works (of any size) to meet an effluent target phosphorus of 1 mg/l P and all filter works to meet a target of 2 mg/l P
7	WFD	All activated sludge sewage works to meet an effluent target phosphorus of 1 mg/l P and all filter works to meet a target of 2 mg/l P and all sewage works greater than 100,000 p.e. to meet an effluent target of 0.3 mg/l.
8	WFD	All activated sludge sewage works to meet an effluent target phosphorus of 1 mg/l P and all filter works to meet a target of 2 mg/l P and all sewage works greater than 100,000 p.e. to meet an effluent target of 0.3 mg/l plus diffuse phosphorus inputs reduced by 25%.

Figure 3 (lower graphs) shows the simulated impact of the Scenario 4 and Scenario 8, compared to no phosphorus control (Scenario 1). Scenario 4 (AMP4 measures) represents the impact of all programmed measures, whilst Scenario 8, represents the highest level of possible control measures that are likely to be available to the regional water company (Anglian Water) and UK regulators (Environment Agency, Defra) in the foreseeable future. Separate phosphorus targets were applied to activated sludge and filter beds because of technical limitations on the effectiveness of chemical P removal at these types of works. Very low effluent P concentrations cannot be achieved at filter works without major investment in tertiary treatment of re-engineering of the works (i.e. conversion to activated sludge).

INVESTMENT COSTS

The regional water company, Anglian Water, provided detailed projected and past expenditure for individual works, which was used to generate generic costs for phosphorus removal by chemical dosing with iron salts. Current control methodologies operated by Anglian Water would fail to achieve the stringent 0.3mg/l P target at the largest works

(Scenario 8) so more advanced treatment to remove suspended solids such as dyno-sand filtration or membrane treatment would be required. Approximate costs of diffuse pollution measures per hectare are presented in Table 2 (European Union, 2003; costs are indicative and require considerable further research and trial work).

Table 2 Estimated costs of diffuse nutrient control measures. (Based on 2002 Sterling to Euro currency rates)

Activity	Cost Euros/ha
Administration of scheme	0.8 (0.15)
Education	3 (0.3)
Costs to farmers (e.g. grant aid costs)	62.5 (94)
Total	*66.3 (94.5)*

Capital costs in parenthesis for establishing system

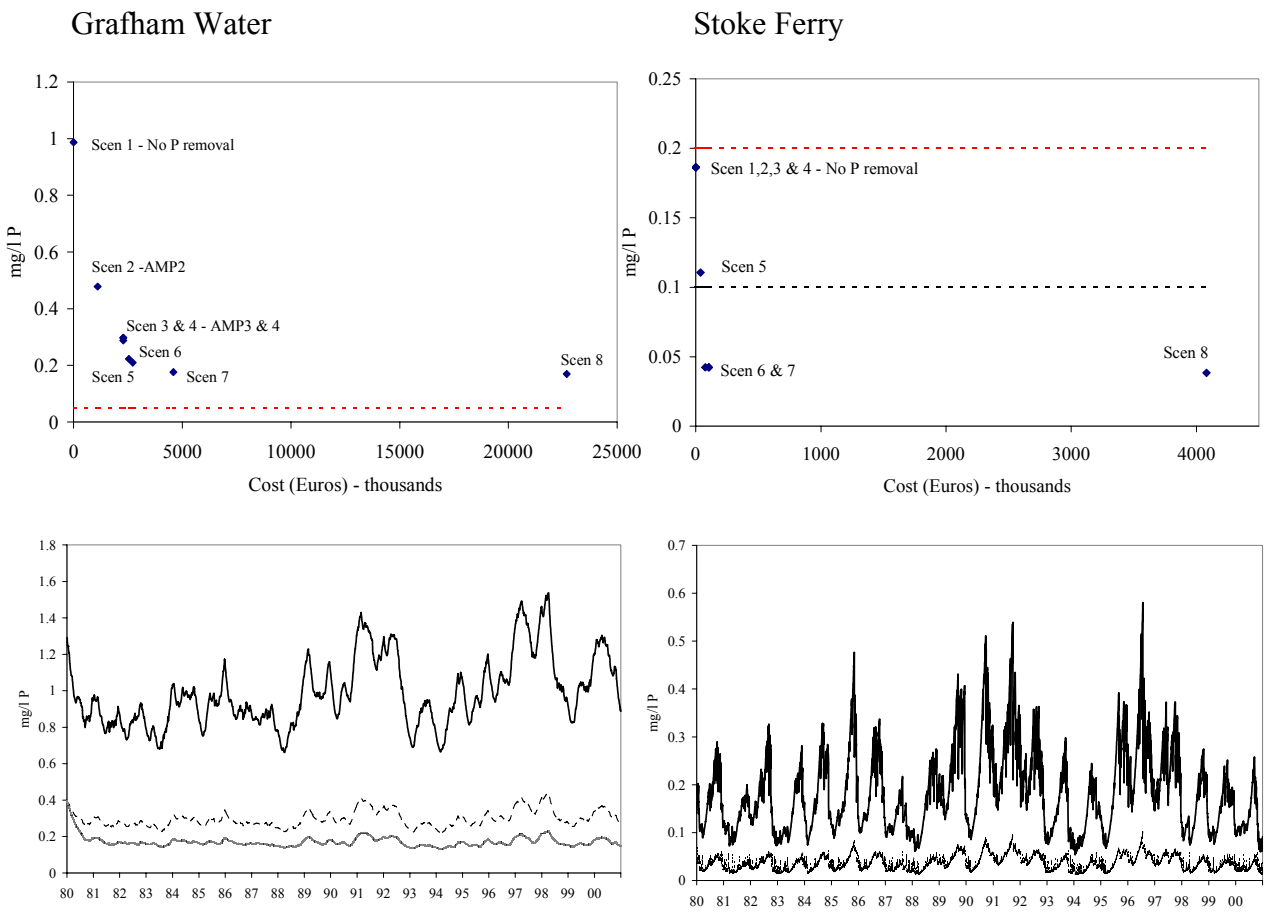


Figure 3 Comparison on impact of phosphorus control measures with estimated annual expenditure (lower time series graphs: upper line = No P removal, middle line = AMP4 measures, lower line = Scenario 8). AMP4 involves no P removal at Stoke Ferry.

COMPARISON OF COSTS AND BENEFITS

The cost effectiveness of the measures depends on whether they achieve reductions in environmental phosphorus concentrations below levels at which ecological benefits occur. Ecological targets for phosphorus in rivers have not yet been established in the UK with regard to implementation of the Water Framework Directive. However, methodologies are established to identify Sensitive Areas for the Urban Wastewater Directive (i.e. total reactive phosphorus concentrations of 0.1 mg/l in running waters and total phosphorus concentrations of 0.05 mg/l in still waters to identify areas sensitive areas). In the absence of more refined targets, two targets were considered in this study; a threshold target of 0.2 mg/l, above which no ecological benefit is likely to occur and a 0.1 mg/l P standard which represents a maximum concentration to achieve good ecological status. For Grafham Water, the lower targets for still waters are applied.

Total upstream costs for the various measures are compared to simulated phosphorus concentrations at Grafham Water and Stoke Ferry in Figures 3. Total costs combine capital and operational expenditure assuming a 30 year life and 2.5% inflation rate for the capital expenditure. The cost effectiveness of the measures with regard to achieving the targets varies greatly between the two sites. At Grafham Water, very large expenditure fails to achieve the target concentrations whereas these can be achieved at Stoke Ferry for much lower costs. Residual effluent phosphorus loads (after P removal) in the catchment upstream of Stoke Ferry are much lower than at Offord, reflecting the lower population density. Higher base flows also provide greater dilution of effluent in the summer, which also makes the target more readily achievable. Marginal benefits at both sites reduce as expenditure increases and phosphorus concentrations decline, reflecting the fact that residual phosphorus in effluent is more expensive to remove as concentrations decline and the fact that measures to control diffuse pollution are expensive.

DISCUSSION

The analysis presented in this paper for Offord and Brandon was repeated across the Wash catchment as part of the full research project (European Union, 2003), which showed that the achievability of phosphorus targets varied greatly across the region depending on population density and catchment hydrology. Even if phosphorus removal using current methodologies (chemical dosing) were undertaken at all sewage works, phosphorus concentrations would fail to meet the standard value in substantial areas of the catchment, due to the very high human population density and the fact that many of the smaller works have filter beds, making achievement of low effluent phosphorus concentrations difficult. Achieving river phosphorus targets would require rebuilding of many of the works, which would entail enormous costs (hundreds of millions of Euros). Clearly, in these circumstances, it might be argued that overriding economic issues make achieving the targets impractical in some parts of the catchment and that investment should be targeted at parts of the catchment where the greatest ecological benefits are likely to be achieved. Phosphorus removal at wastewater treatment works also has some negative environmental impacts associated with the energy costs linked with transporting the chemical to the works and electrical costs involved with chemical dosing. Failure to target phosphorus removal may, therefore, result in a negative overall environmental benefit.

Measures to reduce diffuse losses of phosphorus have little benefit where concentrations resulting from point sources are well above the threshold value. Agricultural measures in particular, might, therefore, be focused on parts of the catchment where control of point sources has been most effective or are likely to be effective in the next round of expenditure. However, this approach raises problems with equitability since the effect on farmers would vary depending on whether they lie within target areas, which will have little relationship with their own actions. Legislation to control diffuse pollution may also be more effective if implemented on a national basis rather than operating on a local scale. Clearly, control of diffuse pollution should be viewed within this wider context, including linkage between the Water Framework Directive and other EU policy measures (such as CAP, Regional structural funds etc).

The Water Framework Directive includes a requirement for economic analysis. Overriding economic issues can also be taken into account when planning environmental measures. However, there is a danger that measures will be implemented in a 'piece-meal' way and that economic issues will only be considered at a late stage. In the UK, OFFWAT, the water industry's economic regulator may be forced to take strategic decisions on economic issues at the end of a five year planning period rather than these being incorporated throughout the detailed planning phase.

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Water quality modelling in large river systems: model complexity, parameter sensitivity and predictive uncertainty

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ABSTRACT

A new dynamic model of water quality, Q^2, has been developed capable of simulating flow and water quality in large river systems using only routinely monitored (regulatory) data. This paper describes the application of Q^2 to the non-tidal River Thames in southern England with a focus on the simulation of DO since this may be regarded as a proxy for the overall health of a river. A generalised sensitivity analysis (GSA) method, using parameter value ranges derived from the literature, is used to identify the key parameters that control model behaviour and the GSA also provides a probabilistic procedure for model calibration and estimating predictive uncertainty. This study shows that, given data limitations, it would be more beneficial to obtain more frequent flow and quality boundary conditions than to obtain improved parameter estimates once reasonable values have been estimated. Furthermore, there is a need to attempt reasonable simulations of all important water quality parameters since attempts to simulate DO without acknowledgement of the simulations of other inter-related determinands will result in greater predictive uncertainty for DO.

INTRODUCTION

The Water Framework Directive requires that integrated water management is needed to counteract deterioration of water quality. This must take into account the acuteness of the problems and the trans-boundary nature, diverse utility, and wide importance of large river systems, and will generate a need for expert advice as to how such objectives might be achieved. This demands an understanding of the processes involved so that necessary improvements can be made without excessive cost, either financially or to other aspects of the quality and function of the system. To achieve this there are many models for studying environmental processes that provide tools for water utility managers and regulators alike. However many of these models are highly complex and have prohibitive data requirements as well as prohibitive financial costs. This is important because the complexity of any model must be consistent with the quality and quantity of data available, and typical time constraints suggest that a highly complex model may not be suitable for many problems (Cox, 2002). In addition, there may be little information regarding the sensitivity of a particular model to

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uncertainty in the (often many) parameters, and it is critical that some understanding is obtained regarding the predictive uncertainty of these models if they are to be used effectively.

Therefore, there is a need for a model to be made widely available, which is suitable for simulating large river systems, requires only data that are routinely available from regulatory organisations such as the Environment Agency, and that has been investigated in terms of parameter sensitivity and predictive uncertainty. In general, a model of intermediate complexity, such as QUASAR (Whitehead *et al.*, 1997), is well suited to simulating water quality in large river systems (Cox, 2003). Furthermore, the model also benefits from a structure that is relatively straightforward to calibrate and which can be used in a stochastic or probabilistic framework as well as for dynamic simulations (Cox, 2003). The utility of stochastic modelling is twofold: (a) for investigations of the errors associated with laboratory analysis and sampling, and the uncertainty associated with imprecise knowledge of model parameters and process mechanisms; (b) for the provision of model results in a form suitable for comparison with the guidelines and standards in the EC Directives and legislation for the UK. However, the QUASAR model itself is no longer available and there are known errors in the published model description, while many of the process descriptions could be improved (Cox, 2002). To this end, a new water quality model called Q^2 is being developed, which is based on QUASAR, but addresses those issues. Furthermore, the new model is being developed as a complete software package, capable of efficient data imports, running multiple simulations and producing report quality outputs.

MODEL DESCRIPTION

Since the Q^2 model is based on QUASAR, the structure is the same CSTR model that describes the dynamic transport and transformation of solutes in branched river systems using zero-dimensional ordinary, lumped parameter differential equations of mass conservation. The linked ordinary differential equations are simultaneously solved using a 4^{th} order Runge-Kutta numerical integration algorithm with a variable-step routine, and this ensures that the equations are solved accurately. The technique has proved to be fairly stable with few numerical problems. The equations used in Q^2 are described in detail in Cox (2002), but the major corrections and improvements from QUASAR that were made are summarised below:

1. The stream depth in each reach is estimated at every time step in Q^2.

2. The temperature model in QUASAR is conservative, but this approach has been found to be poor at simulating summer river temperatures. Thus, a simple heat balance is used in Q^2 with a surface solar heat flux term included to represent solar heating (Chapra, 1997).

3. The nitrate denitrification rate parameter in Q^2 explicitly relates the rate to the bed area available for the removal, since these processes typically occur in the sediments (Toms *et al*., 1975).

4. The BOD decay (oxidation) rate parameter in Q^2 uses empirically derived equations (Hydroscience, 1971), which describe the relationship between the decay rate and the stream depth for shallow rivers (< 2.4 m deep).

5. The net BOD sedimentation rate parameter in Q^2 is not constant, as in QUASAR, but is expressed as a net settling velocity (settling minus re-suspension), which therefore accounts for the stream depth in BOD removal.

6. The SOD rate parameter in Q^2 is related to the bed area available for this oxidation and the first-order relationship with the DO concentration is replaced by a 'Michaelis-Menten' half-saturation term using the method of Lam *et al.* (1984).

7. The method for estimating the DO saturation concentration is that of the APHA (1992) rather than that of Elmore & West (1961).

APPLICATION TO THE RIVER THAMES

The River Thames is relatively long (> 330 km) and changes character along its length. It was therefore desirable that the GSA be carried out on more than one reach. For pragmatic reasons, four reaches distributed along the length of the non-tidal River Thames were chosen in order to ensure that any conclusions drawn would not be valid only for one particular stretch of the river. The reaches chosen are highlighted in Figure 1 and flow between: (1) Castle Eaton and Hannington (close to the top of the system), (2) Oxford and Abingdon (in the upper catchment), (3) Whitchurch and Caversham (Reading) (in the lower catchment), and (4) Molesey and Teddington (at the bottom of the freshwater system).

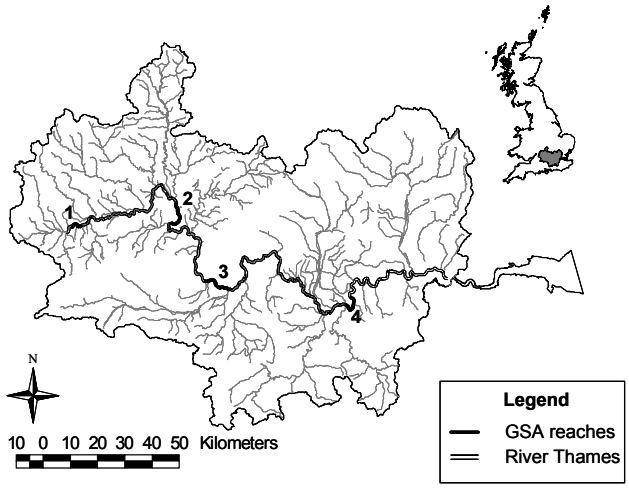


Figure 1 Schematic of the River Thames catchment highlighting the River Thames and the reaches chosen to undergo generalised sensitivity analyses.

The reaches are defined between 'influences' such as weirs, tributary confluences or the sites of major effluent discharges or abstractions, and the reaches chosen all have observed water-quality data available for comparison with the simulations. However, where two influences are close together, more than one may appear in a single reach. Influences are assumed to act at the top of each reach, but the reach itself is sub-divided into equal-sized uniform elements, or sub-reaches, so that no sub-reach will be longer than 3,000 m.

METHODOLOGY

With each of 10,000 Monte Carlo 'shots' a reach was simulated for the whole of 1998 using daily inputs. The input data were extracted from the routine hydrometry and water quality datasets collected by the EA and, because the data available were not as frequent as daily, linear interpolations were made between observations. This limits the temporal resolution of the model to that of the input data, but it is a pragmatic response to data scarcity and may be sufficient if the most important short-term influence is from diurnal meteorological changes, which are simulated by Q^2 through the use of a sub-daily 'internal' time-step.

The criteria for behaviour were based on the observed data and were defined by examining the observed data associated with the bottom of each reach, and identifying a range of values and period of occurrence for the minimum, mean, and maximum conditions. Although it is the DO concentrations that are of primary interest in this study, behaviour criteria were not defined only for the DO time series. This is because the DO model is dependent on flow, BOD, nitrate, ammonium and temperature. However, behaviour criteria were only set for those determinands with mass-balance equations affected by changes in the input parameters, i.e. those that would reflect variations due to the Monte Carlo method.

The ranges of input parameters for the Q^2 model were drawn from distributions derived from values in the literature. These distributions provide a means of incorporating prior knowledge gained from the literature while expressing the uncertainty in parameter values. For the purposes of this study, uniform probability distributions were chosen so that each parameter value had an equal chance of being chosen across the full-range, since it was important for the model to be tested across the whole of the parameter ranges. Thus, the result of each shot consists of the parameter vector itself and the behavioural outcome, i.e. whether the particular parameter vector gave rise to the behaviour (B) or not (N).

RESULTS

The GSA results indicate that seven of the eleven input parameters are sensitive (i.e. significant at the 95% level) in at least one of the reaches, and that the middle reaches are more sensitive than the reaches at either end. The parameters and corresponding (Kolmogorov-Smirnov) $d_{m,n}$ statistics that are significant above the 95% level for each reach were ranked in order of importance and it was clear that the critical parameters are generally those that control nitrate and BOD concentrations.

Differences in parameter sensitivity between reaches reflect different conditions in those reaches. In reach 1, the parameter describing the BOD contribution from dead algae (C_5) is not statistically significant, and this is most likely due to the relatively low algal populations in the uppermost reaches of the Thames. In reaches 2 and 3, it is the SOD parameter that is not significant, and by inference the processes that parameter represents. Furthermore, since the reaeration rate also shows the least separation or is not significant at all in these reaches, it may be inferred that is clear that in-stream DO processes in these middle reaches are generally less important than solute transport is in controlling the behaviour. At the tidal limit of the River Thames in reach 4, it is the BOD parameters (and by inference processes) that are not significant and so, at the bottom of the system, it would appear that transport of BOD is of greater importance to the BOD concentrations than other in-stream processes.

A composite of the four reaches is useful to make a system-wide comparison, of parameter sensitivity, and the mean $d_{m,n}$ statistics over the four reaches show that the most critical parameters overall are the denitrification and nitrification rate parameters, followed by the BOD parameters for settling and decay. Of the DO parameters, those relating to photosynthesis and respiration do not appear as being significant in any of the reaches, but the sediment oxygen demand and reaeration rate parameters were found to be significant above the 95% level for three and two of the reaches respectively and the reaeration rate is ranked fifth most significant overall.

The majority of the model parameters were found to be independent, but there were some co-variance and correlation elements with magnitudes greater than 0.1. The largest magnitude off-diagonal covariance of -0.31 is between the net BOD settling velocity (C_8) and the BOD decay rate parameter (C_2) in reach 1. A negative correlation indicates that interactions between that pair can allow a greater range of values for one if the other takes on a particular value. Thus, increasing the BOD decay parameter while decreasing the BOD settling velocity will still allow for the behaviour criteria are to be met. However, this correlation may be an artefact of the 'decoupling' of SOD from BOD settling in the model. Furthermore, most elements are less than 0.1 in magnitude and as a result it is unlikely that many insights would be obtained from closer inspection of the matrices that are not available from the univariate results (Hornberger & Spear, 1980; Spear & Hornberger, 1980).

To examine the effects of the behavioural criteria on the GSA and the model, the Monte Carlo simulations for each reach were repeated, but with the behavioural criteria now based on the DO criteria alone in order to determine the primary controls on the in-stream DO concentrations. Of the 10,000 simulations, the majority of parameter sets produced simulations that fit the DO behaviour criteria and, because of the low number of non-behaviours, the separations between the cumulative distribution functions are large, reflected by large values of the $d_{m,n}$ statistic. It was also found that, it was generally the non-behaviours that tended to be confined to one or both ends of the parameter range, while the parameter values producing the behaviours were spread more evenly throughout the range. The reaeration rate is the most critical parameter if only DO is considered, and there is a greater consistency between reaches with regard to the rank importance of parameters. The occurrence of non-behaviours arose with values of the reaeration rate at the lower end of the specified range. These low values corresponded to simulated DO concentrations that were too low in the winter and too high in the summer, because the stream was less able to reach levels close to the saturation concentration at any given time.

The parameter ranges used in the Monte Carlo simulations affect the number of behaviours achieved given a set number of model runs, as do the definitions of system behaviour. As such, the interaction of different parameter ranges and behavioural criteria will influence the predictive uncertainty. In Figure 2, only those parameter combinations producing behaviours were considered, but had all of the parameter combinations (those producing behaviours and non-behaviours) been utilised, it is clear that the uncertainty in the output simulations would have been far greater. Figure 3 illustrates the results of such simulations, where the uncertainty bounds are considerably larger, particularly for the simulated nitrate. Thus, it is apparent that when no knowledge of the system is available the choice of parameter values and especially those revealed here to be important, will be critical. However, the relatively low uncertainties revealed when only the parameter combinations producing behaviours are used suggests that once the parameters are approximately correct, it will be the input time-series data that will have the greatest effect on model performance.

CONCLUSIONS

The new Q^2 model has been applied to four reaches distributed along the length of the non-tidal River Thames and, although the model is relatively simple, it has been shown to work well. The model was tested against behaviour criteria based on observed data collected by the EA, and this suggests that the mathematical representation of the system is reasonable, at least as a first approximation. However, the River Thames system is perhaps not sufficiently dynamic to allow complete testing of the internal process representations used in the model. Thus, it would be of benefit to the future design of this model if it were applied to data sets incorporating more significant variations in the in-stream DO dynamics.

The GSA results indicate that nine of the eleven parameters were sensitive in at least one of the reaches if both sets of Monte Carlo simulations are considered. Thus, nearly all of the parameters and, by inference, the processes that they represent, are potentially important and the model probably includes all of the main processes necessary to simulate the DO dynamics (as daily mean values) in the River Thames. The nitrogen model appears to be more sensitive overall than the DO model, and the denitrification rate was the most important parameter when the model was tested against all of the behaviour criteria.

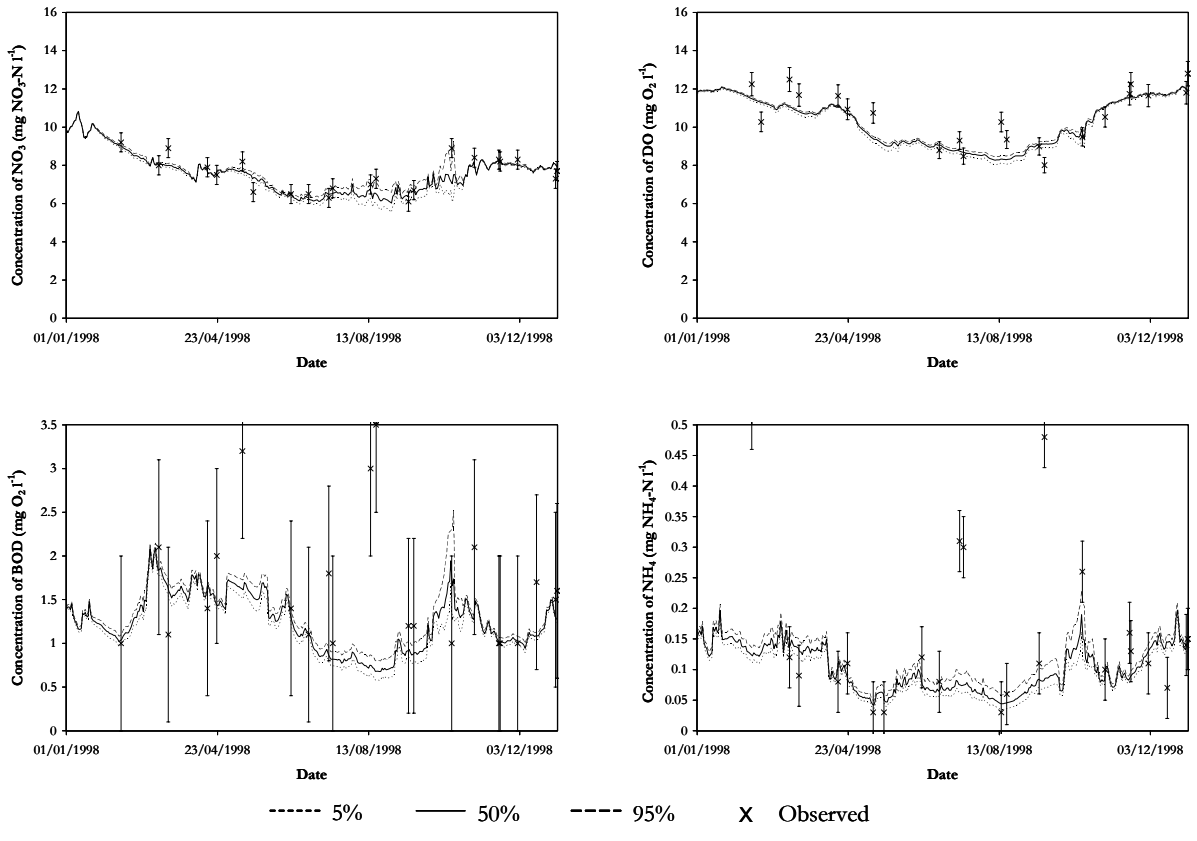


Figure 2 Simulated and observed daily determinand concentratins for reach 4 (Molesey to Teddington). Percentile values are only calculated from behavioural simulations.

The Monte Carlo (behavioural) results can also identify a more specific set of model parameter ranges for the River Thames from that originally drawn from the literature, since following successive Monte Carlo trials, these parameter ranges can be refined to give a more specific (and narrower) range of parameters for each reach. If so desired, further Monte Carlo simulations could be carried out so that these ranges could be increasingly narrowed for calibration purposes, but it is unlikely that a unique set of parameters specific to the particular reach would ever be obtained (Beven & Freer, 2001), and running the model with a range of input parameters allows for the inherent uncertainty in those parameters to be recognised. Furthermore, the results suggest that once the parameters are approximately correct, it will be the input time-series data that will have the greatest effect on model performance.

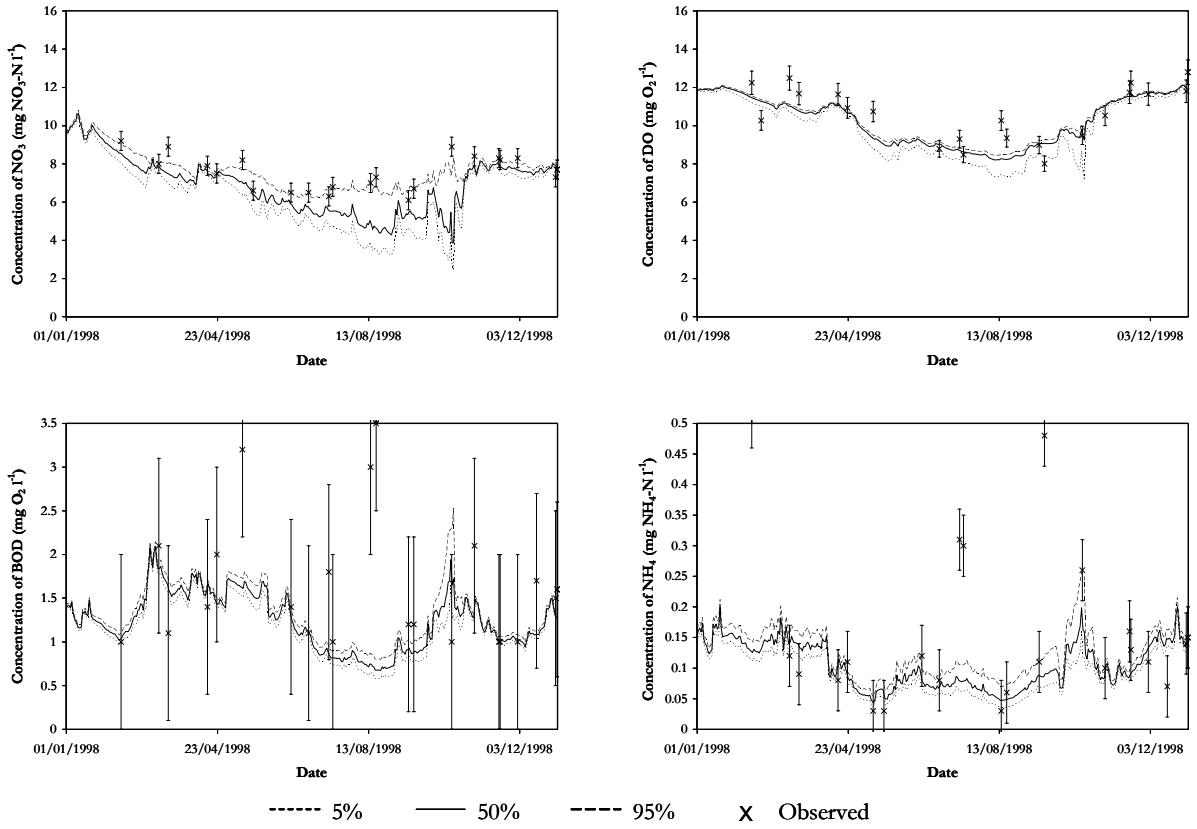


Figure 3 Simulated and observed daily concentrations for reach 4 (Molesey - Teddington). The percentiles are calculated from simulations using all 10,000 parameter sets (i.e. those that produced behaviours and those that did not produce behaviours).

Despite the problems of structural and parameter uncertainty, the work carried out here demonstrates the new model's utility. Namely, the Q^2 model provides a reasonable approximation to the physical, chemical and biological processes occurring in the River Thames, and in particular to those processes that affect the in-stream DO concentrations. Furthermore, this new model provides a basis for formalising the concepts regarding the in-stream processes and thereby allows for hypothesis testing regarding process interaction. The Q^2 model could also be used as a learning tool for examining the dynamics of water quality changes in response to altered flow or water-quality inputs. This is an important attribute given the need to improve the understanding of river system functioning in Europe and around the world for scientific and management purposes.

ACKNOWLEDGEMENTS

This study results from a NERC and EA funded PhD (GT2/97/3/EA), and the authors are grateful to the NERC and the EA for the financial support granted and to Colin Neal (CEH, Wallingford) for co-supervising the PhD. The study was also dependent on significant data sources provided by the EA and CEH, Wallingford, for which the authors are very grateful.

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