Pilot Catchment study of Nutrient Sources - Control Options and Costs

R&D Technical Report P345

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EXECUTIVE SUMMARY

Because the level of nutrients in many rivers is so high, application of the Urban Waste-Water Treatment Directive to relevant point discharges has not, in many cases, achieved the improvements in river water quality which were hoped for. As a result catchment scale plans are required to identify the best place(s) to invest to reduce eutrophication in sensitive stretches of rivers. The study was carried out in three phases: a) identification of any "tools" from the literature that would help to quantify the amount of nutrients (mainly P) entering a river from different sources and predict the effects (eutrophication) which would result; b) on a pilot catchment carry out a test of some tools to identify and quantify the sources of P in the catchment and identify possible management options; c) as part of a CEH funded collaboration, develop the concept of a tool using cost-effectiveness for identifying the best management practices for nutrient control in a river.

Surveys of the literature showed that, although a number of models were available for predicting the concentration of phosphorus in rivers from diffuse agricultural sources, most were too complicated and required too much input data to be acceptable for routine use in a wide range of catchments. The search confirmed that export-coefficient modelling was likely to be the most useful approach for routine use within an organisation such as the Environment Agency.

A case study was carried out on a single catchment to assess the usefulness of "uncalibrated", literature derived export coefficients to estimate diffuse run-off from agriculture. The Warwickshire Avon was chosen as the test location since a very intensive data set had previously been obtained during the UWWTD designation process. Point source phosphorus loads to the River Avon from large sewage treatment works were calculated from discharge data (flow and concentration). Discharges from smaller works were estimated from the contributing population equivalent and an export coefficient value. Phosphorus loads exported in agricultural and urban run-off from agricultural land were estimated for each 50 m x 50 m pixel in the Warwickshire Avon catchment by applying an appropriate export coefficient to each of the 25 categories of land use recorded in the Land Use Map of Great Britain. Similarly, phosphorus loads exported in run-off from livestock were estimated for each 50 m x 50 m pixel from livestock numbers in the MAFF small area statistics. Using a GIS (ARC-INFO) these data were summed to give the total contributing load to each of 50 points on the River Avon upstream of Evesham (catchment outlet). River sampling data were available for each site. In-river total phosphorus loads were calculated from sample data using two methods at each site. By comparing the results it was clear that eight of the sites did not have data from a wide enough range of flows to make reliable estimates of phosphorus fluxes. These sites were not considered in further analysis.

Analysis of the data showed that the proportion from each of the three sources (sewage treatment works, land run-off and livestock run-off), which constituted the total phosphorus load at each point in the river, varied widely. Pictorial presentation of the results needs to be done taking into account the size of flux at each sampling site otherwise the relative importance of the different sources can be misinterpreted. At Evesham 75% of the total phosphorus load was calculated to be derived from point sources. However, following the application of Urban Waste Water Treatment Directive controls to the relevant works the total in river fluxes at Evesham were predicted to be split approximately equally between point and diffuse sources.

The estimated input phosphorus fluxes were within 8% of the measured discharge at Evesham suggesting that, for relatively urbanised catchments, the use of fixed export coefficients is a useful technique, at least for scoping studies of possible phosphorus removal strategies in a catchment. The total phosphorus load in the river at all points was highly correlated with the contributing point source load to that location. Analysis of the residual load following subtraction of the point sources indicated that the cumulative errors in the estimates were too large to identify a need for more catchment specific export coefficients. However, a different analysis suggested that in small catchments (<25,000 ha) the variability of load estimates increased significantly.

Data analysis showed that at least 50 samples at a site, spread over a range of flow conditions, are required for a reliable estimate of mean annual total phosphorus load in a river. A comparison of phosphorus measurement methods suggested that the proportion of total phosphorus in river water, which was in dissolved reactive form, is extremely variable, ranging from 70–95%. As a result it is not very reliable to measure dissolved reactive phosphorus and use a fixed factor to estimate total phosphorus concentrations. Even though it is more costly to perform, the best results will be obtained from measurements of total phosphorus.

A literature survey of models for predicting the extent of eutrophication in rivers resulting from phosphorus dissolved in the water showed that no reliable methods were available for river systems. Unlike most lake systems which are generally dominated by phyto-plankton and, hence, relatively simple to model, in rivers there is a complex mixture of micro and macro plants including planktonic, benthic and epiphytic algae and macrophyte communities. As a result it is, at present, impossible to predict the biological effect of any proposed reduction of phosphorus concentration in a river and develop scientifically based target P concentrations in rivers.

A literature survey was undertaken to identify possible methods of reducing point and diffuse phosphorus losses to rivers. Two main methods of point source P removal were identified, chemical treatment followed by settlement, and biological treatment. For diffuse agricultural sources a large number of Best Management Practices (BMPs) were identified but the detailed methodology was often poorly described. In order to make a first attempt at identifying those with potential for use in the UK a farmer was asked to asses the likely take-up of each method and the type of inducement which might be required for implementation.

A review was carried out of methodologies used to assist in the selection of appropriate P reduction measures in a catchment. Very few usable models were found but cost-effectiveness was proposed as a reasonable approach and a conceptual model was derived with potential for use in the UK. This model combines load information, reduction efficiencies and costs in a model scenario testing methodology. It would be equally applicable to suspended solid, nitrate and pesticide run-off reduction as well as phosphate.

Very little data on the costs of phosphorus removal were found to be either accurate or appropriate to the UK situation. Even when costs were quoted it was very difficult to identify the base data upon which they were based and the model assumptions used. In obtaining cost-effectiveness data there were two major problems: uncertainty in the effectiveness estimates and uncertainty in the cost estimates. In effectiveness estimates definition of the baseline state is always a problem and difficulties in converting short term efficiency measurements to

longer time scales is difficult. However the main problem is the variability between sites resulting from differences in slope, soil type, previous crops, etc. A major problem with cost estimates is which costs should be included. In this study only costs incurred by the farmer are included, i.e. only capital and maintenance costs plus the cost of foregone production.

Almost no modern data was available for estimating the additional costs of P removal from point sources. Almost all modern data includes P removal as part of a total package for a new sewage treatment works. An attempt was made to calculate cost-effectiveness for a sub-set of agreed BMPs. A novel approach was derived using export coefficients to convert from cost per ha to cost per kg P removed. The list of BMPs included: reducing fertiliser input while maintaining fertility; farm pond installation; the use of conservation tillage; the reduction in slope length using break crops; reduced dietary input of P in livestock; implementation of farm waste management plans; moving access points to the top of slopes; moving feeding rings and the cost of public information exercises. Although it is possible to make an estimate of cost-effectiveness there are a number of BMPs for which the efficiency is so site dependent that no cost-effectiveness value could be obtained. The cost effectiveness of P reduction at point sources was found to be considerably more cost effective than that of most diffuse source BMPs.

A short review of the possible economic and policy measures, which could be applied to encourage low P losses, is given. Topics discussed include taxes on products, tradable permits, nutrient accounting and opportunities arising from Agenda 2000. Implementation of one or more of these approaches may increase the uptake of desired BMPs by farmers.

1. INTRODUCTION

The introduction of the Urban Waste Water Treatment Directive (UWWTD) by the European Union in 1991 specified the level of treatment of sewage which is required by law before being discharged into fresh, estuarine or coastal waters. In particular, in article 5, it placed an obligation on member states to identify sensitive areas, within which treatment to more stringent levels than normal is required. These sensitive areas are defined by the regulations as 'natural freshwater lakes and other freshwater bodies, estuaries and coastal waters which are found to be eutrophic or which in the near future may become eutrophic if protective action is not taken'.

Designation as a sensitive area enables the Environment Agency to require nutrient reduction from sewage treatment works serving more than 10,000 people, if it can be demonstrated that nutrient removal will have an impact on eutrophication. However, because the levels of nutrients are so high in some sensitive areas, reductions of nutrients from large sewage works alone will not necessarily bring nutrient levels, particularly phosphorus (P), below the threshold required to reduce the effects of eutrophication. Hence, in order to convince the utility operator as to the need for the costly investment on their part, the regulating agency needs to be able to develop a complete catchment management plan that identifies the combination of investments in the remediation of different sources required to achieve the target level of nutrients.

In a previous report, a draft methodology was given for use in the first phase of any study of the causes of eutrophication in a freshwater system (Hilton et al., 1998). A wide range of possible causes of the observed effects of eutrophication in a river were identified and methods were given by which their importance might be gauged. Once the main causes of eutrophication have been identified (often, but not always, point sources and diffuse run-off from both agricultural land and septic tanks), the Environment Agency (EA) needs to be able to identify appropriate allocations of investment between these sources, in order to develop a realistic strategy for reducing the effects of eutrophication. This requires tools to identify the relative importance of the different major sources of nutrients (particularly P) in the catchment and a means of incorporating and comparing the relative efficiencies and costs of any options to reduce its impact.

In the words of Russo (1991) "As environmental controls become more costly to implement and the penalties of judgement errors become more severe, environmental quality management requires more efficient analytical tools based on greater knowledge of the environmental (and other) phenomena involved to be managed."

The project originally set out to:

Develop a menu of tools to determine:

- Significant nutrient sources in a given catchment.
- Best remedial practice, including cost effectiveness data.

Test the tools on a pilot catchment and recommend appropriate procedures for their use.

However, as the project developed it became clear that there were no easy-to-use tools available "off the shelf". In addition the Agency developed a new strategy for combating eutrophication, and the designation of sites as eutrophic, at least in the first round, was completed before the project started. Hence, it became clear that these initial objectives were not achievable. Through discussions with the Agency the terms of reference were reassessed and the following terms were agreed as the objectives of the contract:

Phase 1. Identify from the literature and other sources a list of tools that might help:

- Quantify the amount of nutrients (mainly P) entering a river from different sources and

- Predict the effects (eutrophication) which would result as a consequence.

Phase 2. Carry out a pilot study to test some tools on a single catchment. The catchment of the Warwickshire Avon was chosen because it was known to have a large data set, including total P measurements, which had been collected previously during the UWWTD designation process. Total P is not measures at most routine river monitoring sites. The tests concentrated on:

- Identification and quantification of the sources of P in the catchment.

- A comparison of estimates of diffuse pollution derived from export coefficients with estimates of the total load derived from measurements in river.

- Identification of possible options for controlling nutrient sources (including public awareness).

Because of the scarcity of cost data for remedial measures it was agreed that IFE would put a significant amount of effort into trying to obtain cost estimates for an agreed list of remediation practices.

Phase 3. As part of an internally funded Centre for Ecology and Hydrology (CEH) science project an attempt would be made to develop a conceptual model (tool) for making cost-effeciveness comparisons to the level of a demonstration, "proof of concept" spreadsheet model. [Completion and reporting of this phase would not be a necessary criterion for completion of this project.]

2. CONSULTATION WITH EA STAFF

2.1 Workshops

Three workshops were held to elicit input from a wide range of Environment Agency staff from a number of the regions. The meetings were held at different venues:

17 December 1997 at Institute of Freshwater Ecology (IFE) East Stoke;
3 March 1998 at IFE, Windermere;
3 April 1998 at Environment Agency, Blandford Forum.

In total 20 individuals from 7 regions of the Environment Agency + HO were involved in the consultation exercise (Appendix 1). The meetings, generally, took the form of a presentation of current thinking on aspects of project development, followed by workshop discussions to elicit Environment Agency staff responses. At each meeting lists of previously identified tools and models were presented and participants invited to add to them. The EA staff agreed with the IFE representatives that there are at present no effective tools or methods for routinely predicting the environmental effects of a given P concentration in rivers.

2.2 Special Diffuse Source Meeting

A special meeting was held at Blandford on 22 October 1998 which concentrated on diffuse sources, their control and economics. P Buckland, R Huggins and R Dils (National Centre for Ecotoxicology and Hazardous Substances) attended from the EA. J Hilton, G P Irons and S McNally attended from the contractor's side.

3. MODELS OF DIFFUSE P LOSS

A number of methods and models for estimating the magnitude of diffuse P inputs from land areas into rivers are reported in the literature. They cover a range of approaches from purely statistical correlations, to complex models that attempt to account for real processes involved in the transport of P. They also cover a range of scales from small plots, through fields and slopes, to entire catchments. Although many models can be applied in a variety of situations, it is important that both the scale of the model and its complexity are fit for the purpose that the results are intended for. Both the spatial scale and the temporal scale need to be considered. Spatial scales range from plot, through field to catchment level. Temporal scales range from single storm events to annual averages. With many models (especially process based ones) there is concern in the scientific community that "over parameterisation" may cause poor results when a model is called on to provide predictions. This is a particular danger when non-specialist staff perform the modelling process. Conversely, over simplified models require a major recalibration exercise at any new site where they are used and can introduce large errors, especially if the predictions extrapolate outside the calibration range. In this section we will consider many of the models reported in the literature and assess their usefulness as the basis of tools which can be easily applied by field officers to estimate the diffuse P run-off load from the catchment.

3.1 Literature Searches

A preliminary literature search was performed using BIDS (Science citation index on line) to identify tools for the prediction of P inputs into rivers. These searches generally consisted of a complex key word search performed on the title, abstract and keyword fields, supplemented by manual inspection of the abstracts to distinguish between relevant and irrelevant hits. There is no single set of keywords that adequately identifies all relevant information. In practice at least four different keyword sets were used.

A number of models were found for predicting P inputs into river systems and various models exist that describe the behaviour of P in rivers. Having identified some relevant models, a second BIDS search was performed which included a list of 36 authors known to have contributed to related work, combined with the keyword "phosph*" in the title, abstract or key words. The search was repeated for combinations of the authors plus a series of other combinations of key words. After a preliminary manual sieve, this search gave 123 references which were studied in detail to produce the summary of the tools and models given in tables 3.1 to 3.4. The list of models found at this stage was presented at the workshops with Environment Agency staff and the delegates present were invited to suggest additional models. This process added no additional references.

3.2 Models from the Literature

Three main groups and one hybrid group of methods/models were identified in the search:

1. <u>P loss models</u> (Table 3.1) estimate the mass of P lost from the land into the river and range from simple correlations to complex process-based models attempting to model the detailed interactions within the system.

2. In stream models (Table 3.2) which deal with the movement of water and or pollutants through a catchment, emphasising dilution and mass balances, and in some cases bio-geochemical processes.

3. Mixed models (Table 3.3) which incorporate both a P loss sub model and an in stream sub model.

4. Discharge correlation approaches (Table 3.4) attempt to assign sources to pollutants using relationships between discharge (flow) and either the concentration or the instantaneous load of the material. For example, a system dominated by point pollution would be expected to show a load that does not rise with flow.

The models listed in the four different groups will be described briefly in the following sections. The majority of this information was derived from three sources: Mainstone *et al.* (1996a), a register of models found on the Kassel University web site (Kassel, 1999), and Donigian *et al* (1991). Other sources have been reference individually.

3.3 P Loss Models

These models are generally one of three types based on: fundamental processes, the soil loss equation or export coefficients.

3.3.1 Process based models

ANSWERS	Areal Non point Source WaterShed Environment Response Simulation (Beasley *et al*, 1969; Beasley and Huggins, 1981). This model simulates the processes of interception, infiltration, surface storage, surface flow, subsurface drainage, sediment drainage, sediment detachment, transport and redeposition for a single rainfall event. The processes are modelled within individual cells and then routed according to topography. P concentrations are simulated by correlation with sediment and run-off. The model requires topographic, drainage, soil, and landuse data. As this model is restricted to simulating single events its main use would probably be the assesment of the efficiency of indiviudal BMP(s) in specific situations.

Table 3.1 Models for predicting diffuse P loss

Model	**Principle/modelled system**	**Additional information**
ANSWERS	predict soil loss and run off volume and assume P loss is proportional.	single event on a catchment scale, soluble and sediment bound P by correlation with soil loss and runoff
CREAMS	predict soil loss and run off volume and assume P loss proportional	losses to ground water lost from model – requires complex calibration/estimation of large number of parameters, US
DEMNIP	models soil processes and predicts P entering groundwater	assumes conditions in Netherlands; no surface water flow.
EPIC	models soil processes and predicts soluble and particulate P in runoff and loss to ground water	designed for predicting crop yields and fertiliser economics requires data from GRASS GIS
VirGIS/SLOSS/PHOSPH	predict soil reaching river and assume P loss proportional	two simple models built on the Virginia state GIS system
USLE	soil loss = ARKLSCP	where A is area, R is rainfall, K is soil erodability, L is slope length factor, S is slope factor, C is crop practice factor and P is conservation practice factor
AGNPS	predict soil loss and assume P loss proportional	single event or continuous - complex - 22 parameters, USA
GAMESP	predict soil loss and assume P loss proportional	tool for estimating effect of management practices on P in run off, complex (10 or more factors) - Canadian
Flux/transmission approach	P export coefficient approach but with one additional parameter to model redeposition during overland flow. The extra parameter may help account for effects arising from varying the distribution of land use. Gives the possibility of modelling buffer strips/zones.	
Generic Export Coefficient (land)	annual total P export estimated as sum of (coefficient x land area under specific land use) for all uses, simple in principle, "just" need areas + export coefficients.	
Generic Export Coefficient (livestock)		similar to above but accounting for livestock numbers
Generic Export Coefficient (unsewered population/unmonitored STW)		similar to above but acounting for human populations
MINDER for rivers	export coefficient approach per unit rainfall	written by WRc, Medmenham

CREAMS	Chemicals, Runoff and Erosion from Agricultural Management Systems (Knisel, 1980). This model predicts runoff, erosion, and chemical transport from a "field" (an area of uniform land-use, soil, rainfall and management). The model is designed to be simple and avoid calibration, while predicting runoff, percolation, erosion, dissolved and particulate bound plant nutrients and pesticides. Two alternative hydrological approaches may be used depending on the data available. As it was developed for agricultural use it ignores groundwater, and any pollutants leaving the base of a soil profile are lost from the system. P exports bound to sediment and free in solution are calculated. A number of models have evolved from CREAMS. GLEAMS - Groundwater Loading Effects of Agricultural Management Systems (Leonard *et al,* 1987) is a companion version with emphasis on movements to groundwater.
DEMNIP	(No known English acronym). This is a group of models developed by Delft Hydraulics that simulates nutrient export from agricultural catchments (Ruygh *et al*, 1990, Vuuren, 1990). There are three relevant submodels DEMGEN (demand generator), NITSOL (nitrogen soil model) and PHOSOL (P soil model). DEMGEN calculates water balances, and other hydrological parameters. while NITSOL and PHOSOL calculate, respectively, nitrogen and P speciation in soil and soil water. These calculations imply a potential model output in terms of dissolved orthophosphate being carried into rivers but the model does not consider suspended solid materials or total P. The model was designed for use in the Netherlands and therefore may contain assumptions that are incorrect in the UK. Data is required on soils, growing seasons, crops and fertilisers.
EPIC	Erosion Productivity Impact Calculator (Sharpley and Williams, 1990) is a mechanistic model designed to assess the long term effects of soil erosion on crop production. It requires inputs from GRASS GIS layers (Geographical Resources Analysis and Support System – developed by US Army Constructional Engineering Research Laboratories), and additional agricultural management information. The main outputs relate to crop yields and the economics of fertiliser use, however the model does compute soluble P lost in runoff, and P transported in sediment.
VirGIS/SLOSS/PHOSPH	VirGIS (Virginia GIS) is the State of Virginia's own GIS system. Two simple pollutant yield models (SLOSS – soil loss; PHOSPH – phosphate) were written to predict non point P sources in a small catchment (1505 hectares), and used to select areas for the implementation and testing of BMPs (TMDL, 1999). The model software is specific for the named GIS system though the principles could be used elsewhere. The models require three soil based parameters (two derived from soil analysis), three that represent aspects of topography and three that represent aspects of management.

3.3.2 Universal Soil Loss equation based models

Many models use the "Universal Soil Loss" (USLE) equation to predict the amount of soil entering a river. The models then use either simple correlations, more complex models or soil analysis to predict the mass of P transported in that quantity of soil.

USLE	Universal Soil Loss Equation (Wishmeier and Smith, 1978) is based on a statistical correlation approach where the predicted mass of soil lost in a given rainfall event is the product of area, rainfall, a soil factor (erodibility), a slope length factor, a slope degree factor, a crop factor and a conservation factor. This equation, or modifications such as MUSLE (Willams, 1975) are widely used in America where there are tables of factors for particular soils, crops, and management practices.
AGNPS	AGricultural Non-Point Source model (Young *et al,* 1989). This model simulates single rainfall events or a continuous time period, using a grid of cells to route water and pollution according to topography. Sediment transported by runoff is calculated within each cell using a version of the universal soil loss equation (including an adjustment for slope shape). Gully erosion is accounted for by user estimates within each cell. P loads are calculated for both dissolved and sediment bound forms using statistical correlations with the runoff volume and sediment load. Data for 22 parameters are required for each cell and include a slope shape factor, a soil erodability factor, a cover and management factor and a support practice factor. The large number of parameters gives considerable scope for over fitting and poor predictive performance in the hands of non-specialist staff. With the aid of specialist staff, and possibly a GIS, a model of this type could play a useful role in both simulating sediment and P loss from a catchment, and the impact of proposed management options.
GAMESP	Guelph model for evaluating the effects of Agricultural Management on Erosion, Sedimentation and P yields. This model was developed in Canada and consists of two components dealing with sediment transport and P. The landscape is divided into land and stream cells of various sizes and shapes, each being small enough to be assumed to be homogeneous. The sediment section uses the universal soil loss equation to estimate the soil lost within each cell, and a delivery ratio to estimate the fraction passed to the next cell. Sedimentation is assumed not to occur in stream cells. The P model assumes that the P is a fixed fraction of the soil lost.

3.3.3 Export Coefficient models

Flux Transmisson approach

An export coefficient approach is used to estimate the P load leaving particular areas. This is then modified by a second model that allows P to be retained on the surface of land between the source and the river.

Generic Export Coefficient (Land)

This is a simple approach usually used to estimate the annual diffuse load into a system from land. An export coefficient is a number that gives the average export per unit area of a particular substance for a particular classification of land. The catchment area is divided into land use types and an export coefficient is obtained for each type (frequently from literature sources). The export for each land-use type is calculated by multiplying the area by the coefficient and the results summed to obtain the total diffuse load. The data requirements are low – export coefficients are available in the literature, though they must be selected with some care to ensure that data adequately match the conditions in the catchment under study.

Generic Export Coefficient (livestock)

This uses the approach described above but adds an additonal diffuse load in each area for each type of livestock that is present. This additional load is simply the product of a constant and the head count for that type of livestock.

Generic Export Coefficient (unsewered population/unmonitored STW)

The point source load for small unmonitored STW can be estimated as the product of an export coefficient and either the population served, or the population equivalent. A similar approach can be used for septic tanks, once the population being served by septic tanks in a given region is known.

MINDER	Model Input Nutrients Determine Eutrophication Risk (Woodrow *et al,* 1994). This is a simple statistical approach using export coefficients expressed in the form of mass of P lost per unit area per unit rainfall. It has been developed by WRc. The availability and commercial status of this model is not known.

3.3.4 In stream models

QUASAR	QUAlity Simulation Along Rivers (Lees *et al,* 1998). This is a Microsoft Windows based in stream model, that requires time series rainfall data. It generates time series output data for a number of WQ parameters (ammonia, nitrate, dissolved oxygen, temperature, E.Coli, pH, BOD and conservative tracers) but does not include a P sub model
QUESTOR	QUality Evaluation and Simulation Tool for River Systems. Questor is an updated version of QUASAR which is being developed at the Institute of Hydrology (Wallingford, Oxfordshire). It is written in Fortran, and operates under UNIX. It is designed to model in stream processes for various pollutants/materials. It generates time series data and requires time series rainfall data. A P specific submodel is currently being developed, in parallel with an export coefficient based diffuse source model. (D Cooper 1998 *personal communication).*
SIMCAT	SIMulate CATchment (Environment Agency, 1995). Performs simple mass balance calculations to describe dilution of point sources, and uses Monte

	Carlo simulations to generate frequency distributions and other statistics at downstream sites. A special feature of SIMCAT is the option of "self calibration" – the automatic generation of any additional sinks/sources required to make the system fit monitoring data. These sources may be allocated as diffuse sources. This model was developed by the Environment Agency in order to aid the calculation of river needs consents.
TOMCAT	Thames CATchment (Environment Agency, 1995). A similar system to SIMCAT, this has the added ability to simulate directly the within day variations in point source discharge quality and river quality. TOMCAT was developed by Thames region of the National Rivers Authority and Thames Water plc in order to aid the calculation of river needs consents.
WASP	Water Quality Analysis Simulation Program (US EPA, 1999) is a framework designed to hold submodels that predict the fate and transport of pollutants. One submodel is Eutro5 which simulates the transport and interaction of up to 8 variables in water. These can include DO, BOD, phytoplankton carbon, chlorophyll a, ammonia, nitrate, organic N, orthophosphate and organic P, in both dissolved and particulate forms.

Table 3.2 Catchment/in stream process models – model the movement of pollutants through river network

Model	Principle/modelled system	Additional information
QUASAR	river modelling downstream of point sources	Windows based - from Institute of Hydrology - continuous time series model does not include P
QUESTOR	in-stream water quality modelling	Fortran programs under Unix - at Institute of Hydrology - continuous time series model; P under development
SIMCAT	river modelling downstream of point sources	mass balance model with Monte Carlo simulation
TOMCAT	river modelling downstream of point sources	similar to SIMCAT but "owned" or partially owned by Thames Water Plc
WASP4	framework for in stream process modelling	has been used for predict phytoplankton dynamics, US EPA

Table 3.3 Mixed models - model the movement of pollutants into and through river network

Model	Principle/modelled system	Additional information
BASINS	HSPF and QUAL2E in integrated environment	a user friendly system with GIS and database facilities set up to automatically extract required data from archives.
HSPF	predict soil loss and run off volume and P is proportional. Can model complex in-stream interactions	very comprehensive but also very data intensive may require a team of modellers if used in its original form (see basins
INCA		

Table 3.4 Discharge correlation approaches – approaches based on monitoring data

Model	Principle/modelled system	Additional information
Hydrograph separation		no definitive method found
Flow concentration/Flow load plots		no definitive method found

3.3.5 Mixed models

BASINS	Better Assessment Science Integrating Point and Non-point Sources (Quenzer *et al,* 1999) is an ArcView based system that brings together within one easily used framework data and models from a number of sources in the US. It has been developed specifically to meet the needs of both State and EPA pollution control staff. It can support the analysis of information at a variety of scales using a range of tools with varying levels of sophistication (including a version of HSPF). The main barriers to developing such a system in the UK are financial and political, with individual government organisations being encouraged to protect their data and maximise the profits they derive from it.
HSPF	Hydrological Simulation Program – FORTRAN (Donigian *et al*, 1984). A highly developed continuous time model able to simulate contaminant run off, and in-stream water quality including sediment interactions. The model has extensive data requirements. A continuous record of rainfall is required, and records of evapo-transpiration, temperature and solar intensity are desirable. Many other parameters can be specified, though default values are available for some. Options can be set up to bypass some sections of the program when data is unavailable. As HSPF can simulate three sediment types (sand, silt, clay) and the hydrolysis, oxidation, biodegradation, volatilisation, and sorption of pollutants, very detailed site monitoring can be required to perform a complete calibration. In order to handle all the details of a full implementation a team may be required. A version of HSPF has been incorporated into BASINS.
INCA	Integrated Nitrogen Catchment Model (Whitehead *et al,* 1998). A processed based nitrogen model that includes both the biogeochemical soil processes of the nitrogen cycle, point sources and hydrological modelling. INCA is currently being development further to include P. (Rachael Dils *personal communication 1999*).

3.3.6 Discharge correlation approaches

There are a number of procedures that attain broadly similar results to the run-off models by studying the combination of flow records and concentration data. These generally make the assumption that most diffuse sources of pollution are rainfall dependent, while most point sources are not. Two general approaches have been noted, hydrograph separation, and line fits. It must be emphasised that the models listed above estimate the loads entering a river, while these methods attempt to allocate the load passing down a river to different sources.

A hydrograph is a plot of flow against time. Hydrographs may represent measured flow over a period of time, the response of a river to a storm, or represent in a generalised/abstract form, the response of a river to rainfall. There are a number of both graphical and mathematical methods (Burt, 1992; Gordon *et al,* 1992) for separating a hydrograph into a set of component hydrographs representing separate sources of water (eg base flow, subsurface flow, and overland flow). Once the base flow is known, monitoring data can be selected that represents the river under base flow conditions, and this data can be used to estimate the pollutant load under those conditions. This load is actually the sum of the load entering in ground water (which is usually assumed to be small) and from point sources. The difference between the base load, and the total load is assumed to be the diffuse load due to run off. One problem with this technique is the lack of a clearly defined and commonly accepted method for performing the hydrograph separation step.

Various alternative procedures exist that produce similar results, with similar assumptions, but avoiding the arbitrary choice of a hydrograph separation. These methods assume that the flows from the point sources are very small compared to the total flow of the river and that the point source loads are not altered by wet weather. The former may not be true in the headwaters of a river, or in an urban catchment. The latter may not be true if storm drains discharge untreated sewage into the river.

The mass balance equation:

$$Q_T C_T = Q_P C_P + Q_D C_D \tag{1}$$

where : Q is flow, C is concentration, and the subscripts $_{T,D,\ P}$ indicate "totals" after mixing, diffuse sources, and point sources respectively.

can be simplified if the point source flow is assumed to be very small compared to the total flow. The total flow then approximately equals the diffuse source flow:

$$Q_T C_T \approx Q_P C_P + Q_T C_D \tag{2}$$

This approximation can be plotted as total load ($Q_T C_T$) against total flow (Q_T). If a straight line is then fitted to the data, the gradient gives an estimate of the concentration entering from the diffuse sources, and the intercept gives an estimate of the point source load. Alternatively, it can be rearranged to the form:

$$C_T \approx \frac{Q_P C_P}{Q_T} + C_D \tag{3}$$

and plotted as concentration against 1/flow. Again, a straight line may result in which case the intercept gives an estimate of the concentration entering from the diffuse sources and the gradient gives the load entering from the point sources. Equations (2) and (3) are very closely related however (3) tends to emphasise data associated with low flows, while equation (2) emphasises the data associated with high flows. As in real life the concentration of a pollutant entering a system from a diffuse source is likely to be related to the flow, an empirical extension to equation (2) may be required so that:

$$Q_T C_T \approx Q_P C_P + k(Q_T)^n \tag{4}$$

where k is a constant and $C_D = k\ (Q_T)^{n-1}$. The exact theoretical meaning of such a fit is unclear, however the intercept does still represent the point source load, as long as the assumptions listed earlier remain true. It is logical to assume that in heavy rain, soil, fertilisers and manure are more likely to get washed into a river, and this type of effect can be fitted by equation (4).

3.4 Model Selection/Discussion

It is a general principle in modelling that the model should be kept as simple as possible, to prevent the dangers of over-fitting (figure 3.1). Unfortunately, only professional judgement can decide when a model is an over simplification. Models must be tested thoroughly when they are first set up in a catchment to avoid erroneous results, which inexperienced staff could accept without question "because the computer said so".

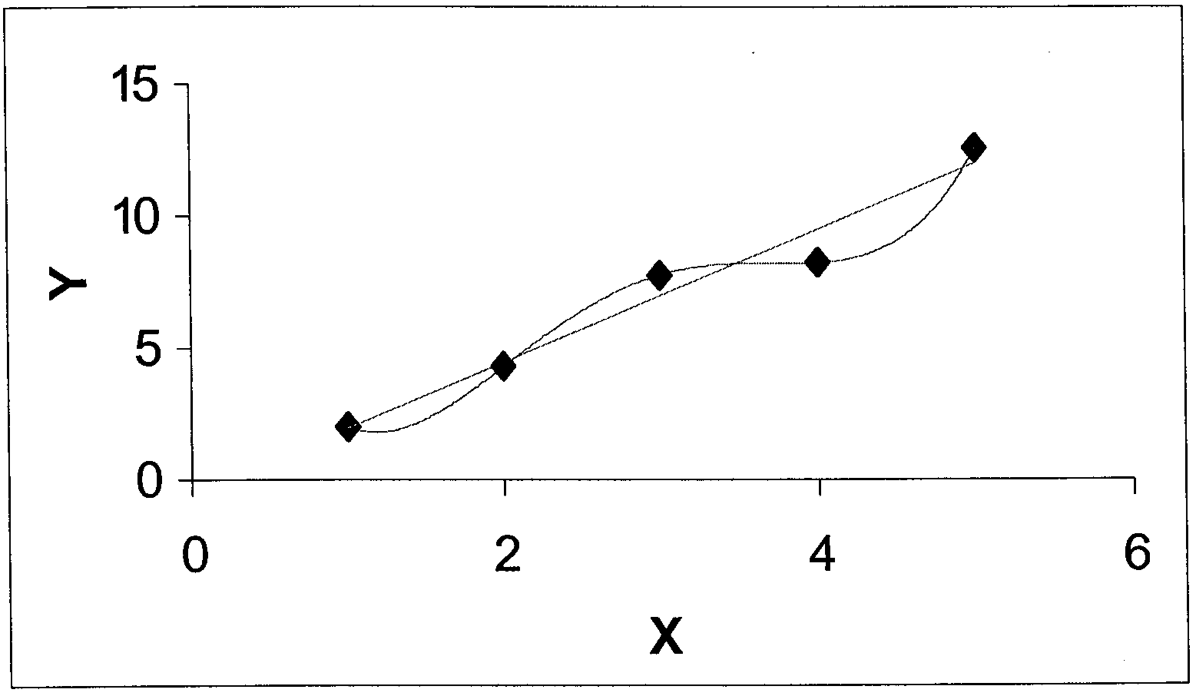

Figure 3.1 **A simple example of over fitting a straight line plus added noise. Simple linear regression ($y = 2.50x - 0.54$, $R^2 = 0.96$); polynomial regression ($y = 0.25x^4 - 2.71x^3 + 10.05x^2 - 11.99x + 6.40$, $R^2 = 1.00$).**

The list of models presented above is probably not complete, however it should be representative. Clearly in any given situation, one model is likely to perform better than the others, however more than one may well give acceptable results. In the absence of an extensive knowledge of the field of application and every model that could be available, it is impossible to create a clear set of rules defining when each model should be used. The only alternative is to follow the procedure outlined below, with reference to the data requirements and outputs of the various models found to be available, on each occasion that a modelling exercise is required. It is unrealistic to attempt to supply a definitive list of models, their requirements and outputs and status as they will be out of date by the time they are published.

1. List the information that must be produced from the modelling exercise.
2. List the simplest model/models that can provide that output.
3. List the data requirements of the models.

4. Compare the data requirements of the models to the data available for setting them up, and ensure that there is ADDITIONAL INDEPENDENT data available to check the model predictions.

5. If one of the simple models can't give the level of detail required in time or space obtain the assistance of specialist staff to set up one of the more complex models.

On the basis of recommendations from a previous Environment Agency report (Mainstone *et al*, 1996b), it was assumed from the outset that a GIS approach using run-off coefficients would be the most useable approach to diffuse source load estimation. This assessment of models available in the literature confirmed this assumption since all other approaches either required a large number of extra parameters or were significantly more complex than export coefficients while containing the same assumptions. If seasonal load variations are required then an approach similar to that of MINDER may be required, where water volume is taken into account.

The assessment also suggests that the discharge correlation approach applied to in-stream data is also simple to use and needs limited data resources (see later) compared to process models. It has potential as a method for checking the export coefficient approach. However, any comparison must be interpreted with care as the diffuse input models give the amount of material entering the river, while the monitoring data gives the amount of material carried by the river.

When in-stream data is used it is important that total P data are used. In the absence of a reliable model to predict changes in P speciation, it is the only form of P which is conservative and is therefore the only form that can be used in a mass balance. It is important that the staff involved in such a project have a clear understanding of what forms of P are measured by each of the Environment Agency methods.

4. PILOT STUDY – THE WARWICKSHIRE AVON

4.1 Background

On the basis of previous reports (Mainstone *et al*, 1996a,b) it was assumed from the outset that a GIS approach using P run-off coefficients would be the most likely approach to diffuse source load estimation. Assessment of the models available in the literature and consultation during the workshops confirmed this approach, as all other approaches either required a large number of extra parameters or were significantly more complex while containing similar assumptions.

In most inland waters P is the limiting nutrient, ie the nutrient limiting biological productivity (Vollenweider, 1968). In the few freshwaters where this is not the case, either a known natural source of P (Marl lakes) is present or a very large man-made (point) source has introduced a large excess of P. As a result something else becomes limiting (often nitrogen) but, in the latter case, a reduction of the P concentration will make P limiting again. The supplied data set for the Warwickshire Avon catchment included a measurement similar to dissolved reactive P at most sites on most occasions and total P at less sites and less frequently. However, relatively complex process based models are required to predict transformations between soluble-reactive ortho-P and total P in lake systems, and such models are unavailable for rivers. As a result only total P was included in the models since it can be considered to be conservative in river systems. To ensure that the model tested could be easily applied elsewhere without expensive field by field surveys on a catchment scale, existing data sets, described below, were used throughout.

4.2 Catchment Overview

The area of the Warwickshire Avon catchment studied, ie the area upstream of Evesham, covers 2200 square kilometres. It contains approximately 100 known point sources (STWs), only 7 of which qualify for controls under the UWWD (see figure 4.1). [Of these 7 qualifying works, 5 have installed P removal facilities, subsequent to the collection of the data set used here.]The majority of the catchment is pasture or arable land (see figure 4.2), though Coventry and its satellite towns represent a considerable urban population.

4.3 Methods

An export coefficient approach was used to generate estimates of diffuse sources upstream of each sampling point as outlined below. This used export coefficients from the literature (May *et al*, 1996; Johnes *et al*, 1996) to estimate the annual total P export from each land cover type in the Institute of Terrestrial Ecology (ITE) land cover map (see below). Export coefficients are crop specific, therefore a weighted mean was used for the ITE land class "arable" based on the breakdown of different crop types given in the MAFF small area statistics for Warwickshire. Stock densities were available at a comparatively course geographic scale within Warwickshire from the MAFF small area statistics. These were used (see below) to develop a "stock" export coefficient specific to each MAFF parish group. Areas outside Warwickshire were assigned an average stock export coefficient based on the stock density within Warwickshire. For each site these calculations gave estimates of land derived diffuse P and stock derived diffuse P which together represent total diffuse P (neglecting septic tanks).

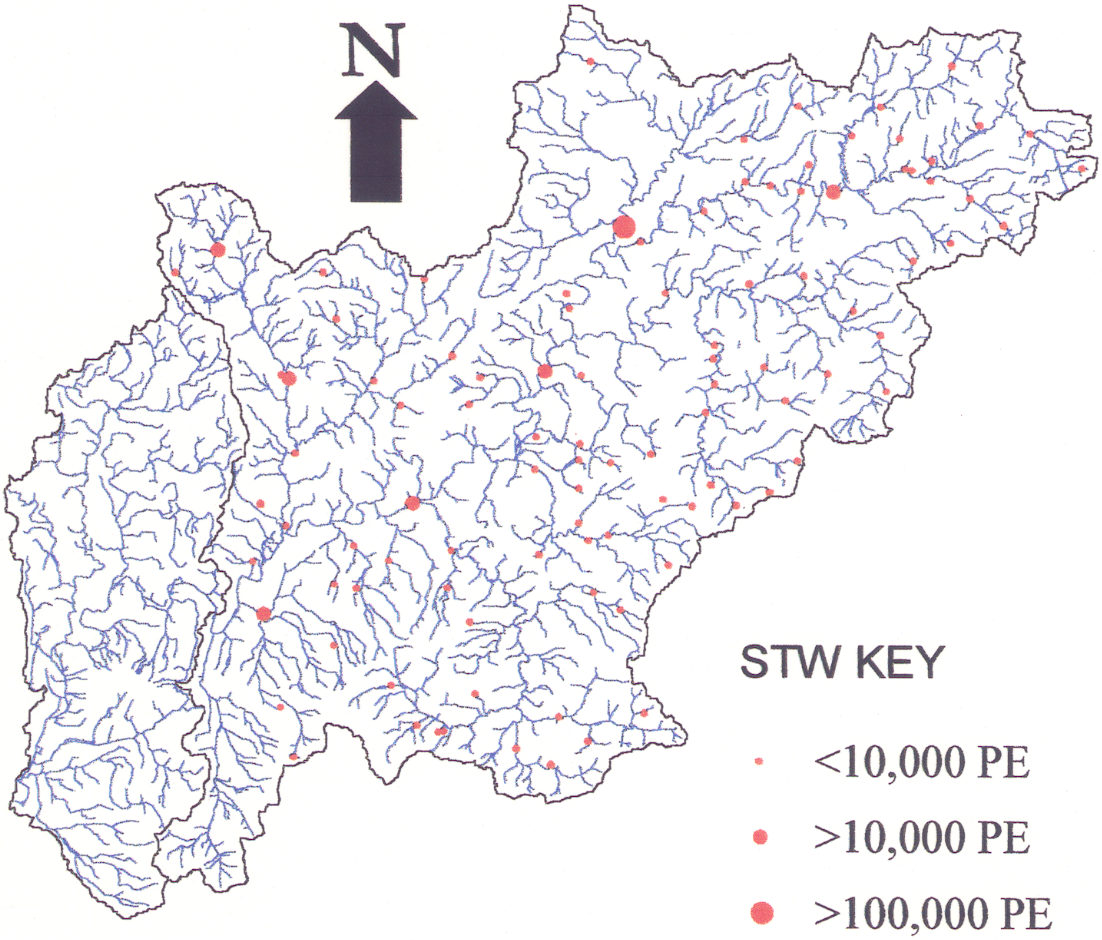

Figure 4.1 Outline of Warwickshire Avon catchment, including catchment boundaries upstream of Evesham. Point STW sources shown

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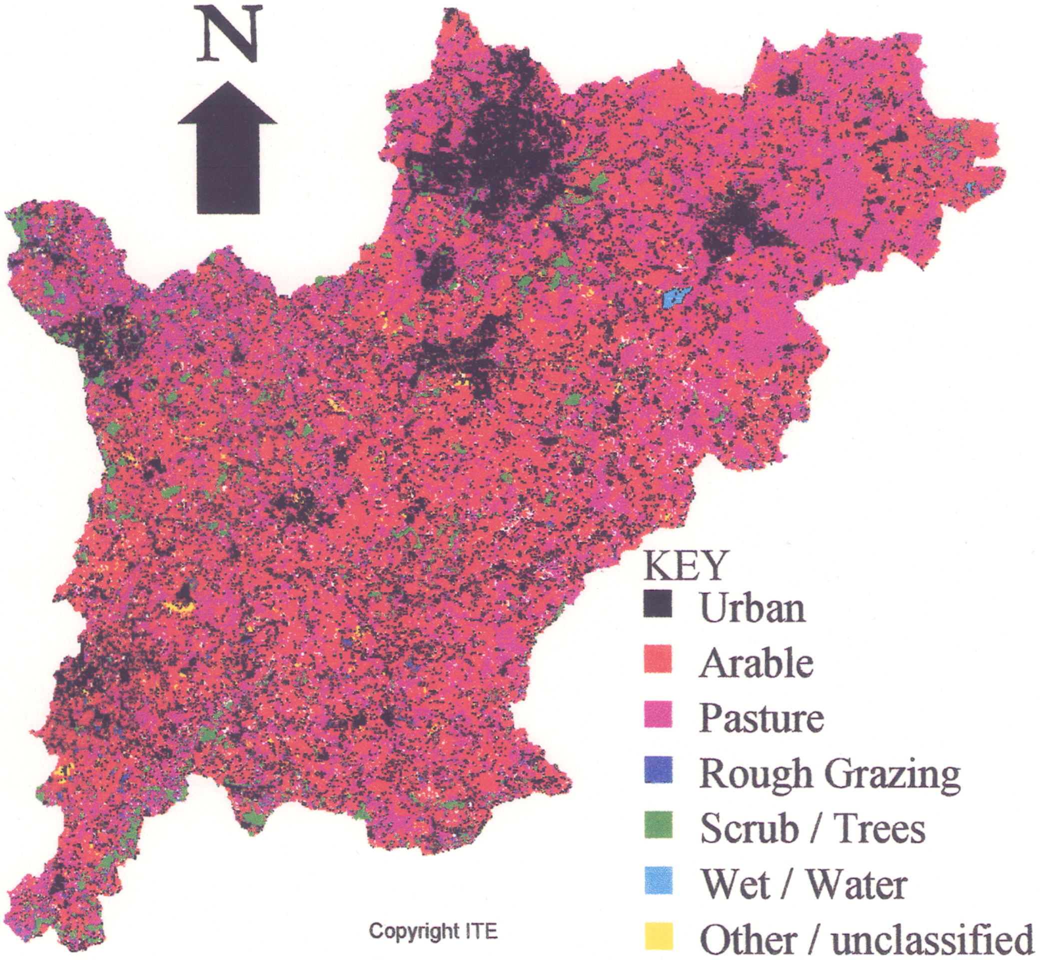
4.4 Summary of Data Sources

4.4.1 ITE Land Cover Map of Great Britain

The ITE Land Cover Map of Great Britain was produced in 1993 from a combination of summer and winter satellite images. It has a resolution of 25 metres and records 25 cover types (+ unclassified) including 18 types of semi-natural vegetation, sea, inland water, beaches, bare ground, developed and arable land. This type of data is required for many of the tools that can be used to predict the diffuse sources of pollution within a catchment, especially the likely contributions of arable farming. An abridged version of this data set is shown in figure 4.2.



Copyright ITE

Figure 4.2 Institute of Terrestrial Ecology Landcover Map (displayed as 7 classes for clarity) for the Warwickshire Avon catchment

4.4.2 Land surface water flow direction map

Fifty metres resolution – generated from the Institute of Hydrology (IH) digital terrain map, this map shows the direction of surface flow in 50 m pixels. It is useful for defining catchment boundaries, and other simple hydrological functions.

4.4.3 Ministry of Agriculture Fisheries and Food (MAFF) small area statistics

Lists among other statistics the numbers of various types of livestock, for groups of parishes. These may be used to refine the estimates of farm related diffuse P sources obtained from the Land Cover data, and paper maps of parish/agricultural district boundaries.

4.4.4 Chemical, flow and sewage treatment works data

Chemical, flow and STW data were collected by the EA for a previous project called "Eutrophication in Controlled Waters in the Warwickshire Avon Catchment" (Foster *et al*, 1998). This data set was used to check the modelling predictions and to identify the data requirements for future studies. The sites listed in this data set are shown on figures 4.1 (STW) and 4.3 (monitoring).

4.4.5 Daily average flow data from the National Water Archive for 9 sites in the Avon catchment required for some types of load calculation

These sites are shown on figure 4.3.

4.4.6 Parish-Line Marketeer 1991 census civil parish boundaries data set

This data and the boundaries used to locate the MAFF data are shown on figure 4.4.

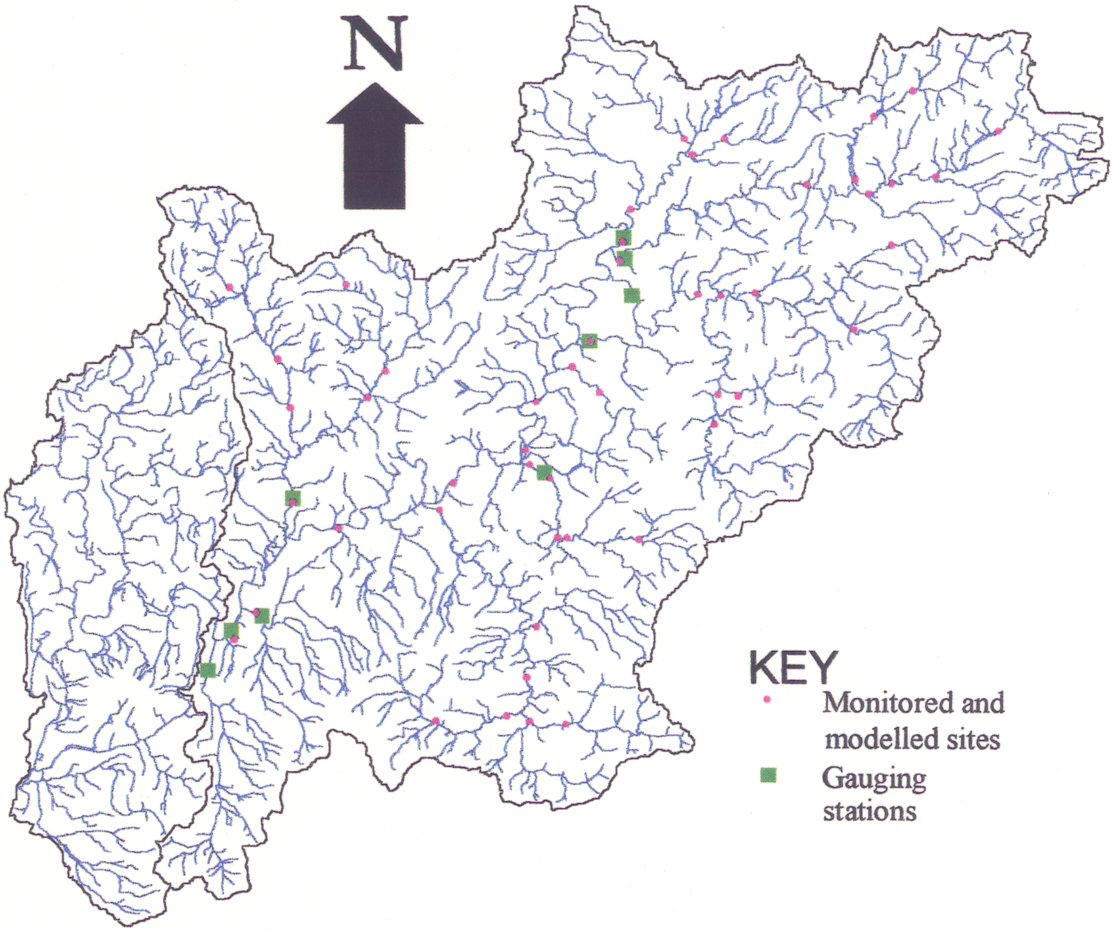


Figure 4.3 **Warwickshire Avon catchment boundaries with monitoring sites and gauging stations**

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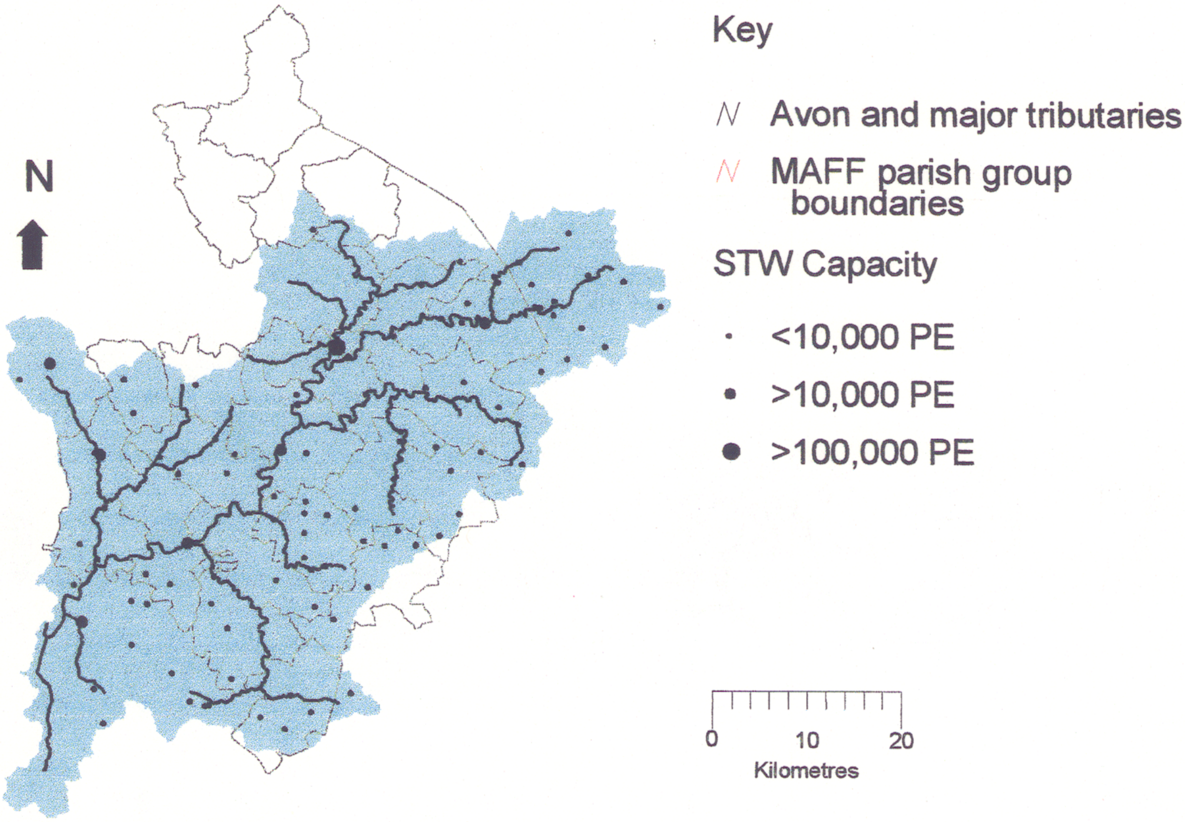

Figure 4.4 **Parish boundaries and boundaries used for locating MAFF parish group data in the Warwickshire Avon catchment.**

Some features of this map are based on spatial data licensed from the Institute of Hydrology, includes material based on Ordnance Survey 1:50000 maps with the permission of the Controller of Her Majesty's Stationery Office © Crown copyright. Some features of this map are based on digital boundaries that are ED-Line Copyright and Crown Copyright and are derived from the 1991 Census of Population

4.5 Preliminary Data Processing/Construction of GIS Data Sets

4.5.1 Diffuse load estimation

In order to take maximum advantage of IFE expertise in the area of diffuse run-off estimation using GIS, Dr Gordon Irons spent one week in Edinburgh discussing the most appropriate GIS approach with Dr L May (IFE). Particular areas covered in the discussions included: aspects of data acquisition, licensing and the scientific limitations of some data sets.

An area of the land cover map of Great Britain was obtained under licence from the ITE, and installed within the GIS system (figure 4.2). It was used as supplied (with 25 m pixels allocated to one of 26 land cover types – including unclassified).

A licence was negotiated to use some Institute of Hydrology (IH) data stored for other purposes at IFE on this project. This covered the IH River Centre Line Network, and sub sets of the IH digital terrain map. These were already installed on the system, and the relevant subsets were extracted to a new workspace to simplify their use.

Preliminary checks indicated that the IH River centre line data correctly showed the Itchen entering the Leam downstream of Marton (confluence grid ref 440545 269002). However, the flow direction grid, which was used to define the catchment area up-stream of each sampling point, contained a minor problem near the confluence of the Itchen with the Leam. This grid routed the flow of the Itchen into the Leam at the wrong point (upstream of Marton). This type of error is caused by the interpolation process used to generate this type of GIS data, in which the contours of a map are digitised and used to interpolate the shape of the land. In the area where the error occurred there are very few contours and the interpolation appears to have produced erroneous results. When modelling the whole of the Warwickshire Avon catchment an error of this type has little or no impact. However, the Environment Agency data included a monitoring point on the bypassed section of the Itchen. In order to obtain sensible predictions for this reach eight 50 m pixels were modified redirecting the Itchen into its correct course in a copy of the data that was used in this project.

Permission was obtained to use MAFF small areas statistics. These were supplied in MS Excel spreadsheet files, which were imported into a relational database constructed in MS Access to simplify the indexing and data extraction. The paper maps supplied by MAFF were not at a suitable scale to be digitised, and it was not clear who owned copyright on them. Therefore the data set for the 1991 boundaries for civil parishes within Warwickshire were purchased from "The Data Consultancy" (URPI Group Ltd). This data set provided the vast majority of the boundaries required for allocating the Warwickshire MAFF small area statistics to land areas. Three areas were listed in the MAFF statistics that were not explicit in the "Data Consultancy" parish data, Nuneaton Unitary Authority (UA), Rugby UA and Leamington Spa UA. Leamington Spa was completely surrounded by other parishes in Warwickshire, which identified the UA area. Boundaries for the two other Unitary Authorities were interpolated from the 1:50,000 OS and added to the GIS layer (see figure 4.4). This required the addition of three boundaries, and removal of redundant boundaries, to leave the MAFF agricultural districts and parish groups. A list of parish names in each group generated from the GIS system agreed with the list generated directly from the MAFF data.

The stocking density for each type of stock listed in the MAFF statistics were calculated for each parish group using the total head of that type from the MAFF statistics, and the area derived from the GIS system. Summary data for the ITE land cover types in each parish group were also calculated. The resulting data set was divided into training and test sets by randomly selecting half the parishes. Principle component regression was used in an attempt to develop a model of stocking density as a function of land cover types. No model was found that could provide useful predictions in the test set, therefore it was decided to use average stocking densities from Warwickshire in those areas of the catchment that fall outside of the county.

A summary table of the stock numbers, and areas for each parish group in Warwickshire was exported from the GIS system, and transferred to MS Excel. These data were used to calculate the estimated average export per hectare in each parish and in Warwickshire as a whole using the export coefficients listed in table 4.1. The expected animal exports per hectare for each parish group were then transferred back to the GIS system, and scaled to give the expected diffuse P run-off from animals in kg $(50\ \text{m})^{-2}$ year^{-1}. The data were required in this form since later steps involved calculations that "count" 50 m pixels. Areas outside Warwickshire were allocated a value equivalent to the within Warwickshire mean diffuse P run-off coefficient from animals. Since the contributions from stock were comparatively small, it was concluded that the errors caused by both this approximation and the approximate boundaries mentioned above were insignificant.

Table 4.1 Livestock annual P exports

Stock type	Export coefficient	Source
Cattle	0.22 kg P year^{-1}	By calculation from Johnes (1996)
Pigs	0.14 kg P year^{-1}	By calculation from Johnes (1996)
Sheep	0.045 kg P year^{-1}	By calculation from Johnes (1996)
Fowl	0.0054 kg P year^{-1}	By calculation from Johnes (1996)

4.5.2 Point source load estimation

The Environment Agency monitoring data, and point source lists were supplied as MS Excel Spreadsheets. These were imported into a relational database (MS Access), to simplify indexing, and data extraction. A list of point sources was generated, and the load associated with each source calculated. Four methods of point source load calculation were used and are summarised below:

1. A figure based on population equivalents using a literature value for the export per population equivalent for typical sewage, including detergents and both primary and secondary sewage treatment (0.38 kg year^{-1}person^{-1} – Johnes *et al*, 1996).

2. An estimate based on monitoring data for the 17 sewage treatment works where it was available, and using the population equivalent calculation where it was not. The average instantaneous monitored load (AIML) was obtained from monitored data using the equation:

$$\mathrm{AIML} = \frac{M\sum_{i=1}^{n} Q_i C_i}{n}$$

Where M is a multiplier to convert from the units of instantaneous load into kg year^{-1}, Q_i is the instantaneous flow at the time of the ith sampling, C_i is the concentration of the ith sample, and n is the number of samples.

3. An estimate based on monitoring data as above but with the concentration capped at 2 mgl^{-1} for works that qualify under UWWTD (works with a population equivalent >10,000). All the monitored concentrations that exceeded the cap value for qualifying works were replaced by the capped value, and calculation 2 was repeated.

4. An estimate based on monitoring data as above but with all qualifying works capped to the lowest concentration that they qualify for under UWWD - works with a population equivalent >100,000 capped at 1 mgl^{-1}.

In each case where STW monitoring data was used to calculate an annual load, the average instantaneous load was scaled to units of kg year - this is equivalent to "WRc method 3" (Foster *et al,* 1998).

4.6 Extraction of Relevant Data

A GIS data set was prepared from the point source data by exporting the relevant data including grid references from MS Access as CSV (comma separated value) files and importing them into the Arc/Info. The GIS system was then used to calculate for every 50 m pixel in the catchment the following summary data –

1. Contributing area upstream under land cover type n (for land cover types 0 to 25).
2. The total expected P export from animals upstream, based on the mean animal P exports per 50 m pixel calculated earlier.
3. Total of contributing point source loads upstream for load calculation scenarios 1 to 4.

The grid references of the EA monitoring points were exported from MS Access as a CSV file, and imported into the GIS system, with the aid of a specially prepared set of macros. These macros initially imported the sites as point locations in OS coordinates, but then enabled the user to interactively select an appropriate grid cell on the line of cells representing the river in the GIS. As some points had to be moved quite large distances to locate on the river, much further than the expected 50-100 m, a checklist was generated containing EA supplied grid references, grid references used, and site descriptions.

This was supplied to the EA to be checked by local staff. All the GIS locations were accepted as being good representations of the true positions. (The two points that moved the greatest distances appear to have been affected by digit swapping during data inputting.) A series of macros were then used to extract from the GIS system the summarised data (see above) that was relevant to each site. This data was transferred to MS Excel for further processing.

4.7 Calculations

4.7.1 In-river loads estimated from sources

The diffuse source P load from each land type at each EA monitoring point is simply calculated as the product of the contributing area upstream – extracted from the GIS system above, and the relevant export coefficients - summarised in table 4.2.

The total P input into the system upstream of each sampling point was estimated as the sum of the sewage works loads (method 2 above), and the diffuse sources.

Table 4.2 Export coefficients by land cover type

	Source	Value kg P ha^{-1} $year^{-1}$	Rationale
1	Sea/Estuary	0.00	Not a contributing area
2	Inland water	0.00	Not a contributing area
3	Beach and Coastal Bare	0.00	Not a contributing area
4	Saltmarsh	0.00	Not a contributing area
5	Grass Heath	0.02	0.02 – Rough grazing – source 1
6	Mown/Grazed turf	0.2	Average of 0.1 and 0.2 – Permanent and temporary grass source 1
7	Meadow/Verge/Semi-Natural	0.2	As type 6 above
8	Rough/Mash grass	0.02	0.02 – Rough grazing – source 1
9	Moorland grass	0.02	See 8
10	Open shrub moor	0.02	See 8
11	Dense shrub Moor	0.02	See 8 and 15
12	Bracken	0.02	See 8
13	Dense shrub heath	0.02	See 8 and 15
14	Scrub/Orchard	0.02	See 8 and 15
15	Deciduous Woodland	0.02	0.02 – Woodland – source 1
16	Coniferous Woodland	0.02	0.02 – Woodland – source 1
17	Upland Bog	0.00	Possible wetland trapping P
18	Tilled land	0.66	Weighted mean of cereals (0.65), root crops (0.8), rape (0.65), field veg.(0.65) – weighted by % areas given in MAFF statistics
19	Runderal Weed	0.02	This land type includes areas of bare ground, being colonised by annual and short lived perennials. Selected the value for rough grazing – source 1
20	Suburban/rural development	0.83	Source – 2
21	Continuous urban	0.83	Source – 2
22	Inland bare ground	0.70	Weighted mean of urban and tilled land according to % area found in land cover map
23	Felled forest	0.02	See 15
24	Lowland bog	0.00	Possible wetland trapping P
25	Open shrub heath	0.02	See 8 and 15
	Unclassified	0.48	Weighted mean of coefficients listed above according to % areas classified in the land cover map.

Source – 1 (Johnes 1996)
Source –2 (May *et al.* 1996)

4.7.2 In-river loads estimated from river monitoring data

There are several load calculation procedures described in the literature, many of which are known to give potentially biased results. The simplest (average concentration multiplied by average flow) is known to underestimate load, as high flow events are comparatively rare and are generally underestimated in monitoring schemes. The AIML calculation outlined above is generally reported to be unbiased given good data (very frequent or continuous monitoring is required to ensure that a representative sample of high flow events is sampled). The AIML approach was used to produce the first estimates of measured load.

As the monitoring data was variable in quantity and many sites did not have enough data to have caught the high flow events an alternative "rating curve" approach was also tested. This is closely allied to the flow correlation approaches discussed in chapter 3. An empirical curve was fitted to the data for those days on which both flow and concentration (load) data were available. This curve was used to predict the local load at each site from the average daily flows at a nearby gauging station. The curves were extrapolated to estimate the loads during high flows. In some cases, particularly where data at high flows was limited, this introduces considerable uncertainty into the estimates. However in the absence of special monitoring triggered at high flows, or very frequent sampling there is no alternative. The procedure used is described below.

One gauging station was identified for each site to supply flow data: this was the gauge that maximised the correlation between the daily average gauge flow, and the instantaneous flows measured during site visits. The measured data at each site were fitted to a model of the form given below using a numeric optimisation method to minimise the sum of square deviations between the function and the measured instantaneous loads.

$$L_i = a + b(Q_i)^c$$

where, L_i is the instantaneous load calculated from the monitoring data on the i^th day, (in units of kg/day).

Q_i is the average daily flow at the most appropriate gauging station on the i^th day and a, b, c are constants for each site obtained by a numeric optimisation to minimise the SSE (sum of squares error) where:

$$\mathrm{SSE} = \sum_{i=1}^{n} (Q_i C_i - L_i)^2$$

where C_i is the concentration measured on the i^th day. The intercept and gradient of a preliminary linear regression were used as starting values for the constants a and b while c was started at 1 (linear). As numeric optimisation was used (rather than an analytical solution) there is no simple method to verify that the best-fit curve has been found, however in most cases the fit appeared reasonable to the eye.

The rating curve was used to predict the load over a 730 day period from the individual daily flows recorded at the nearest appropriate flow gauging station, and the average results scaled to give an annual load.

$$\mathrm{load} = \frac{\sum_{i=1}^{730} \left[a + b(Q_i)^c \right]}{2}$$

4.8 Results

Appendix 2 tabulates, for each sampling point, estimates of the following parameters: average instantaneous monitored P load in the river; total load in the river estimated from the rating curve, estimated total P load entering the river (point and diffuse sources), being the sum of: the best estimate of point source P input load, P load exported from land, P load exported from stock; estimated point source derived P assuming STW >10,000PE capped at 2 mg P l^{-1} and STW >100,000 PE capped at 1 mg P l^{-1}.

The output also shows a scatter plot of instantaneous P load against gauge flow; a time series plot of instantaneous P load; pie charts of the relative contributions of different P sources at each site; the relative proportions of different land types in the catchment; and the predicted annual load, split by season and modified to reflect the effect of the UWWTD.

The key results are summarised using pie charts on catchment maps in the discussion below.

4.9 Discussion

4.9.1 Relative importance of point and diffuse sources within the catchment

The split between estimated total diffuse P load (export coefficients from land and livestock), and total point load P (calculated using method 2 above) upstream of each sampling point was plotted using pie charts on a map of the catchment, figure 4.5. Some points are suppressed, as there is some doubt as to the validity of calculations at these sites (see 4.9.2). The P load for the catchment as a whole above Evesham is about 75% point source derived. However, there are subcatchments dominated by both point sources and diffuse sources. Without a detailed local knowledge of the distribution of eutrophication symptoms, it is impossible to clearly define which sources are likely to control those symptoms. Simple observation of figure 4.5 suggests that, since a large number of sampling points are dominated by diffuse inputs, losses from agricultural land are important sources. However a different picture emerges if the relative size of in-river loads is considered.

To further investigate the relative importance of the different types of P sources, the total P load at each site was added to the plot – represented by the radius of the pie charts, figure 4.6. This emphasises the sites with a high load that represent larger catchment areas. This plot shows that the areas that are dominated by diffuse sources are small, headwater catchments and that the loads within these areas are very small, suggesting that in the context of the whole catchment the relative importance of diffuse P sources is minimal. Hence, when considering eutrophication in the catchment as a whole, the priority management approach is likely to focus on point sources. However, any eutrophication problems within the diffuse source dominated sub catchments would need to be controlled by reducing diffuse sources of P.

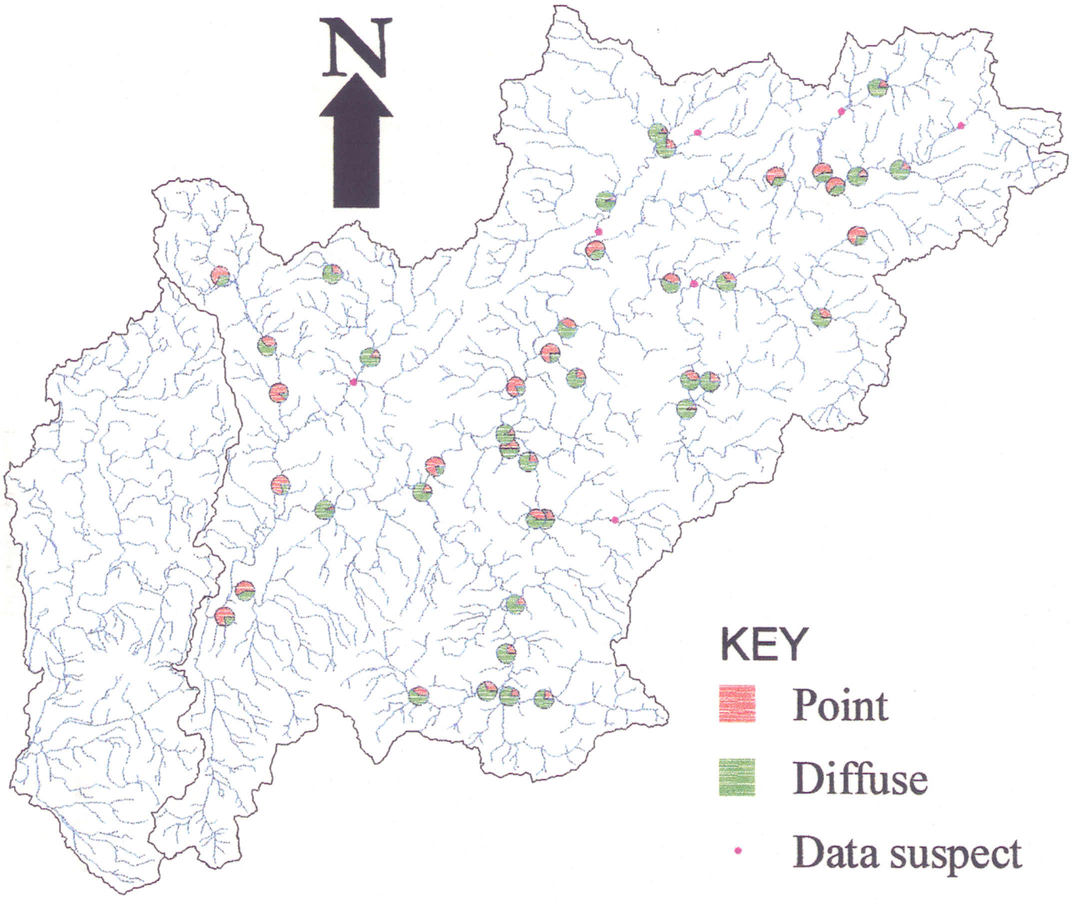

Figure 4.5 **The relative importance of point and diffuse source P in different areas of the Warwickshire Avon catchment**

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A further map was then plotted showing the total predicted P load at each point under the influence of the full implementation of the UWWTD, figure 4.7. To aid comparison, the ratio between radius and total load at each point was kept the same as in figure 4.6. This map demonstrates that even after the full implementation of the UWWTD, point sources have a major impact on P loads over the majority of the Warwickshire Avon catchment. Point sources still contribute over half the total P load at Evesham, due to the large number of small point sources that are not controlled by the directive.

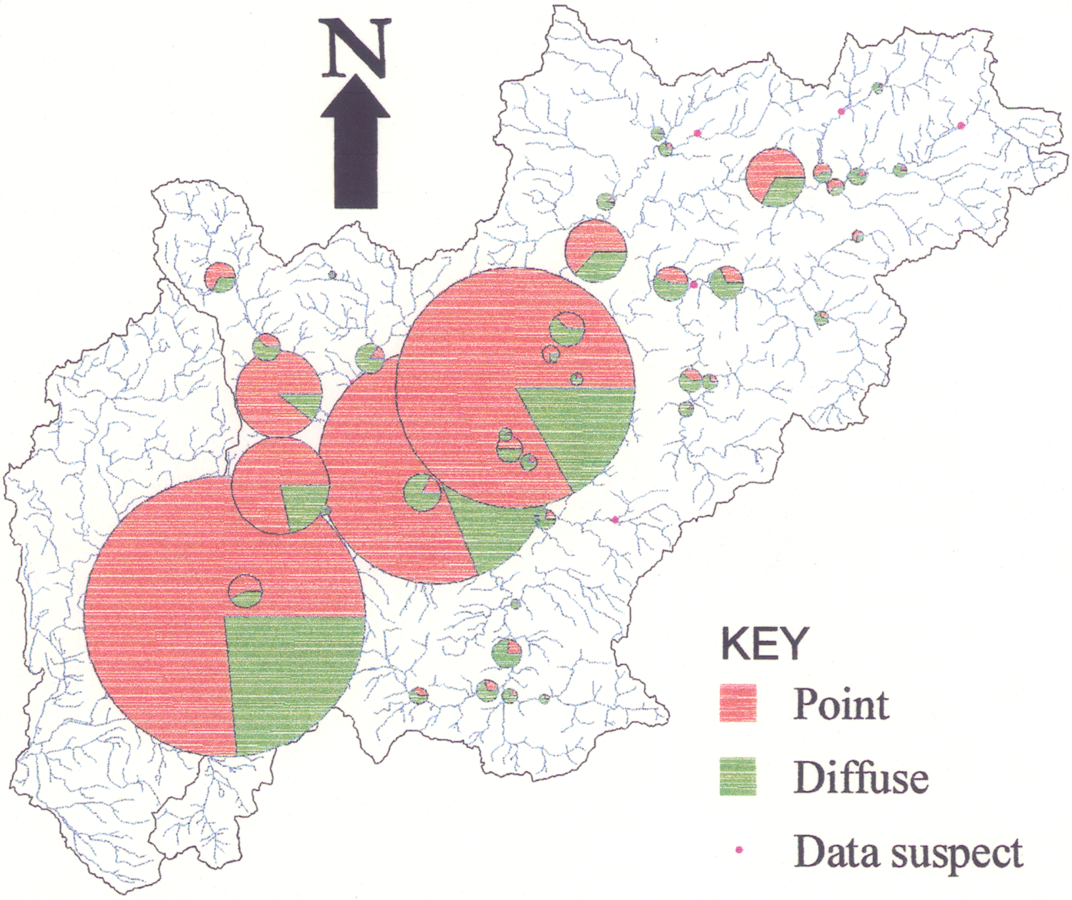

Figure 4.6 **Map showing the relative importance of point and diffuse source P in different areas of the catchment. The diameter of the pie charts is proportional to the average instantaneous phosphate load**

Some features of this map are based on spatial data licensed from the Institute of Hydrology, includes material based on Ordnance Survey 1:50000 maps with the permission of the controller of Her Majesty's Stationery Office © Crown copyright

These maps highlight a clear difference between this catchment and the majority of catchments studied in the literature. The population density in the Warwickshire Avon catchment is about 400 people per square kilometre whereas most catchments that have been assessed for diffuse source pollution have far lower population densities. The Windrush catchment studied by Johnes *et al* (1996) is typical of these with approximately 50 people per square kilometre.

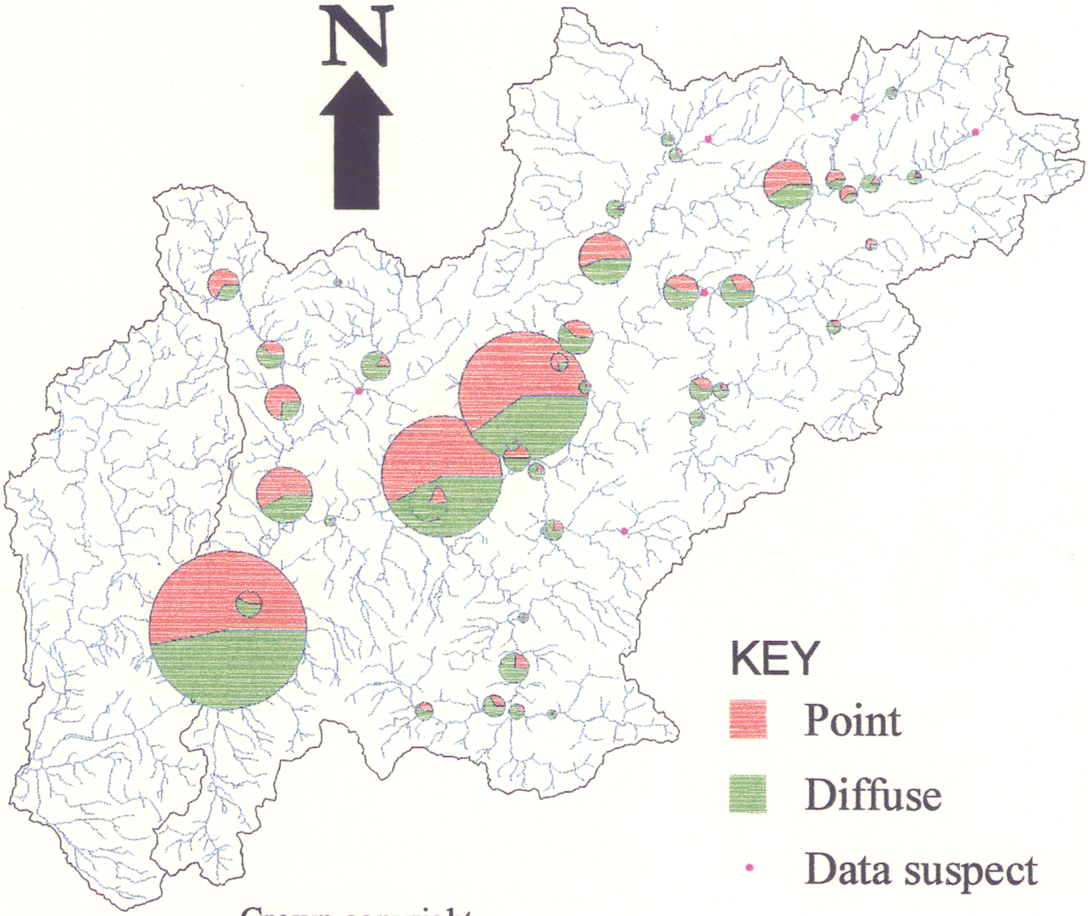

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Figure 4.7 **The relative importance of point and diffuse source P in different areas of the Warwickshire Avon catchment assuming full implementation of the UWWTD. Pie chart diameters are proportional to the average instantaneous P load**

Some features of this map are based on spatial data licensed from the Institute of Hydrology, includes material based on Ordnance Survey 1:50000 maps with the permission of the controller of Her Majesty's Stationery Office © Crown copyright

4.9.2 A comparison of the in-river load estimates obtained by different methods

The rating curve estimates of load were inspected, and the result from site 11053180 - Itchen (Marton, conf. Leam) was discounted, as it was clearly unrealistic. This site demonstrates the danger of the rating curve approach. With only 38 data points concentrated at low flows there was no way of producing a good estimate of load at this sampling station in the system. This point was therefore removed from the data set before any further analysis was performed.

The two sets of estimates of the in-river loads at each site were then compared on a simple x-y plot, figure 4.8. For ease of comparison a line indicating y = x was added. This plot clearly shows the majority of points fall near the line although the rating curve approach generally gave a slightly higher result. This is expected as the approach attempts to account for the higher loads that are exported during high flow events.

There are 6 points (marked a-f) that could be considered outliers since they do not fall near the line. These sites all had less than 30 samples, with no, or very few, points at high flows so that predicted loads were estimated from a model which had been extrapolated well beyond its calibrated region. Point (a) is the Alne at Little Alne, site 8101500, and has only 20 valid measurements. As all of these were recorded at low flows, it is impossible to predict the true total load through this site. Point (b) is the Sowe at Stoneleigh. This site is totally dominated by sewage effluent so that the main process at high flows is dilution. The use of the rating curve approach in this situation tends to be dominated by the dilution at high flows, rather than the increase in particulate P from diffuse sources. Points (c) to (f) correspond to comparatively larger catchment areas. Since flows are higher at these points, the difference between the two calculation procedures is expected to be proportionally higher.

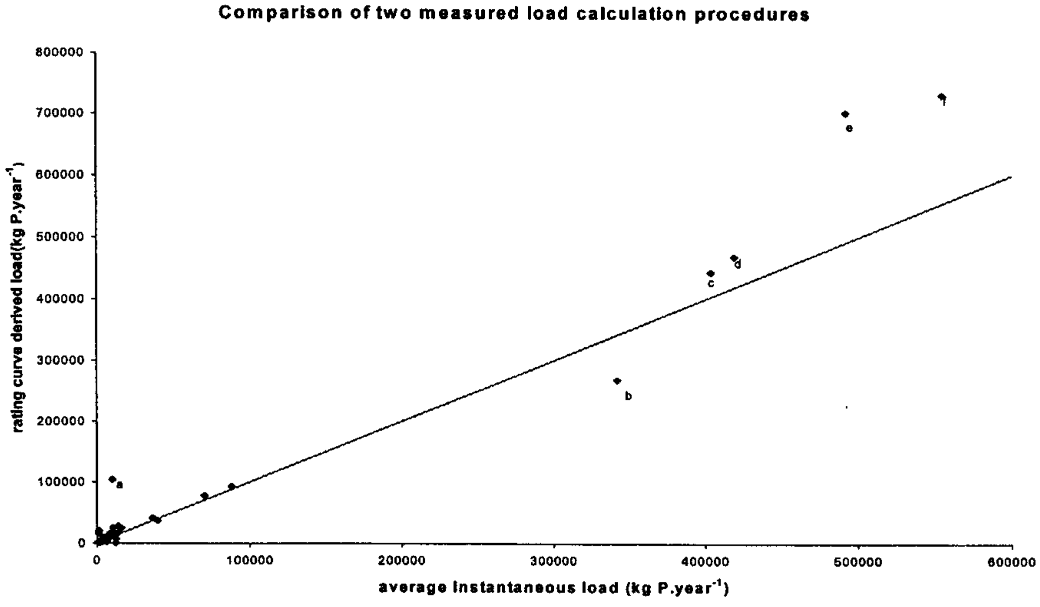


Figure 4.8 **Comparison of two load calculation procedures**

In order to obtain a better comparison of the two methods, a second plot was prepared using the loads normalised by catchment area (figure 4.9). Four new points fall clear of the ideal expected line. Again these correspond to sites where the rating curve has been extrapolated. Site g is the Avon at Kilworth which has 38 data points all at flows within the bottom third of the flow rage. Site h is the Dene at Kineton with 18 data points, all falling within the bottom fifth of the flow range. Site i is the Smite Brook at Coombe Abbey with 16 samples all within the bottom half of the flow range and site j is the Swift at Bransford Bridge with 19 samples all within the bottom half of the flow range.

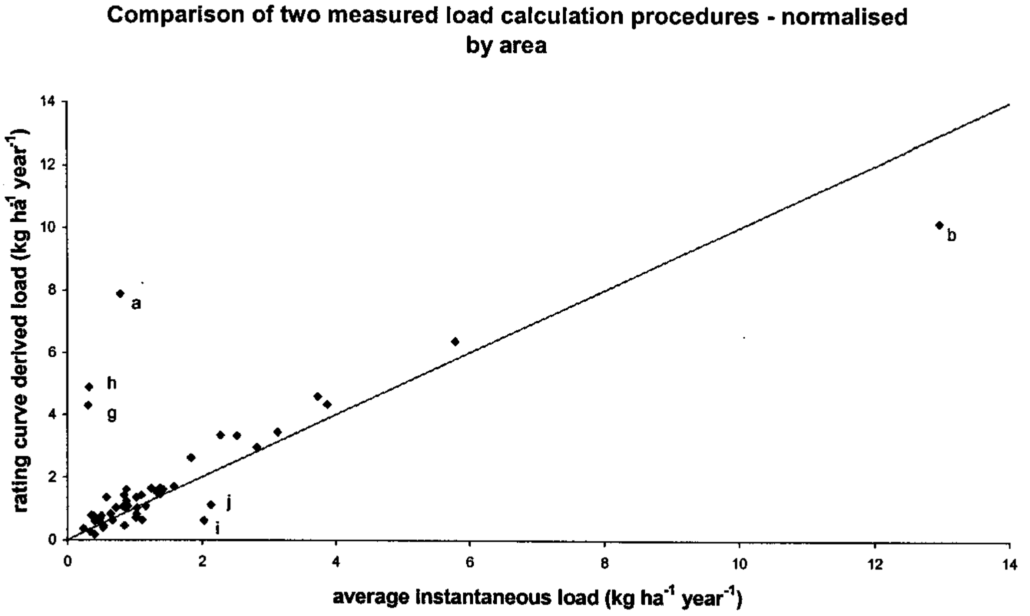


Figure 4.9 **Comparison of two load calculation procedures, with figures normalised for catchment area**

In the following discussion, the data relating to the sites identified above have been removed (sites marked a-j above). There are many other sites with the same restricted numbers of samples, where the two methods agree, as the data covers a wider range of flow.

4.9.3 A comparison of predicted and measured in-river loads

The in-river load estimates of the residual sub-group (i.e. those remaining after closer scrutiny of the data) were then compared to the load estimates of P inputs into the river system, derived from land use and other data. Comparisons were made both for individual classes of input (diffuse from land, diffuse from animal, point sources) and for their total. These results are summarised in figure 4.10 a-d. In each case, the diagonal line on the plot is a geometric regression line. This is a "least squares line" that gives as descriptive relationship between the two variables rather than a predictive one. The advantage of geometric regression is that it assumes that uncertainty exists on both axes. These lines have the equation:

$$R = aI + b.$$

where

R is the rating curve estimate of load
I is the input estimate being compared
a is the gradient and b is the intercept, both constants.

the constants for the lines in figure 4.10 a-d are summarised in table 4.3.

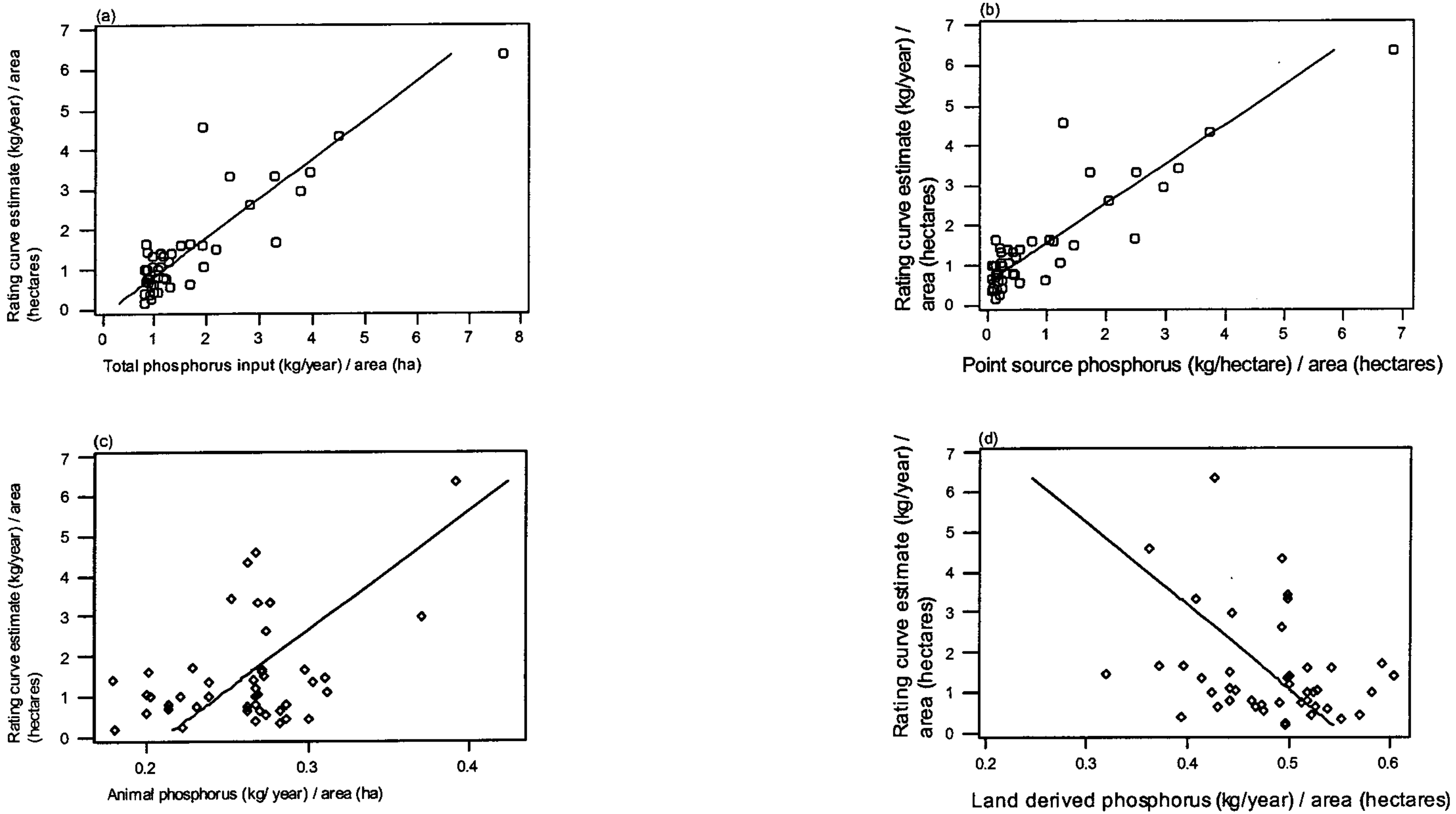


Figure 4.10 Comparison of measured load to full model and various sub models

Table 4.3

Input	**Figure**	**Correlation**	**Intercept, b** kg hectare^{-1} year^{-1}	**Gradient, a**
Ideal model		1	0	1
Total P inputs	4.10 a	0.83	-0.13	0.98
Point source P	4.10 b	0.84	0.59	0.99
Animal P	4.10 c	0.49	-6.32	29.92
Land derived P	4.10 d	-0.36	11.42	-20.65

Clearly there is considerable scatter in the plots, however the descriptive line for the total estimated P input compared to the measured load in the river does approximate the ideal line. There is a clear visual similarity between the plot for all inputs and the plot for just point sources figure 4.10 (a) and (b). There is a slight improvement in the correlation when diffuse sources are excluded (figure 4.10 (b)), which may suggest that the detailed between site variations in point load are not being adequately modelled by the export coefficient approach. This also demonstrates that variations in total load between sites in the catchment are dominated by point sources. However if the diffuse sources were neglected this would lead to an underestimate in the load calculations of approximately 0.6 kg hectare^{-1}year^{-1} which corresponds to approximately 20% of the total load at Evesham. It is therefore essential that some account of diffuse sources is included, to avoid underestimating the total load. A flat rate diffuse source applied to the whole catchment could be used in simple estimates to obtain a first impression of the likely importance of diffuse agricultural inputs in a large catchment. However, it would not predict any local variations in diffuse inputs, which would tend to be more important in small sub-catchments.

4.9.4 Comparison of predicted and measured diffuse loads

The export coefficient results were investigated further to clarify their performance in predicting local variations in diffuse load. The difference between the point source inputs and the measured load in the river was plotted against the export coefficient estimate of diffuse load – both normalised by catchment area (figure 4.11). Error bars representing the absolute minimum level of uncertainty that could be associated with each axis are included. The horizontal error bars represent a 1% net uncertainty in the export coefficient estimate of the diffuse load. This figure is extremely conservative as export coefficients are frequently determined to only 2 significant figures. The vertical error bars represent 5% uncertainty in the river loads and in the point source loads, based on the assumption that the flow measurements that these estimates depend on are unlikely to have an uncertainty smaller that this. Again, this is extremely conservative as it completely ignores the uncertainties in the load calculation methods. Error bars up to an order of magnitude greater than these could be considered reasonable, given the difficulty of obtaining reliable estimates of load and the uncertainties associated with extrapolating export coefficients from one site to another.

It is clear from figure 4.11 that even with the small error bars described above, there is considerable overlap of the error bars. This suggests that the differences in diffuse exports per unit area of catchment are too small, compared with the large load estimation errors, to justify further analysis. It is therefore not possible in the Avon catchment to demonstrate any significant variations in diffuse exports between sub-catchments with different land use. In the absence of significant variations in diffuse exports within the catchment, it is impossible to either confirm of deny the assumption that the export coefficients are modelling local variations in P inputs resulting from local land uses and stock densities. The only possible conclusion from this approach to testing is that the export coefficient approach is giving values that are correct within the wide margins allowed by the measurements available. It must be re-emphasised this catchment has far greater population density than typically found in catchments investigated for diffuse sources in the literature (see 4.9.1).

4.9.5 Comparison of prediction quality to catchment area

An alternative approach was used to investigate the quality of predictions for all sources as a function of catchment area. This comparison is awkward, as the quality of the load estimate itself is likely to be a function of the number of samples used, however it can be instructive. The percentage deviation between the estimate of P inputs, and the measured load using the rating curve approach, was plotted against catchment area, with the number of measurements represented by the area of the markers (figure 4.12).

The values of the highest deviations found appear to be approximately inversely related to the area of the catchments being studied, ie the biggest deviations occur in the smallest catchments. This pattern is consistent across catchments with different numbers of samples. This suggests that the model as a whole is failing to predict local variations in the smaller sub catchments, though there is no simple method of determining the causes of these variations. They could be due to either point or diffuse sources or to poor estimates of the in-river load. Catchments with an area greater than 25 thousand hectares typically have deviations of less than ±100%.

4.9.6 Comparison of load measurement quality to data quality

A similar approach was to investigate the quality of the load estimate, as a function of the number of samples. This comparison is complicated by the potential effects of catchment area – a small catchment has less processes occurring within it and will react more quickly to external events. This should make small catchments more sensitive to differences in monitoring and calculation procedures. The percentage difference between the rating curve load, and the average instantaneous load was plotted against the number of samples, with catchment area represented by the area of the markers (figure 4.13).

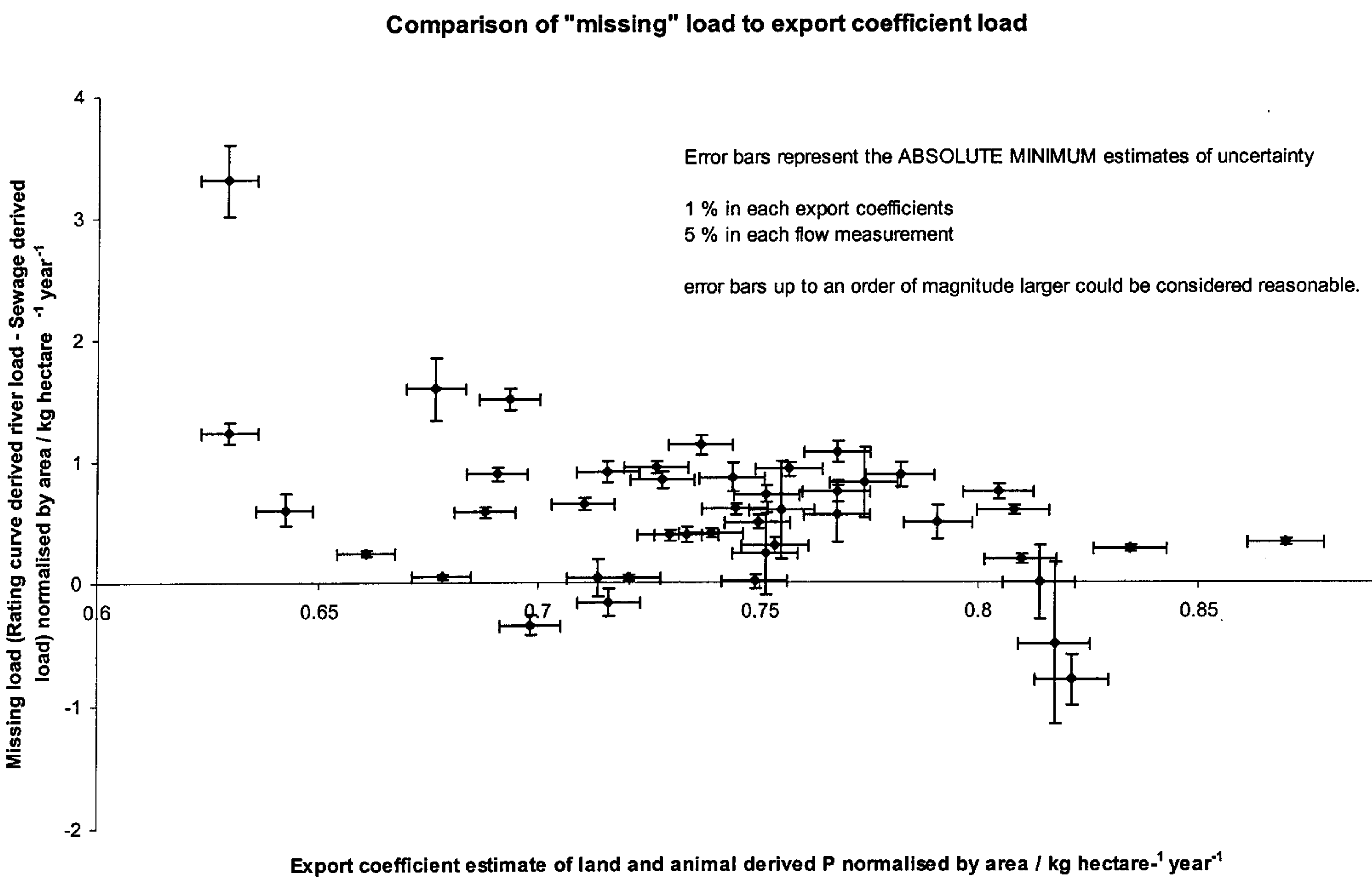


Figure 4.11 Comparison of predicted and measured diffuse loads

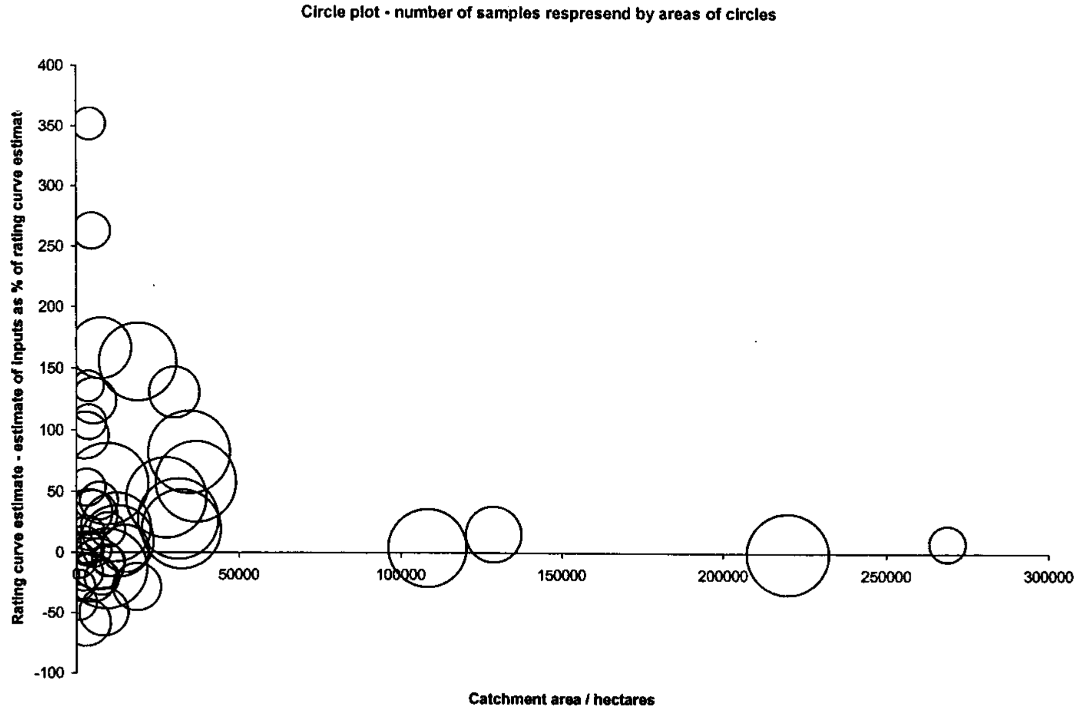


Figure 4.12 Deviation between modelled and measured loads as a function of catchment area. Circle diameter represents number of samples taken

Clearly, the points with more than approximately 50 samples have deviations in the range ±50%. This suggests that the estimates of load do not improve dramatically after a year of weekly sampling, or 2 years of fortnightly sampling. With fewer than 50 samples there is dramatic rise in the magnitude of the deviations between the different methods of calculating the total load. It is possible that better estimates of the total load could be obtained with fewer samples using a sampling program specifically designed for measuring the total load. This would have to aim to sample under the full range of flow conditions and would decrease the number of samples, at the expense of an increase the complexity of logistics and unpredictability of workloads for laboratories.

4.9.7 Phosphate speciation as a function source

Simple modelling procedures such as the ones used here can only predict total P loads or concentrations, which is a major limitation as a variable fraction of total P actually contributes to eutrophication. This proportion is known as "bioavailable" P (BAP). Alternative more complex process models can sometimes predict the forms in which P will occur, but they require very large data sets to run, and are open to misinterpretation, and misunderstanding. It is therefore important for all staff interpreting P data to be aware of the different types of P measurement available, and understand how these relate to each other in real samples.

A number of phosphate measurement protocols were used to generate the monitoring data used in this study (Foster, 1998).

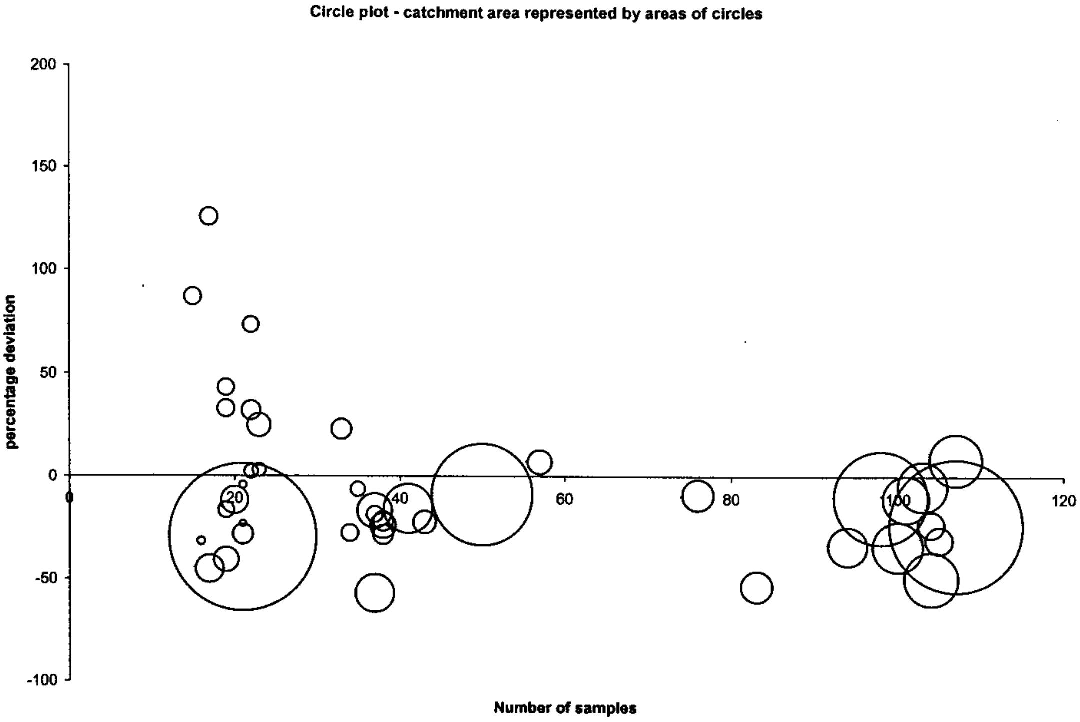


Figure 4.13 **Percentage deviation between the two methods of calculating load as a function of number of samples. Catchment area represented by area of circles**

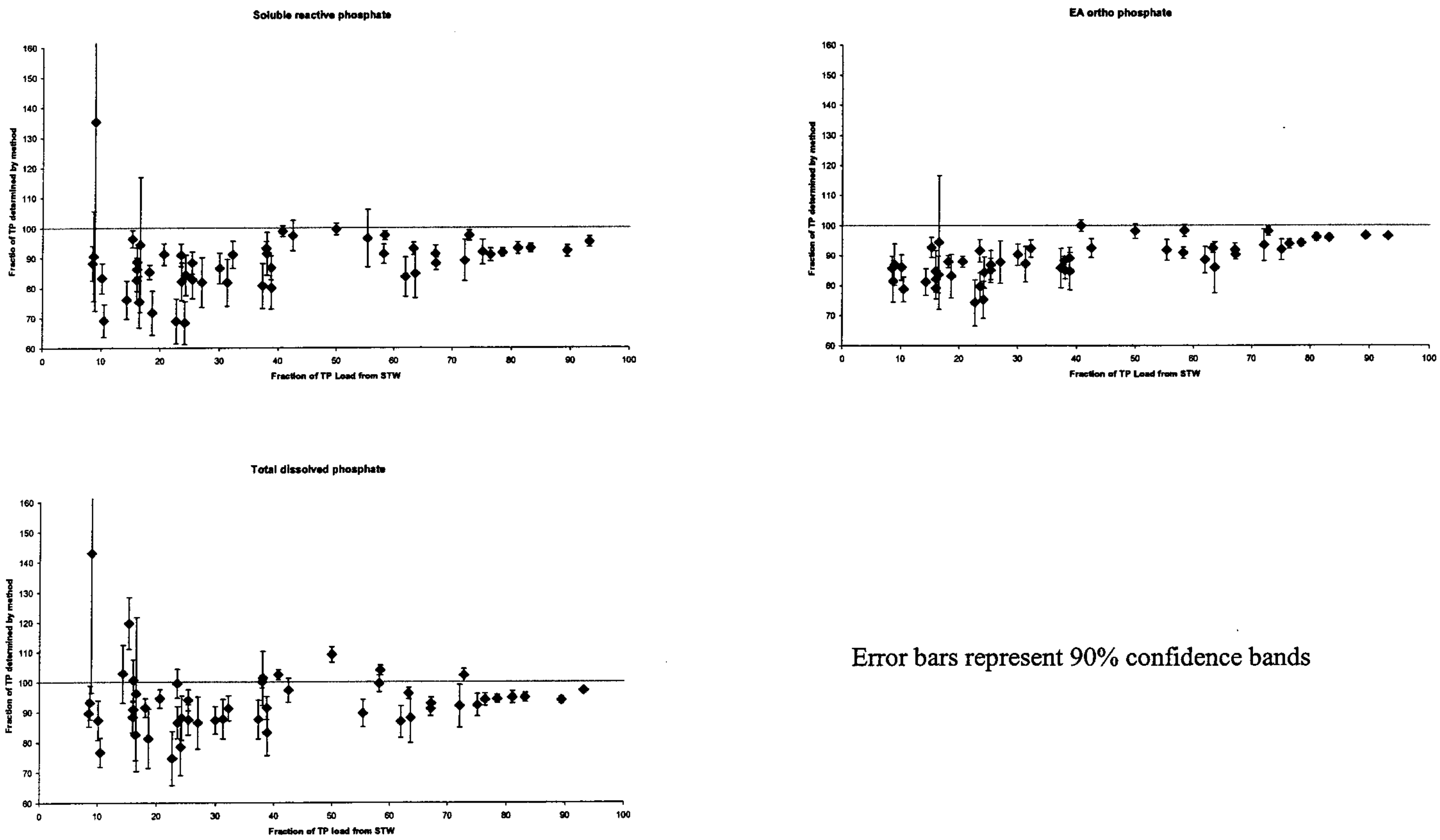
SRP	Soluble Reactive Phosphorus - the quantity of P contained in the sample that can pass through a 0.45 µm membrane filter, and reacts with the chemicals used to detect phosphates. This is mostly but not exclusively ortho phosphate $(PO_4)^{3-}$.
EA "ortho P "	EA documents and data sets frequently report "ortho phosphate" results, these are similar but not identical to SRP. The filtration step in the SRP protocol is replaced by a settling step.
TDP	Total dissolved phosphate - the total quantity of P in the sample that passes through a 0.45 µm membrane. The filtered sample is digested, to ensure that all P present after filtration is converted into a form that reacts with the chemicals used to detect phosphates.
TP	Total Phosphorus – a whole water sample is digested to ensure that all P present is in a form that can react with the chemicals used to detect phosphate.

A number of other protocols have been described in the literature of measuring BAP, including bioassays using algae (Gerdes and Kunst 1998) and Ferric oxide strips (Sharpley *et al*, 1995) but these are seldom used in routine monitoring programmes.

To investigate the relative quantities of P reported by the EA method listed above, the results for all protocols reported were expressed as a fraction of the TP measurement for each sample. These fractions were then summarised by site, and plotted in figure 4.15 against the modelled fraction of P load at each site derived from sewage. The error bars represent 90 % confidence intervals. Clearly all methods give similar results in waters dominated by sewage treatment works. This agrees with the results of Gerdes and Kunst (1998); showing that the majority of P in sewage effluent occurs in a bioavailable form (ortho phosphate). In waters dominated by diffuse sources far more complex behaviour is found. In all cases the SRP protocol should give the lowest detected fraction. At some sites the SRP and TDP methods appear to detect more P than the total P method. This is likely to be due to contamination during the filtration step, incomplete digestion or sample preservation problems. It should be noted that the variations in the detected fraction between sites increases as the diffuse sources become more important (low % load from STW). This is expected as a number of different P sources contribute to the "diffuse" load, including among others soil erosion, fertilisers, livestock (excreta and wastes) and subsurface flow. Soil erosion would be expected to contribute comparatively little P to the dissolved phase. Therefore, if a site's main source of P was soil erosion, then only a small fraction of the total P present would be expected to show up in the SRP, "EA ortho phosphate" and TDP methods. Conversely if the diffuse inputs are dominated by fertiliser run-off or animal wastes then the proportion of bio-available P (SRP, ortho-P, etc) would be expected to be a high proportion of the total P.

It should also be noted that the variability of the P fractions found at a single site can become very high when diffuse sources dominate. This is predicted as different diffuse P sources would be expected to react differently to rainfall and hydrological conditions. In general the dissolved phosphate forms would be expected to dominate the system during base flow conditions, as a result of subsurface flow, with other sources becoming more significant during rainfall as soil particles and or animal wastes are mobilised.

Clearly any model or procedure that assumes a fixed ratio of SRP to TP is using very large approximations, and cannot give precise predictions when applied to real problems. There may however be situations where such assumptions are justified as other factors also limit the quality of predictions. For example it may be appropriate to weight the results of simple export coefficient studies using bio available fractions, especially when trying to assess cost effectiveness in terms of eutrophication. This data also clearly demonstrates that although "EA ortho P" gives a reasonable description of the P content of sewage effluents, it does not give clear quantitative information on diffuse P sources. Therefore, the EA must consider incorporating total P measurement into any sampling programme that aims to quantify diffuse sources.



Error bars represent 90% confidence bands

Figure 4.14 **Analytical bias as a function of source type**

4.10 Conclusion

The modelling exercise has demonstrated that the majority of the Warwickshire Avon is dominated by point source P from sewage treatment works. Approximately 75% of the total P load and an even higher proportion of the bioavailable P load is associated with point sources. Initially policies to reduce P loads in this catchment should therefore concentrate on phosphate removal at point sources. A significant percentage of this point source load will be unaffected by UWWTD as it is derived from smaller works, and actions to control these will probably have to result from negotiation with the water companies.

The population density in the Warwickshire Avon catchment (~400 PE km^2) is significantly higher than that of the Windrush (~50 PE km^2 Johnes *et al,* 1996) from which the export coefficients are derived. Consequently the P from sewage works in the Warwickshire Avon catchment obscures the contributions of other (diffuse) sources. The subtraction of one large uncertain figure from another generates large uncertainties in the estimates of residual or non point source loads (figure 4.11). It is therefore not possible to validate the detailed predictions made by the export coefficient approach. A simple overview of the whole catchment, does suggest that the export coefficient estimates are likely to be correct. There is not enough evidence to conclude that the export coefficient approach has failed, and this is despite an extensive and expensive sampling program.

In a wider context, this failure to obtain definitive results from monitoring data may indicate that the export coefficient approach should be used as a screening procedure. There is evidence to support the use of the export coefficient approach to obtain approximate values for diffuse sources, and it is comparatively easy to estimate loads for point sources. This data could then be used to identify those catchments where the P sources are clearly dominated by one or other type of input. Additional resources and research could then be allocated to the accurate and precise measurement of total and bio available P loads in systems where the situation is less clear. With the limitations of current knowledge, such a monitoring program must include total P determinations, "Ortho P" is not adequate for any form of diffuse source phosphate monitoring. Total P is a comparatively expensive procedure in any laboratory, therefore more sophisticated sampling strategies should be considered to minimise the number of samples requiring total P analysis. This may require the use of flow proportional sampling or continuous monitoring, and may also involve more complex and detailed modelling than has been attempted here. It may also be worth investigating other automated/instrumental methods of determining P, to improve the Agency's ability to measure and understand P sources and loads.

Whatever the context it must be emphasised that with the current state of knowledge it is essential that model results are treated with a degree of caution. No matter how powerful the computer or how impressive the output, if the model contains any invalid data or assumptions the results will be unreliable.

5. THE ROLE OF MACROPHYTES AND BENTHIC ALGAE IN THE P DYNAMICS OF RIVER SYSTEMS

5.1 Introduction

Eutrophication is, in fact, a qualitative description of observed effects in a water body. Prior to the late '80s the majority of interest focussed on lake systems and it is only relatively recently that the description of eutrophication in rivers has been a topic of major research. There is clear evidence (Vollenweider, 1968) that, in almost all freshwater systems, P is the nutrient limiting the amount of vegetation (biomass) developed. In lakes the biomass usually takes the form of phytoplankton and it is now possible to model the processes involved in phytoplankton growth and decay in lakes using relative simple models (eg Hilton *et al*, 1992). However, in rivers the visible expressions of eutrophication are much more varied. The following have been proposed as indicators of eutrophic conditions in rivers:

a. too much planktonic algae (deep rivers).

b. too much benthic algae (shallow rivers).

c. excessive amount of filamentous algae.

d. too many macrophytes (highlighted in flood control).

e. reduced diversity of plants present.

f. move from macrophyte dominance to algal dominance.

g. regular DO dips due to excessive plant growth.

As a result, it has proved very difficult to model eutrophication in rivers. However, if scientifically defensible limits are to be set on acceptable nutrient levels in rivers, reliable models are a required to link nutrient levels with observable indicators of eutrophication in rivers. A literature survey was carried out to assess the present state of the art in these types of model.

5.2 Literature Search on the Role of Macrophytes

Literature quantifying the role of macrophytes in the P dynamics of river systems is difficult to find. There is no clear set of key words that is associated only with the relevant literature, therefore the following strategy was used to perform a literature search on the role of macrophytes:

1. A list of relevant authors was generated from papers already held at IFE River Laboratory, and a BIDS search was performed to find all papers by these authors with the word "phosp*" in the title, keywords or abstract. This resulted in 117 hits.

2. A BIDS search was performed using the combinations of up to four groups of key words for example:

   (phosph*) and
   (water or submerged or floating or emergent) and
   (macrophyte or plant or diatom) and
   (intake or uptake or flux or dynamics or model* or estimate*).

   Where * is used as a "wild-card" i.e. indicates any other letter, or groups of letters in the alphabet.

3. Abstract searches performed by library staff in:

   Aquatic Sciences and Fisheries abstracts and
   Water resources abstracts.

4. Workshops with Environment Agency staff.

The resulting lists of papers were transferred to local computers in a format that included abstracts. A preliminary manual sieve reduced the number of papers to 121 potentially relevant references.

A list of macrophyte models located by these searches was presented at the first two workshops with Environment Agency staff, who were invited to add to them; no additional models were put forward.

5.3 Discussion

The factors that are thought to influence macrophyte growth, and which have been used in existing models, have been reviewed by Carr *et al* (1997). These include light, temperature, nutrient availability (N, P and C) and water velocity. The authors also discussed the loss of biomass (eg. due to washout, death and decay). However, they note that these processes were usually modelled using empirical equations, due to a lack of mechanistic information in the literature.

Many papers detail the growth patterns or nutrient uptake kinetics of individual species of plants and algae. Although these are of use when setting parameters within models estimating biomass development in rivers and lakes, they are not sufficiently general for use in model development. Very few models reported in the literature include macrophytes (see below). Most of those that do, relate to lakes rather than flowing waters and seldom include or predict the effect of macrophytes on the nutrient load.

5.3.1 Lake models

MEGAPLANT (Scheffer *et al,* 1993) (Model Evaluating General Aquatic Plant Laws and New Theories) is a relatively simple generic simulation model for the dynamics of submerged macrophytes in lakes. The authors suggest that, although the model's qualitative results are useful, quantitative predictions are sensitive to the parameter values used. As reliable values for these parameters were not available from the literature, the prospects of accurate predictions were poor. This model does not include any explicit nutrient calculations. It simply assumes that they are present in excess of requirements. As such, this model is unlikely to be of use to the present study.

P C Lake (Janse, 1997) describes the competition between phytoplankton and macrophytes within the framework of closed nutrient cycles in a lake system, including the upper sediment. Top-down effects *via* the food web were included. A second model that includes nutrient cycling, macrophytes, phytoplankton and sediments has also been described (Asaeda and VanBon, 1997). Although these models may be relevant to the prediction of eutrophication effects in canalised rivers, the level of work required to extend their use to a system such as the Warwickshire Avon is beyond the current project.

5.3.2 Flowing water models

Existing models for flowing water are probably of limited use because they do not model the relationship between macrophyte growth and P concentration. For example, STREAM – Segment Travel River Ecosystem Autograph Model (Park and Lee, 1996) is a river system model that uses the moving segment approach in a cell-in-series system to model river flow. Although the model includes macrophytes and phosphate, there appears to be no interaction between the two.

Plant development in unshaded streams has been reviewed within the context of a conceptual habitat model (Biggs, 1996). The author postulates that hydraulic stability over a series of years governs the relative dominance of periphyton, bryophytes and macrophytes, while hydraulic variation on a shorter timescale affects periphyton biomass. This may be a starting point for deciding which groups of producers should be modelled in any given river system. A further consideration is the result of a survey of Florida spring runs (Duarte and Canfield, 1990) which revealed that the standing crops of submerged macrophytes were not significantly correlated to either total nitrogen or total phosphate. They were, instead, correlated with the percentage surface area of the water shaded by marginal vegetation. As might be expected, the maximum rates of daily production were significantly correlated with the standing crop and inversely correlated with the degree of shading.

A little more information was found in relation to modelling the growth and development of benthic algae. Benthic algal blooms have been studied in Danish streams with a pronounced seasonal change in discharge (Sand-Jensen *et al*, 1988). They were found to grow during a period in spring when the light reaching the sediment surface exceeded 8 E m^{-2} d^{-1}. The light level was controlled by water depth and turbidity in winter and by macrophyte shading in summer. A rapid exponential loss of benthic algae was noted during the summer, caused by sedimentation. However, there was little information on the exchange of nutrients between benthic algae and the overlying water. This is because all nutrients were assumed to be in excess of requirements and, therefore, their interaction was not modelled in this study. Some insight into this subject was provided by Cerco and Seitzinger (1997). These authors used a combination of modelling and field studies to examine the role of subtidal benthic algae in the eutrophication processes of shallow estuarine systems. Their results indicated that benthic algal uptake retained phosphate in the sediments when the light intensity at the sediment surface exceeded 150 μE m^{-2} s^{-1}.

It is not clear from the literature which factors and interactions are most important for modelling the removal of P from flowing waters by macrophytes and benthic algae. A number of studies have indicated that macrophytes are rarely nutrient limited and are, instead, controlled by light limitation, including the shading effects of periphyton and plankton (Phillips *et al*, 1978; Sand-Jensen and Borum, 1991). Estimates of the quantities of nutrients removed from the water by macrophytes are confounded by their ability to use nutrients from the sediment as well. Although an empirical model for estimating the relative importance of roots in P uptake has been proposed (Carignan, 1982), its validity may be questioned. The TP content of the macrophyte biomass has been estimated as the product of the plants' P content (by chemical analysis) and estimates of the standing crop from surveys (eg Casey and Westlake, 1974) and has been found to constitute a small fraction of the annual load. In contrast to the above, the release of P from dead vegetation has been studied in detail (Jewell, 1970).

5.4 Summary

Although various fragments of models were found that described one of the following sub problems

1. the transport of P in a river
2. the interaction of P with sediment
3. the uptake of P by a plant/algae
4. the growth of a plant or algae

no model or combination of models were found that could predict the "eutrophication" responses of a river from the water chemistry. There are some possibilities that may be worth investigation in a future project but they will require considerable time and effort before becoming useful tools. One approach would be to extend the work of Biggs (1996) to the UK, in an effort to predict in general terms which types of autotrophs to model. An alternative approach is to use a statistical framework to reinterpret the plant data collected nationally in recent surveys, to identify the controlling variables.

6. METHODS FOR REDUCING P INPUT TO RIVERS

6.1 Introduction

In order to develop a strategy for reducing P levels in a given river, a range of P removal techniques must be considered for use either on point sources or diffuse sources. In this chapter as complete a list as possible of the potential methods for reducing P is presented and the methods are critically assessed as to their usefulness in the near future.

6.2 Methods for Reducing Point P Sources

Brett *et al* (1997) listed five main types of approaches for removing P from sewage effluents:

1. Physical removal following chemical precipitation.
2. Modified biological treatment.
3. Crystallisation.
4. Alternative sewage treatment systems (fluidised bed reactors).
5. Sludge treatment for disposal.

Within each grouping a number of different variations of the technique were described. Within the main groupings the crystallisation approaches are only realistically applicable to effluent streams containing high P concentrations. A pilot scale system for normal sewage effluents has been tried in the Netherlands but it is still experimental. Similarly, although there are a number of less traditional methods being tried, such as fluidised bed reactors, they are still at the experimental stage and are not widely used. There are also a number of methods available for reducing the P content of sewage sludge but they are not appropriate for the reduction of P in sewage treatment works effluents. The result is that there are, in effect, only two approaches which have been shown to work reliably at the present time: chemical precipitation and modified biological treatment.

Both processes can be applied in a variety of different ways, each with a different efficiency and cost. Historically the Environment Agency and its predecessors have rarely specified the type of sewage treatment process required in order to ensure compliance with the relevant Environmental Quality Standards. Although it is likely that as biological techniques are used in more locations and are subsequently developed further, they will become the Agency's preferred long term option for P removal, the Agency does not currently recommend one P removal technique to the exclusion of another. Hence, it was agreed with EA staff, that work would concentrate on obtaining information on retrofitted treatment of the biological treatment stage effluent with ferric (or ferrous) salt, followed by settlement. This system is known to be able to operate reliably, is relatively simple to retrofit and can be used as a benchmark, against which dischargers may compare other systems they wish to use. Details of biological systems would not be specifically sought.

P.1 Chemical precipitation of P from sewage effluents

The process is extremely simple, consisting of the addition of a solution of either alum (aluminium sulphate), ferric chloride or a ferrous salt (chloride or sulphate)) directly into the effluent stream followed by simultaneous flocculation and settlement in the secondary tanks. In trickling filter systems the chemical is added immediately following the filters; whereas, following activated sludge treatment, it is generally added after secondary settlement in order to reduce the amount of chemical required and the amount of sludge produced. There are a number of plants in the UK regularly using this method. Removal efficiencies of about an average of 80% (range 70-90%) can be reliably achieved in long term operation by good operators. Bowker and Stensel (1990) give details of the operation costs in the US. Cooper *et al.* (1995) gives UK specific data.

P.2 Biological removal of P

Bacteriological removal of phosphorus is only practical in the activated sludge process, not in trickling filters. The bacteria are induced to take up "luxury" P in the aerobic process section by incorporating an anaerobic stirred tank at the beginning of the process. In this tank, bacteria in the recycled sludge are mixed with the in-flowing sewage feed and, in the anoxic conditions, the bacterial cells take up large amounts of short chain fatty acids. When these bacteria are subsequently subjected to aerobic conditions the cells utilise the stored fatty acids to grow very quickly. In the process they take up excess P from solution. P is lost from the system by removing excess sludge at a relatively slow rate. Efficiencies of 80% are reported as being easily achievable (Brett *et al*, 1997). The two main benefits of this approach, compared to chemical precipitation are: 1) significant reductions in chemical costs, even if inflow P concentrations rise; and 2) reductions in the amount of sludge produced, compared to chemical treatment. Bowker and Stensel (1990) give details of costs.

6.3 Methods for Reducing Diffuse P Sources

When considering how best to reduce the run off of P from diffuse, generally agricultural sources, it is prudent to identify where erosion occurs most frequently (MAFF, 1997) and where P concentrations in the soil are highest (Harper and Pacini, 1995b). Since the import of P to farms is generally 50% greater than the exports of P from farms in saleable products (McIntosh, 1994), the control strategies for reducing the transport of P to freshwaters can best be considered under three headings:

1. control/minimisation of P inputs;
2. soil erosion control (particulate transport);
3. control of P transported in dissolved forms.

A number of sources, given in the reference list (Appendix 3), were consulted with respect to management techniques for reducing diffuse sources of P to freshwaters. Within the UK a significant literature has developed on the methods to reduce the transport of suspended solids to rivers. Since a significant proportion of the total P is contained in particles, any techniques to reduce suspended solids in run-off will also reduce particulate P in run-off. A combination of US and UK literature were used for identifying methods of reducing P inputs.

The list in table 1 has been compiled from the sources and arranged so as to put together BMPs, which have the same underlying philosophy within a given farming type. A more detailed description of each BMP is given in Appendix 3.

It is important to remember that, as Tippett and Dodd (1995) pointed out, BMPs are not necessarily additive when used in conjunction. When a given BMP is used it may have an effectiveness of, say, x%. When the same BMP is applied in conjunction with another BMP, with an effectiveness y%, it may have a significantly lower effectiveness than the sum (x + y)%. Hence there is a limit on the cost effectiveness of adding additional BMPs. The most effective approach is likely to be a suite of BMPs covering different aspects of the farm operations, not a number all applied to the same aspect.

A subset of BMPs was agreed with Environment Agency staff (see chapter on cost effectiveness) which would be studied in greater detail to try and ascertain realistic cost and efficiency estimates. However, it was felt that there would be some benefit in obtaining a view from the farming community on the practicality of applying the BMPs in the UK situation. As a result, on 26 April 1999, the Authors paid a visit to Mr Alastair Leake of CWS Agriculture, who kindly gave his opinion on the practicality and likely uptake in the UK of the individual BMPs. These opinions were based on his own experience as the manager of a very large farming unit and from his experience as a national adviser to UK government on the interaction between the environment and farming. The results are recorded in Table 6.1a-f.

On the basis of these comments it is clear that a number of BMPs can be ignored as not being appropriate to the UK situation. Others will need either education, at the very least, and/or some financial incentive to the farmer to carry out relevant BMPs. Using these data as a basis for defining questions, it might be prudent for the Environment Agency to carry out a wider survey amongst farmers to confirm the generality of these findings.

6.4 Discussion

6.4.1 Non-transferability of US experience

It is clear from comparing Environment Agency pollution prevention and diffuse pollution control reports and, particularly, US reports of BMPs, that effectively none of the Agency reports give clear descriptions of BMPs (even in cases where US names imply a completely different BMP when translated into UK English, eg terracing). In consequence, lists of BMPs are appearing in the UK literature, which are not applicable to the UK situation or, for which there is no evidence of any reduction in P due to the application of a given BMP. In part this is a result of reports being written for different parts of the Agency with different areas of interest, such as:

- erosion/suspended solids control,
- nitrogen reduction,
- P reduction,
- pesticide reduction.

Many BMPs appear in a list for a pollutant for which it has minimal removal efficiency because it is efficient for another pollutant. (Grassed channels would be a case in point which are not efficient at P removal but may be efficient for suspended solids removal.) It is also clear that both efficiency estimates and cost estimates are often not appropriate for transfer (in many cases even after some modification) to the UK situation.

6.4.2 A direction forward

In order for the Agency to move forward, we recommend that the Agency carry out a structured survey on behalf of all four of the main groups interested in diffuse pollution as follows:

1. Using Agency reports and other sources (this report may be a starting point) make as complete a list as possible BMPs of recorded in the literature.

2. By visiting sites in the US and Europe and obtaining the relevant literature, obtain good descriptions of the BMP, how they are implemented and how effective they are.

3. Identify as far as possible published estimates of the efficiency of each BMP for each of the four pollutants separately and assess the transferability of that estimate to UK conditions. It may be prudent, where possible, to consider the efficiency of reducing dissolved and suspended components separately and the likelihood of increasing groundwater pollution as a result of decreasing surface water pollution. Where no efficiency estimates are available it may be possible to put some bounds on the efficiency from a knowledge of the operating principles of the BMP.

4. Consult with UK farmers and farming institutions (in an enhanced study along the lines reported herein) as to the applicability/likelihood of uptake of each BMP in the UK situation and factors, such as financial incentives, which might affect uptake rates.

5. On the basis of this information compile an authoritative list of BMPs which are applicable and reasonably effective, in the UK situation, to each of the four pollutants considered.

6. Use this list as a basis for:

    a. Re-assessing the need for UK trials to measure efficiency.

    b. Carrying out research work to quantify the effect on efficiency estimates of the main factors considered to influence variability in efficiency estimates, such as previous crop, slope, etc.

7. When it is clear which BMPs are appropriate to the UK situation and likely to be effective in reducing one of the four pollutants, carry out an assessment of costs to determine a cost-effectiveness estimate.

8. On completion of a study of this type it will be possible to move forward constructively to develop a useable tool for assessing the appropriate spread of resources between point and diffuse sources in order to reduce nutrient pollution in a surface water at a given site. It would also be possible to compile a subset of the best approaches to diffuse pollution control in the UK. Research work in the US (McCann and Easter, 1998) suggests that concentrating effort on a few well tested methods with good reduction efficiencies which are acceptable to farmers as part of their operational practices is more likely to be effective in reducing diffuse sources. It would then be possible to organise successful campaigns for farmers to introduce these appropriate methodologies.

Table 6.1a A list of BMPs for reducing the input of P to agricultural systems. The letter and figure in column 2 refer to a more detailed description of the method given in Appendix 3

Method	Ref.	Notes	Farmer's comments	Likelihood of uptake	
				Large farms	Small farms
Reduce fertiliser input while maintaining fertility	A1	Regular soil P testing + knowledge of crop requirements.	Fertiliser companies used to do free testing but now farmers have to pay for this. For some farmers, there has been a tendency to build up reserves of P. This is less likely at the moment with low agricultural incomes. High probability that testing would be used as a management tool if it were free to the farmer.	With incentives	With incentives
Use of machinery to locate fertiliser close to plant roots	A2		There are two main problems with this – the high cost of precision equipment; the need to reformulate fertiliser for different crops; so it is a high cost technique usually done by a contractor. The technique may be economical for high value crops such as potatoes. However, for lower value crops such as wheat, the technique will not be cost effective.	When financially appropriate otherwise no.	Very small
Better timing of inorganic P application	A3	To be available when plant needs it but to avoid storms immediately after application.	Normally done already by good farmers. When applying inorganic fertiliser because of cost.	Done already	Done already
Nutrient Management plans	A4	A method of accounting for all sources of nutrients to the soil, particularly manure, when balancing uptake and usage.	Data is widely available to arable farmers to calculate nutrient budgets but is rarely done in practice.	Improved uptake may require subsidised advice.	Improved uptake may require subsidised advice.

Method	Ref.	Notes	Farmer's comments	Likelihood of uptake	
				Large farms	Small farms
Liming of acid soils to utilise existing P reserves.	A1	Neutralisation of acidity allows more breakdown of organically bound P.		Generally done when appropriate	Generally done when appropriate

Table 6.1b. A list of BMPs for reducing the run-off of dissolved P from arable land to water. The letter and figure in column 2 refer to a more detailed description of the method given in Appendix 3.

Method	Ref.	Notes	Farmer's comments	Likelihood of uptake	
				Large farms	Small farms
Buffer strips	B1	30-100% reduction for nutrient. Have the potential for sorption sites on the soil to become saturated. Under these conditions the strip becomes ineffective at reducing P. Not effective where land is drained or in groundwater recharge areas.	This could be encouraged by the Countryside Stewardship Scheme but at the moment 50%+ of applications are rejected due to shortage of funds. Also, it would help if farmers were allowed to include land around the edges of fields to meet the "Set-Aside" obligations.	possible, particularly with incentives	possible, particularly with incentives
Wetland restoration	B2	Active encouragement of former wetland areas, particularly farm ponds to retain water and encourage settling of particles and biological use of nutrients by plants.	Sometimes it is in the interest of farmers to create such features. Large, ecologically aware farms draw up wildlife plans. This could involve restoring a wet area to a pond. This is partly to improve the scenic value of the area. In cases where wetland creation requires taking valuable crops out of production, there will need to be some external incentive (or regulation).	possible, particularly with incentives	Possible, particularly with incentives.
Artificial reed beds at the end of drainage channels	B3	Active planting of reeds in natural or artificial wetland areas. The combination of anaerobic mud with micro-aerobic areas in the root zone remove P.	Have been developed in CWS to deal with waste generated by a large dairy herd, but very expensive - £60k for a 200 cow unit	Possible for effluent treatment. Costs high for run-off treatment.	Too expensive.

Method	Ref.	Notes	Farmer's comments	Likelihood of uptake	
				Large farms	Small farms
Oxidation ponds	B4	Use of pond systems to settle out particles and encourage P uptake by plants and algae.	As for wetland restoration.	possible, particularly with incentives	possible, particularly with incentives
Critical area planting/integrated crop management.	C23	Selective choice of crops to suit soils and slopes, so as to minimise particulate and dissolved run-off of P.	See below: arable- particle run-off.	With some incentives.	Very small

Table 6.1c. **A list of BMPs for reducing the run-off of particulate P from arable land to water. The letter and figure in column 2 refer to a more detailed description of the method given in Appendix 3**

Method	Ref.	Notes	Farmer's comments	Likelihood of uptake	
				Large farms	Small farms
Reduce slope gradient/length					
Contour ploughing	C5	Ploughing along contours rather than up and down slope.	From the farmer's point of view, this is normally inconvenient and time-consuming. This is a result of the common UK field pattern on hills where they are generally long up and down the hill but fairly narrow. Hence it requires more frequent, more careful (and cumbersome) manoeuvring of tractors than other methods of cropping.	No take up likely.	No take up likely.
Contour cropping	C2	Sowing crops in rows along contours rather than up and down slope.	See contour ploughing	No take up likely.	No take up likely.
Hedgerow planting and management	C3, D3	Intercepts run-off since root line tends to be slightly higher than surrounding land. Plants use nutrients.	Hedgerow management has been highlighted as an issue by a number of farmer/ ecologist for a. Some incentive to replant degraded hedgerows would help.	Good with incentives.	Good with incentives
Terracing	C4	Intercepting run-off by creating ditches with high down-slope side.	Not practical in this country.	No take up likely.	No take up likely.
Intersperse grass banks or ditches	C1	To reduce speed of any overland flow.	Practical difficulties in operating machinery around such systems.	No take up likely.	No take up likely.
Strip cropping	C1	Intersperse strips of different crops so that low growing plants reduce run-off from areas above.	From a practical point of view, fields are not big enough for this practice. It would be very difficult to manage two crops in one field.	No take up likely.	No take up likely.

Method	Ref.	Notes	Farmer's comments	Likelihood of uptake	
				Large farms	Small farms
Permanent or temporary strips of grass sown parallel to the contours	C1		See strip cropping.	No take up likely.	No take up likely.
Intercept flow at the bottom of a slope by a grass buffer zone	B1		Possibility if narrow grass strips are effective otherwise it results in two crops, effectively being grown in the same field (see immediately above)	Likely to be acceptable with incentives	Likely to be acceptable with incentives.
Cover bare ground as quickly as possible –					
Triple cropping	C6	Keep a crop growing all the time to provide ground cover.	Management is too difficult.	No take up likely.	No take up likely.
Green manure cropping	C7	Rapidly re-sow bare ground with a fast growing crop that reduces erosion and can be ploughed in to provide a manure crop before re-seeding.	Only organic farmers tend to bring green manure crops onto the farm. For arable crops directly drilled in the autumn/winter, cereal stubble can be left in the soil. The view was taken that often this will do as well as planting a green crop in preventing runoff of P. For crops that can't be planted in winter such as potatoes and sugar-beet, often land is left fallow in the winter. It was suggested that some incentive could be given to plant green crop in the winter for these cases.	Education required if cereal stubble is not efficient at stopping erosion. (Experts believe the latter is the case.)	Education required if cereal stubble is not efficient at stopping erosion. (Experts believe the latter is the case.)
Crop residue management	C9	Leaving crop residues on the soil, e.g. chopped straw.			
Cover crops	C10	Effectively the same as green manure cropping.	See comments for green manure.		

Method	Ref.	Notes	Farmer's comments	Likelihood of uptake	
				Large farms	Small farms
Timing of sowing.	C11	Establishment of growth of winter wheat before winter protects against erosion. Maize is normally planted in the spring already.			
Under sow cover crop	C10	e.g. with maize sow clover underneath to give ground cover.	Considered impractical because of problems of managing two crops in one field.	Unlikely	Unlikely
Temporary grassland	C8				
Low disturbance tillage methods					
Conservation tillage or Non-inversion tillage.	C13	Examples of the types of practice include: Avoiding fine seed beds; not rolling seed beds; chisel plough and direct drillings.	Highly favoured. They allow more water retention, reduce P loading and develop a better soil structure. The equipment is very expensive (about £35,000); financially attractive on large farms. There are a lot of time savings involved. Smaller farmers may not be able to justify large capital cost (unless they are part of a farming co-operative equipment pool). An additional disadvantage is where herbicide resistant weeds are present. In this case, conventional ploughing is needed about once every 3 years. Again, larger farms may find it easier to bear the cost of running two systems. Dr. Vic Jordan has done some research on the benefits of these practices.	Already used.	Costs too high for use without major incentives.

Method	Ref.	Notes	Farmer's comments	Likelihood of uptake	
				Large farms	Small farms
Compaction management; Includes: Establishing tramlines after emergence; Shallow tine behind wheel; Using old tramlines and low ground pressure tyres.	C14	Evidence to show that erosion often begins on compacted tramlines.	Only recognised as a problem by a few farmers.	Education needed.	Education needed.
Maintain good soil structure –					
Use bulky organic manures	C16	Addition of organic manures improves soil.	If the farmer has it they will use it. Liquid slurries are less effective but cheaper to handle than straw/ slurry mixes.	Depends on availability of manure.	Depends on availability of manure.
Incorporation of straw residues	C16	Organic matter added in form of chopped straw residues.	If the farmer has it they will use it. Higher yielding crops generally generate more straw.	Only possible if cereals grown the previous year.	Only possible if cereals grown the previous year.
Grassland rotation	C8	Put a grassland break into the crop rotation pattern.	Only feasible where there is a use for the grass		
Buffer strips	B1	+ reduces particle run-off	See above.		
Wetland restoration	B2	+ reduces particle run-off	See above.		
Critical area planting / integrated crop management.	C23	Selective choice of crops to suit soils and slopes, so as to minimise particulate and dissolved run-off of P.	Not a problem for big farmers. Problem for small farmers under severe economic pressure, e.g. small dairy farmers who a) may find it profitable to convert from dairy to arable (thus ploughing up pasture) or b) are forced to grow forage crops (e.g. maize) on inappropriate locations.	With some incentives.	Very small

Table 6.1d. A list of BMPs for reducing the run-off of P from livestock production to water. The letter and figure after each method refer to a more detailed description of the method given in Appendix 2

Method	Ref.	Notes	Farmer's comments	Likelihood of uptake	
				Large farms	Small farms
Reducing sources					
Reduction in the dietary intake of P in intensive livestock rearing areas	A5	most nutrient import to farms comes in the form of inorganic fertilisers	Comment from feed supplier. P intake is considered to be a major factor in maintaining fertility. Farmers unlikely to change without good evidence to show it is workable without yield reductions or fertility problems.	Need results of major on-going trials to convince farmers.	Need results of major on-going trials to convince farmers.
Nutrient Management plans	A4, E5, A1	The nutrient content of livestock slurry should be included in considerations of nutrient requirement rather than just considering slurry a waste		Uptake depends on free analyses and advice.	Uptake depends on free analyses and advice.
Reassess the need for continued maintenance dressings of P.	A1	Especially when manure is added but also to take account of residual soil P following cropping.		Uptake depends on free analyses and advice.	Uptake depends on free analyses and advice.
Better timing of manure application.	A3	To be available when plant needs it but to avoid storms immediately after application.	This is a problem since most farms do not have sufficient storage for slurry until when plants need it and/or until weather conditions are good. See nutrient management plans – arable.	New codes of practice forcing change but costs high.	Unlikely without considerable financial assistance.

Method	Ref.	Notes	Farmer's comments	Likelihood of uptake	
				Large farms	Small farms
Dissolved run off					
Farm waste management plans	E5	A general plan to minimise waste generation on farms and its impact on the environment.	Spreading livestock manures is a very big concern. Farmers often don't have enough storage to spread the waste when the crops would benefit from its nutrient content. Instead manure is viewed as a waste. It is often spread on land at times when crops cannot use is such as in winter. The major drawback of transferring manure from one farm to another would be the cost of transport. Costs would also include storage on the other farm and spreading the waste on the land. Farms recently upgrading facilities must follow the Water Code resulting in a major cost of storage facilities (to allow 8 months storage).	Being forced to comply by codes of practice when upgrading facilities but costs high.	Unlikely without considerable financial assistance.
Create buffer zones to slow down run-off at critical places on a slope	B2				
Particulate run-off					
Location	E7 E3 E2	Locate outdoor pig units to minimise risk of erosion; Relocate livestock trails away from watercourses; Regular relocation of feeding troughs during out-wintering of stock.	Pig units should be located away from streams and steep slopes. Relocation of livestock trails can have health benefits for dairy cows but economics less of a driver for beef herds.		
Changing land use	C23	Introduce grass into the rotation;	Needs incentives	Possible if financial	Possible if financial

Method	Ref.	Notes	Farmer's comments	Likelihood of uptake	
				Large farms	Small farms
		Change worst affected areas to permanent grassland or woodland; Consider set-aside.		incentives	incentives
Create buffer zones to slow down run-off at critical places on a slope	B1	As a last resort, construct embankments or settlement areas to trap silt laden water (C17, B2, B3, B4) Plant new hedges or dig ditches to restrict run-off or redirect it from areas of high erosion(C18, C20,C21)	See comments above.		
Grazing land management	E4, E2 E6	Keep stocking densities low, particularly in wet weather, to stop overgrazing and maintain a strong sward; Reduce the occurrence of strip grazing fodder crops, particularly in winter	Kale crops for winter fodder can be under-sown with grass or clover but apart from the costs due to reduced yield + seed + management it also means that herbicides can't be used to control weeds.		
Bank erosion prevention and consequent reduction in direct manure inputs to rivers	E1	Fence river banks except for small watering area	Considered to be very important. Stewardship grants can mistakenly encourage poaching in areas which are hard to fence, e.g. banks of meandering rivers.		

Table 6.1e. **A list of one-off and general maintenance BMPs for reducing the run-off of P from farms to water. The letter and figure after each method refer to a more detailed description of the method given in Appendix 2.**

Method	Ref.	Notes	Farmer's comments	Likelihood of uptake	
General maintenance good practice.					
Ditch maintenance	D2 C19 C15	Clean ditches regularly; Grass waterways; return soil removed from ditches to eroding areas	All except grassing waterways generally considered a normal management practice.	Done already.	Done already.
Irrigation management	D1	Caution when irrigating slopes to avoid run-off.			
Incorporate small field margins into field regime.	C22	Effectively small buffer zones.	As for buffer zones.	Good with incentives.	Good with incentives.
One-off modifications.					
Intercept roof run-off to soak away	D8	Separate rainwater from roofs from polluted farmyard run-off.			
Intercept farmyard run-off	D7	Intercept and treat highly contaminated farmyard run-off before it reaches streams.	This is a big issue due to the pollution it can cause (mainly BOD). Most farms should now have it but very expensive. Even a small yard produces a lot of water and the smaller treatment units cost £8K+.	High due to high fines for polluting streams	High due to high fines for polluting streams
Reduce run-off onto fields from farm roadways	D4 F2 D5 D6	Direct run-off to drains; Culvert streams at crossings; Relocate access points to top of slopes rather than the bottom; Control vehicle movements.			

Table 6.1f. **A list of BMPs for reducing the transfer of P to water by bank erosion or from other land use types. The letter and figure after each method refer to a more detailed description of the method given in Appendix 3**

Bank erosion

Bank erosion is generally a minor source of sediment (10%) compared to agricultural run-off (Understanding buffer strips).

1. Stream banks should be stabilised (F1). A number of techniques can be used as described later
2. Fence banks when livestock are grazing, except for small watering area (E1)
3. Reduce overgrazing near riverbanks (E4)

Erosion in uplands

Avoid destruction of plant cover by livestock (E4) or recreation activities

Limit poaching around winter feeding areas (E2, E6, and E1)

Protect eroding areas by encouraging regeneration of appropriate vegetation (C6-C10)

Erosion in woodlands

In short rotation coppice, woodland or forestry reduce run-off and soil erosion

Keep cover plants or trash where possible

Avoid compaction by machinery or equipment

Take care when installing ditches, roadways and stream crossings

7. COST-EFFECTIVENESS ANALYSIS AS AN AID TO DECISION MAKING IN UWWT PROBLEMS

7.1 Possible Management Tools

Over the last decade and a half, the possibility that diffuse sources can be major contributors to eutrophication has been recognised. Two areas where this has been in the forefront of thinking are the United States and Australia. However, as a result of their legislative and funding structures, the majority of published studies from both countries have focussed on different aspects of the problem, rather than the problem of which strategy to adopt in a specific river rehabilitation project.

Over the last few years Australia has begun to spend large sums of money on river restoration. As a result, a number of published studies (eg those contained in Bunn *et al.*,1993; Raine and Gardiner, 1995) consider the problem of the policy and scientific framework of decisions on river rehabilitation. However, the lack of a legal framework in Australia, which requires action on specific objectives, results in the major effort being put into group decision making through "Landcare" groups, whose objective is to look at a wide range of problems (which generally include eutrophication and siltation but also salination, among others) and identify the approach(es) which will be acceptable to the majority of users. Because there are few direct uses for Australian rivers per se, the justification for a project is based on a cost benefit analysis, a process which is unnecessary in UWWT Directive decisions. Rutherford *et al.* (1998) outlined the whole development process, working from goal setting through identifying constraints on achieving goals, strategy development, designing the methodology and choosing the tools for implementation and evaluation, including eventual maintenance schemes. They, like us, highlighted the difficulty in setting goals from a scientific basis but found the major problem in Australia, to be the assessment of need. This is different from the situation in the UK where the case is established by law. Hence there is little published work from Australia on the balance of cost and effort between reducing nutrient losses from point and diffuse sources.

Rutherford *et al.* (1998) suggested one approach, Multi-Criteria Analysis which might be applicable in the UK. In this case each factor is assessed in terms of its economic, social and environmental importance. The scores are then weighted on the basis of focus group discussions. In Rutherford's work the final weightings were 40% economic, 20% social and 40% environmental. Multi-criteria analysis is an alternative to cost-benefit analysis for setting policy objectives. It is less useful as a tool for deciding the best mix of methods to implement to achieve the objectives of a policy, such as the UWWT directive. In the more specific area of the search for alternative proposals to solve the problem(s), Stewardson (in Rutherford *et al.*, 1998) specifically states that "no systematic method is currently available to select the optimum restoration design".

In the US soil erosion, leading to high sediment loads in rivers, has been the major area of concern with the concurrent transfer of nutrients and pesticides as secondary, though still important issues. As a result, there has been an inherent assumption in all planning that diffuse sources of materials must be reduced (eg US EPA, 1985 and references therein). Although in many sub-catchments this can be shown to be the major source of nutrient or pesticide pollution, the importance of tackling sources of diffuse pollution can be magnified when the source of funding becomes important. For example, Holland and Klimek (1985) reported an offer by one State government such that "If local government take certain actions to reduce non-point run-off in their jurisdiction then State government might not have to require P removal at local government treatment plants". Hence in most US work there is an implicit assumption that diffuse sources must be reduced. The balance of effort between point and diffuse sources is seldom discussed in the literature.

However, a few authors have discussed the methodology required to apportion effort between point and diffuse source control. Baker and Skavroneck (1985) reported the development of an approach in Wisconsin which included five steps:

1. Problem identification.

2. Analysis of the relative contribution of point and non-point pollution sources to the water quality problem.

3. Definition of water quality improvement objectives with various combinations of point and non-point source controls.

4. Assessment of the ability to meet water quality improvement objectives with various combinations of point and non-point source controls.

5. Development of a management plan recommending appropriate point and non-point source controls.

This is a useful summary of the major steps and will be considered further in this section.

Day (1992) suggested cost effectiveness at the catchment level as an appropriate framework for water resources planning and management. He suggested that the least cost concept is one way of integrating ecology, economics, technology and institutions into a framework for cost effectiveness analysis. He envisaged this approach kicking in once a series of targets (both quantitative and qualitative) for water quality at the location under study have been identified. He then proposed five steps:

1. Use aquatic biologists to define what improvements in appropriate aquatic water quality indicators would be required to achieve the targets, eg how much habitat would need to be rehabilitated to improve the fishing by a certain amount; what maximum turbidity or e-coli count is acceptable before swimming is possible, etc.

2. Use water quality modellers to define how the indicators identified above are linked to those pollutants that need to be controlled, eg secci disc and water clarity; P concentration and algal growth.

3. Use models to determine what reduction in pollutant discharge is required to bring about the required change in indicator levels.

4. Divide the catchment into the sub-catchments of relevant tributaries. Identify the point and non-point discharges of relevant pollutants in these sub-catchments and their discharge patterns. For each of the major discharges estimate the costs of

reducing pollutant discharges by different amounts and convert to costs per tonne of pollutant reduced for each discharge reduction approach.

5. Select the least cost combination of measures to achieve the level of discharge reduction specified for the target level. The general approach would be to identify the reduction method with the lowest cost and keep adding reduction methods with the next lowest cost until the in-river target level of pollutant is reached.

Day continued by suggesting that this information could then be used to help set policy by including within the decision the target level that the general public would be willing to pay to achieve the desired outputs in the receiving waters compared to the use of the money elsewhere. He gave an example of the use of this approach in the Fox/Wolf river watershed in north-east Wisconsin and Lake Michigan. The analysis showed that:

a. A 50% reduction of P entering Green Bay (Lake Michigan) could not be achieved by reducing point sources alone.

b. A small segment of the agricultural land area contributed the majority of P.

c. As a result, the cost of reducing P and suspended solids from agricultural sources was 1% of the cost of removing them from point sources.

In a slightly different approach, Johnson *et al.*, (1994) defined four steps in the development of a management strategy:

1. Define a set of desired objectives for water quality/resource use.

2. Project a number of economic/demographic trend scenarios onto the conditions in a base year (chosen arbitrarily) to produce a selection (four – in order to incorporate some appreciation of the likely variability) of predictions of conditions at some time in the future (Green Bay: base year 1990, predictions 2010). These predictions assume no additional funds will be spent on water resources/water quality management beyond the level of spending in the base year. This development is not, at this stage to develop a management strategy which will meet the 2010 objectives for, say, water use, water quality, water based recreation and habitat protection but, rather, to make sure that management strategies identified in later stages will be effective under the future conditions.

3. Estimate the effects of projected pollutant loads under the alternative projections and compare them with the desired water quality (and other) objectives.

4. If the desired objectives are not going to be met then develop a least-cost management strategy to achieve the objectives.

The project concluded that cost-effectiveness was a much better approach to the problem of resource allocation than cost-benefit analysis, since many benefits are not easily quantified in monitory terms. However, there is still a requirement to justify the initial water quality and resource objectives, which may require the use of cost-benefit analysis. They also found that the objectives must be clearly defined. Hence, in their study, water quality objectives were clearly defined in easily quantifiable terms. Conversely, habitat objectives, although apparently clear, were actually found to be inherently vague and difficult to quantify so that cost-effectiveness analysis was not possible.

An important feature implicit within the analysis was that the final decision should not be based solely on the cost-effectiveness of a P reduction method but on the source load weighted cost effectiveness, ie the total cost of a given P reduction method is given by:

Total method cost = total load from a source x cost per kg P reduction,

Which for a BMP becomes:

Total method cost = land area x kg P lost per hectare x % reduction from BMP x cost per kg reduced

The final combination of methods may also rest on the "quantum effect" of modifications to a plant, ie the effluent from a plant can only be treated or not, so that the costs of partial treatment may be almost the same as for full treatment. Similarly, Johnson *et al* (1994) showed that it is also important to include any upstream pollutant traps in a system when estimating most cost-effective management plans. In other words if a pollutant, say suspended solids, is partially or completely trapped by a freshwater feature, eg a lake, then BMP application upstream of this feature will be of little benefit downstream of the feature.

7.2 A Way Forward

By combining the approaches proposed above, the following framework can be obtained:

1. Identify the problem and its cause, eg eutrophication results from too much P.

2. Define a set of desired objectives for water quality/resource use in the river.

3. Estimate the relative contribution of point and non-point pollution sources to the water quality problem.

4. Given the types of discharges in the catchment, identify for the major discharges (point and diffuse) which P reduction approaches (BMPs, chemical treatment of point sources, etc) are worthy of consideration given practical considerations (farmer acceptability, practicality, etc).

5. For each of the major discharges (point and diffuse) estimate the costs of reducing pollutant discharges by the different methods and convert to costs per tonne of pollutant reduced for each discharge reduction approach.

6. Rank the methods in terms of cost effectiveness.

7. Starting with the most cost-effective method, assess its ability to meet water quality targets in the river after removing a realistic proportion (ie using realistic values for the process removal efficiency) of the P load in the effluent (point or diffuse).

8. Continue down the list of ranked reduction methods until the in-river quality target has been achieved.

9. [If required, project a number of economic/demographic trend scenarios onto the conditions in the base year to see if the approach has a significant lifetime.]

10. Develop a management plan recommending appropriate point and non-point source controls.

Step 3 to 9 can be incorporated into the conceptual management model shown in figure 7.1. It consists of three main sections, point sources, diffuse sources and the river system. The point source load in most systems can be estimated as the sum of the loads from the individual sewage works. The individual loads can be calculated from either the number in the contributing population, or from regular monitoring data. In practice the best estimates are likely to result from the latter approach, where the data exist. At every works there are theoretically many management options to reduce the quantity of P in the final effluent, which simultaneously, increase the cost of treatment.

Similarly, the P load from diffuse sources can be estimated as the sum of the contributions of pollutant exported from a number of different land uses and management types. For diffuse sources there are two different levels of management, relating to changes between and within land uses. A change between land uses would indicate a change in crop type or farming type, say from maize growing to permanent pasture, resulting in a change in the total phosphorus exported from diffuse sources. Changes like this would probably have to be brought about by the use of subsidies or regulations. This approach will have associated direct and indirect costs. However, these changes are not under the direct control of the Agency (although the Agency may influence government policy) and will not be considered further in the discussion of this model approach (a brief overview of the methods is given in chapter 9).

At the second level the details of the land management (BMPs) under an existing land use can be altered, again with associated costs. From the estimate of the P contribution from the two source types (point and diffuse) and, knowing the annual flow in the river, the concentration in the river can be estimated. This can then be compared with a target concentration in the river to assess the effectiveness of achieving the target by a combination of remediation measures on both point and diffuse sources. (It should be noted that implementation of even a very cost-effective BMP may need to be encouraged by the use of subsidies or regulation if it is not seen as appropriate by farmers and land owners.)

A simple consideration of previous chapters of this report shows that the expression of eutrophication in rivers is very variable, unlike the case of a lake where planktonic algae are generally the dominant effect. As a result, at the present time, no models are available which are capable of making realistic predictions of the distribution of the effects of eutrophication between planktonic algae, macrophytes, benthic algae and filamentous algae for a given P load/concentration in the river. It was agreed at the workshops that this is a key parameter limiting the use of management models for directing P control strategies. It was felt that the development of this link is beyond the scope and time scale of this project. Some data from the Mean Trophic Rank (MTR) program indicate that in-river concentrations must be below 0.1 mg/l P to make any difference to observed effects (F H Dawson, Pers. Comm.). However, below this trigger level it is not possible to set more detailed targets. Research developments could be used in a future programme to rationalise the setting of phosphate targets in rivers but, for the present, it will be assumed that a target level will be reached by a combination of political expediency and local knowledge.

By using a model such as this it would be possible to try different combinations of approaches (derived either from political considerations, eg the cost should be equally shared between diffuse and point sources, or from the addition of BMPs ranked according to cost effectiveness). However, no such model exists at the present time. We suggest that the Agency should consider commissioning the development of such a model for use in nutrient, pesticide and suspended solids applications.

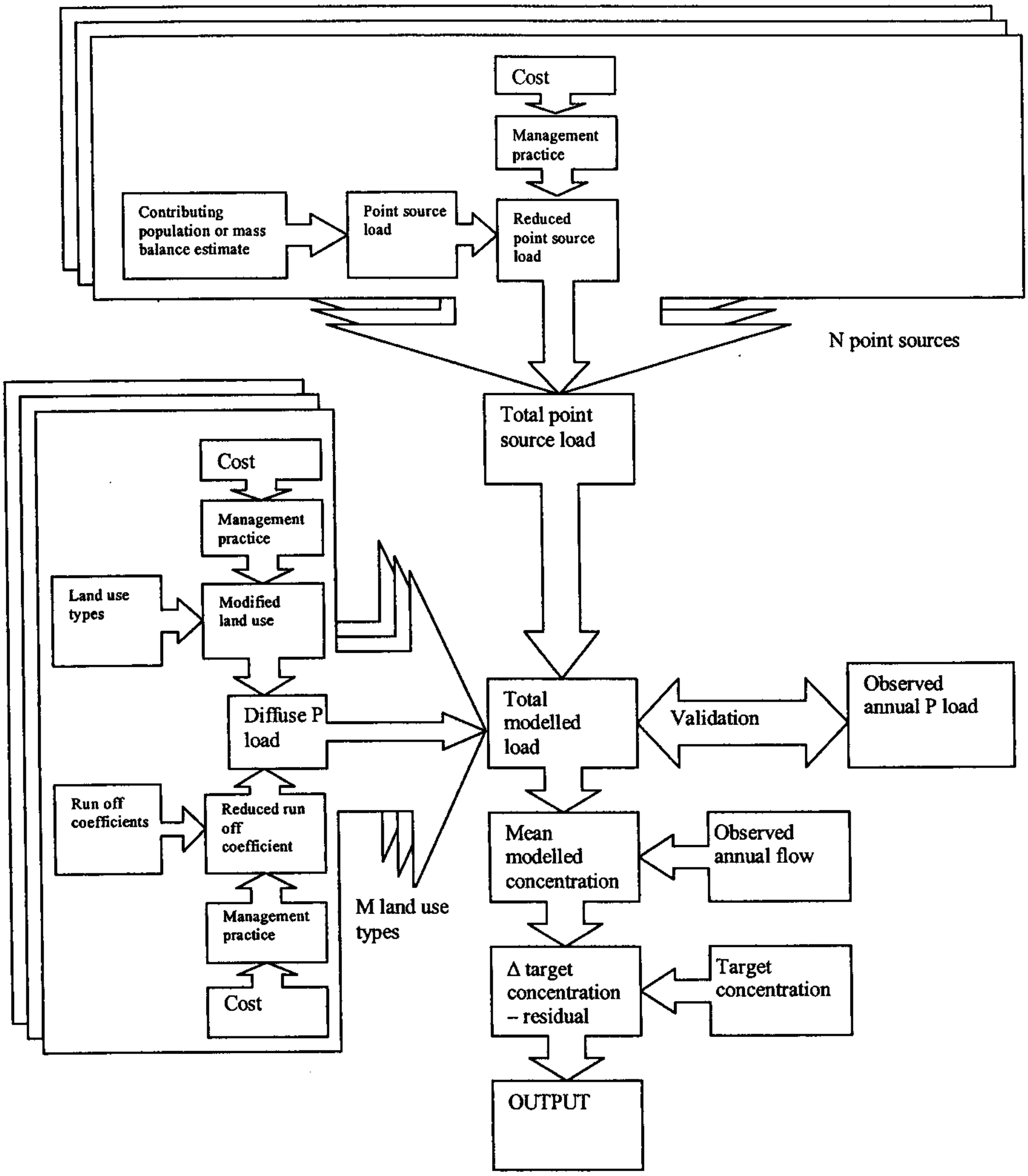


Figure 7.1 Conceptual management model

8. COST-EFFECTIVENESS OF MEASURES TO REDUCE PHOSPHORUS

Reducing P pollution cost-effectively involves implementing measures that give maximum efficiency for least cost. Ideally, the ratio of costs to effectiveness would be computed for each measure. For example, for each agricultural BMP measure one would obtain the annual cost per hectare divided by kilograms reduced per hectare. P-reduction methods could then be compared on a common basis to select the measure that would reduce P most cost-effectively.

Very little data is available in the literature which is either accurate or appropriate to the UK situation. Tippett and Dodd (1995) reported calculations of cost effectiveness for a number of processes in the Tar-Pamlico Basin in the NE USA. These included a number of BMPs for reduced loss of P from animal manure disposal such as: land application at agronomic rates compared to excessive rates of addition; anaerobic lagoons, followed by agronomic rates of land application; storage ponds; composters for dead bird carcasses; dry stacking of poultry manure prior to spreading. Calculations for the first two make some unexplained assumptions which, do not appear to be mathematically correct. The assumptions include: a) spreading at agronomic rates compared to higher rates reduces the amount lost per hectare whilst, simultaneously increasing the number of hectares required in proportion giving no overall improvement; b) figures of 50% reductions in the P content of slurry following anaerobic digestion, whereas, P is conservative under these conditions (N is not, however). The other two methods are very specialised and will not be discussed further. They also gave costs for the implementation of water control structures which, in their systems, produced reductions in N and P run off by maintaining high water levels in canals to reduce the threat of drought stress. The authors of the original study on which Tippett and Dodd based their work (Evans *et al.*, 1991) suggested that the nutrient reduction was attributable to decreased outflow, increased crop uptake, increased denitrification and increased sedimentation of P. However it is not very effective for P. P tends to decrease in systems where the outflow is via surface run-off but increases losses where they are via the soil profile (Tippett and Dodd, 1995).

Even when estimates of costs have been found it has been extremely difficult to find the basis for the estimates. Water Quality Management Ltd (1995) reported the same problem. In order to be able to reassess costings on a regular basis, as new developments require, the Agency should create and maintain a "recommended costing" archive which gives the basis, including any assumptions and base data, upon which cost estimates have been made. Without this archive the Agency will continually have to reinvent the wheel when a question is asked about the basis of a costing or an update is needed.

8.1 Uncertainties

In part the lack of data available for cost-effectiveness estimates results from the uncertainty in the estimates of efficiency and the fact that both the costs and effectiveness of measures are highly site-specific. Tippett and Dodd (1995) highlighted the uncertainties associated with their (and all) estimates of BMP cost effectiveness. In animal waste management practices they identify two main problems: a) the difficulty in deciding what is the baseline against which to assess P run-off reduction. They used agronomic rates of manure to land but this is likely to change depending on the crop to be grown. In the States it is also defined in terms of

N, not P; b) the reduction in P is only as good as the farmers' adherence to the rates recommended by nutrient management guidelines. Without incentives this will probably not be maintained at the optimum level.

Almost all efficiency estimates are site dependent. Key parameters determining apparent efficiency are the pre-BMP crop and the pre-BMP rate of fertilisation. However, surface slope and soil type are major factors, as are distance from the stream and climatic conditions. Without some attempt to obtain an indication of the relationship of these variables and reduction efficiencies, reliable, portable cost efficiency estimates will remain elusive.

Very few data exist which unequivocally give an estimate of reduction efficiency, even allowing for the uncertainties introduced by the factors highlighted above. This is because, in natural systems it is very difficult to isolate the effect of a given BMP from other effects, particularly when, as is often the case for practical considerations a number of BMPs are implemented in a package over a period of time. In addition many estimates of efficiency are based on model assessment rather than actual measurements. Hence the reliability of the estimates is a totally dependent on the ability of a model to accurately describe the processes involved both before and after BMP introduction.

Camacho (1990, 1992) indicated that many in-field BMPs that reduce nutrient run-off by increasing the infiltration route lead to significant amounts of the nutrients being lost to groundwater. This effect needs to be taken into account when assessing effectiveness.

Many studies assess effectiveness over short time scales, often only a single storm event, at best a few weeks. However, it is likely that the efficiency of BMPs will reduce with time. This effect is not incorporated into any efficiency estimates or BMP lifetime estimates.

The information requirements of knowing exactly what measures are cost-effective are immense. There are many specific problems in addition to the problems with efficiency estimates outlined above. Not least, there are equivalent problems with estimating costs which are also highly farm specific.

In particular, there is the question of what exactly we mean by costs. One can talk about financial costs, which are costs incurred by the farmer or the Environment Agency. One might also talk about economic costs, which is the resource or opportunity cost to society. For example, from the farmer's point of view, any foregone agricultural production as a result of implementing pollution control measures, is a cost. However, in a broader social (and economic) perspective, foregone agricultural production is a lower cost because this is heavily subsidised by the taxpayer.

In this study, we will be concerned with the financial cost of BMPs to the farmer and the Environment Agency. We do not consider any savings in agricultural subsidies that may result as a consequence of introducing BMPs. From the farmer's perspective, the cost of a BMP is the direct costs of implementation (ie capital cost and maintenance cost) and the opportunity cost of foregone production. In some cases, one or both of these costs will be zero.

In this study, various point and diffuse measures are selected and an attempt is made to compute average costs and effectiveness. We first deal with point source pollution, where measures of cost are more straightforward. Then we discuss the issues in reducing pollution from diffuse sources. During a second workshop held at Blandford, on 22 October 1998, attended by P Buckland, R Huggins R Dils, J Hilton, G P Irons and S McNally, EA staff agreed that data on both the effectiveness and costs of almost all BMPs were not in a readily accessible form, even when they were available. The meeting also recognised that although many data were actually known, in specific research centres, they were not generally available. As a result, an example set of BMPs was agreed for which data on effectiveness and costs would be found, either from literature sources or from equipment/service suppliers. This allowed the project to proceed with the concept development, without diverting too many resources to data collection. The agreed list was:

a. Reduced fertiliser input while maintaining fertility.

b. Farm ponds.

c. Conservation tillage.

d. Reduction in slope length using break crops.

e. Reduced dietary intake of P in livestock.

f. Farm waste management plans.

g. Low stocking density.

h. General maintenance/one-off precautions.

8.2 Point-Source Pollution

Data on the costs of point source pollution are quite difficult to obtain because of the commercially sensitive designation assigned to it by water treatment companies. There is some data in the literature, a selection of which is given in table 8.10. However, it should be recognised that very little of the data is immediately applicable since costs and efficiencies generally refer to total plant costs, not the more usual additional costs of P removal. Similarly often the additional data required to derive cost effectiveness (£ per kg P removed) are unavailable. As a result the best data we could obtain were from Cooper *et al* (1994, 1995) using some small assumptions (table 8.10).

Even the data format of Cooper *et al* (1994, 1995) is such that a significant number of assumptions are required to recalculate the cost-efficiency of P removal at point sources. Hence the authors recommend that, when a common format for storing the base data, its source and the assumptions in the calculations is agreed (see later) that one or more of the UK experts in this field should be commissioned to draw together data in this format.

8.3 Diffuse Sources of Pollution

8.3.1 Reducing fertiliser input while maintaining fertility

Reducing fertiliser input while maintaining fertility involves regular soil P testing and knowledge of crop requirements. A series of case studies undertaken by ADAS suggests that not only could inorganic fertiliser application be reduced on many farms by a better knowledge of the nutrient content of livestock manure but the farmer would also achieve a substantial cost savings (MAFF, 1997). On the ten case-study farms, the content of major nutrients in livestock manures was generally underestimated and this led to excess application of inorganic fertiliser to supply plant nutrients. ADAS found a cost, in terms of fertiliser which need not have been purchased of between £300 and £1,300 per farm. When farmers over-apply P, they often do so because of a perceived benefit rather than an actual benefit.

In England, the method used by ADAS to establish adequate P levels in soils is the "Olsen P Test". The recommended level is partly based on what crops are grown but also on cropping history and soil type. Olsen P indices relevant to different crops are given in table 8.1.

Table 8.1 Recommended soil P levels for different crops

Crop	Soil P index	Recommended soil P levels
Grassland, most arable crops	Index 2	16-25 mg P/litre extractant*
Potatoes, vegetables, maize	Index 3	26-45 mg P/litre extractant
	>= Index 4	No additional inorganic P required

* Extractant = $NaHCO_3$, pH 8.5

In assessing the net cost of this BMP, the issues are as follows: how many soil tests are required per year? how many soil tests are required per hectare?; what are the cost implications of such tests?; what is the likely saving to the farmer? Paul Withers of ADAS provided the following information.

How many tests and what cost? Depending on the type of rotation, soil sampling would need to be done once every three to five years. For arable rotations such as cereals and oilseed rape, once every five years would be sufficient. These crops are not very sensitive to P application. For potatoes, sugarbeet and vegetables, a sample would need to be taken about once every 3 years. However, for simplicity, and to put an upper bound on the cost an annual test has been assumed in the following calculation.

The requirements would be about one sample every 15 ha (about one sample per field). The cost of the analysis (which would be for P, Potassium and Magnesium) would be £7. The time taken to sample the field would be, on average 30 minutes per field. At an average charge for an ADAS consultant at £300 per day (7.5 hours), this would come to £20 per field (assumed equivalent to 15 ha.). Giving a total cost of £27 per field or £1.8 per ha. However, many farmers take their own samples giving a reduced cost per sample of £7 (nominally £0.5 per ha.).

These costs may be turned to cost-effectiveness value using the export coefficient approach. Assume costs of £1.8/ha/year and a reduction in run-off coefficient from the average tilled land value of 0.66 kg P/ha/year to 0.43 kg P/ha/year (half way between tilled land and semi-natural grassland, table 4.2). Since the reduction in P loss is equivalent to 0.23 kg P/ha/year, the cost effectiveness becomes £7.8/kg P for an annual test, £7.8/x /kg P tested every x years.

Saving to farmers? The saving that arises from doing these tests and reducing fertiliser use as a consequence depends very much on what crops are growing and the P content of the soil. Provided the P in the soil is at an adequate level, there is no economic benefit to adding additional fertiliser. In this case, the farmer can only save money by reducing P. For example, normally a cereal crop uses about 50 kg P per ha. For an average field of 15 ha, this is an application of 750 kg P. If all this application of P was unnecessary due to sufficient P reserves in the soil, the saving to the farmer would be £225 per field (ie 750 kg P x 30p; On average P costs about 30p per kg). In the case of potatoes, where 200 kg/P/ha may be required, the equivalent saving would be £900 (ie 200 kg x 15 ha x 30p) if all additional P application proved unnecessary. In order to make a useful estimate of potential P reductions from this method further work would need to be done, possibly on a regional basis, to assess the "typical" amount of over fertilisation at present. The availability and cost of zero P or low-P fertilisers requires addressing.

8.3.2 Farm ponds

In the case of ponds, or indeed other types of wetland, it is sometimes in the interest of the farmer to create these features. For example, artificial reedbeds have been developed on a Co-operative Wholesale Society (CWS) farm to deal with waste generated by a large dairy herd. CWS have also drawn up wildlife plans and this could, for example, involve restoring a wet area to a pond. Of course, such initiatives are costly in cases where wetland restoration/ creation requires taking valuable crops out of production, but subsidies may reduce the economic impact of such schemes.

In establishing the direct costs of a pond, of relevance is the commercial costs for excavation and the removal costs for the spoil. The former would normally by about £1.50 per cubic metre of volume extracted, whereas the latter costs about £5 per cubic metre (Jeremy Biggs, Pond Action, pers. comm.).

The type of pond created depends on whether the issue is to remove particulate P or dissolved P. If it is the former, the retention time is much lower - a couple of hours compared to 2-5 days for dissolved P removal. Work in LOIS (Land Ocean Interaction Study) suggests a 50:50 split of dissolved to particulate runoff - so a two hour retention time would only be 50% efficient, at best. With regard to rainfall, a figure of 5mm/hour in upland (non-agricultural) areas is the type of rainfall likely to produce surface run-off (Jim Hudson, Institute of Hydrology, pers. comm.). Information on the maximum design loadings and minimum retention time for sedimentation basins are obtained from the Institute of Water Pollution Control (1973). Information on dissolved removal is based on Reynolds (Personal communication).

On the basis of all this information, two estimates can be given for the pond volume (table 8.2).

Table 8.2 Load information for sedimentation pond/wetland formation

	Sedimentation	*+Dissolved*
Maximum possible efficiency (but likely to be much less)	50%	100%
Run off per ha land surface @ 5 mm/h rain	50 m^3/ha/h	50m^3/ha/h
retention time (minimum) (see text for sources)	2 hours	2 days (4 days)
	48 hours (96 h)	
Maximum loading	30 m^3/m^2/d (1.25 m^3/m^2/h)	-
Surface area (=max. flow/max loading)	40 m^2/ha	-
Min. depth = max load x retn (absolute min. =1.5 m to avoid resuspension)	2.5 m	-
Volume = run-off x retn	100 m^3/ha	2400 m^3/ha (4800 m^3/ha)
Costs		
Excavation costs = £1.50 per m^3 of volume excavated	£150/ha	£3,600/ha (£7,200/ha)
Moving costs (= £5 per m^3), 50% moved, remainder used for banks.	£250/ha	£6,000/ha (£12,000/ha)
Total construction cost	£400/ha	£9,600/ha (£19.200 /ha)
Over a 20 year lifetime	£20/ha/year	£480/ha/year (£960 /ha/y)

The figure for the sedimentation lagoon is about twice as large as an estimate of £230/ha from Water Quality Management Ltd. (1995) but no allowance was made in the latter for transporting the spoil. They assessed annual operating costs of £20/ha/year which can be added to the figures above giving £40 and £500/ha/year, respectively.

These costs can be turned to cost-effectiveness using the annual P export coefficient (say 0.66 kg p/ha/year, see table 3.2). The resulting estimates are £120/kg P (£40/[0.66 x 0.5] for simple sedimentation at 50% efficiency for P removal using solids removal alone and £758/kg P (500/0.66) for 100% P removal efficiency.

8.3.3 Conservation tillage

The potential benefits of conservation tillage are very widely acclaimed. Low disturbance tillage methods allow more water retention, reduced P loading and facilitate a better soil structure. ECOtillage, a management system developed by Monsanto and Simba, is claimed

Given such large savings, it may be difficult to see why all farmers do not immediately adopt this management system. One reason why not all farmers have adopted this system is the high capital cost involved. For large farms, this new capital equipment may be more readily purchased and justified (larger absolute potential savings). For smaller farms, it may be more difficult to justify this large expenditure – though the problem might not arise if the farm is part of a co-operative that purchases such equipment. Apart from the large cost, the other disadvantage is in the case where herbicide resistant weeds are present. In this case, conventional ploughing is needed about once every three years (Alastair Leake, pers. comm.). Again, larger farms may find it easier to bear the cost of running two systems.

These costs can be turned to cost–effectiveness values by using the export coefficient for bare soil (0.7 kg P/ha/year, table 4.2). Assuming, in the absence of other data, a) 10% reduction in run-off of particles and b) 50% of P loss is in particulate form, then reductions in P loss equate to 0.035 kg P/ha/year. By using Ecotillage the farmer has an apparent saving of (£60 - £12 =) £48 per ha/year compared to traditional methods. Hence, for each £1371 (= £48 / 0.035) saved one kg P less is lost, ie a negative cost. The calculation is of course bounded by the fact that the maximum P reduction is limited by the total catchment area and, since P reductions are small per hectare, the BMP may not make a significant contribution to reductions in P load in the river.

8.3.4 Reduction in slope length using break crops

There are various techniques wherein break crops, such as grass, are used as a way of cutting down the length of a slope. The main costs are the opportunity cost of production and the direct cost of the seed. With regard to the latter, an estimate for a mixture of grass seed is about £65 per ha (CWS Agriculture).

The opportunity cost depends on what the land might be used for. Assuming that a one hectare "grass break" displaces one hectare of crops, a measure of the opportunity cost would be the gross margin per ha (excluding direct subsidy). Gross margin is defined here as the value of grain, subsidy and straw, less the cost of seeds, fertilisers, sprays and other expenses. It is likely that farmers would be able to keep Arable Area Payments with grass breaks, hence this is not included as an opportunity cost. Estimates for various crops are presented below in table 8.4.

Table 8.4 Costs of different crops and P run-off rates

Crops	Gross margin per ha	Gross margin per ha (excl. subsidy)	kg P lost per ha per year (Johnes, 1996)	Minimum cost effectiveness £/kg P
Winter wheat	£455-£675	£213-£433	0.65 - 0.3 = 0.35	1300
Spring wheat	£395-£590	£153-£348	0.65 - 0.3 = 0.35	1129
Winter barley	£395-£570	£153-£328	0.65 - 0.3 = 0.35	1129
Spring barley	£385-£535	£143-£293	0.65 - 0.3 = 0.35	1100
Maincrop potatoes	£715-£2205	N/A	0.8 – 0.3 = 0.5	1430

Ranges are low to high estimates from Nix (1998)

Assuming that the P export rates following intorduction of crop breaks would reduce from those of a cereal crop (0.65 kg P/ha) to those of temporary grassland (0.3 kg P/ha), then the difference between the two run-off rates is the reduction in P run-off rate (table 8.4). Hence the minimum cost effectiveness of the technique would be derived by dividing the gross margin cost to the farmer by the reduction in kg P/ha/year (table 8.10 and 8.4). Even if these measures were cost-effective, a further difficulty for farmers would be in managing these fields. Fields would have to be very big for this sort of practice. As with contour cropping, this practice is perceived by farmers as inconvenient and time-consuming, and has additional operational costs since it requires more frequent and careful manoeuvring of tractors than other methods of cropping.

8.3.5 Reduced dietary input of P in livestock

There is a large number of complicating factors to consider when trying to assess the cost effectiveness of P reduction in concentrates and in implementing such a change, should it be shown to be cost effective.

1. In general, P concentrations in concentrates are higher in the US than in the UK (Europe) hence, there may be more scope for P reduction in concentrate in the US than in the UK.

2. A source in the feed industry stated that, in fact, reduced fertility and poor bone structure are perceived by farmers to be the major problem with reduced P intake, not reduced yield. Since it is long-term chronic effects which are considered to be important, long-term trials will be required to convince farmers that this is not the case (assuming it is not the case!). Such trials are ongoing in the US, UK and other European countries, eg The Netherlands, Republic of Ireland and Germany but will take several years to come to conclusion. At the present time, the industry feels that there is no significant over-feeding of P to cows in the UK.

3. Cost estimates are very complicated because dairy cows are fed different proportions of concentrate (including sugar beet pulp, wheat and manufactured feed mixes) and forage crops (grass/silage/hay) depending on their position in the lactation cycle.

4. With pigs and poultry the proportion of digestible P needs to be included in any calculations. However, cost estimates are generally simpler since the animals are fed totally on concentrates.

5. Information on the effects of reducing the dietary input of P to livestock is limited. In addition, research findings are controversial. ADAS research shows that it is possible to reduce dietary input without any impact on livestock yield. However, research in Holland shows that below a certain threshold, reduced dietary input can have dramatic effects (Paul Withers, ADAS, pers. comm.).

ADAS compared feeding 90 g of P per day to 60 g per day to cattle. The results indicate that it possible to reduce P by about 20 or 30 g per day without an effect on yield, depending on what the forage contains. The latter depends on the P content of the soil, so that the cost – effectiveness of reducing P in concentrate depends in part on the analysis results from soil and grass tissue tests. Conversely a nutritionist in an animal feed company stated that near lactation, when maximum concentrates are being given, a cow would, typically be given 10 kg concentrate per day at 0.7% P (= 70 g P per day) + forage, which typically supplies an additional 25 – 30 g P per day. Hence, to reduce P intake by 30 g per day effectively means reducing the P content of the concentrate by 50%. It is interesting to compare this P reduction with the export coefficient of 0.22 kg P /y /cow quoted for cows (Johnes, 1966). This is equivalent to 0.6 g per day, ie the run-off rate is about 0.7% of daily intake. Since the cow remains in good health it is highly likely that the cow will continue to extract from the feed the same amount of P, ie the amount required for maintenance and growth, and that the excess will be reduced, ie all the 30g per day reduction would have gone straight through the animal. Hence the reduction in run-off could be much larger than the 30% reduction in the feed.

There would be cost advantages to such a reduction (P containing materials are relatively expensive and usually have a significant impact on the price of concentrate, ie high P concentrates are more expensive than low P concentrates). However, the manufacturer felt that it would be difficult to reach 0.35–0.4% P in feed. If the required P concentrations can be achieved by reducing the P containing components, which are the most expensive, then the saving will be in the reduced price of the feed. However, the price would certainly not be halved by a 50% reduction in P but might be reduced by a maximum of 25%. This would bring the cost of feed down from £200 /tonne to £150 /tonne [(200-150)/1000= 5 p/kg]. Feeding at 10kg per day this is equivalent to a saving per cow of 50p /cow /day. If we assume a 50% reduction in P run-off to 0.3 g/day, then the cost effectiveness is [50p / 0.3 g P =] £1.7 /g P = £1700 /kg P saving.

However, this rate of feeding only lasts for a short time. In summer 4-5 kg concentrate per day is a more likely figure so that average annual savings will be much lower than the calculation below suggests. As a result this estimate can only be considered as indicative.

An alternative approach using data provided by Paul Withers (ADAS) and John Metcalf (Borregaard, UK) can be used to examine the potential saving associated with reducing the dietary input of P. We compare a situation where a farmer is feeding livestock 90 g P per day (max.) with a situation where he/she is feeding livestock 60 g P per day (max). It is assumed that a) P is limiting so that additional concentrate is required to increase the intake of P and b) P can be reduced with no effect on economic yield (live weight gain, l/d milk, etc as appropriate). It is also assumed that most of the P requirements of cattle are met through the purchasing of concentrate. In this case, a farmer providing livestock with 60 g P per day would give them 12 kg concentrate per day maximum (1 kg concentrate gives 5 g P). A farmer providing 90 g P per day would feed cattle a maximum of 18kg concentrate per day.

Cows have different feed requirements at different stages of the lactation cycle. Two feeding regimes are considered, which are assumed to relate to the same point in the lactation cycle: a) feeding at a maximum of 18 kg per day (at a certain stage) b) feeding at a maximum of 12 kg per day. In this example the saving to the farmer is the difference between Scenario A and B.

	Scenario A	Scenario B
First 14 weeks:	12 kg/day	18 kg/day
Mid-cycle:	6 kg/day	9 kg/day
Late lactation:	3 kg/day	4.5 kg/day
Annual Total	2.1 tonnes	3.15 tonnes
Total cost	£409.5 - £556.5	£614.25-£834.75

Saving: difference between A and B: £204.75 - £278.25 per cow per year.

- Costs are from Nix (1998). One tonne of concentrate for dairy cows costs between £195 and £265 per tonne.

In reality this is a rather artificial calculation since the farmer would feed the same amount of dry matter concentrate with a lower P concentration. However, this very crude approach gives similar savings per year as the previous method. However, both calculations require a significant number of assumptions and the outcomes should only be considered as indicative of potential savings at maximum feeding rates.

8.3.6 Farm waste management plans

In *The Water Code (MAFF, 1998)*, farmers are advised how to draw up a Farm Waste Management Plan to decide when, where and at what rate to spread manure, slurry and dirty water on the farm.

The stages of a Farm Waste Management Plan can be summarised as follows:

1. Identify land on the farm where waste should not be spread at any time. Leave an untreated strip at least 10 metres wide on both sides of watercourses. Irrigation systems should work so that there is no chance of spray coming within 10 metres of the watercourse.

   To reduce risk of polluting groundwater, livestock manures and other organic wastes should not be applied within 50 metres of a spring, well or borehole that supplies water for human consumption or is used in farm dairies.

   No waste should be spread on very steep slopes or on land with special designations (eg SSSIs).

2. Calculate how much land is needed to take the total nitrogen in all the waste that has to be spread on the farm at a rate of 250 kg N/ha or less. If there is not enough land, arrangements need to be made to spread the excess material on suitable land elsewhere.

   The farmer should take account of the P content of manures when working out manure application rates and how much fertiliser is needed.

3. Identify land where waste should not be spread at certain times or where the spreading rate should be limited.

4. Calculate the largest amount of waste that may need to be stored, before spreading it on the land. In some situations, it may be possible to reduce the volume of waste produced. If more storage space is needed, the fifth stage is to choose and design a suitable storage system to meet the needs of the farm.

Some of the costs that may be incurred in implementing such a plan are as follows:

Costs of advice: The day rate for an ADAS consultant is £300. Lowe *et al.* (1992) also cost a system of free (to farmers) farm visits to establish a farm-specific waste handling plan. They estimate that allowing for two visits per holding, such a scheme would cost the supplier £400 per farm. It should be noted that the farmer may be able to get free advice in preparing a Farm Waste Management Plan. For example, in the *MAFF News Release 1996*, there was an announcement of six new areas targeted in an intensive campaign, giving farmers free ADAS advice in the preparation of their own farm waste management plans. In an earlier pilot farm waste plan study in 1992, farmers volunteering to take part were asked to prepare a farm waste management plan on the basis of simple instructions. The plans were then checked by ADAS. 75% of the farmers in the pilot areas took part, of whom more than 50% drew up their plans with little or no assistance from ADAS. An evaluation by ADAS showed that over 80% of participants had incorporated the plan into their farm waste management practices in some way.

Costs of P analysis for every field: Each year an analysis of P and N content in the soil will be required if reliable management plans are to be achieved. Similarly the P and N content of all manure sources must be measured. Costs for P analysis are estimated in section 8.3.1 and, with an additional charge for N analysis, are about £27 + £7 (N analysis) = £34 per 15 ha field or £2.27 per ha. Obviously the total cost will be farm specific.

Renting land: If farmers need to rent land in order to spread waste, and if it is feasible to do this, average rents are as follows:

Table 8.5 Average rent value by rented area

Total rented area of holding	All agreements (£ per ha)
Less than 10 ha	179
10 to 49.9 ha	163
50 to 99.9 ha	140
100 to 249.9 ha	128
250 ha and above	105

Source: Nix (1998)
Note: there is also significant regional variation in rents.

However, possibly farmers would not have to rent land in order to spread manure (if the other farmer uses this as a nutrient). The more important cost is likely to be that of transporting the waste to another farm - which depends on the distance, the quantity of waste to be transported and the type of waste (eg slurry v farm yard manure). The other farm would need to have adequate storage facilities.

Storage facilities: *The Water Code (MAFF, 1998)*, gives details on typical volumes of waste produced by different livestock and typical amounts of bedding material used. Professional help will be needed in designing a new storage system. Of course, the precise storage requirements (and therefore costs) depends on the adequacy of existing storage and also how much waste is likely to be produced by the particular farm. Hence, this cost is highly farm specific. However, where major improvements are required, the capital expenditure can be very high. For example in a report to the NRA by Water Quality Management Ltd (1995), the unit cost of improvements to farm waste handling facilities was quoted as £25,000 per farm (based on advice of NRA, Rural Land Use Group). Unit costs for waste storage are given in table 8.6.

Table 8.6 Costings for waste storage (Nix, 1998)

Waste storage	Per m^3 stored
Lined lagoon with safety fence	£20
Glass-lined steel slurry silo	
Small (400 m^3)	£33
Medium (1200 m^3)	£24
Large (3600 m^3)	£19
GRP below-ground effluent tank, encased	
Small (12 m^3)	£365
Large (36 m^3)	£290

For a 100 ha farm, where capital costs are discounted over 10 years and no extra land is required for disposal, costs can be calculated as follows:

Farm waste management plan (p.85)		£400	
Soil sampling cost per ha	+	2.27 x 100	
Improvements to waste handling facility	+	25,000/10	
	=	3127/100 ha	= £31.27 /ha/y.

Assuming a 50% reduction in run-off giving a run-off coefficient per cow of 0.11 kg P/y/cow (table 4.1) and a stocking rate of two cows per hectare (IAX stocking rates, NFU, Wareham) a cost-effectiveness of (£31.27 / 0.22) = £142 / kg P.

8.3.7 Reduce stocking density

The cost of this BMP is the opportunity cost of foregone production. This is highly farm specific since it depends on 1) the type of livestock; 2) the number of livestock; 3) how a reduction in livestock numbers would affect entitlement to subsidies.

With regard to the last point, farms entitled to claim under the Suckler Cow Premium Scheme and the Beef Special Premium (BSP) Scheme are only allowed to claim subsidies for a limited number of livestock. In the former case, the farmer has a quota and cannot claim subsidies in excess of this (though he/she may have more livestock on the farm). For the BSP scheme, the farmer cannot claim subsidies for more than 90 animals. If the farmer has a lot more livestock than that for which he/she can claim subsidies, reducing stocking density will not involve any reduction in direct payments. However, this would not be the case where the stocking levels are close to the level on which the farmer can claim direct payments. There is a further complication in that if farmers reduce the stocking rate sufficiently, they become eligible for an extensification premium. Details of subsidies are provided in table 8.7.

Even apart from the subsidy issue, the economic impact of reducing stocking density depends on the characteristics of the farm. This is illustrated below for the case of dairy cows (table 8.8).

Table 8.7 Subsidy payments

Type of Livestock	Payment	Restrictions
Sucker Cow Premium	£112.41 per head	Max. stocking rate: 2 livestock units per ha; Cannot claim in excess of quota (but can purchase quota)
Finished male cattle. Beef Special Premium	£84.32 per head	Limited to 90 animals per producer per year; Max. stocking rate: 2 livestock units per ha
Extensification premium (1)	£27.93 per cow	Max. stocking rate: 1.4 livestock units per ha
Extensification premium (2)	£40.34 per cow	Max. stocking rate: 1 livestock unit per ha
Sheep annual premium	£14.50 per ewe	Limited to individual quota (but can be purchased). LFAs received supplement of £5.15 per ewe in 1998
HCLAs Cows	£47.40 per cow in SDAs £23.75 per cow in DAs	Various restrictions apply. Limit on total HCLA payment of £88.7 in SDAs and £60.85 in DAs
HCLAs Sheep	specially qualified stocks: £5.75 per head (SDAs); £2.65 (DAs)	Various restrictions apply. Limit on total HCLA payment of £88.7 in SDAs and £60.85 in DAs

HCLA = Hill cattle livestock area; SDA = Special designated area; DA= designated area;
LFA = less favoured area data from Nix (1998)

Table 8.8 Performance levels for dairy cows at a) 2 cows; b) 2.25 cows per forage area

Performance level (at 2 cows per forage area)	Low	Average	High	Very high
Gross margin per cow (excl. BLSA) £s	610	711	812	900
Gross margin per ha. £s	1220	1420	1625	1800

Performance level (at 2.25 cows per forage area)	Low	Average	High	Very high
Gross margin per cow (excl. BLSA) £s	593	694	795	883
Gross margin per ha £s	1320	1540	1765	1960

BLSA = Breeding Livestock Appreciation; data from Nix (1998)

The average foregone profits for reducing livestock of other types is presented in table 8.9. Further detail in Nix (1998).

Table 8.9 Gross margin per head and P export rates for livestock.

	Gross margin per head (£)	Export coefficient per animal kg P/y[a]	Gross margin per forage ha (£)	Hectare per animal[b]	Erosion loss kg P/y[c]	Total export kg/hd/y
Beef: single suckling (per cow) lowland	243-297	0.22	310-320	0.78	0.31	0.53
Beef: single suckling (per cow) upland and hill	200-232	0.22	230-320	0.87	0.35	0.57
Beef: double suckling (per cow)	238-275	0.22	450-510	0.53	0.21	0.43
Beef: multiple suckling (per cow)	332-384	0.22	600-670	0.55	0.22	0.44
Sheep: lowland spring lambing (per ewe)	33.7	0.045	370	0.091	0.036	0.081

Hd = head; cost data from Nix (1998)

[a] derived from Johnes (1996)

[b] gross margin per head divided by gross margin per forage area

[c] assumes bare earth losses (0.7 kg P/ha/y) reduced to temporary pasture losses (0.3 kg P/ha/y) (table 3.2 and Johnes, 1996). The difference is 0.4 kg P/ha/y which is multiplied by number of hectares per animal.

P run-off is reduced by two mechanisms if stocking rates are reduced: a) by the loss of the run-off from animal faeces; b) a reduction in the erosive loss of soil made bare by over stocking. The former is estimated simply from the run-off coefficient per year estimated by Johnes (1996). The latter can be estimated, as a maximum, by assuming all the land required to feed this animal is bare at the higher stocking rate and is reduced to temporary pasture at the lower stocking rate. This includes an additional assumption, namely that the area of land required to keep the lower rate animals does not change its run off rate and remains at the temporary pasture loss rate. Summing these loss rates gives the total loss rate in table 8.9. By dividing the gross margin per head by the kg P lost per year the cost effectiveness can be estimated (table 8.10).

8.4 General Maintenance/One off Precautions

8.4.1 Moving access points to the top of slopes

This BMP identifies locations where field boundaries and access points can be altered to reduce diffuse pollution.

The direct costs for this are estimated as follows:

Estimate for installing a 5-bar gate into a field: £550.

This is inclusive of a charge of £80 for the gate itself. The remainder is the labour charge.

8.4.2 Feed trough location

These act as a focus for livestock activity increasing intensity of use in a small area of the field. Relocating feed and water troughs can reduce the effect of intensive livestock movements.

There are no obvious benefits to farmers of moving troughs around. One way of moving the trough is to use livestock feed trailers. By having two on a turn-around basis, there is no extra time cost but there is a capital cost. Also, feed trailers are more expensive then ring-feeders. Ring-feeders are also more difficult to move - in which case a time element should be taken into account.

Feeding ring (size unspecified)	£110
Feeding ring (18" trough)	£96
Feeding ring (24" trough)	£120
Feeding trailer (standard, painted frame):	£1,100
Feeding trailer (heavy duty, galvanised frame)	£1,600

The efficiencies of both these BMPs are likely to be very site specific and no information is available for calculating cost effectiveness estimates.

8.5 Public Information

Cost estimates for a public information campaign (table 8.10) are given by WQM (1995). Since there is no information on the effectiveness of these campaigns in either reducing P inputs or increasing the acceptability of increased P loads no estimate of cost effectiveness can be made.

8.6 Discussion

Given the quality of the input data the estimates of cost-effectiveness can only be considered as a first attempt, but they do demonstrate that, for agricultural BMPs, the run-off coefficient is a very useful means of normalising cost data to give cost-effectiveness estimates. However, there are still a number of BMPs, such as moving feeding rings and moving access points to tops of slopes for which the efficiency is so site dependent that no efficiency data are available. Hence no cost effectiveness estimate can be made. A significant number of BMPs fall into this category and, in addition, many of the estimates of effectiveness will be very site (at farm or even field level) dependent and hence, inherently difficult to assign reliable cost-effectiveness estimates. It may be possible to reduce the variability in some of these estimates to acceptable levels in catchment scale assessment exercises by obtaining relationships between, for example, soil type, slope and efficiency. However, for others, such as the ring feeder BMP it is unlikely that an estimate applicable at the catchment scale will ever be available.

It is interesting to note from table 8.10b that the cost effectiveness of point source reduction is at least an order of magnitude better than the cost effectiveness of using almost all BMPs. Only a free soil testing/nutrient management plan service and the ecotillage approaches are cheaper. This is contrary to comments in most of the US literature. It should be remembered that the cost benefit for the ecotillage system is obtained from the manufacturer and is probably over optimistic. However on large farms it is obviously cost effective, even excluding any benefit in P run-off reductions, but is unlikely to be taken up by small farmers.

Table 8.10a Point source treatment costs

Method	Source	Efficiency	Lifetime	Cost		Cost (£)	
		%	Years	element	10,000pe	100,000pe	1,000,000pe
Chemical treatment of sewage (mixed)	WQM Ltd.	90	Unknown	Capital	40,000	100,000	400,000
				Annual Op	5,000	45,000	319,000
By subtraction of Denitrifying activated sludge plant costs from full plant costs including chemical treatment (Ferrous chloride)	Cooper	90	21y capital depreciation	total	-	37,500	-
						20,000 pe	200,000 pe
Ferric chloride	Brett *et al*	80-95	Unknown	Capital		43,400	77,100
				Annual Op		26,400	244,000
Biological treatment of sewage	WQM Ltd.	80	Unknown	NVP @ 6%		2,300,000	
By subtraction of Denitrifying activated sludge plant costs from full plant costs including biological P removal.	Cooper	78	21y capital depreciation	total	-	34,000	-
Public Information (one campaign)	WQM Ltd.	Unknown	Unknown	Annual Op	4,000	16,000	80,000

Table 8.10b Cost effectiveness of methods

Method	Source	Efficiency	Lifetime	Cost	£/kg P retained	Notes
		%	Years	element	'in the soil.	
Additional cost of removing P from sewage using chemical precipitation	Cooper	90	21y capital depreciation	total	12.5	For 100,000 pe plant
Soil test to check to reduce P over-use	This report	35	3-5 year test		1.6-2.6	
Settlement pond removing solids with and equivalent P reduction efficiency of 50%	This report	50	20y capital depreciation		120	
Settlement pond removing solids with and equivalent P reduction efficiency of 100%	This report	100	20y capital depreciation		758	
Ecotillage	This report	5	10y life	1000 ha	-1371	
Grass breaks in crops on sloping fields	This report					
- Winter wheat	This report	54	Annual	Minimum	1300	
- Spring wheat	This report	54	Annual	Minimum	1129	
- Winter barley	This report	54	Annual	Minimum	1129	
- Spring barley	This report	54	Annual	Minimum	1100	
- Main crop potatoes	This report	63	Annual	Minimum	1430	
Reduced dietary intake in livestock	This report	50	continuous		-1700	
Farm waste management plans	This report	50	10y capital depreciation	100 ha farm	142	
Reduced stocking density	This report					
Beef: single suckling (per cow) lowland	This report	58	Annual	minimum	458	
Beef: single suckling (per cow) upland and hill	This report	62	Annual	minimum	351	
Beef: double suckling (per cow)	This report	47	Annual	minimum	553	
Beef: multiple suckling (per cow)	This report	48	Annual	minimum	755	
Sheep: lowland spring lambing (per ewe)	This report	11	Annual	minimum	416	

9. POLICY ISSUES FOR DIFFUSE POLLUTION ARISING FROM NUTRIENTS

Economists generally advocate the use of economic instruments (ie taxes and permits) for pollution prevention and control. Such instruments have efficiency advantages over a more regulatory approach because they allow flexibility in how different individuals/firms respond to a pollution reduction measure[1]. Unlike the case for a regulatory approach, the responsible government agency does not need information on farm specific abatement costs in order to implement an efficient solution. However, in the context of non-point source pollution, the case for economic instruments is less clear-cut. As this study shows, eutrophication arising from P can arise for a range of different reasons - so it might not be clear what to link the tax or permit to. Furthermore, the pollution impact might just as easily arise from management practices (eg land-spreading of manure in winter) as from the actually quantity of P used on the farm. Nonetheless, economic instruments are discussed in the context of non-point source pollution in the UK (eg DETR, 1997) and although they do not provide a complete solution to the P problem, they may still be relevant in the context of a package of measures to reduce P. In this section, some of the issues concerning economic instruments are briefly reviewed. There is also some discussion of mineral accounting and the proposals of relevance under Agenda 2000.

9.1 Product Charges

The issues surrounding a tax on fertiliser are frequently discussed in the literature (eg Burrell, 1989). In principal, a tax could be linked to the P content of fertiliser. Such a tax may have several effects: reducing the quantity of P fertiliser applied; more careful application of P fertiliser; substitution of chemical fertiliser for animal manure.

However, research has shown repeatedly that a very high tax would be needed to stimulate a significant reduction in the use of chemical fertiliser. For example, Dubgaard (1990) suggested that a tax of 200% on N in commercial fertilisers would be necessary to achieve the Danish target of a 30-35% reduction in use of inorganic N. The low responsiveness of fertiliser use to a change in the price (ie a small elasticity) may be explained because the cost of fertiliser is a low percentage of the total value of production. In addition, manure is not perceived to be a very good substitute for chemical fertiliser as the P content is often not accurately known, and the handling is more difficult.

With regard to livestock, the issue is different since organic waste is produced on the farm, rather than purchased. However, there are various types of product charge that might help in reducing pollution generated by the presence of livestock. For example, the OECD (1991) considers a charge based on purchased feedstuffs (or their nutrient content) or a charge based on livestock density per ha. Also, in Belgium, it has at times been proposed that there be a charge levied on the suppliers of animal feed, based on its N and P content (Declerq and Sennesael, 1991). There

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[1] Producers with lower abatement costs take most action and this allows overall efficiency in meeting the environmental standard.

are a few points that could be made in favour of such an approach. First, it would stimulate suppliers to provide feed with lower proportions of such nutrients.[2] Secondly, if this tax is reflected in relative prices, there may be some substitution away from feed with a high nutrient content. Thirdly, it might be easier to have a significant impact than in the case of chemical fertiliser since concentrated feedstuffs account for a high proportion of production costs for livestock producers

While the environmental impact of product charges may be quite small (if the tax is not set at a high enough level or is not closely linked to the cause of pollution), there will be some advantages of including them within a broader approach of reducing eutrophication. Apart from the signal given to farmers and suppliers of fertiliser/animal feed, the revenue generated could possibly be earmarked to pay for the more costly measures which may be deemed necessary within vulnerable zones (eg free farm visits).

One of the problems with product charges is that the impact of P reduction measures may be very location specific. However, charges must be implemented at a national level. DETR (1997) discuss a rebate approach as a possible solution. Under such a system, there would be a national charge with rebates available in relation to certain local conditions or activities. For example, a farmer with proof of purchase and area in production could apply for a rebate if the quantity per ha. did not exceed a given amount. However, this is much less of an issue if the product charge is fairly small.

An alternative approach for controlling nutrient pollution is the establishment of a tradable permit system. Unlike the case of product charges, there is no direct financial cost (though there may well be an opportunity cost).

9.2 Tradable Permits

If the nutrient problem can be linked to over-use of one particular input, a tradable permit system would allow a more efficient outcome than an alternative based on mandatory reduction of the input (eg livestock; manure; fertiliser) on every farm. The advantage of this approach is that it allows flexibility as to which farmers should reduce input application. This is particular relevant when farms are very different and are likely to face very different opportunity costs of reducing input use.

An example of this approach is as follows: if the main problem driving the eutrophication problem was chemical fertiliser, a restriction on fertiliser use for the whole catchment might be put in place. Permits could be allocated to farmers allowing them to use only a stated amount of fertiliser per hectare. If these permits were tradeable, then farmers who had a high return per unit of fertiliser applied would find it profitable to purchase additional permits and those who had a low return would have an incentive to sell. A similar principal could be applied to land use permits or permits based on stocking rates.

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[2] It is possible for suppliers to reduce the nutrient content of animal feed. Bennett (1991) has noted that in the Netherlands, efforts have been made to draw up voluntary agreements with industry to reduce the concentrations of P and N in animal feed.

DETR (1997) consider a permit applied to stocking rate schemes wherein the permit would be applied to stocking rates on livestock (ie number of animals per hectare) based on an assessment of the acceptable nutrient loading. This loading would then be expressed in terms of the number of livestock units per hectare for different livestock.

In comparison with a tax, advantages of this approach include the fact that it can be applied within a particular geographic area (rather than the whole country) and the environmental outcome may be less uncertain than the case of a product charge. While the idea of tradeable permits may seem fairly radical or unlikely to be implemented, it should be remembered that in the agricultural sector there is already considerable experience of tradeable permit systems. In addition to the well-known case of milk quotas, under both the Suckler Cow Premium Scheme and the Sheep Annual Premium Scheme, there are farm specific quotas for the amount of premiums farmers may claim on livestock. These quotas are tradeable.

There are some examples of schemes to allow trading between point and non-point sources in the US. For example, in the Tar-Pamlico Basin, North Carolina, a scheme was introduced in 1989 to allow point sources to contribute to an agricultural fund to cover the costs of reductions in non-point inputs. However, no "trades" have taken place (DETR, 1997). There are other examples of schemes which have a similar principal - the schemes in the Dillon Reservoir and the Cherry Creek Reservoir respectively (both in Colorado). However, none of these schemes has yet proved very active in the number of trades that have taken place (DETR, 1997).

9.3 Nutrient Accounting

In several European countries, farmers are obliged to keep some form of mineral accounting system (Rude and Frederiksen, 1994). While the details of implementation (or proposed implementation) vary between countries, the principal is that farmers keep some record of minerals used on farms. For example in the Netherlands, the accounts are partly normative figures on the excretion of manure from livestock and figures on the amount of fertilisers and feed bought, as well as the agricultural products sold on the market. These elements are then transferred into minerals. It would be possible to consider some charge on surplus balances. A tax on the total mineral production on farms was proposed in the Manure Action Plan for Belgium (Rude and Frederiksen, 1994).

While such systems may seem excessively administrative for regions where eutrophication is not a primary concern, it is a way for the government agency to find out more about what is happening on farms. However, this very time consuming type of record-keeping may be far less efficient than a simple product charge.

9.4 Opportunities in Agenda 2000

While the introduction of any of the above measures may be politically difficult, there may be an opportunity to introduce more control measures in the context of Agenda 2000 (European Commission, 1997). Some of the ideas in Agenda 2000 of greatest relevance are as follows:

- ***Cropping***: Member States can make granting of direct payments for arable crops and set-aside conditional on the respect of environmental provisions.

- ***Beef***: "The Commission will reflect on how incentives to extensify production can be strengthened with a view to improving their effectiveness in relation to environmental objectives"

- ***Differentiation and ceilings for direct payments***: The introduction of an individual ceiling covering all direct income payments. Member States to introduce differentiation criteria according to agreed rules.

- ***Agri-environmental policy***: The Commission will make a proposal enabling Member States to make direct payments conditional on the respect of environmental provisions.

These various forms of cross-compliance might allow easier introduction of some of the control measures suggested in this report. For example, BMPs such as buffer strips, types of wetland restoration and critical area planting are among the measures that could be made obligatory in certain agricultural contexts or regions of the country. Hence government agencies could be advocating the most cost-effective of these options.

10. CONCLUSIONS

A. The Warwickshire Avon

A1. 75% of the total P load at Evesham is derived from point sources.

A2. Following the application of Urban Waste Water Treatment Directive controls to the relevant sewage treatment works, the total in-river fluxes at Evesham were split approximately equally between point and diffuse sources.

A3. The total P load at all points in the River Avon was highly correlated with the contributing point source load to that location.

B. Catchment run-off models

B1. The majority of models that predict the run-off of P from agricultural land are too complex or require too many calibration parameters for routine use within the Environment Agency. Export coefficients are the most user-friendly approach.

B2. Pictorial presentation of the proportion of diffuse and point source input to P flux in a river needs to take into account the size of flux at each sampling site otherwise the relative importance of the different sources can be misinterpreted.

B3. In urbanised catchments, the use of fixed export coefficients is a useful technique for estimating diffuse inputs of P in scoping studies of possible P removal strategies.

B4. Analysis of the residual load following subtraction of the point sources in the Warwickshire Avon indicated that the cumulative errors in the estimates were too large to identify a need for more catchment specific export coefficients.

B5. In small catchments (<25,000 ha) the variability of load estimates increases significantly.

C. Estimation of in-river loads and eutrophication effects

C1. At least fifty samples at a site, spread over a range of flow conditions, are required for a reliable estimate of mean annual P load in a river.

C2. The proportion of total P in river water in the Warwickshire Avon catchment, which is bioavailable, is extremely variable, ranging from 70–95%. As a result it is not very reliable to measure dissolved reactive P and use a fixed factor to estimate total P concentrations. Even though it is more costly to perform, the best results will be obtained from measurements of total P.

C3. A review of models predicting the relative growth of mixtures of planktonic algae, benthic algae, epiphytic algae and macrophytes in rivers showed that there were no useful operational models available at present. As a result there are no ways of defining in-river standards for P scientifically.

to reduce grass weed management, reduce plant establishment costs, save time and maximise yields. The basic principle is to allow weed seeds and volunteers enough time to germinate and then to control them prior to establishing the next crop. An overview of the average cumulative percentage saving over a conventional system is outlined below. The data in table 8.3 are averages taken from the cost calculator produced by Monsanto (Colin Stride, pers. comm.).

Table 8.3 Ecotillage costs

1. Base system on medium to heavy soils
(ie Plough and press; two passes of power harrow; air drill; roll)

Progressive adoption steps	Cumulative % savings	
	Time	Cost
Replace air drill with a cultivator drill	30	10
and replace power harrow with double press	40	20
and replace plough with two passes of discs	50	40

2. Base system on light to medium soils:
(ie Plough and press; springtine cultivator; air drill; roll)

Progressive adoption steps	Cumulative % savings	
	Time	Cost
Replace air drill with a cultivator drill	30	5
and replace plough with 1-2 passes of disc and press	40	35

Soil type		*Savings (average, £/ha)*	
	Establishment[1]	*Agronomic*[2]	*Total*
Light to medium	£25	£35	£60
Medium to heavy	£45	£45	£90

1. Establishment savings result from reductions in manpower, etc, compared to traditional methods required to prepare the seed bed and plant the seeds.
2. Agronomic savings consist of the following: lower seed rates with the system; reduced slug or aphid control costs; reduced herbicide costs (after 1-2 years); reduced manganese /magnesium deficiency through consolidation; better/quicker crop establishment.

Capital cost: £60,000 approx. (i.e. for 3 metre disks + double press + cultivator drill). If we assume a 10 year life for the equipment with maintenance costs of £6000 per year, then, for a 1000 ha farm costs are £12/ha/year.

D. Phosphorus removal processes at point sources

D1. There are only two common methods of P removal at sewage treatment works: chemical precipitation and biological treatment.

D2. Very few explicit data exist for the additional costs of treating point source P. Most modern costings are for complete works, including P reduction.

D3. P reduction at point sources was found to be considerably more cost effective than that achieved by most diffuse source BMPs.

E. Phosphorus removal from diffuse sources (BMPs)

E1. A large number of BMPs have been proposed for reducing diffuse pollution from agriculture and are listed in the report.

E2. Application of two or more BMPs together will probably not result in a P run-off reduction equal to the sum of the efficiencies of the individual methods. Hence there is likely to be a level of diminishing return on investment as more BMPs with the same objective are applied.

E3. Many of the BMPs suggested in the literature are presently unacceptable/ inappropriate to the farming community as a result of either increased costs or increased complexity of management.

E4. There is a lack of clear descriptions of BMPs, particularly where their use has been suggested by US sources.

E5. Implementation of one or more economic and policy measures may increase the uptake of desired BMPs by farmers.

F. Cost-effectiveness

F1. Cost-effectiveness is a better basis than cost-benefit for making management decisions between different P reduction technologies.

F2. Very little data on the efficiency and costs of P removal techniques, particularly those from the US, were found to be either accurate or appropriate to the UK situation.

F3. It is not easy to find the basis on which previous estimates of costs have been based.

F4. There is a lack of reliable data on the effectiveness of P reduction methods and almost no attempt to model the major sources of between site variability.

F5. A major problem with cost estimates is assessing which costs should be included.

F6. Export coefficients can be used to convert from cost per ha to cost per kg P removed.

F7. It is possible to make an estimate of cost-effectiveness for many BMPs but there are a number of BMPs for which the efficiency is so site dependent that no cost-effectiveness value could be obtained.

G. Management models

F1. There are very few models in the literature which can form the basis of a model approach to the design of catchment scale P reduction strategies.

F2. There are no off-the-shelf tools for comparing different P removal options which include cost, efficiency and scenario testing and are easy to use.

F3. It should be possible to make catchment scale cost estimates for different P reduction scenarios using a combination of GIS and a cost-effectiveness based model.

F4. A model structure is proposed to incorporate P load, reduction efficiency and cost into decision making.

F5. A model of this type could easily be applied to water quality problems related to suspended solid, nitrogen and pesticides.

11. RECOMMENDATIONS

1. The use of fixed export coefficient based models is recommended for routine estimation of diffuse run-off of P from catchments greater than 25,000 ha in scoping studies designed to identify the main P reduction strategies in a catchment.

2. The Agency should require a minimum of 50 samples covering a time span of no more than 5 years in order to cover a wide range of flows when making estimates of total P flux in a river.

3. Total P should be the preferred measurement in studies requiring a comparison of P loads since, at the moment, there is no good P- biological effect model.

4. The Agency should invest significant finances in exploring a number of alternative means of predicting the effects of eutrophication in rivers from P concentrations. This would develop the scientific basis for setting nutrient targets in rivers. Given the early stage of development of these models, an independent assessment of the different approaches used by different groups should be made.

5. The Agency should carry out a larger survey of farmers to ascertain the usefulness/ applicability of different BMPs and the types of inducements (financial or otherwise) which might be needed to aid their introduction.

6. The following work programme on diffuse pollution is proposed for Agency funding in order to reduce the overlap of work between four sections of the Agency, *viz.* suspended solids reduction; P reduction; nitrogen reduction and pesticide reduction in run-off.

    a. Using Agency reports and other sources (this report may be a starting point) make as complete a list as possible of BMPs in the literature.

    b. By visiting locations in the US and Europe particularly, obtain good descriptions of the BMPs and how they are implemented.

    c. Identify as far as possible published estimates of the efficiency of each agricultural BMP for each of the four pollutants separately and assess the transferability of that estimate to UK conditions. It may be prudent, where possible, to consider the efficiency of reducing dissolved and suspended components separately and the likelihood of increasing groundwater pollution as a result of decreasing surface water pollution. Where no efficiency estimates are available it may be possible to put some bounds on the efficiency from a knowledge of the operating principles of the BMP.

    d. Consult with UK farmers and farming institutions (in an enhanced study along the lines reported herein) as to the applicability/likelihood of uptake of each BMP in the UK situation and factors, such as financial incentives and preceived loss of productivity, which might affect uptake rates.

e. On the basis of this information compile an authoritative list of BMPs which are applicable and reasonably effective, in the UK situation, to each of the four pollutants considered.

7. The Agency should commission the development of a cost effectiveness based model for use in optimising investment in management practices to reduce nutrient loads, pesticides and suspended solids in rivers.

8. f. Use this list as a basis for:

(1) Re-assessing the need for UK trials to measure efficiency.

(2) Carrying out research work to quantify the effect on efficiency estimates of the main factors considered to influence variability in efficiency estimates, such as previous crop, slope, etc.

g. When it is clear which BMPs are appropriate to the UK situation and likely to be effective in reducing one of the four pollutants, carry out an assessment of costs to determine a cost-effectiveness estimate.

9. Cost-effectiveness, rather than cost-benefit, should be used as a basis for allocating resources to P reduction strategies.

10. The Agency should pursue the development of a combined P load, cost-effectiveness model as an aid to developing P removal strategies in catchments.

11. The same model should be developed for water quality improvements in nitrogen, suspended solids and pesticides.

12. The Agency should begin to develop an archive of cost-effectiveness for each P (and other pollutant) reduction method containing the exact methodology and cost base upon which the cost-effectiveness estimate has been made.

13. The Agency should invest in research to try to ascertain the main causes of site to site variability in BMP effectiveness and develop models to develop generic export coefficients which can be calibrated using ecological measurements (eg slope, soil type, time period of measurement, etc).

14. The Agency should invest research in improving the cost–effectiveness estimates of BMPs relevant to the UK situation.

15. BMPs for which a cost-effectiveness estimate cannot be made should not be recommended by the Agency.

16. The Agency should consider the potential benefits which might accrue were Government to introduce one or more economic measures to encourage implementation of BMPs.

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Appendix 1

Workshop Attendees

Date and Location	Name	Region	Comment
17 December 1997	D Harvey	Southern	Low attendance- snow
IFE, East Stoke			
3 March 1998	J McEvoy	Thames	
IFE, Windermere	J R Haines	Thames	
	S Brierley	Anglian	
	J Pitt	Anglian	
	R J Hemsworth	Welsh	
	H Millband	Welsh	
	A Hicklin	South-West	
	P Mitchell	South-West	
	P Bryson	South-West	
	D C McIlroy	Midlands	
	D Foster	HO	
	R F Prigg	North-West	
	P Wittred	North-West	
	S Jones	North-West	
	I Dunhill	North-West	
3 April 1998	R Huggins	South-West	
Environment Agency,	P Bryson	South-West	
Blandford Forum	R Robinson	South-West	
	P Mandeville	Southern	
	S McNally	ITE	

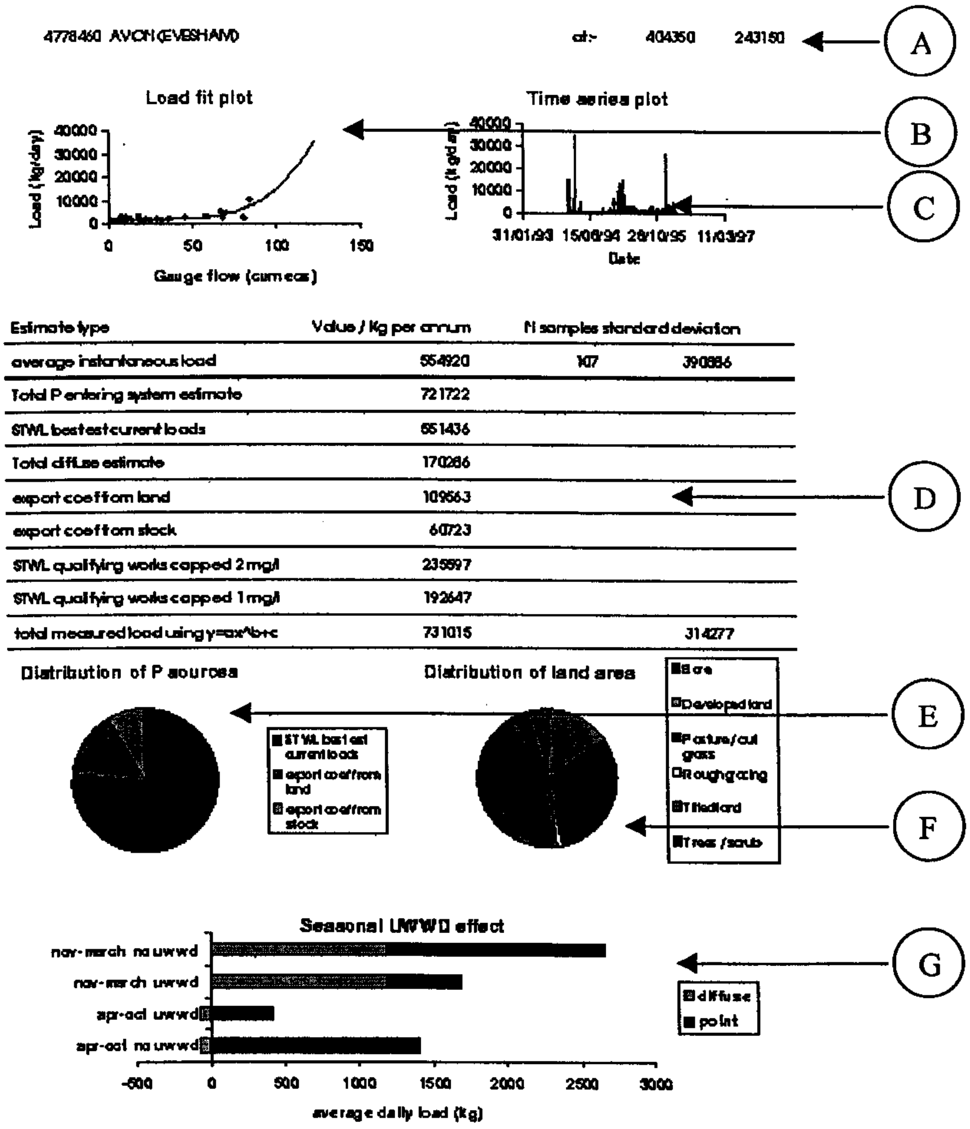
Notes

1. P Buckland (EA); J Hilton (IFE) and G P Irons (IFE) in attendance at all three.

2. IFE staff consulted on separate occasions: Dr.C S Reynolds; Dr.F H Dawson; Dr A House.

Appendix 2
Case Study Results

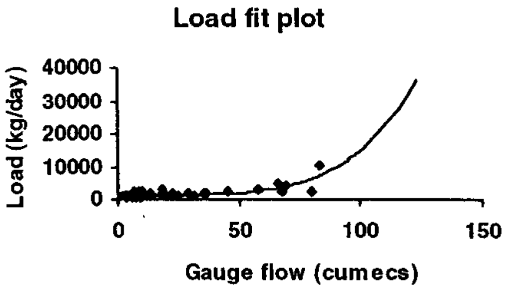
The following pages contain a summary of the results for 50 river monitoring sites. Each page follows the format shown below where:
(All loads refer to total phosphorus and are reported as kg P/day.)



Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	554920	107	390886
Total P entering system estimate	721722		
STWL best est current loads	651436		
Total diffuse estimate	170286		
export coeff from land	109563		
export coeff from stock	60723		
STWL qualifying works capped 2 mg/l	235597		
STWL qualifying works capped 1 mg/l	192647		
total measured load using y=ax^b+c	731015		314277

A	EA site code, site description and grid reference in meters.
B	scatter plot of instantaneous load vs gague flow. The line is a numeric fit for $y = ax^b + c$
C	time series plot of instantaneous load. The line is predicted load using the curve shown in B
D	table summarising predictions.
E	Relative contributions of different P sources
F	Relative land areas in catchment
G	Predicted annual load, spit by season and modified to reflect the effect of UWWD.

4778460 AVON (EVESHAM) at:- 404350 243150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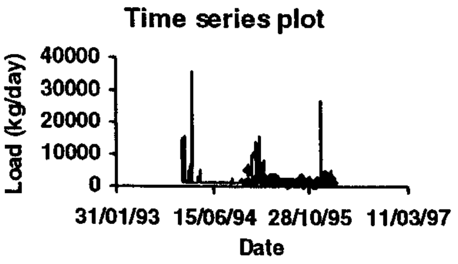
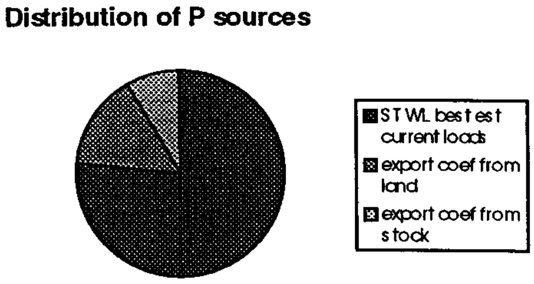
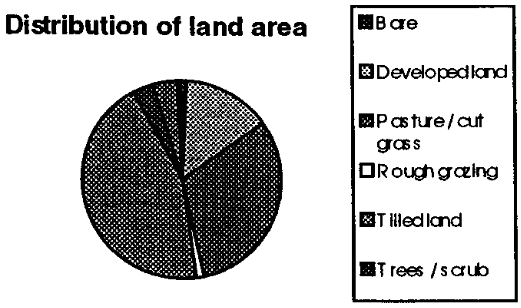
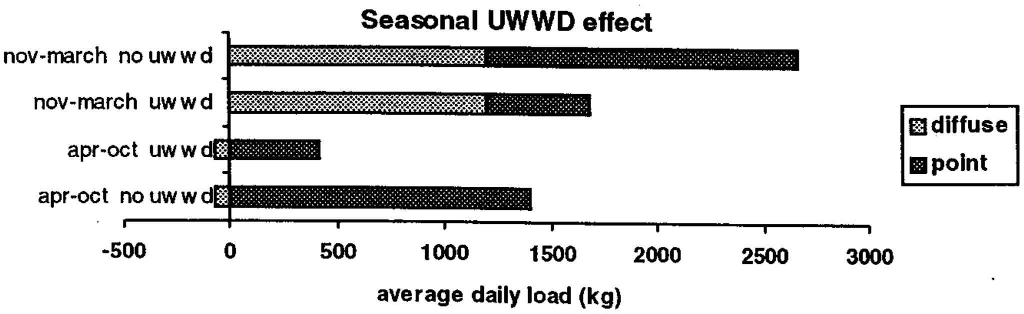

Time series plot

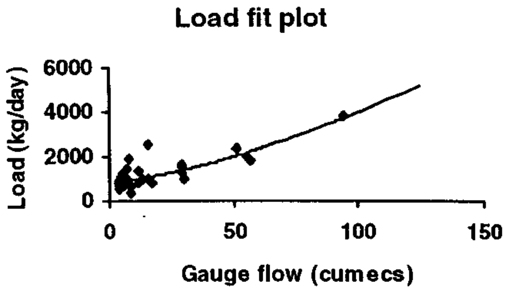
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	554920	107	390886
Total P entering system estimate	721722		
STWL best est current loads	551436		
Total diffuse estimate	170286		
export coef from land	109563		
export coef from stock	60723		
STWL qualifying works capped 2 mg/l	235597		
STWL qualifying works capped 1 mg/l	192647		
total measured load using y=ax^b+c	731015		314277

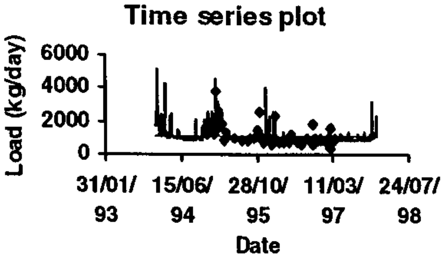

Distribution of P sources


Distribution of land area

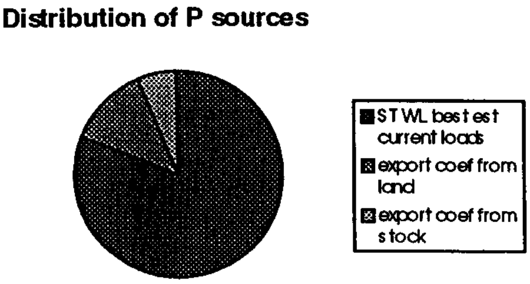

Seasonal UWWD effect

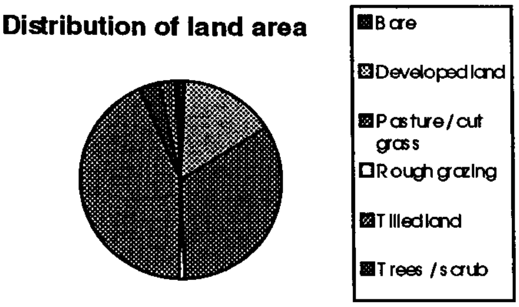
4787440 AVON (STRATFORD) at:- 420600 254800

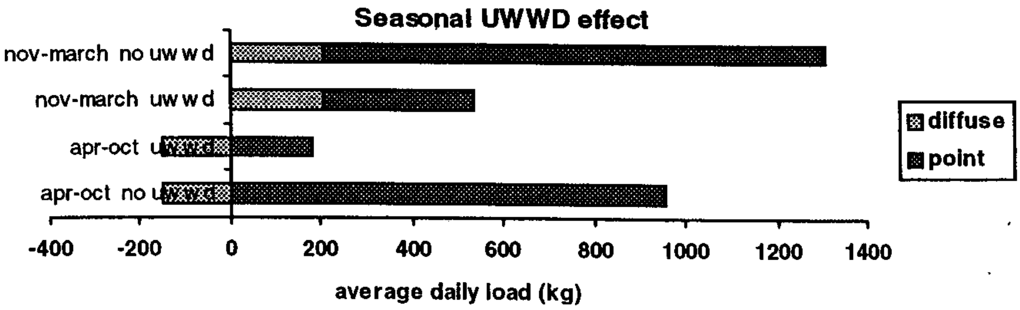




Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	403978	50	216955
Total P entering system estimate	510078		
STWL best est current loads	413157		
Total diffuse estimate	96921		
export coef from land	64289		
export coef from stock	32632		
STWL qualifying works capped 2 mg/l	174492		
STWL qualifying works capped 1 mg/l	131542		
total measured load using y=ax^b+c	443790		126265

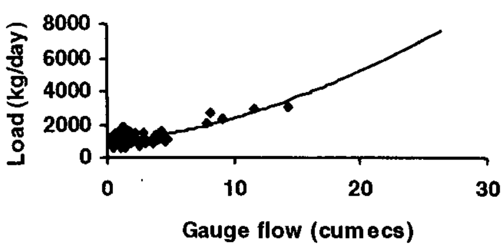




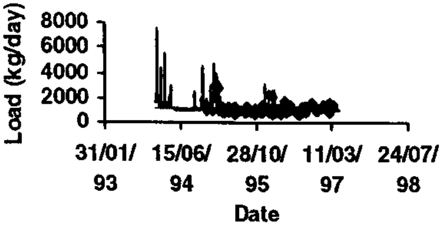


4790980 AVON (BARFORD) at:- 426800 260900

Load fit plot



Time series plot



Estimate type	Value / Kg per annum	N samples	standard deviation
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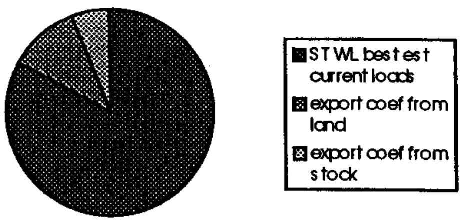
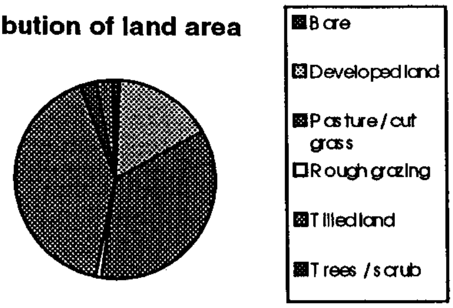
Total P entering system estimate	486146		
STWL best est current loads	404342		
Total diffuse estimate	81804		
export coef from land	53327		
export coef from stock	28477		
STWL qualifying works capped 2 mg/l	165677		
STWL qualifying works capped 1 mg/l	122727		
total measured load using y=ax^b+c	468868		95442

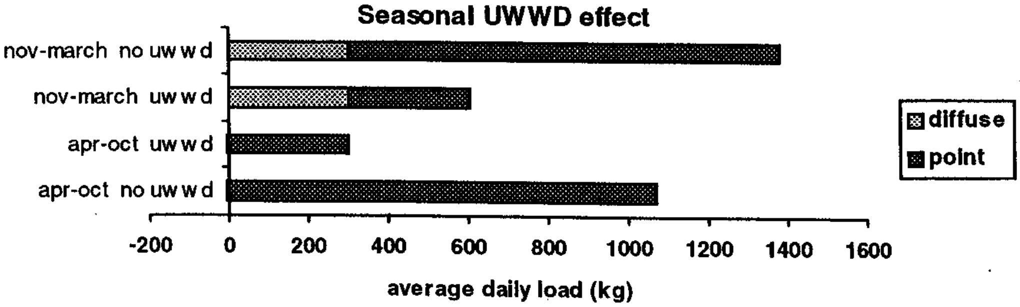
Distribution of P sources



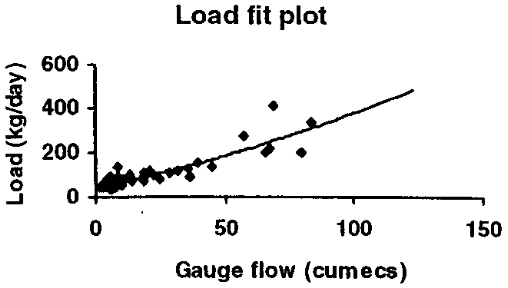
Distribution of land area

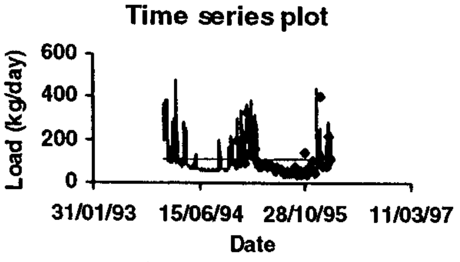


Seasonal UWWD effect

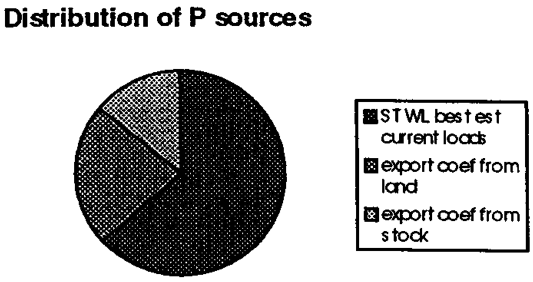


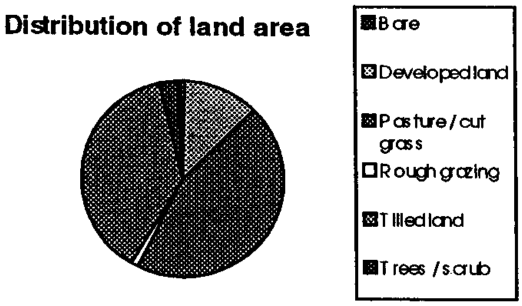
4796740 AVON (STARE BRIDGE) at:- 433000 271400

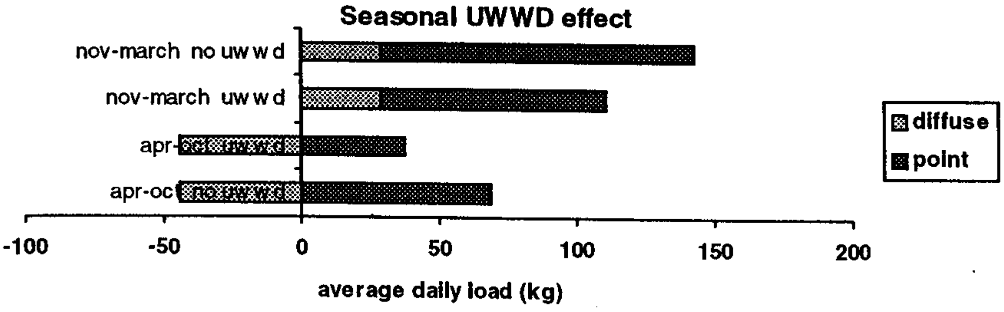




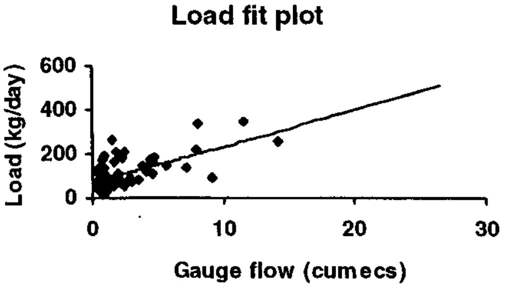
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	39980	107	28457
Total P entering system estimate	67344		
STWL best est current loads	42586		
Total diffuse estimate	24758		
export coef from land	15444		
export coef from stock	9315		
STWL qualifying works capped 2 mg/l	30997		
STWL qualifying works capped 1 mg/l	30997		
total measured load using y=ax^b+c	37003		11909

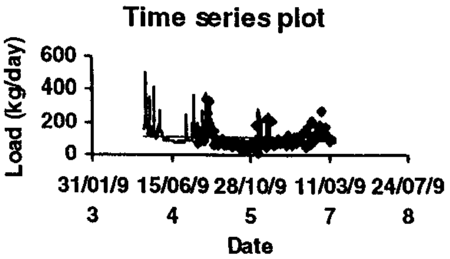




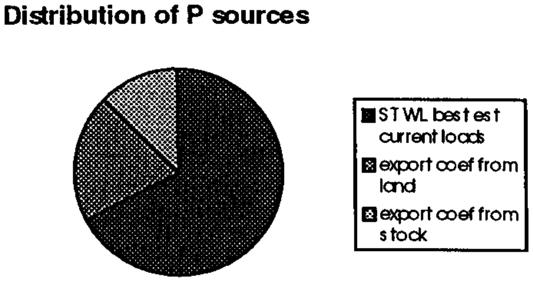


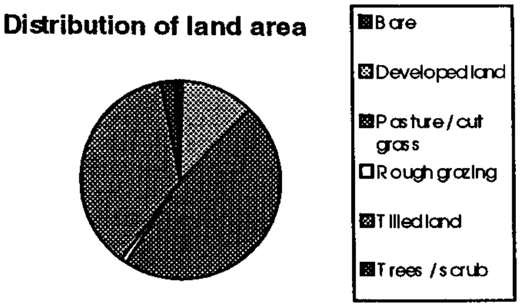
4799980 AVON (LAWFORD) at:- 446890 277130

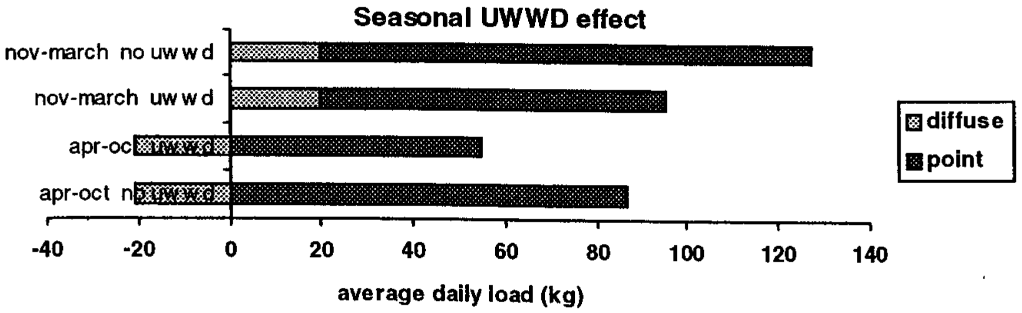




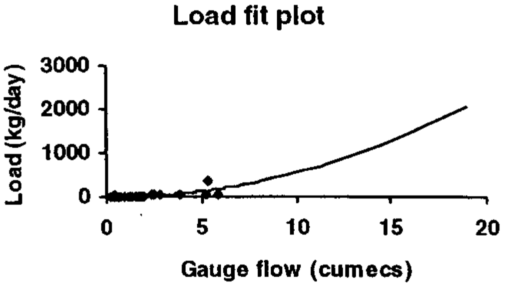
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	36803	101	21384
Total P entering system estimate	59903		
STWL best est current loads	40219		
Total diffuse estimate	19684		
export coef from land	12168		
export coef from stock	7516		
STWL qualifying works capped 2 mg/l	28630		
STWL qualifying works capped 1 mg/l	28630		
total measured load using y=ax^b+c	41407		16660

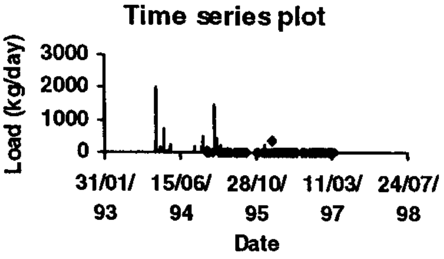




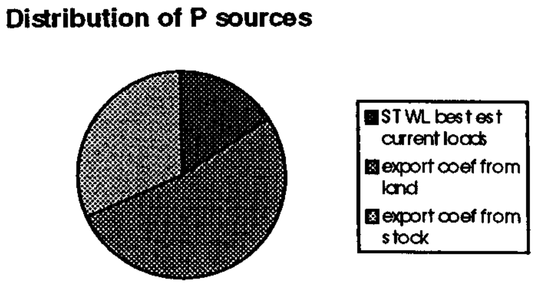


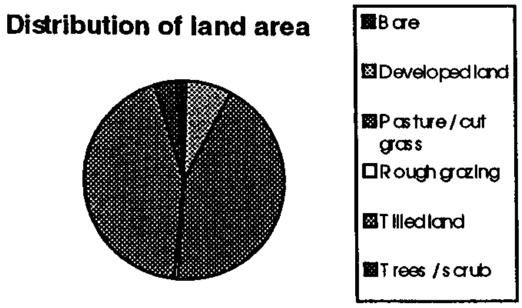
4805580 AVON (CLIFTON) at:- 453200 277200

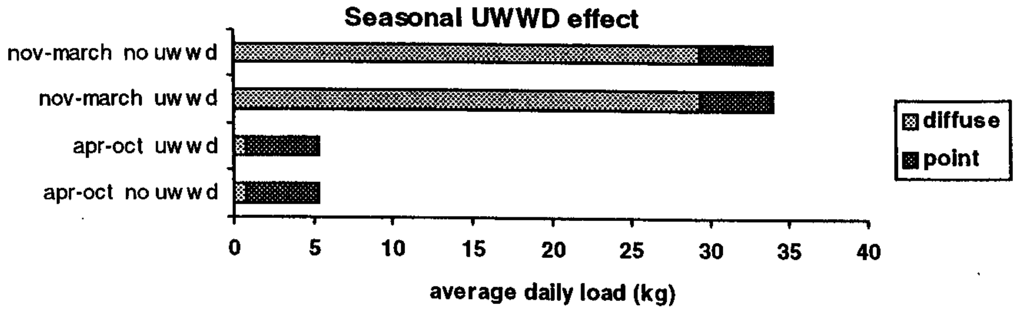




Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	4490	83	15692
Total P entering system estimate	10595		
STWL best est current loads	1699		
Total diffuse estimate	8896		
export coef from land	5534		
export coef from stock	3362		
STWL qualifying works capped 2 mg/l	1699		
STWL qualifying works capped 1 mg/l	1699		
total measured load using y=ax^b+c	9733		11589

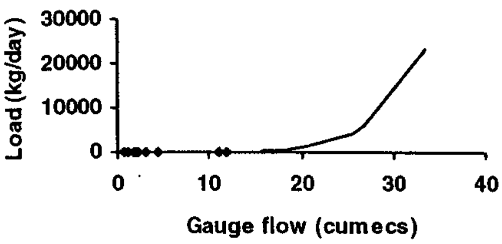




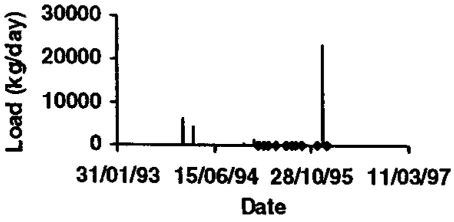


4809150 AVON (KILWORTH) at:- 461130 281160

Load fit plot

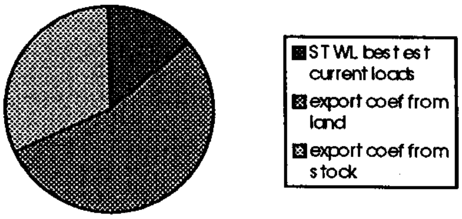


Time series plot

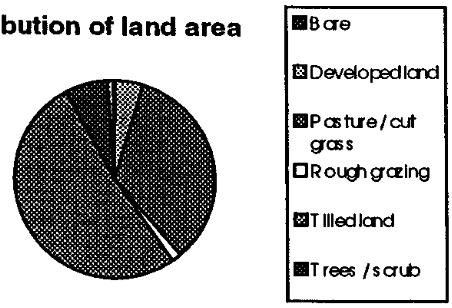


Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	1472	38	2935
Total P entering system estimate	3974		
STWL best est current loads	567		
Total diffuse estimate	3407		
export coef from land	2135		
export coef from stock	1271		
STWL qualifying works capped 2 mg/l	567		
STWL qualifying works capped 1 mg/l	567		
total measured load using y=ax^b+c	20391		1504

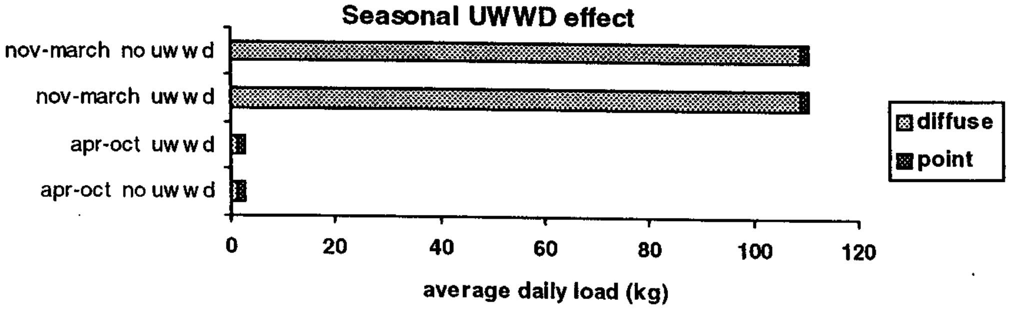
Distribution of P sources



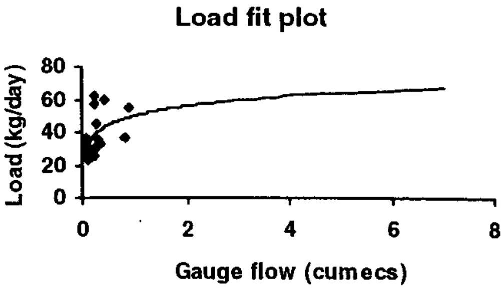
Distribution of land area

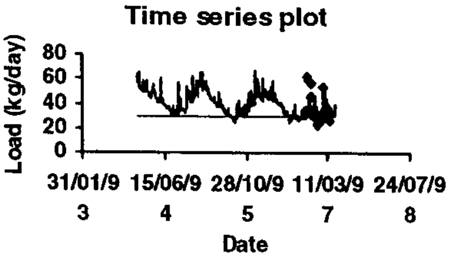


Seasonal UWWD effect

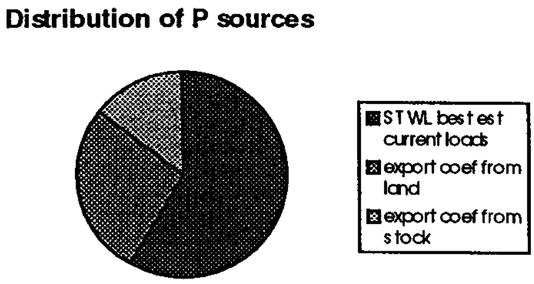


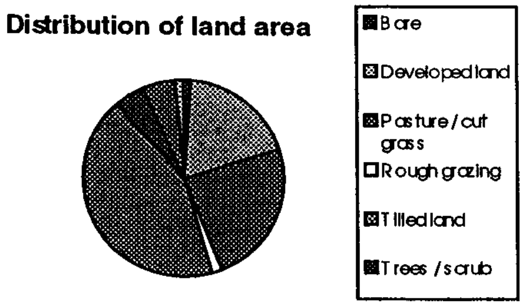
7687380 BADSEY BROOK (OFFENHAM) at:- 405950 245150

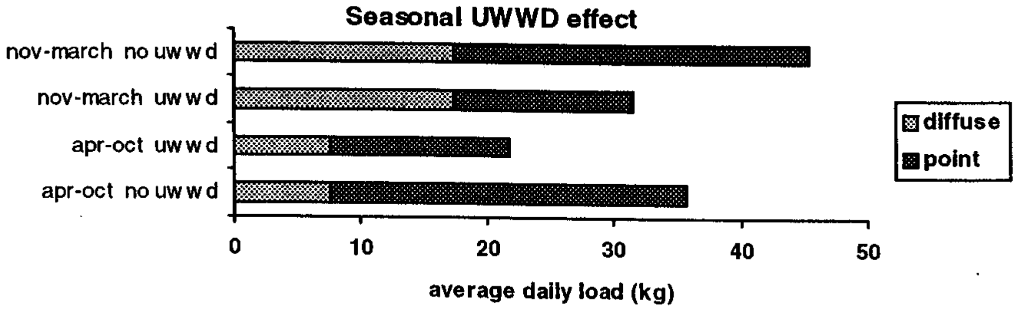




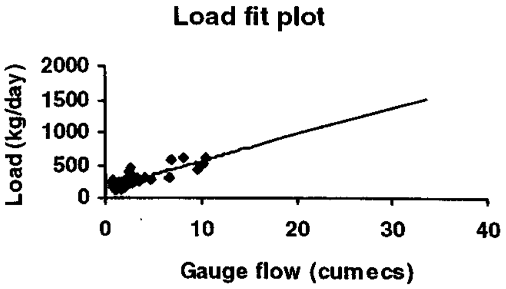
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	13450	20	4366
Total P entering system estimate	18059		
STWL best est current loads	10539		
Total diffuse estimate	7520		
export coef from land	4932		
export coef from stock	2588		
STWL qualifying works capped 2 mg/l	5468		
STWL qualifying works capped 1 mg/l	5468		
total measured load using y=ax^b+c	15249		4176

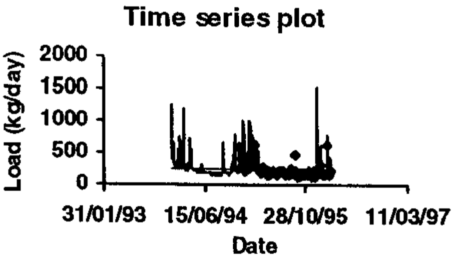




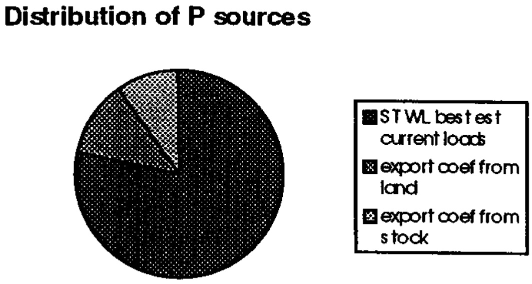


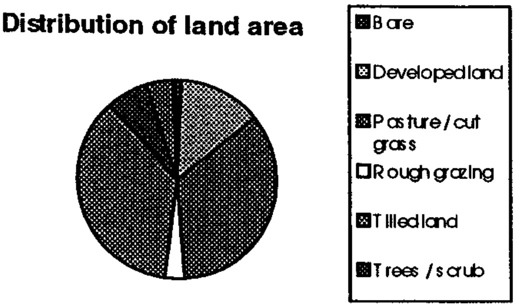
8017335 ARROW (BROOM) at:- 408680 253320

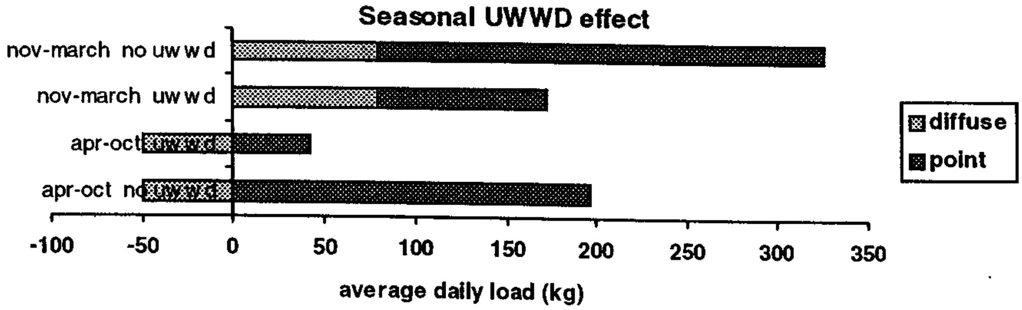




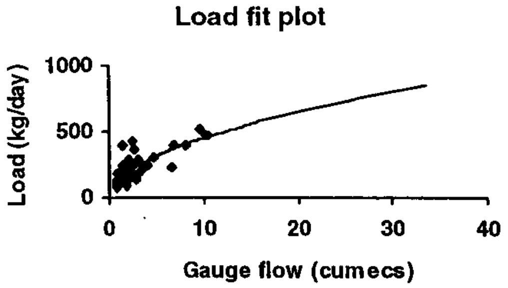
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	88076	103	40244
Total P entering system estimate	117983		
STWL best est current loads	92543		
Total diffuse estimate	25440		
export coef from land	13849		
export coef from stock	11590		
STWL qualifying works capped 2 mg/l	36343		
STWL qualifying works capped 1 mg/l	36343		
total measured load using y=ax^b+c	92635		24659

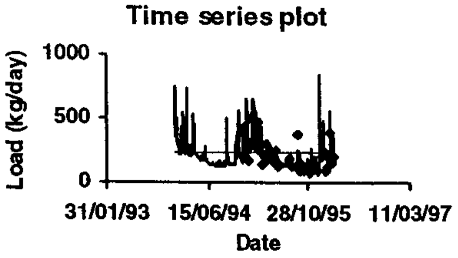




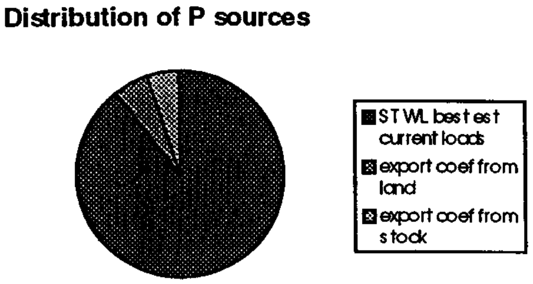


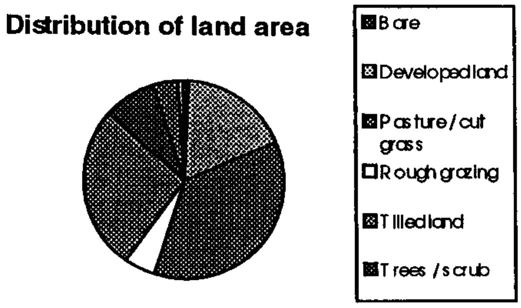
8018960 ARROW (COUGHTON FORD) at:- 408550 260350

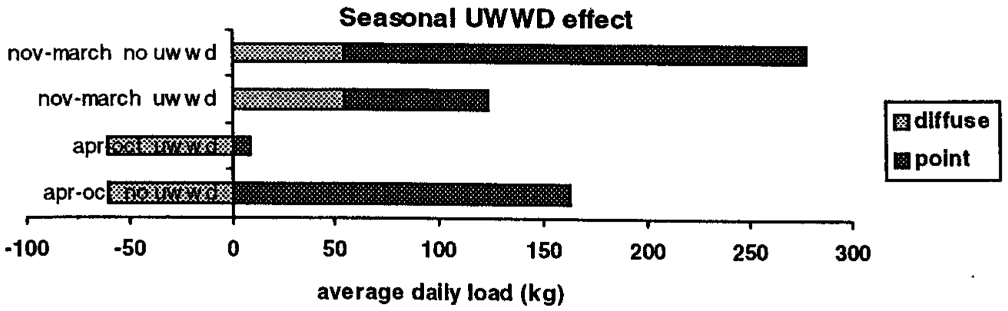

Load fit plot


Time series plot

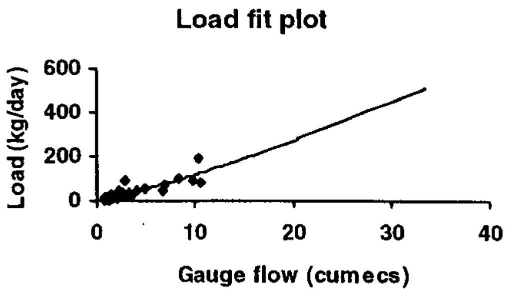
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	70591	76	37478
Total P entering system estimate	93644		
STWL best est current loads	83672		
Total diffuse estimate	9972		
export coef from land	5193		
export coef from stock	4780		
STWL qualifying works capped 2 mg/l	27472		
STWL qualifying works capped 1 mg/l	27472		
total measured load using y=ax^b+c	77672		26700

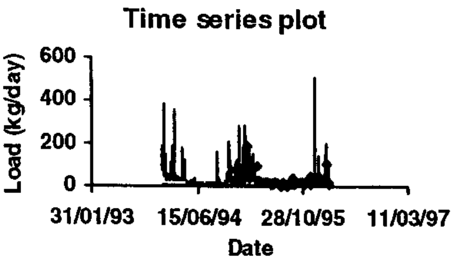

Distribution of P sources


Distribution of land area

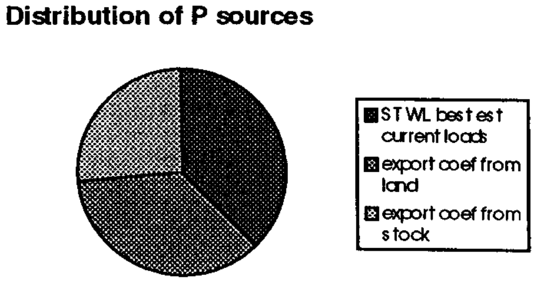

Seasonal UWWD effect

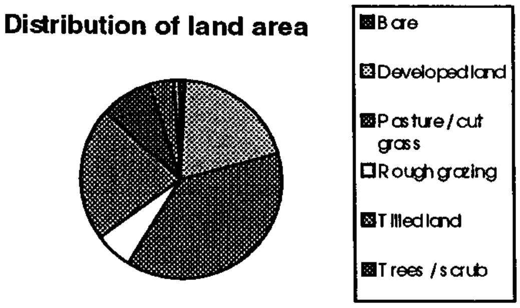
8019580 ARROW (STUDLEY) at:- 407630 263930

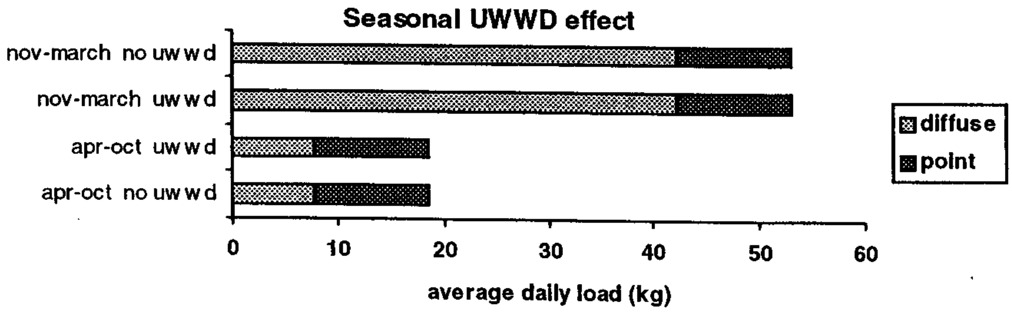




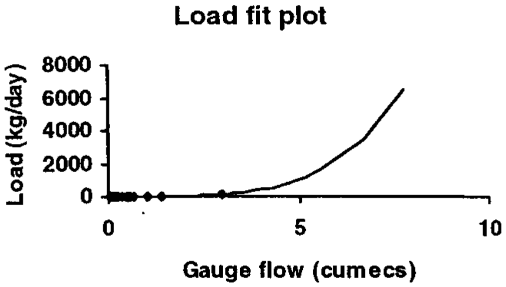
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	9470	104	9829
Total P entering system estimate	10660		
STWL best est current loads	4041		
Total diffuse estimate	6619		
export coef from land	3814		
export coef from stock	2806		
STWL qualifying works capped 2 mg/l	4041		
STWL qualifying works capped 1 mg/l	4041		
total measured load using y=ax^b+c	12382		5851

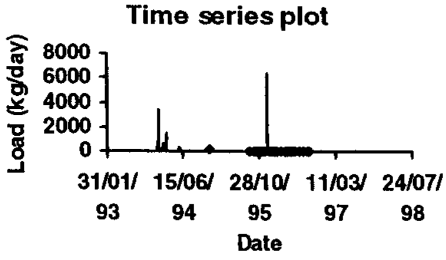




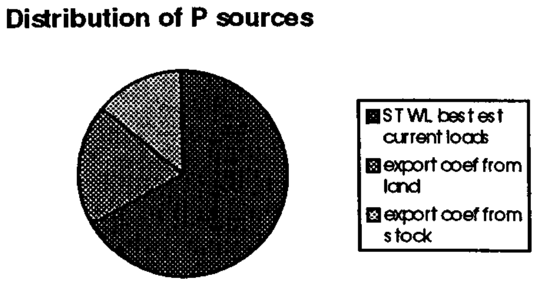


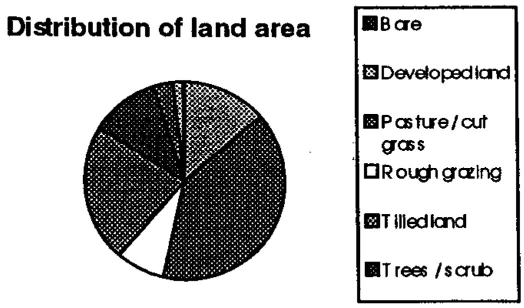
8022000 ARROW (BORDESLEY) A441 ROAD BRIDGE at:- 404050 269250

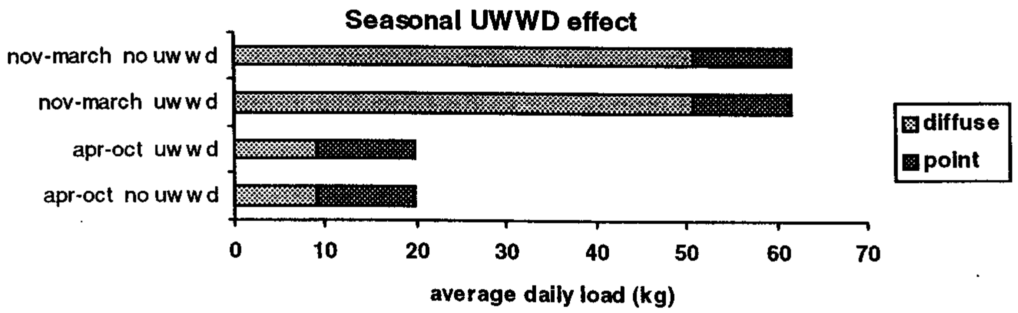




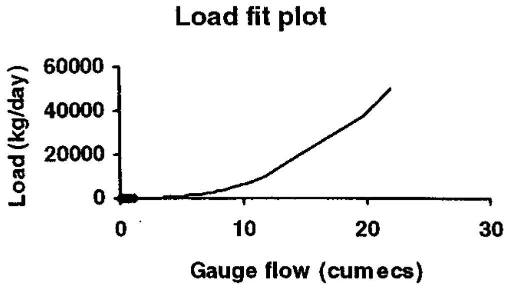
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	11765	37	13022
Total P entering system estimate	6018		
STWL best est current loads	4036		
Total diffuse estimate	1982		
export coef from land	1138		
export coef from stock	844		
STWL qualifying works capped 2 mg/l	4036		
STWL qualifying works capped 1 mg/l	4036		
total measured load using y=ax^b+c	14462		2005

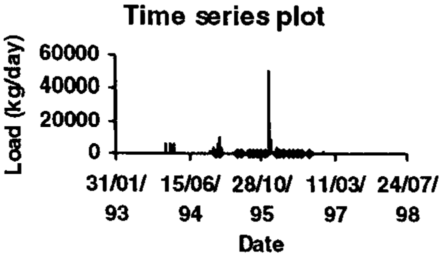




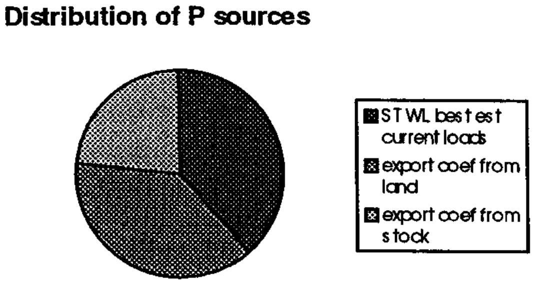


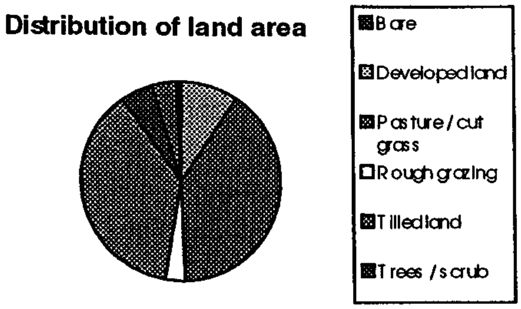
8101500 ALNE (LITTLE ALNE)   at:-   414250   261150

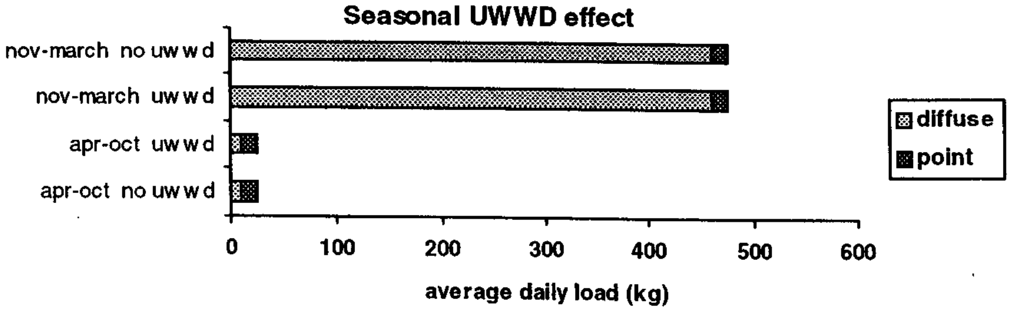




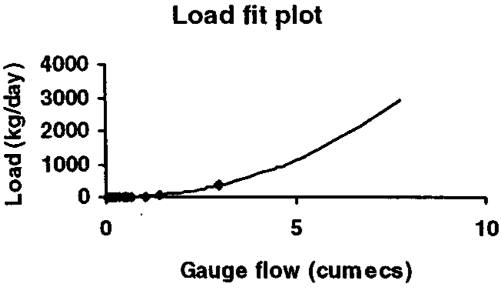
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	10400	20	7296
Total P entering system estimate	14950		
STWL best est current loads	5789		
Total diffuse estimate	9161		
export coef from land	5631		
export coef from stock	3530		
STWL qualifying works capped 2 mg/l	5789		
STWL qualifying works capped 1 mg/l	5789		
total measured load using y=ax^b+c	104032		1771

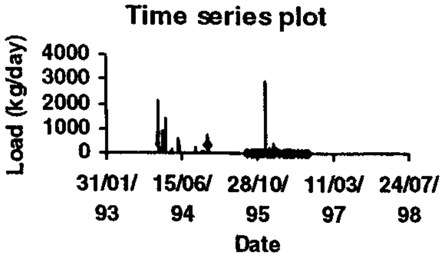




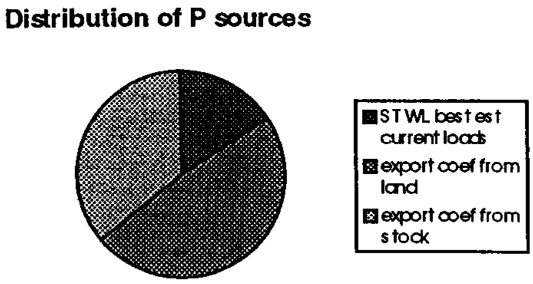


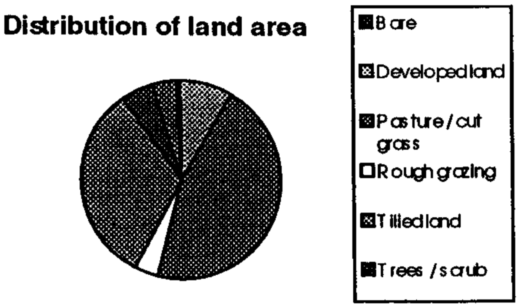
8102620 ALNE (WOOTTON WAWEN) at:- 415600 263100

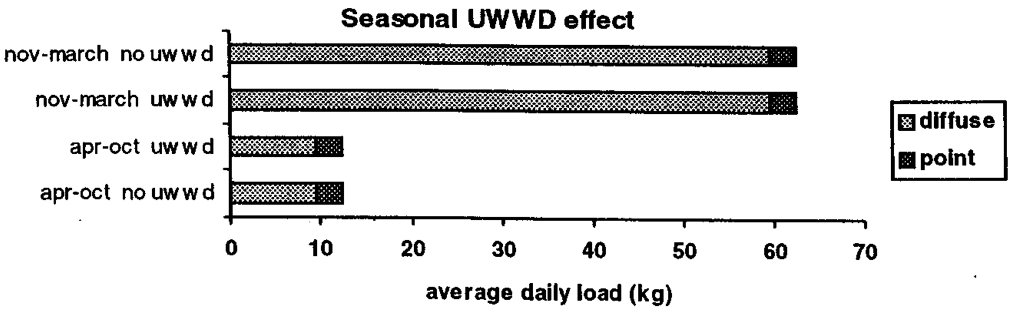




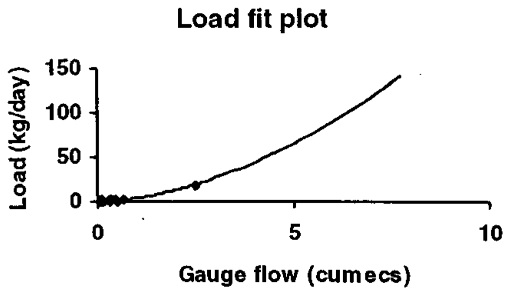
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	10428	38	21210
Total P entering system estimate	6928		
STWL best est current loads	1110		
Total diffuse estimate	5818		
export coef from land	3316		
export coef from stock	2502		
STWL qualifying works capped 2 mg/l	1110		
STWL qualifying works capped 1 mg/l	1110		
total measured load using y=ax^b+c	13738		1229

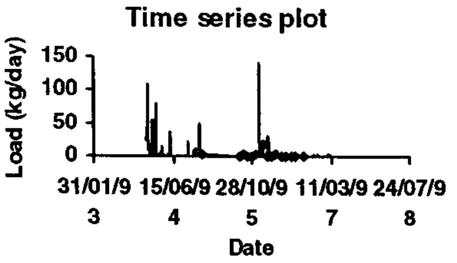




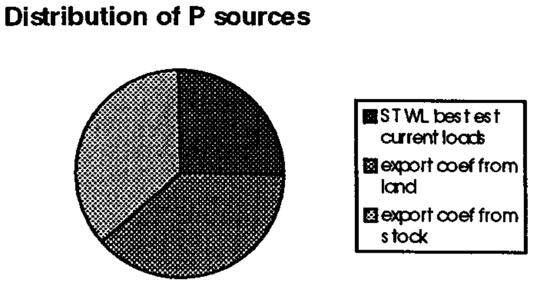


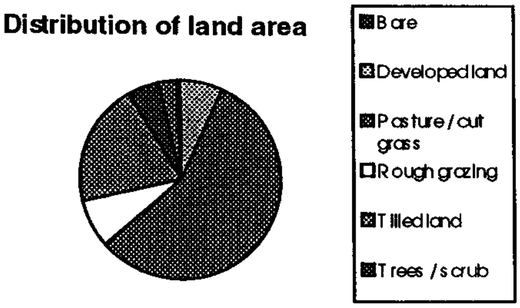
8104600 ALNE (DANZEY GREEN) at:- 412700 269480

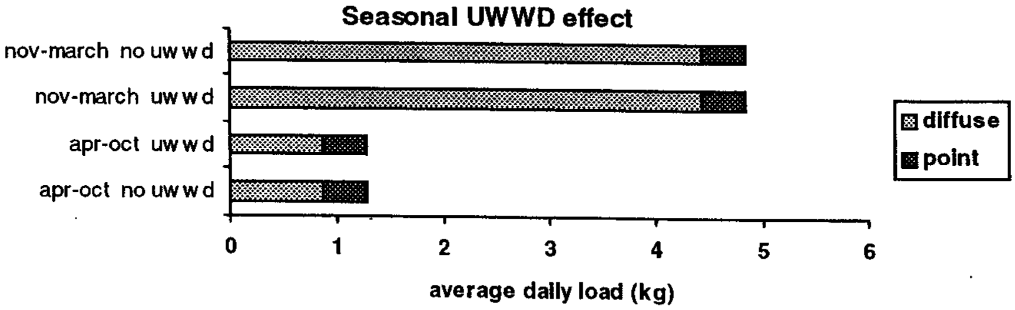




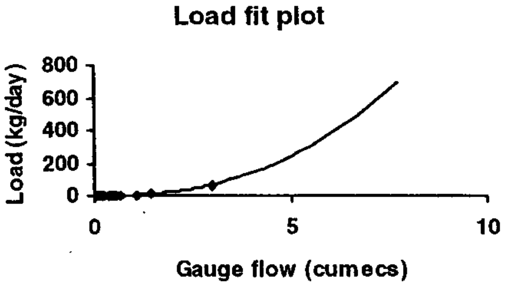
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	1007	21	1639
Total P entering system estimate	615		
STWL best est current loads	156		
Total diffuse estimate	459		
export coef from land	233		
export coef from stock	226		
STWL qualifying works capped 2 mg/l	156		
STWL qualifying works capped 1 mg/l	156		
total measured load using y=ax^b+c	1055		207

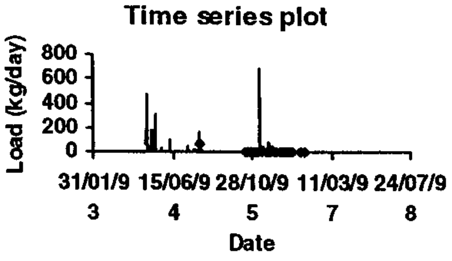




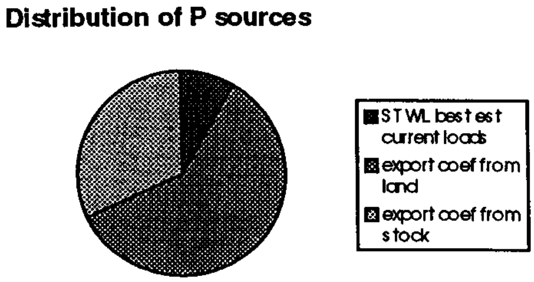


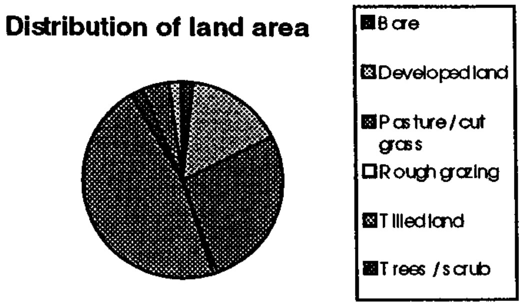
9253100 NOLEHAM BROOK (WELFORD PASTURES) at:- 412100 251400

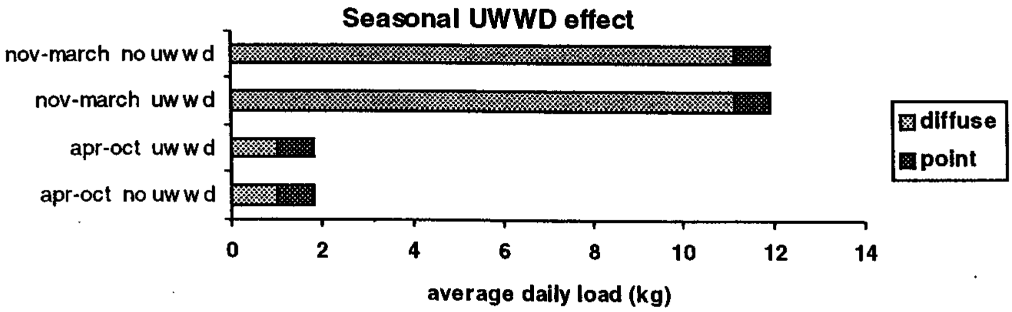




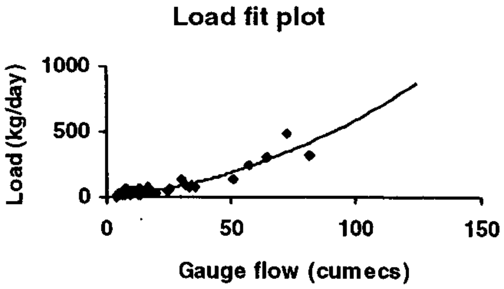
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	1892	34	4252
Total P entering system estimate	3479		
STWL best est current loads	295		
Total diffuse estimate	3184		
export coef from land	2070		
export coef from stock	1113		
STWL qualifying works capped 2 mg/l	295		
STWL qualifying works capped 1 mg/l	295		
total measured load using y=ax^b+c	2616		339

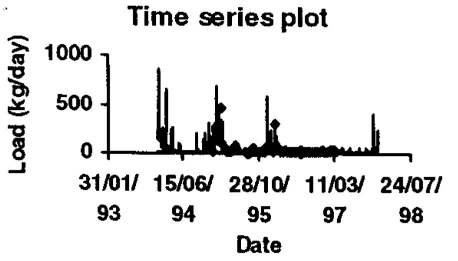




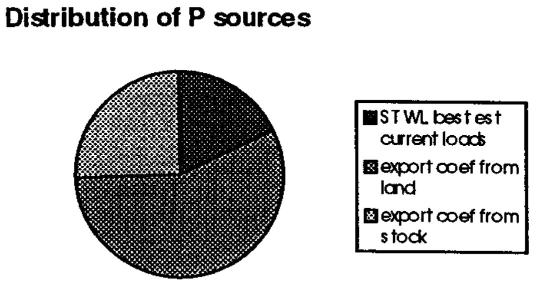


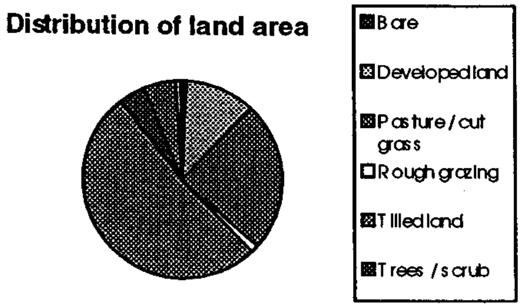
9563180 STOUR (CLIFFORD CHAMBERS) at:- 419600 252800

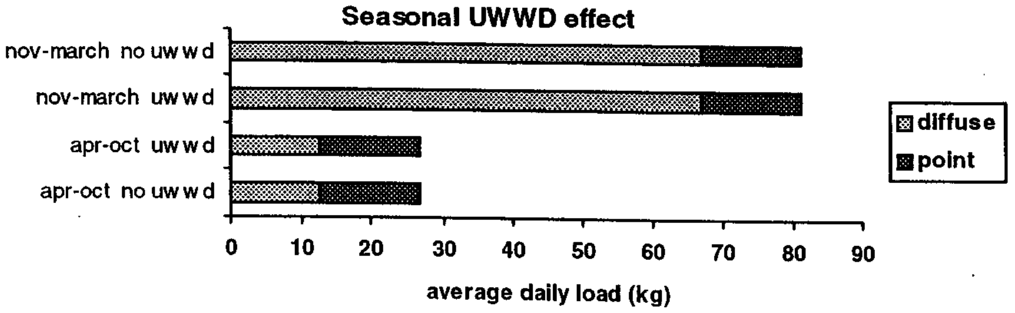




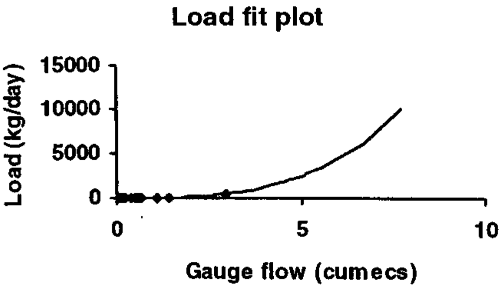
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	16155	100	24278
Total P entering system estimate	29302		
STWL best est current loads	5314		
Total diffuse estimate	23988		
export coef from land	16524		
export coef from stock	7464		
STWL qualifying works capped 2 mg/l	5314		
STWL qualifying works capped 1 mg/l	5314		
total measured load using y=ax^b+c	24635		8189

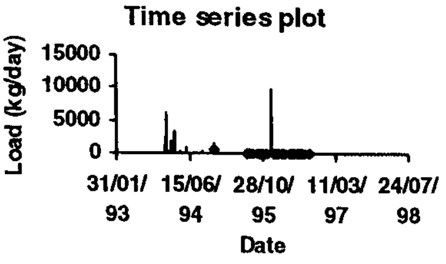




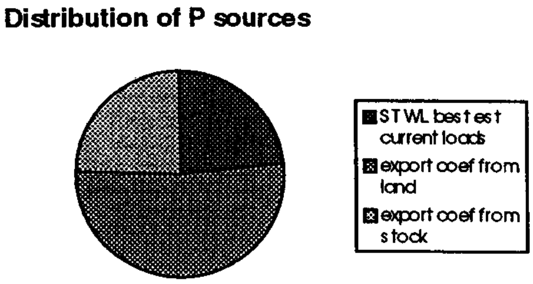


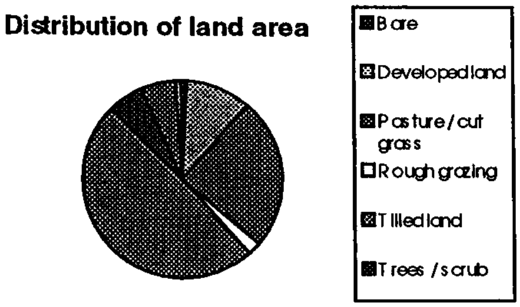
9567610 STOUR (SHIPSTON) at:- 426100 240400

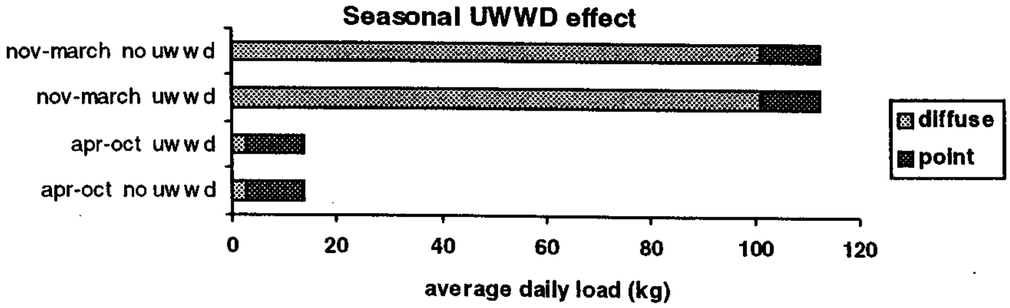




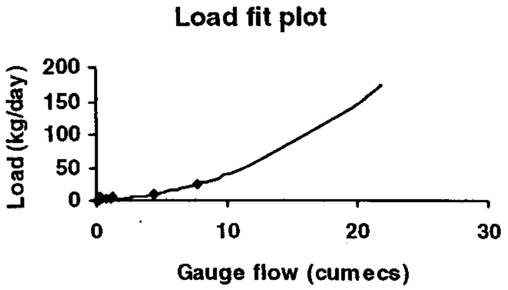
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	10693	37	29643
Total P entering system estimate	17846		
STWL best est current loads	4204		
Total diffuse estimate	13642		
export coef from land	9222		
export coef from stock	4420		
STWL qualifying works capped 2 mg/l	4204		
STWL qualifying works capped 1 mg/l	4204		
total measured load using y=ax^b+c	25045		2498

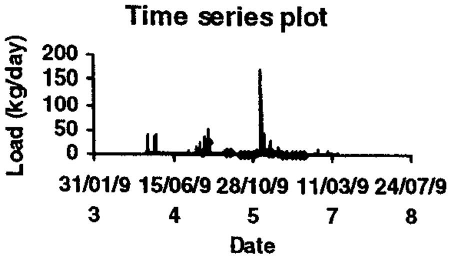




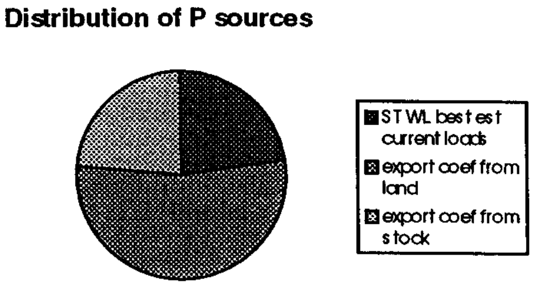


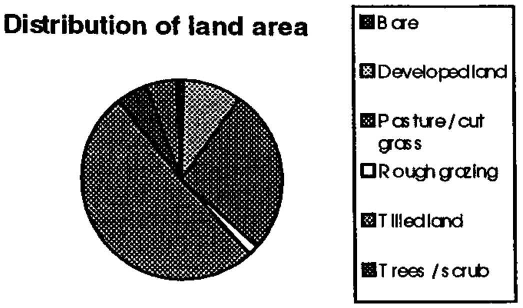
9569250 STOUR (CHERINGTON) at:- 429000 236900

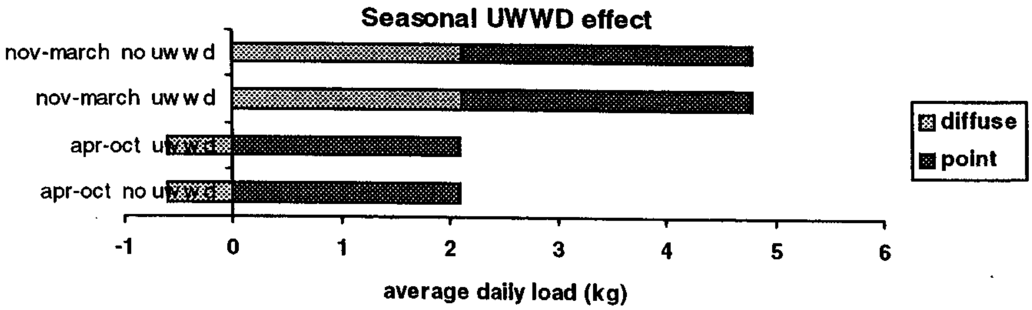




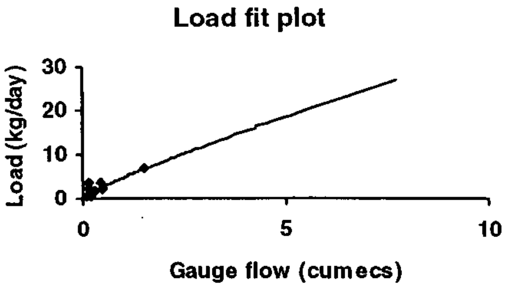
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	1609	22	2336
Total P entering system estimate	4426		
STWL best est current loads	1005		
Total diffuse estimate	3421		
export coef from land	2361		
export coef from stock	1060		
STWL qualifying works capped 2 mg/l	1005		
STWL qualifying works capped 1 mg/l	1005		
total measured load using y=ax^b+c	1221		430

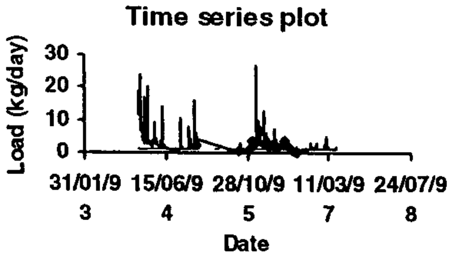




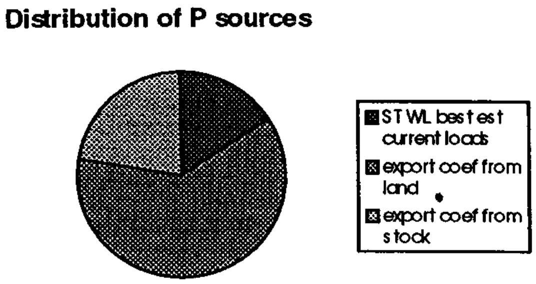


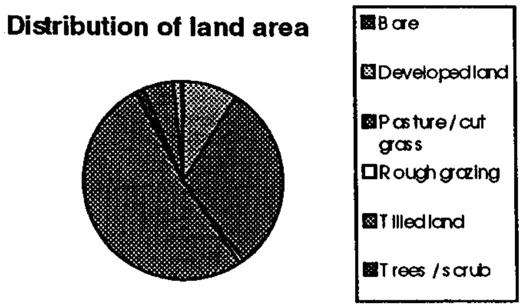
9693120 WYNYATES BROOK (CONF. STOUR) at:- 426800 244200

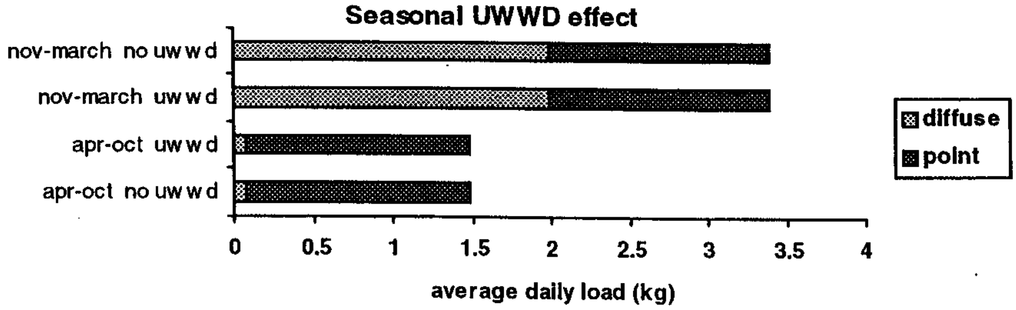




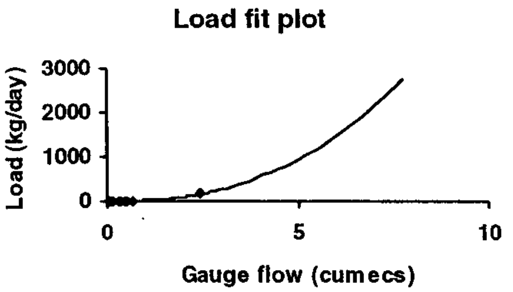
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	1651	17	3062
Total P entering system estimate	3305		
STWL best est current loads	527		
Total diffuse estimate	2778		
export coef from land	2038		
export coef from stock	740		
STWL qualifying works capped 2 mg/l	527		
STWL qualifying works capped 1 mg/l	527		
total measured load using y=ax^b+c	732		304

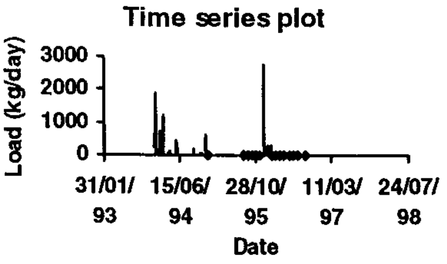




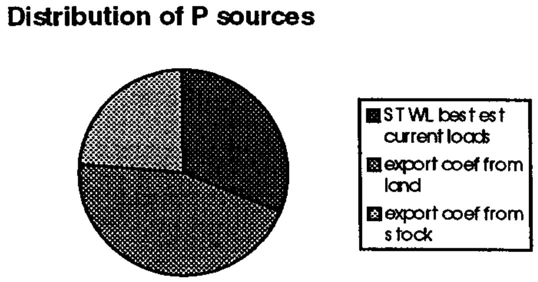


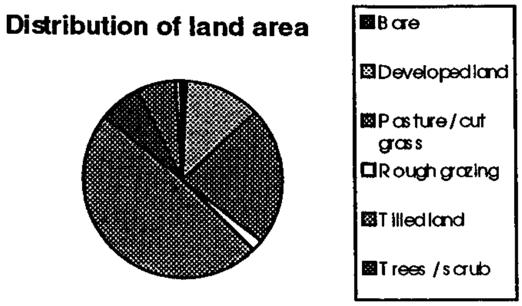
9834100 KNEE BROOK (HIGH FURZE) at:- 424600 237500

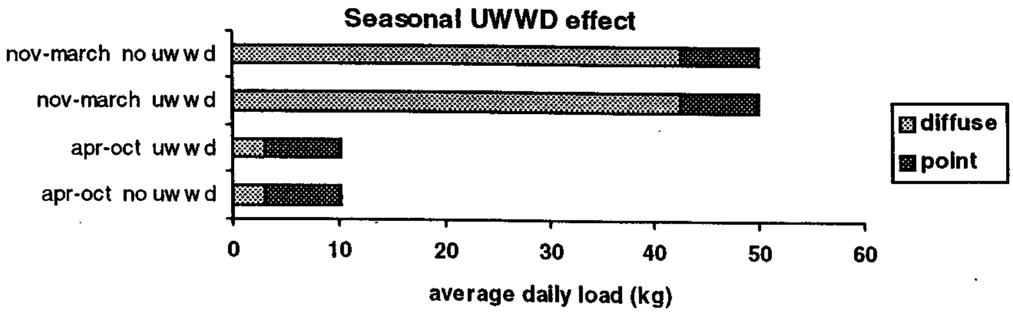




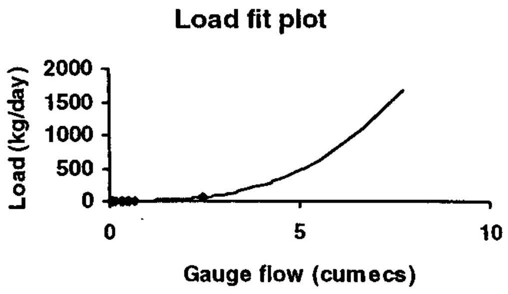
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	6566	19	13506
Total P entering system estimate	8741		
STWL best est current loads	2733		
Total diffuse estimate	6008		
export coef from land	3918		
export coef from stock	2090		
STWL qualifying works capped 2 mg/l	2733		
STWL qualifying works capped 1 mg/l	2733		
total measured load using y=ax^b+c	11114		868

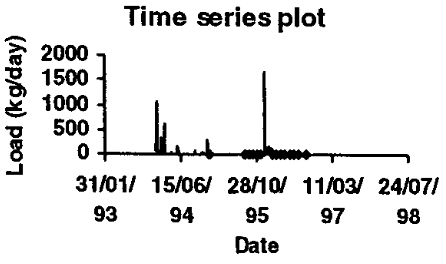




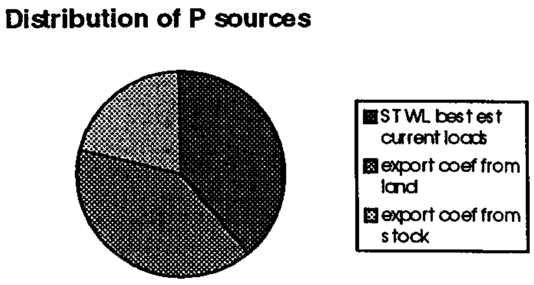


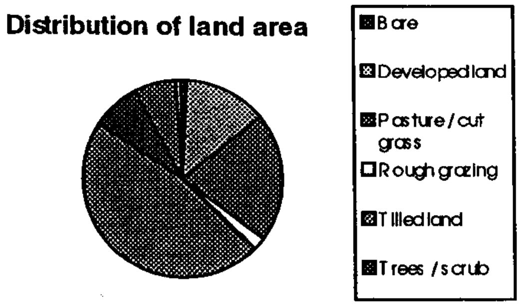
9835650 KNEE BROOK (PAXFORD) at:- 419300 237100

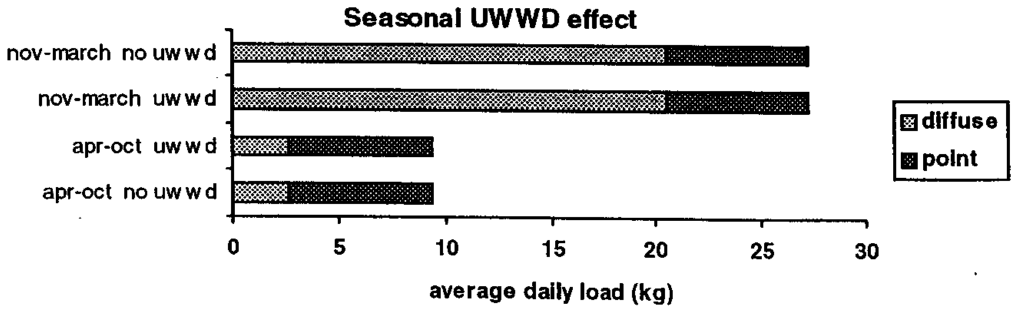




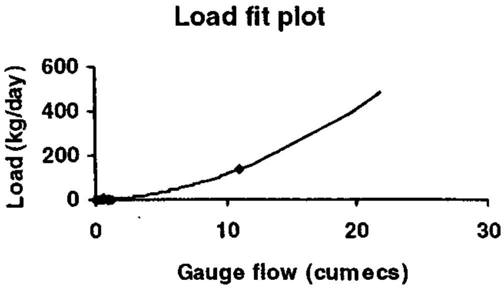
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	4559	21	4850
Total P entering system estimate	6511		
STWL best est current loads	2527		
Total diffuse estimate	3984		
export coef from land	2595		
export coef from stock	1389		
STWL qualifying works capped 2 mg/l	2527		
STWL qualifying works capped 1 mg/l	2527		
total measured load using y=ax^b+c	6365		602

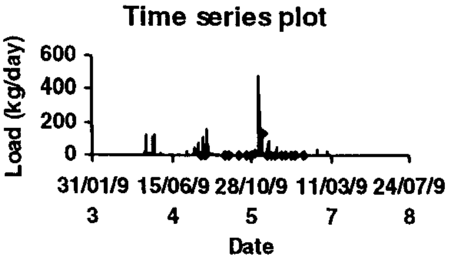




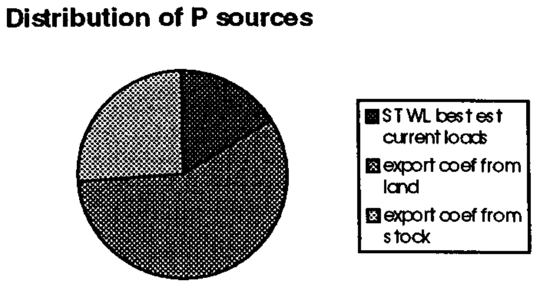


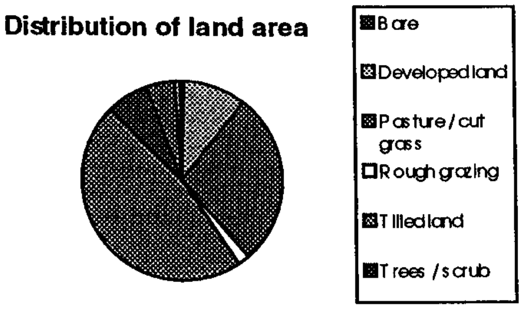
9983000 NETHERCOTE BROOK (MITFORD BRIDGE, CONF. STOUR) at:- 426300 237100

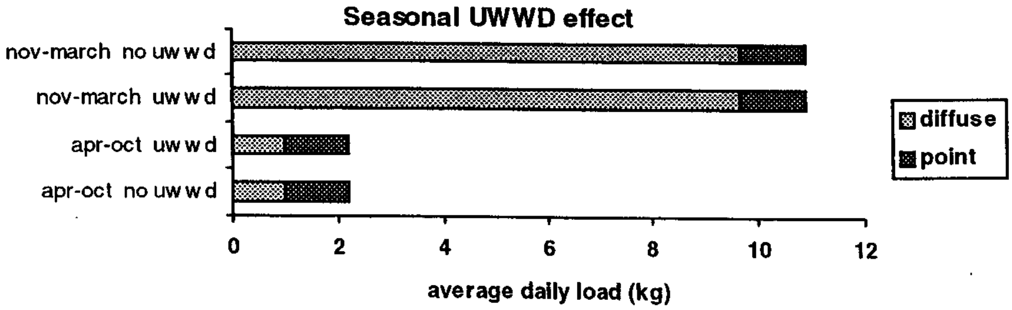




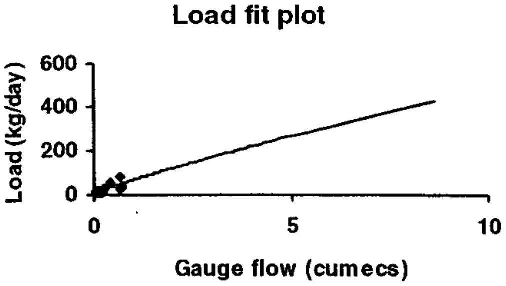
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	3508	19	11389
Total P entering system estimate	2838		
STWL best est current loads	466		
Total diffuse estimate	2372		
export coef from land	1632		
export coef from stock	740		
STWL qualifying works capped 2 mg/l	466		
STWL qualifying works capped 1 mg/l	466		
total measured load using y=ax^b+c	2453		431

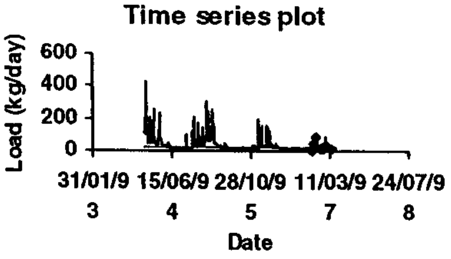




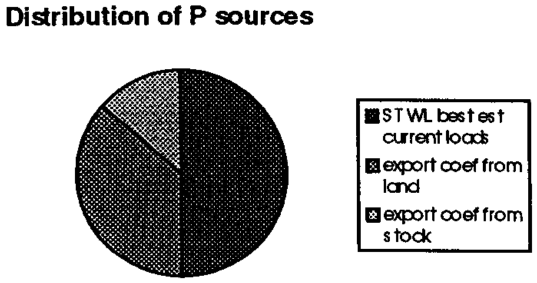


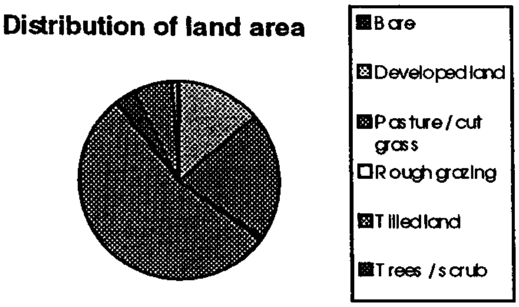
10324230 DENE (CHARLECOTE) at:- 426300 256200

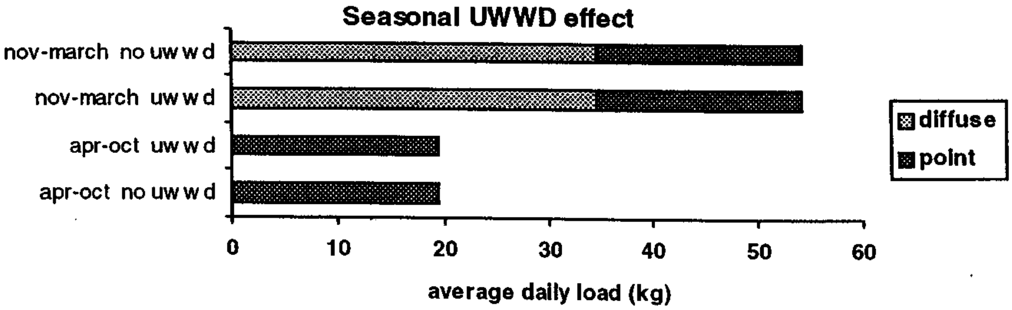




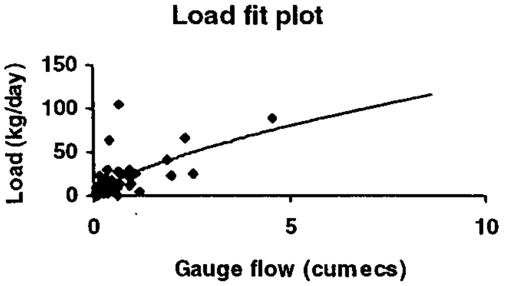
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	8621	17	7042
Total P entering system estimate	14641		
STWL best est current loads	7307		
Total diffuse estimate	7334		
export coef from land	5349		
export coef from stock	1984		
STWL qualifying works capped 2 mg/l	7307		
STWL qualifying works capped 1 mg/l	7307		
total measured load using y=ax^b+c	15794		5759

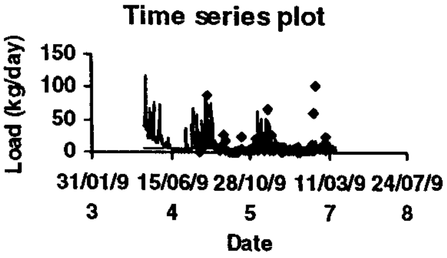




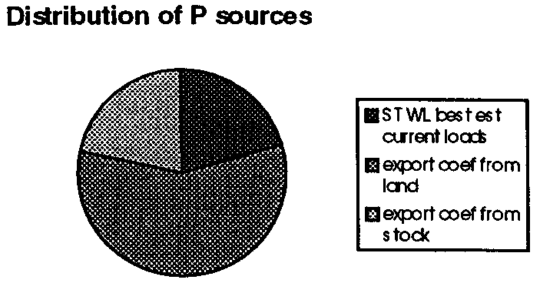


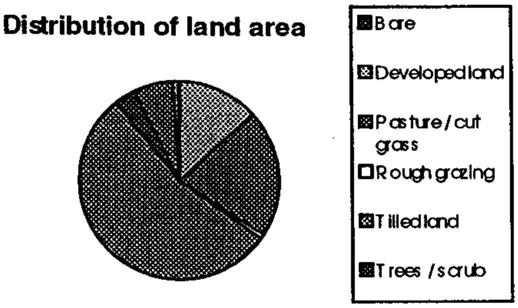
10324320 DENE (WELLESBOURNE) at:- 427800 255200

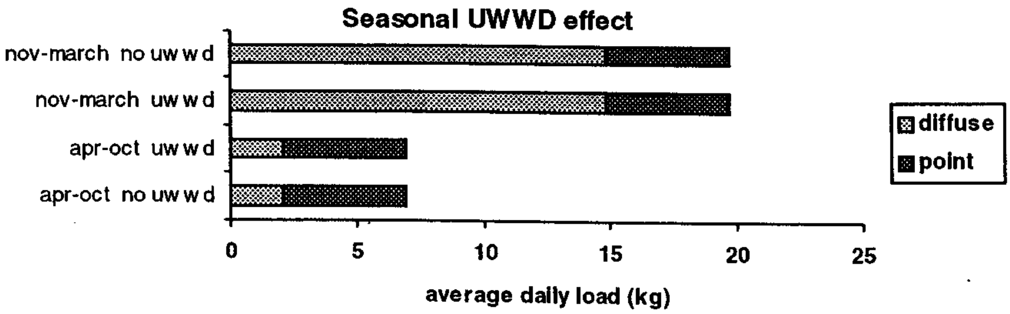




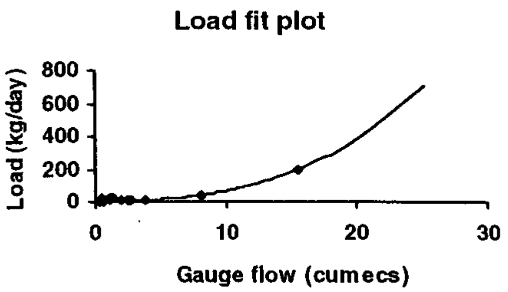
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	3911	105	5935
Total P entering system estimate	8873		
STWL best est current loads	1825		
Total diffuse estimate	7048		
export coef from land	5136		
export coef from stock	1912		
STWL qualifying works capped 2 mg/l	1825		
STWL qualifying works capped 1 mg/l	1825		
total measured load using y=ax^b+c	5683		4326

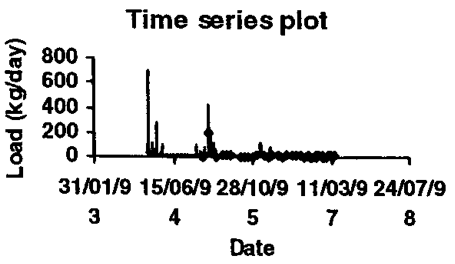




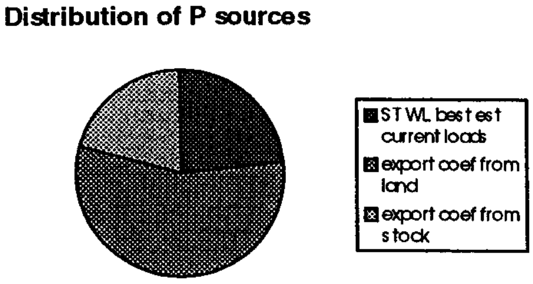


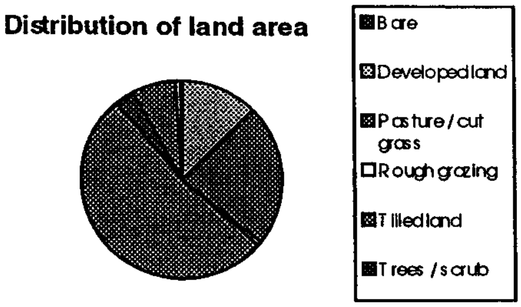
10325060 DENE (FOSSE WAY) at:- 429100 250800

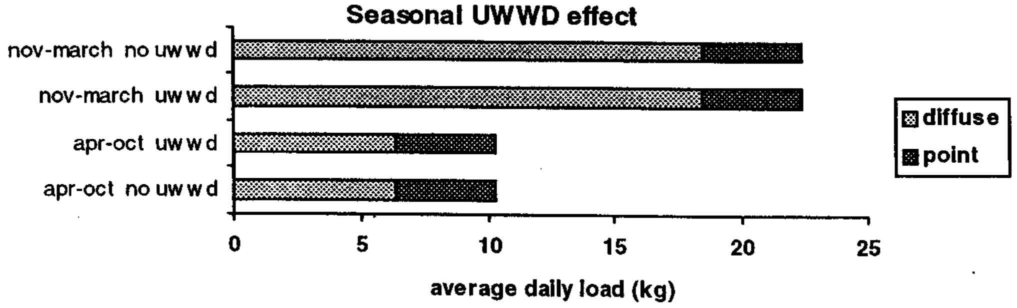

Load fit plot


Time series plot

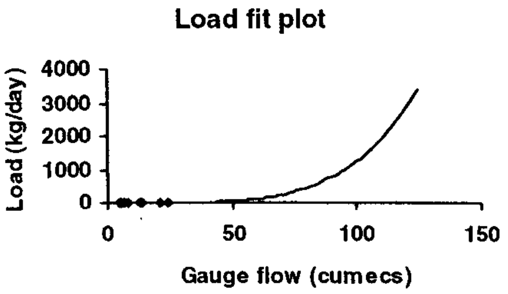
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	5488	43	10792
Total P entering system estimate	6341		
STWL best est current loads	1490		
Total diffuse estimate	4851		
export coef from land	3518		
export coef from stock	1333		
STWL qualifying works capped 2 mg/l	1490		
STWL qualifying works capped 1 mg/l	1490		
total measured load using y=ax^b+c	7072		2069

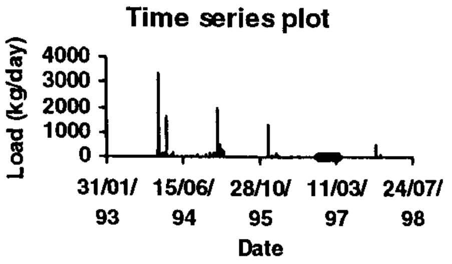

Distribution of P sources


Distribution of land area

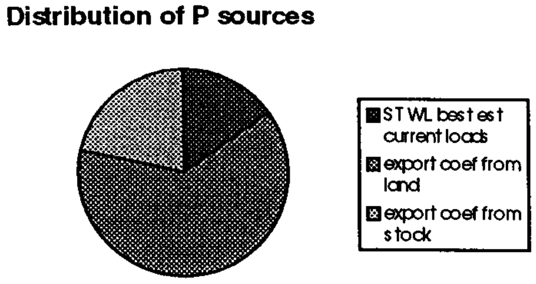

Seasonal UWWD effect

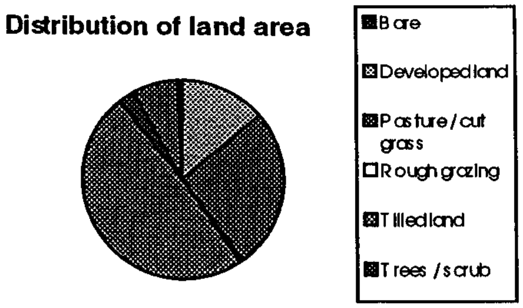
10325700 DENE (KINETON) at:- 434400 250700

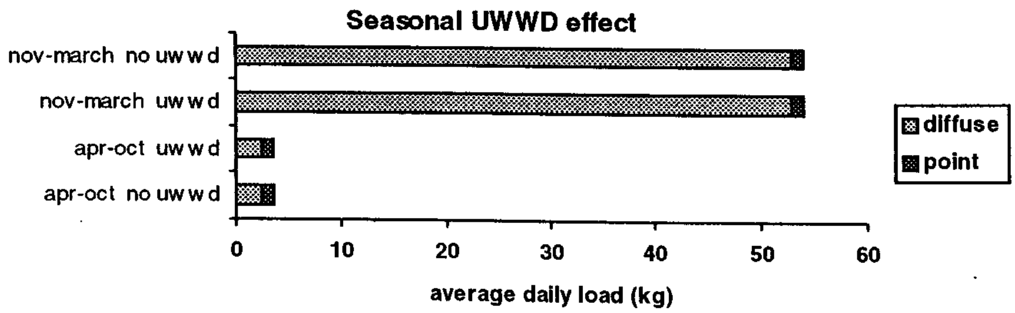




Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	1041	18	341
Total P entering system estimate	2708		
STWL best est current loads	413		
Total diffuse estimate	2295		
export coef from land	1709		
export coef from stock	587		
STWL qualifying works capped 2 mg/l	413		
STWL qualifying works capped 1 mg/l	413		
total measured load using y=ax^b+c	15997		321

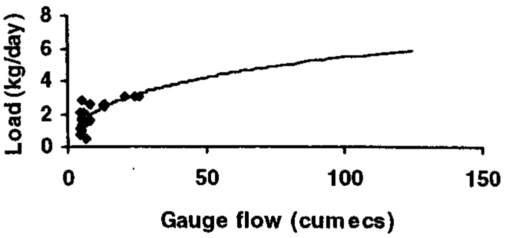




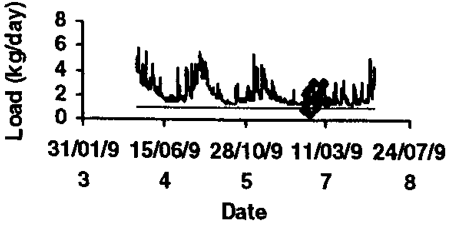


10353020 ETTINGTON BROOK (WELLESBOURNE ROAD) at:- 428400 250800

Load fit plot

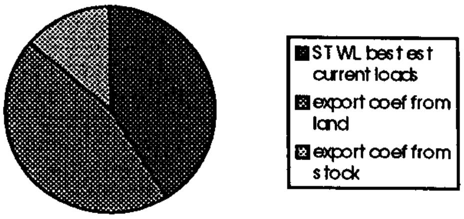


Time series plot

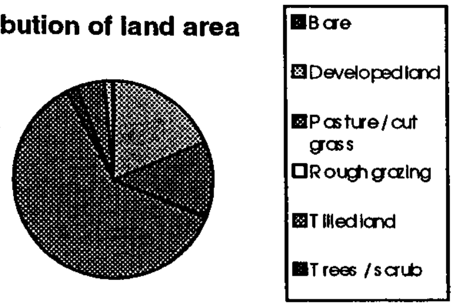


Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	679	21	284
Total P entering system estimate	823		
STWL best est current loads	335		
Total diffuse estimate	488		
export coef from land	376		
export coef from stock	112		
STWL qualifying works capped 2 mg/l	335		
STWL qualifying works capped 1 mg/l	335		
total measured load using y=ax^b+c	885		205

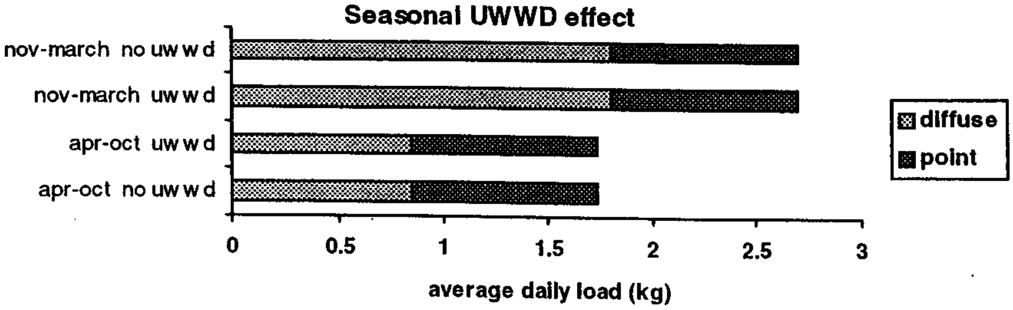
Distribution of P sources



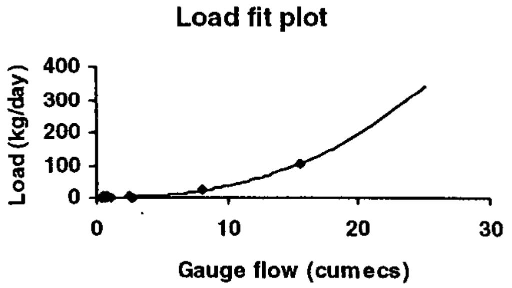
Distribution of land area

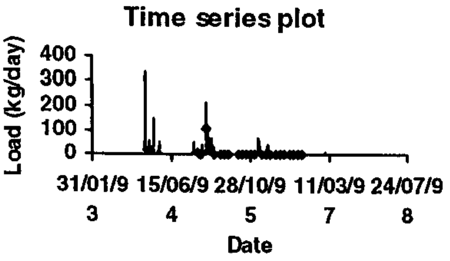


Seasonal UWWD effect

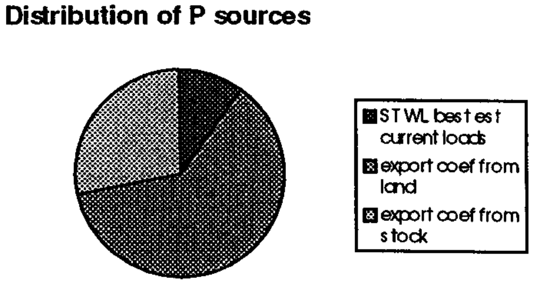


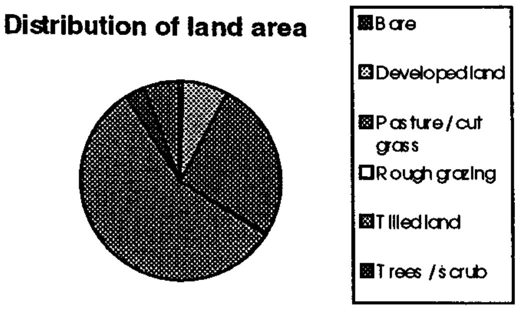
10579020 THELSFORD BROOK (HAMPTON LUCY) at:- 426000 257300

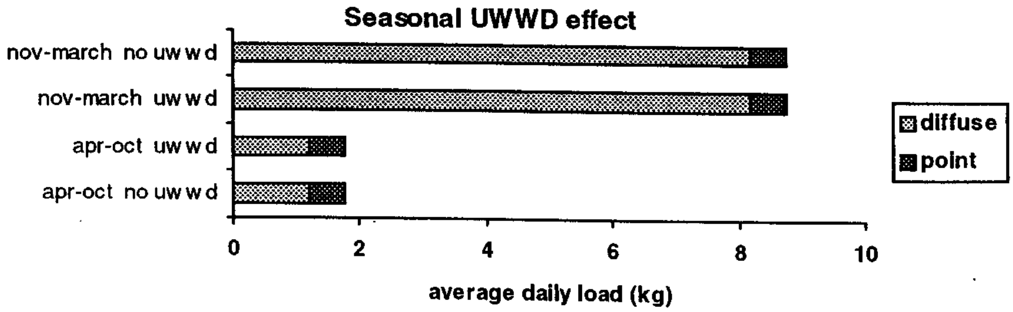




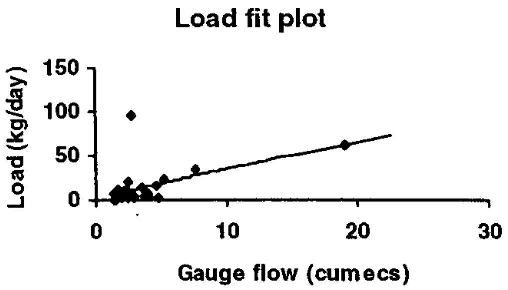
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	2602	22	8060
Total P entering system estimate	2118		
STWL best est current loads	214		
Total diffuse estimate	1904		
export coef from land	1302		
export coef from stock	602		
STWL qualifying works capped 2 mg/l	214		
STWL qualifying works capped 1 mg/l	214		
total measured load using y=ax^b+c	2551		367

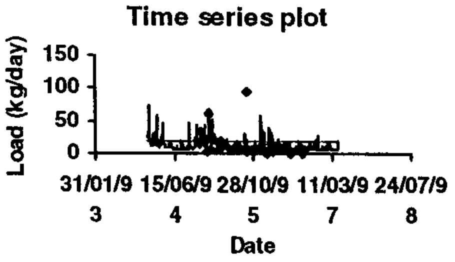




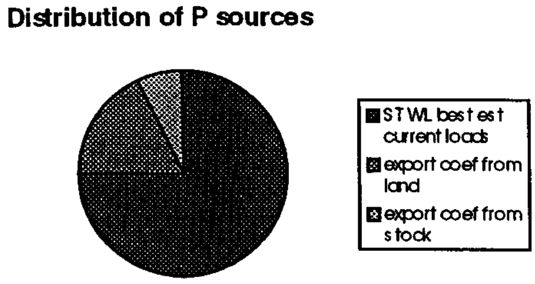


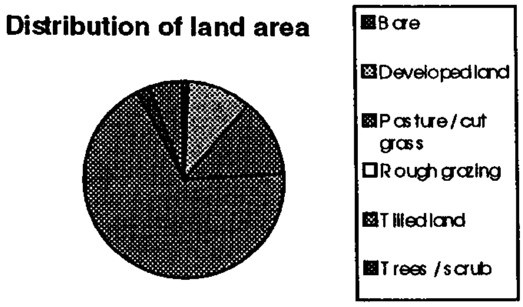
10863220 TACH BROOK (A41 ROAD BRIDGE) at:- 429500 263500

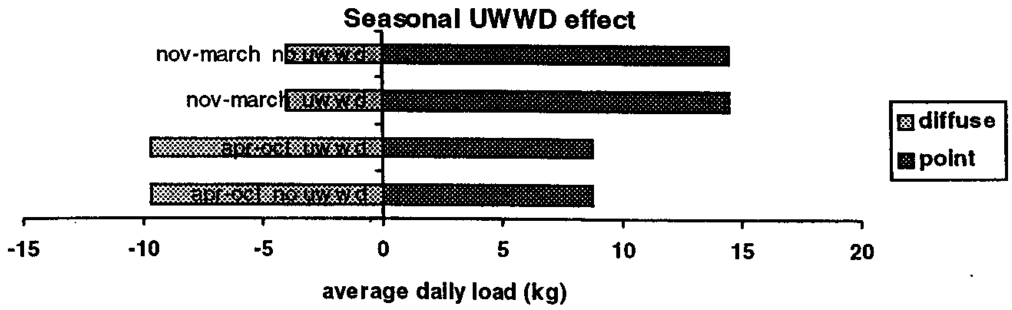

Load fit plot


Time series plot

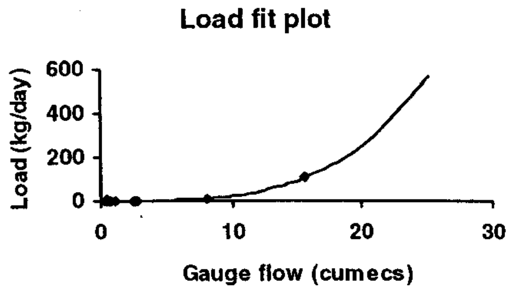
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	4419	35	6716
Total P entering system estimate	9230		
STWL best est current loads	6930		
Total diffuse estimate	2300		
export coef from land	1658		
export coef from stock	642		
STWL qualifying works capped 2 mg/l	6930		
STWL qualifying works capped 1 mg/l	6930		
total measured load using y=ax^b+c	4722		5954


Distribution of P sources


Distribution of land area

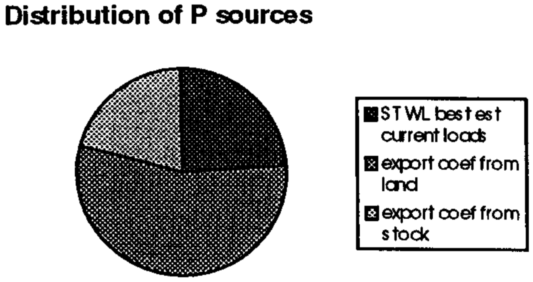

Seasonal UWWD effect

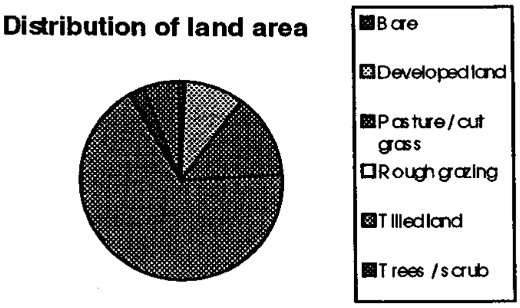
10863900 TACH BROOK (A452 ROAD BRIDGE) at:- 431500 261600

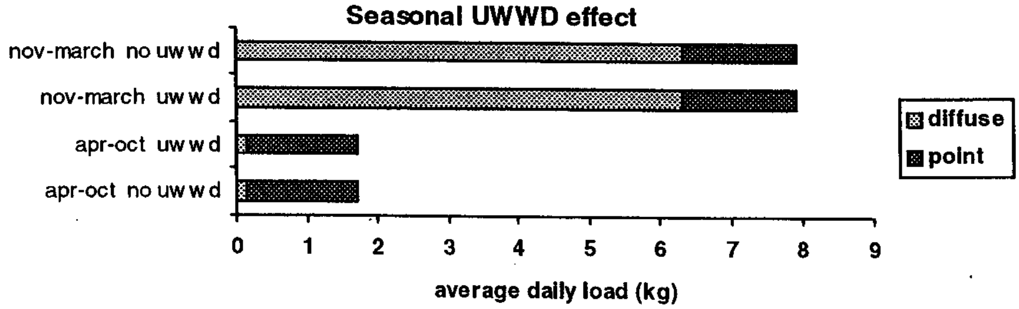

Load fit plot


Time series plot

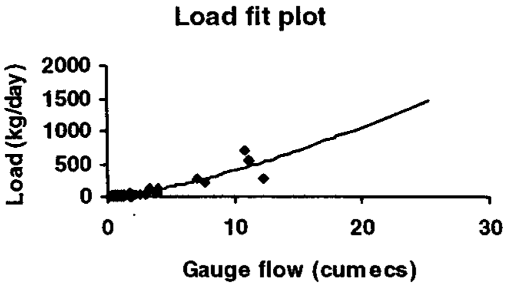
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	2391	23	8020
Total P entering system estimate	2465		
STWL best est current loads	598		
Total diffuse estimate	1867		
export coef from land	1352		
export coef from stock	515		
STWL qualifying works capped 2 mg/l	598		
STWL qualifying works capped 1 mg/l	598		
total measured load using y=ax^b+c	2324		657

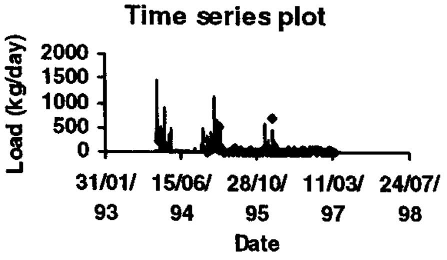

Distribution of P sources


Distribution of land area

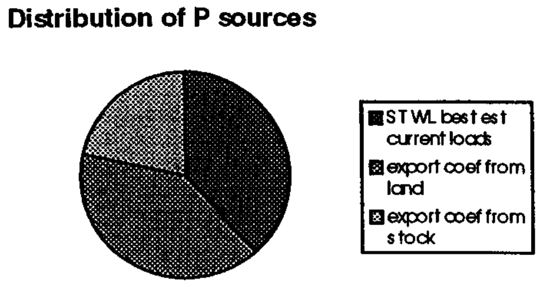

Seasonal UWWD effect

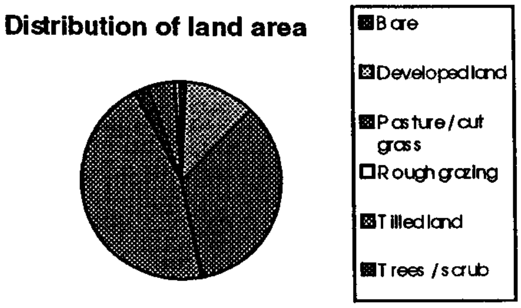
10927160 LEAM (PRINCES DRIVE) at:- 430800 265400

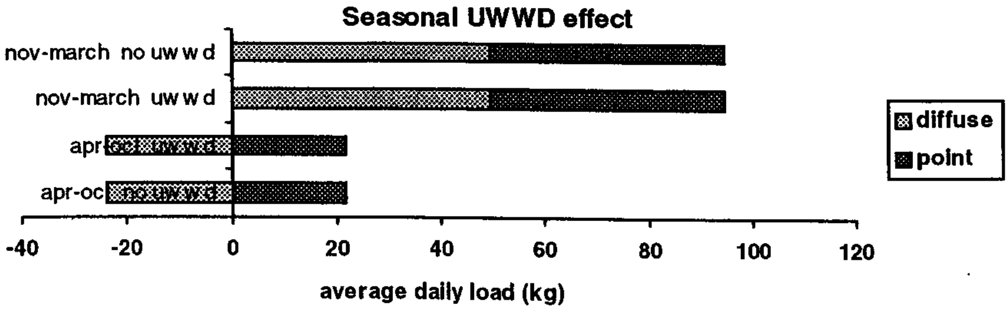




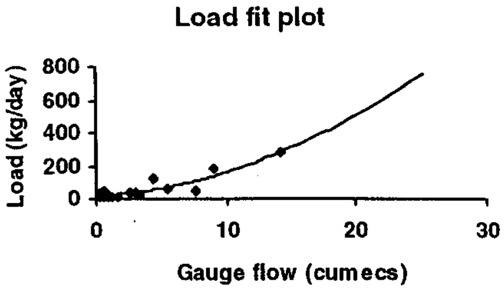
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	14128	104	34320
Total P entering system estimate	44615		
STWL best est current loads	16967		
Total diffuse estimate	27648		
export coef from land	18002		
export coef from stock	9647		
STWL qualifying works capped 2 mg/l	16967		
STWL qualifying works capped 1 mg/l	16967		
total measured load using y=ax^b+c	28243		14950

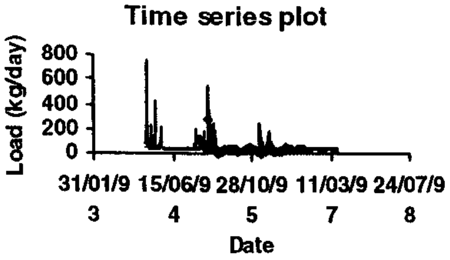




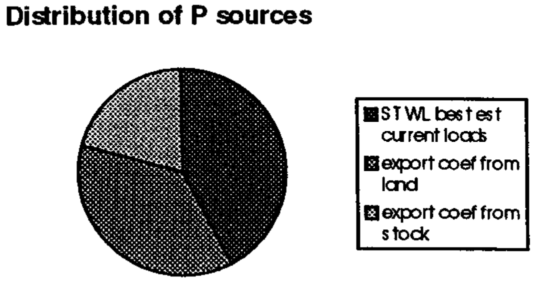


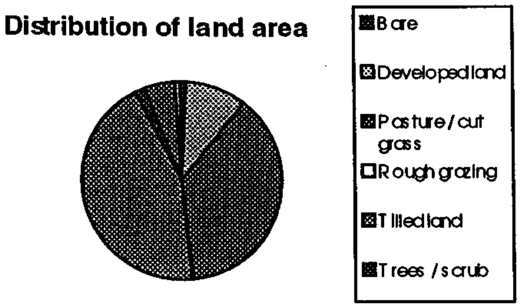
10929750 LEAM (EATHORPE) at:- 438800 268900

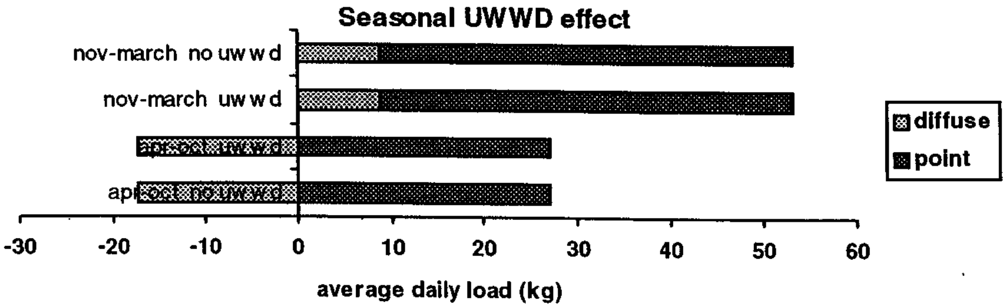




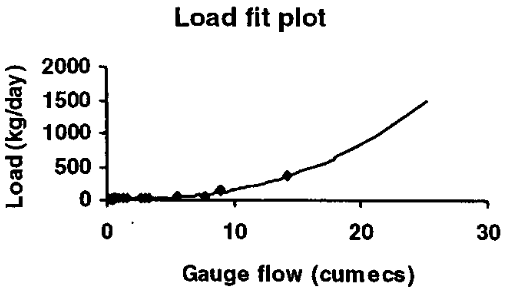
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	14302	41	17773
Total P entering system estimate	39165		
STWL best est current loads	16631		
Total diffuse estimate	22534		
export coef from land	14272		
export coef from stock	8263		
STWL qualifying works capped 2 mg/l	16631		
STWL qualifying works capped 1 mg/l	16631		
total measured load using y=ax^b+c	17006		6913

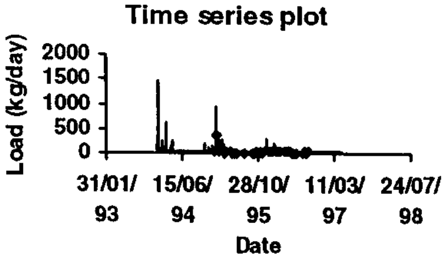




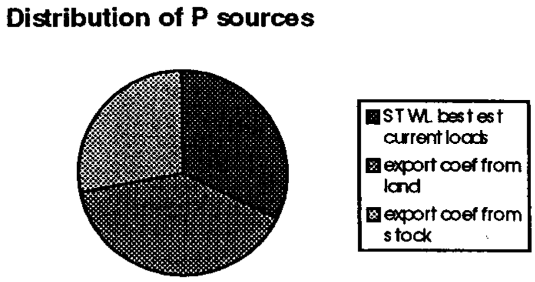


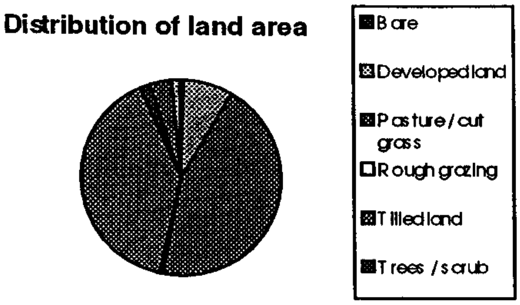
10930970 LEAM (BIRDINGBURY) at:- 443100 269000

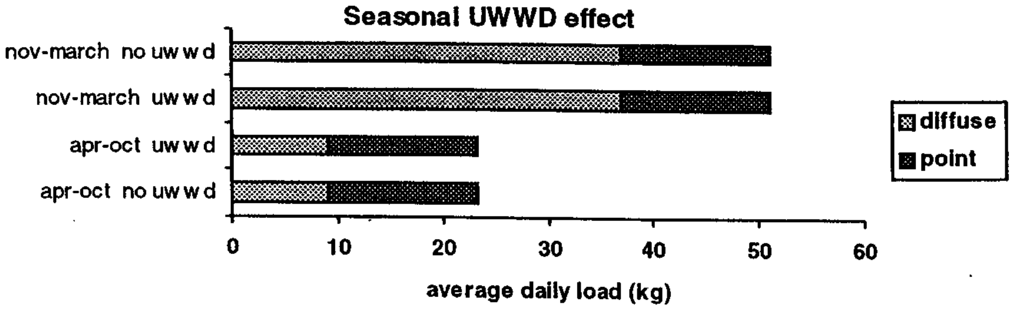




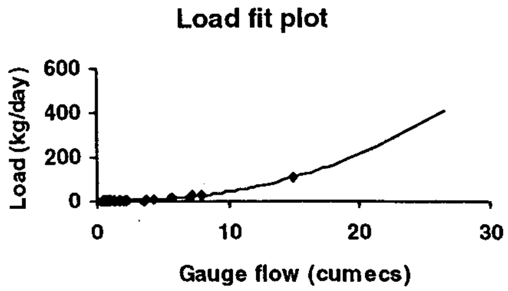
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	13371	37	21006
Total P entering system estimate	16531		
STWL best est current loads	5316		
Total diffuse estimate	11215		
export coef from land	6570		
export coef from stock	4645		
STWL qualifying works capped 2 mg/l	5316		
STWL qualifying works capped 1 mg/l	5316		
total measured load using y=ax^b+c	16100		3769

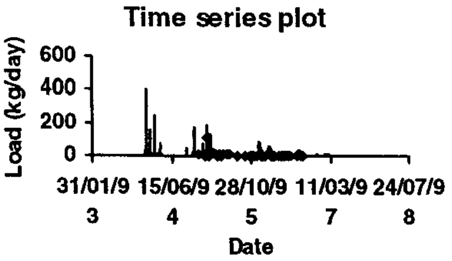




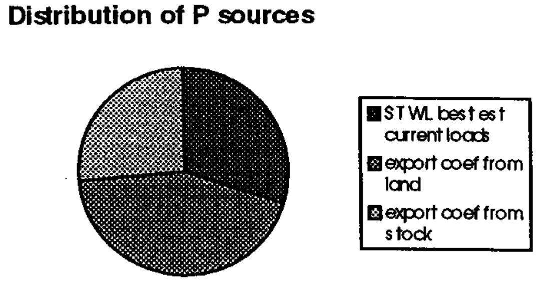


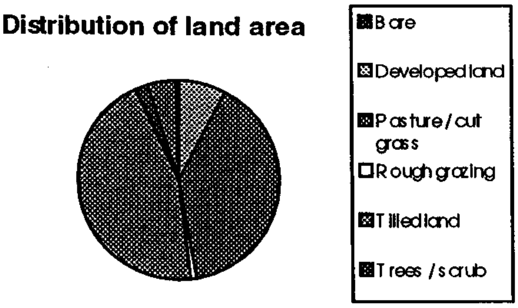
10932420 LEAM (SAWBRIDGE) at:- 450400 266300

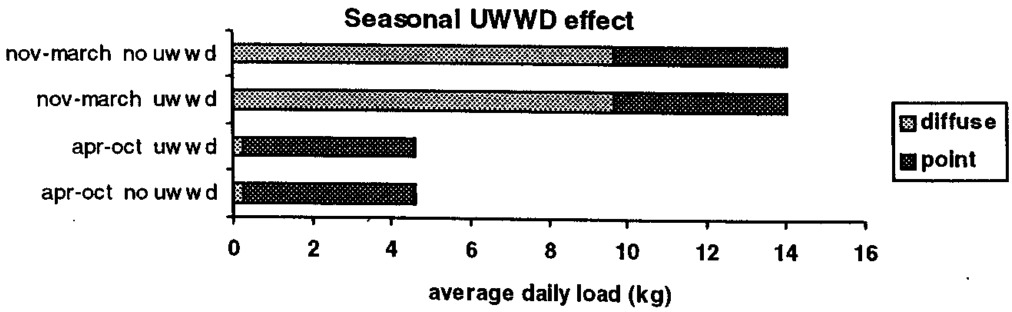




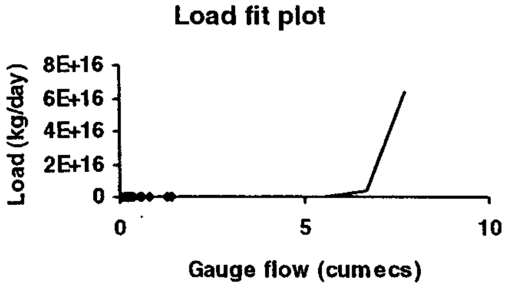
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	3251	38	6375
Total P entering system estimate	5443		
STWL best est current loads	1632		
Total diffuse estimate	3811		
export coef from land	2353		
export coef from stock	1458		
STWL qualifying works capped 2 mg/l	1632		
STWL qualifying works capped 1 mg/l	1632		
total measured load using y=ax^b+c	4166		673

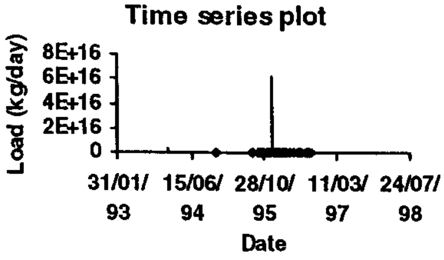




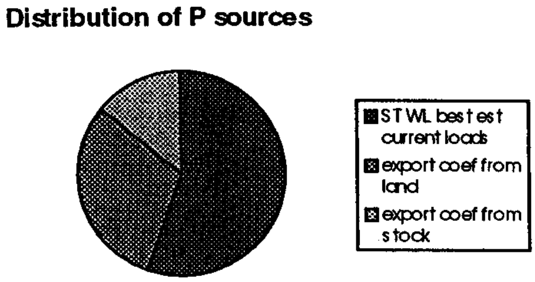


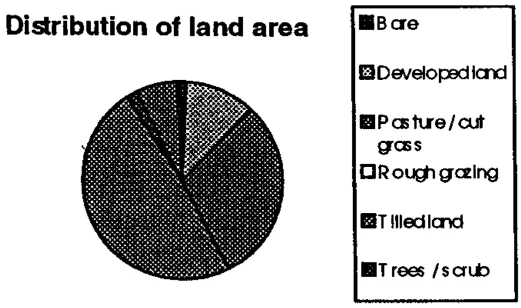
11053180 ITCHEN (MARTON, CONF. LEAM) at:- 440500 268800

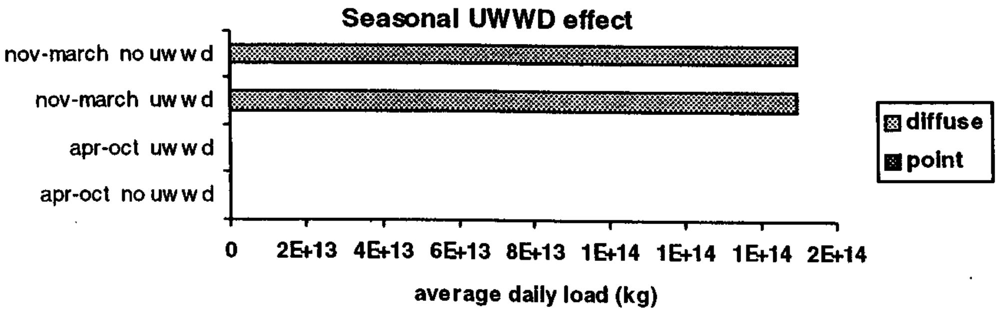




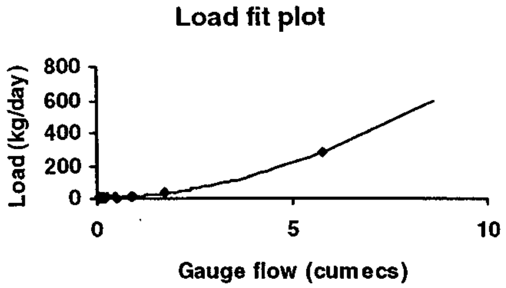
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	12460	38	12848
Total P entering system estimate	20449		
STWL best est current loads	11315		
Total diffuse estimate	9134		
export coef from land	6317		
export coef from stock	2817		
STWL qualifying works capped 2 mg/l	11315		
STWL qualifying works capped 1 mg/l	11315		
total measured load using y=ax^b+c	33043936724257000		2129

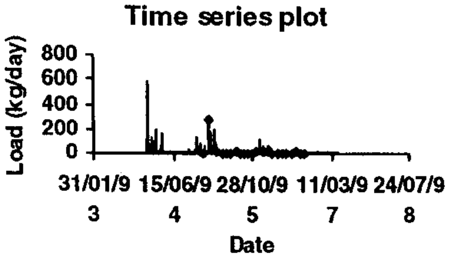




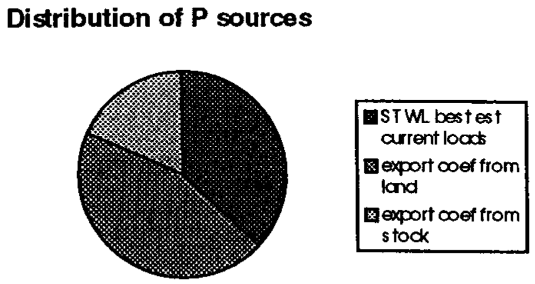


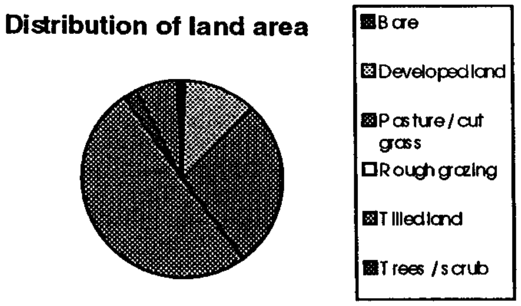
11054980 ITCHEN (A425) THORPE BRIDGE at:- 440300 261500

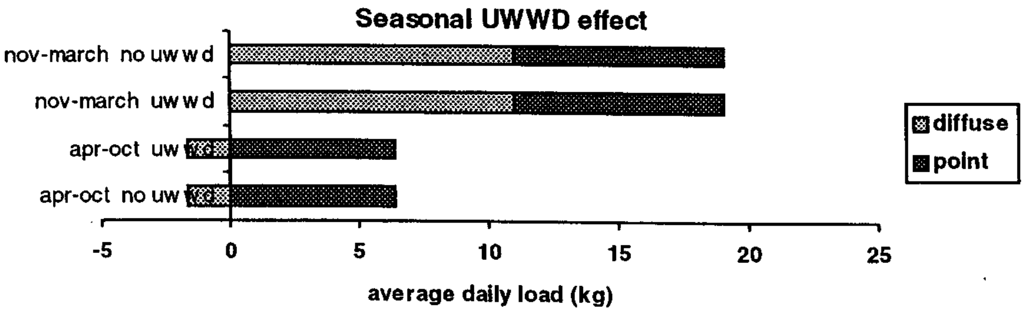




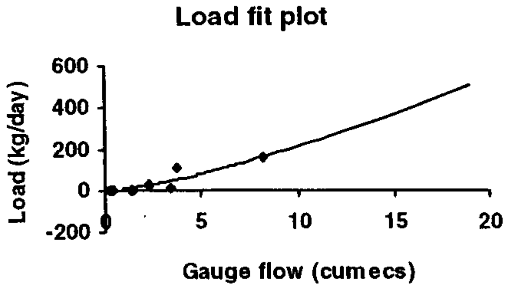
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	7137	23	20701
Total P entering system estimate	8130		
STWL best est current loads	3032		
Total diffuse estimate	5098		
export coef from land	3610		
export coef from stock	1488		
STWL qualifying works capped 2 mg/l	3032		
STWL qualifying works capped 1 mg/l	3032		
total measured load using y=ax^b+c	5722		1015

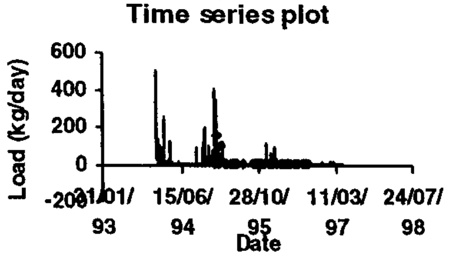




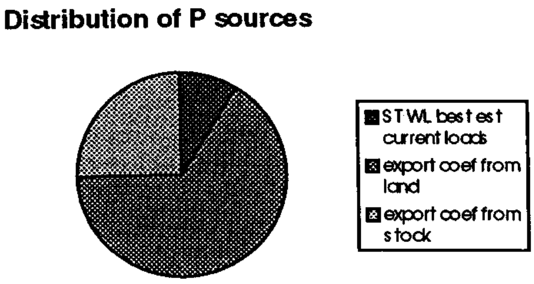


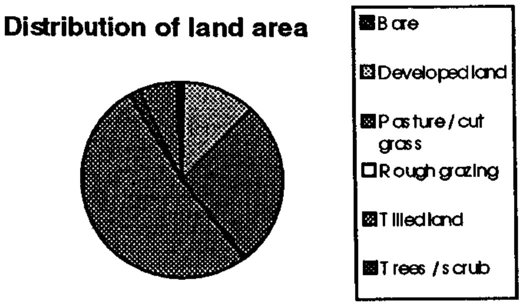
11055720 ITCHEN (DEPPER'S BRIDGE) at:- 440000 259300

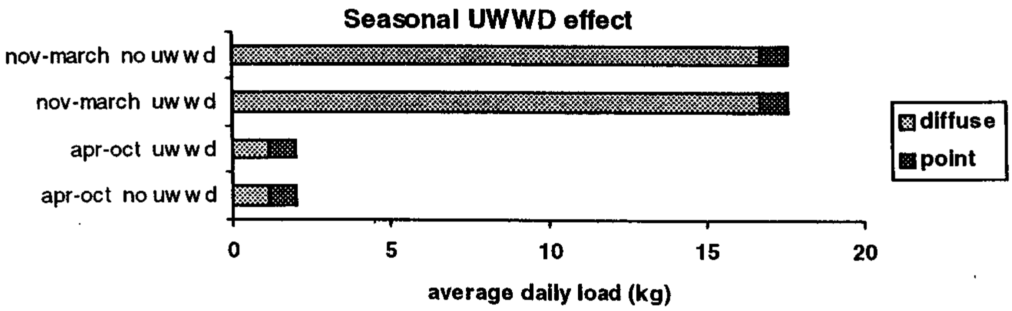




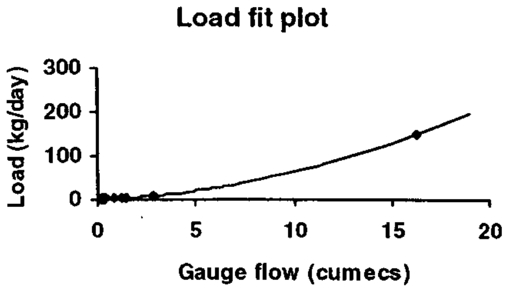
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	3542	38	10841
Total P entering system estimate	3898		
STWL best est current loads	346		
Total diffuse estimate	3552		
export coef from land	2561		
export coef from stock	990		
STWL qualifying works capped 2 mg/l	346		
STWL qualifying works capped 1 mg/l	346		
total measured load using y=ax^b+c	4960		3928

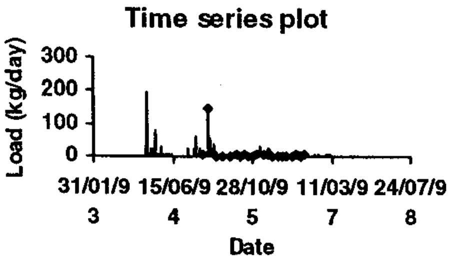




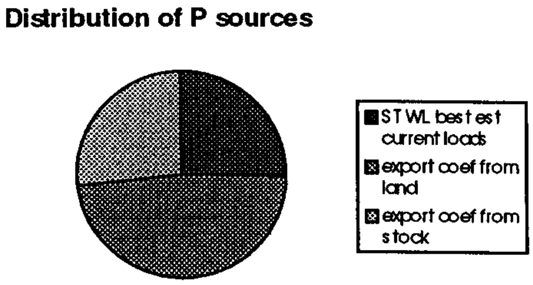


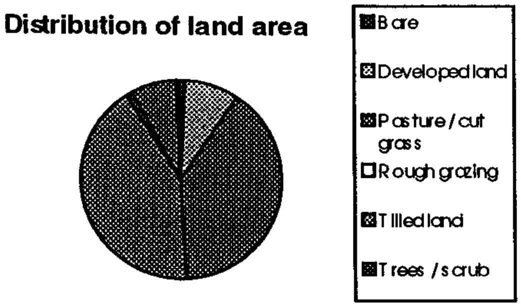
11091180 STOWE (SOUTHAM, BROWN'S BRIDGE) at:- 441800 261400

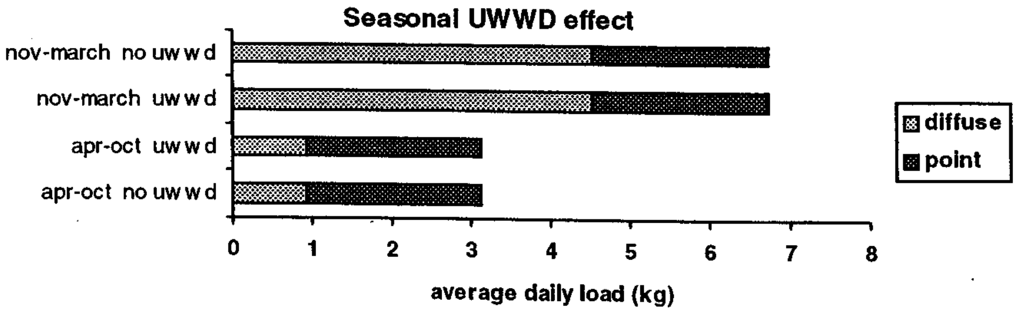




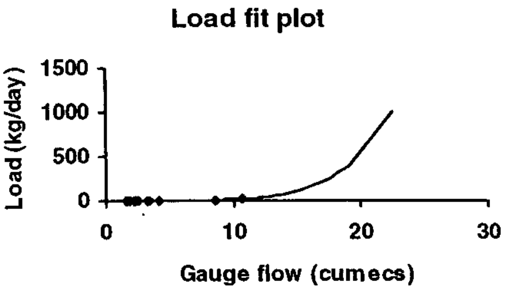
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	3668	22	11431
Total P entering system estimate	3242		
STWL best est current loads	824		
Total diffuse estimate	2418		
export coef from land	1545		
export coef from stock	873		
STWL qualifying works capped 2 mg/l	824		
STWL qualifying works capped 1 mg/l	824		
total measured load using y=ax^b+c	2119		477

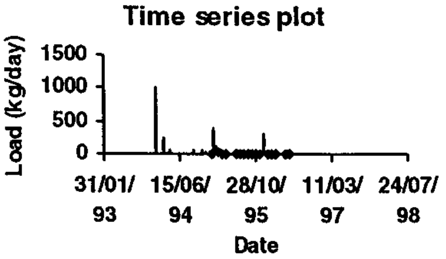




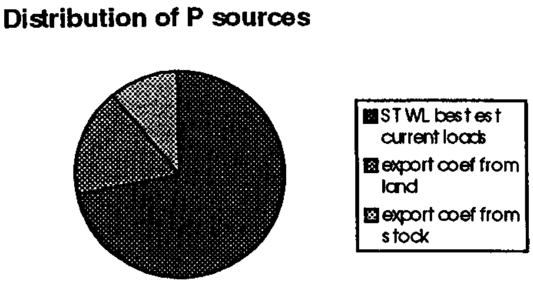


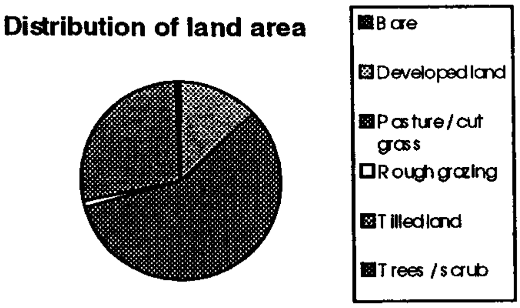
11296050 RAINS BROOK (BARBY LODGE) at:- 453200 272600

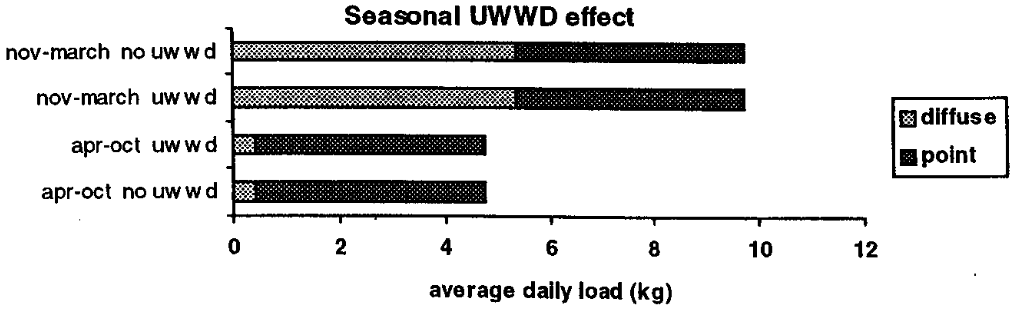




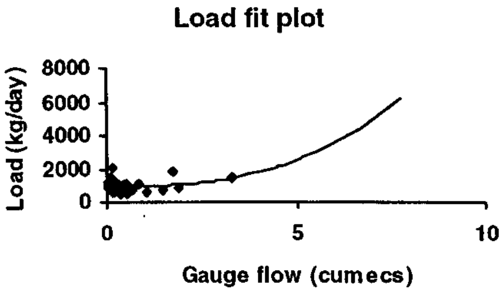
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	2128	16	1516
Total P entering system estimate	2267		
STWL best est current loads	1633		
Total diffuse estimate	634		
export coef from land	381		
export coef from stock	253		
STWL qualifying works capped 2 mg/l	1633		
STWL qualifying works capped 1 mg/l	1633		
total measured load using y=ax^b+c	3123		522

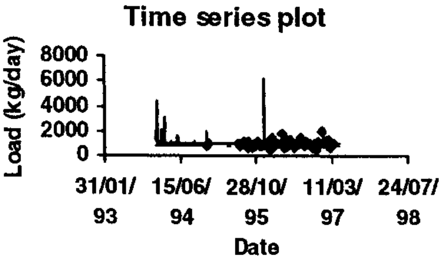




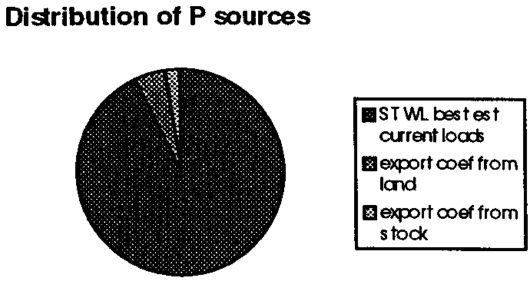


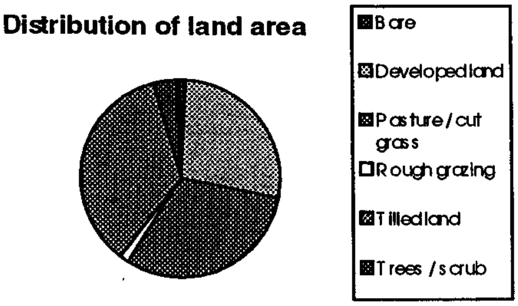
11559060 SOWE (STONELEIGH) at:- 433200 272800

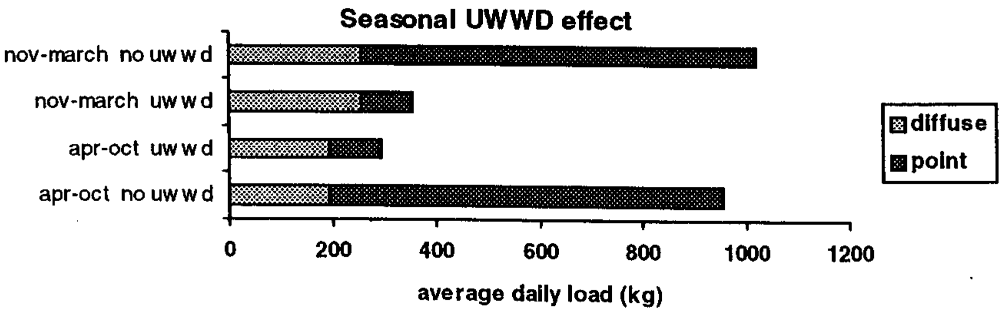




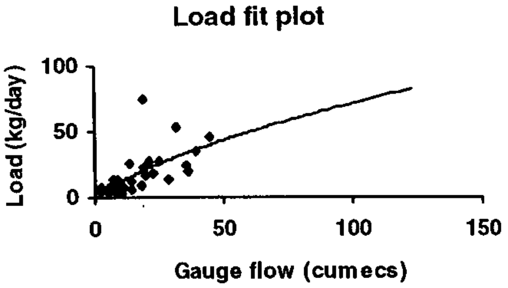
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	342634	107	81159
Total P entering system estimate	306797		
STWL best est current loads	285724		
Total diffuse estimate	21073		
export coef from land	14071		
export coef from stock	7003		
STWL qualifying works capped 2 mg/l	87395		
STWL qualifying works capped 1 mg/l	44445		
total measured load using y=ax^b+c	268264		62383

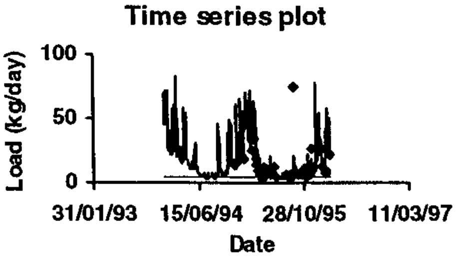




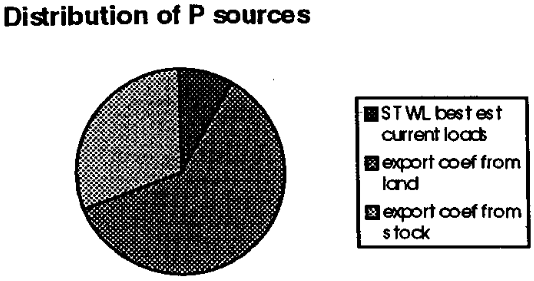


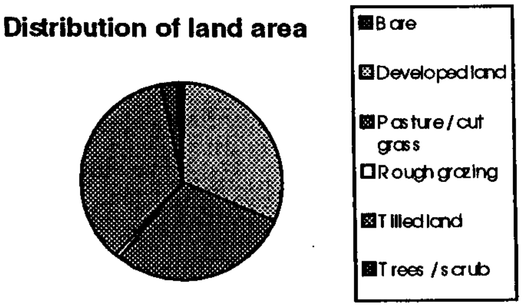
11559580 SOWE (BAGINTON) at:- 433800 275200

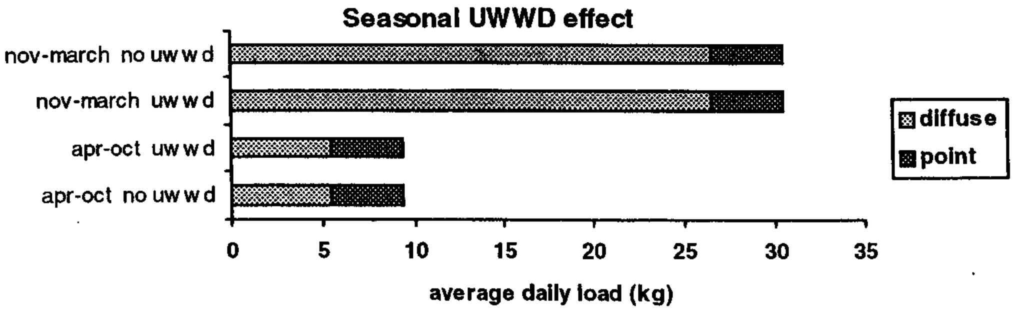




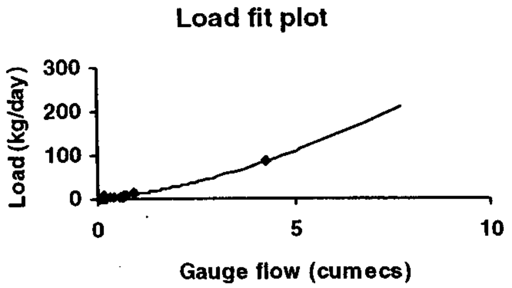
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	4447	94	4128
Total P entering system estimate	17214		
STWL best est current loads	1496		
Total diffuse estimate	15718		
export coef from land	10395		
export coef from stock	5323		
STWL qualifying works capped 2 mg/l	1496		
STWL qualifying works capped 1 mg/l	1496		
total measured load using y=ax^b+c	6746		3960

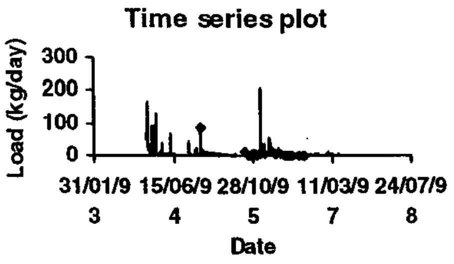




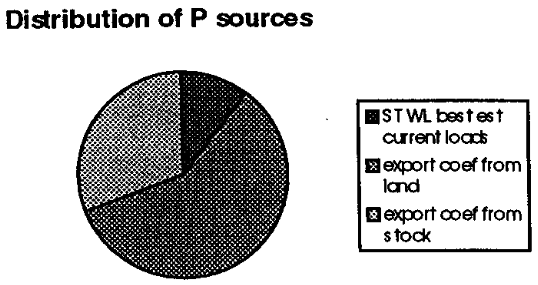


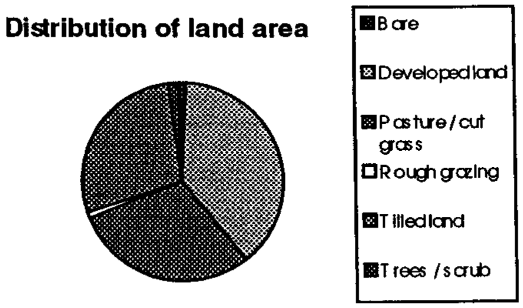
11561520 SOWE (WALSGRAVE) at:- 437800 280500

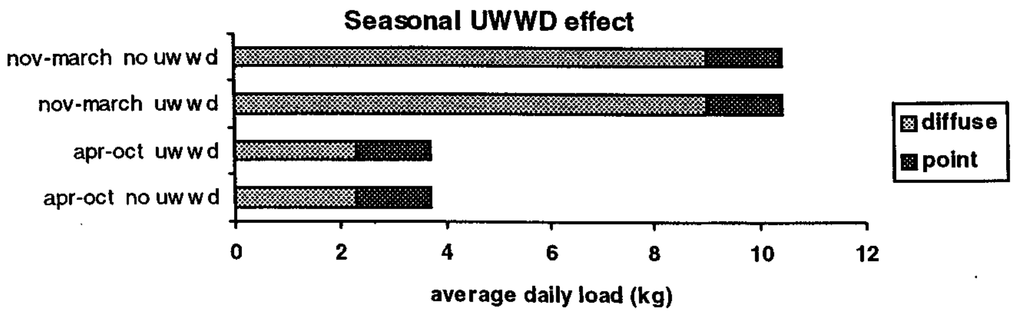




Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	2823	33	5405
Total P entering system estimate	5135		
STWL best est current loads	537		
Total diffuse estimate	4598		
export coef from land	3010		
export coef from stock	1588		
STWL qualifying works capped 2 mg/l	537		
STWL qualifying works capped 1 mg/l	537		
total measured load using y=ax^b+c	2297		547

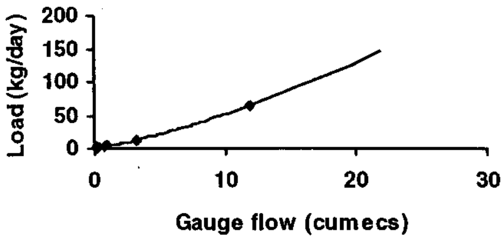




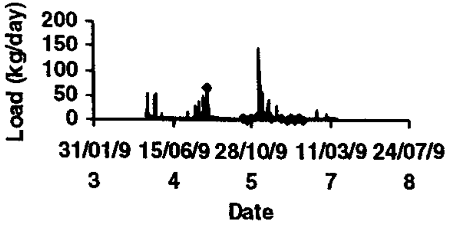


11827820 COOMBE POOL (OUTFLOW) at:- 438430 279310

Load fit plot

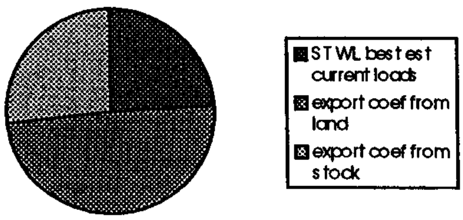


Time series plot

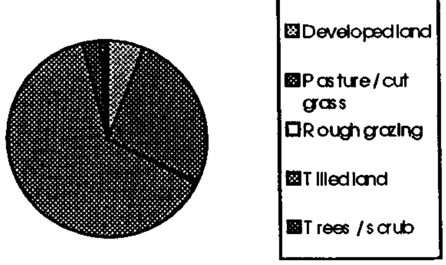


Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	3154	15	5929
Total P entering system estimate	3978		
STWL best est current loads	959		
Total diffuse estimate	3019		
export coef from land	1949		
export coef from stock	1070		
STWL qualifying works capped 2 mg/l	959		
STWL qualifying works capped 1 mg/l	959		
total measured load using y=ax^b+c	1690		474

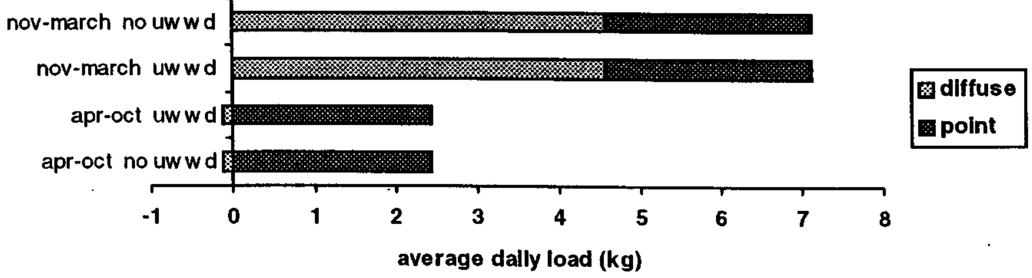
Distribution of P sources



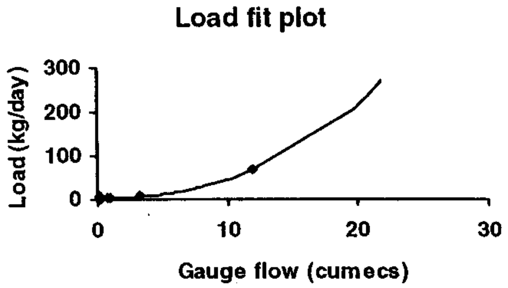
Distribution of land area

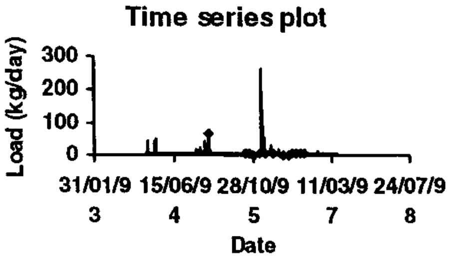


Seasonal UWWD effect

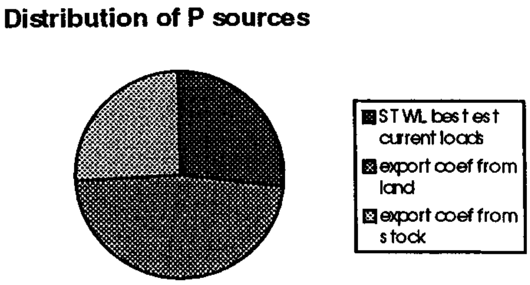


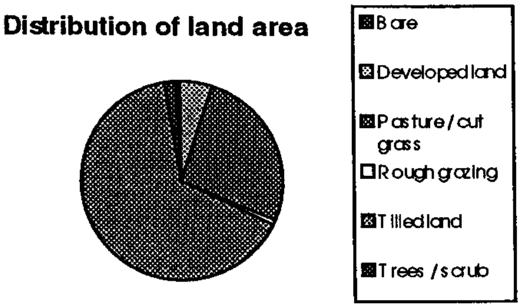
11828390 SMITE BROOK (COOMBE ABBEY) at:- 440800 280500

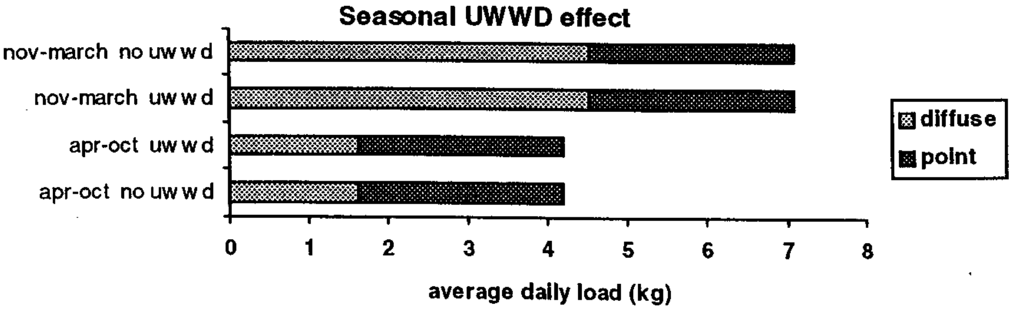




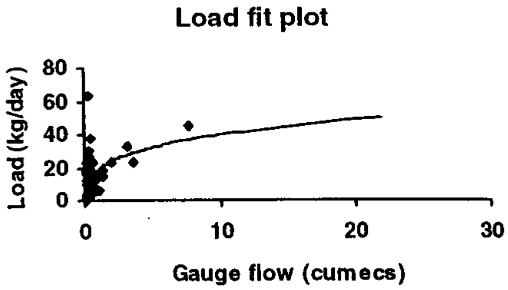
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	6450	16	14414
Total P entering system estimate	3549		
STWL best est current loads	959		
Total diffuse estimate	2590		
export coef from land	1677		
export coef from stock	913		
STWL qualifying works capped 2 mg/l	959		
STWL qualifying works capped 1 mg/l	959		
total measured load using y=ax^b+c	1961		550

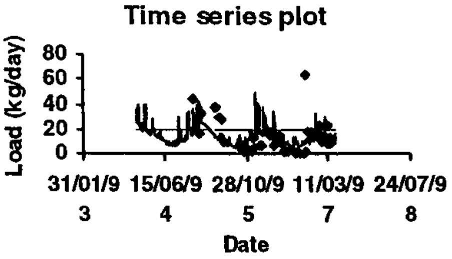




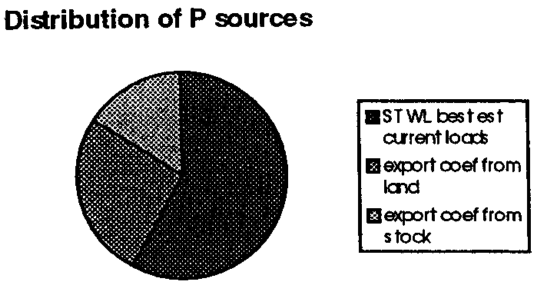


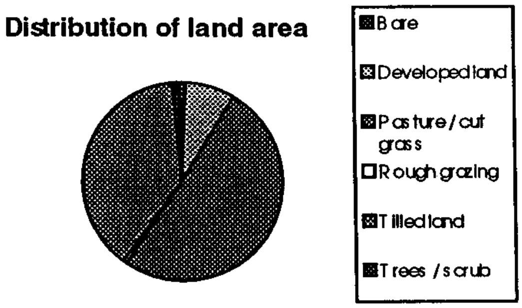
12178180 SWIFT (BROWNSOVER HALL) at:- 450500 277500

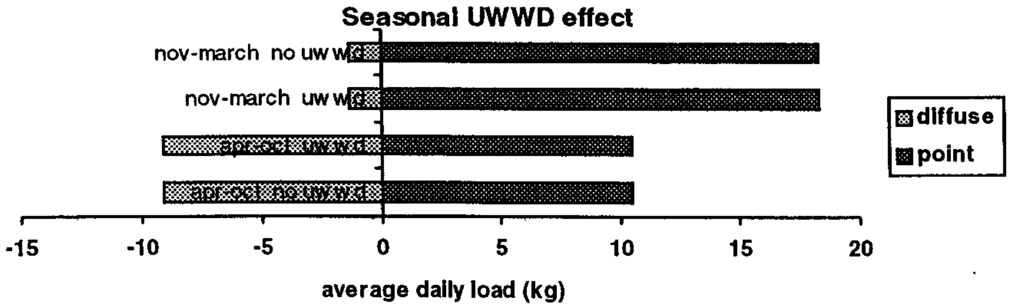




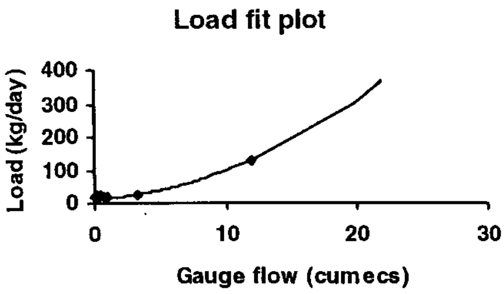
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	5075	57	4590
Total P entering system estimate	12646		
STWL best est current loads	7355		
Total diffuse estimate	5291		
export coef from land	3243		
export coef from stock	2048		
STWL qualifying works capped 2 mg/l	7355		
STWL qualifying works capped 1 mg/l	7355		
total measured load using y=ax^b+c	4749		3534

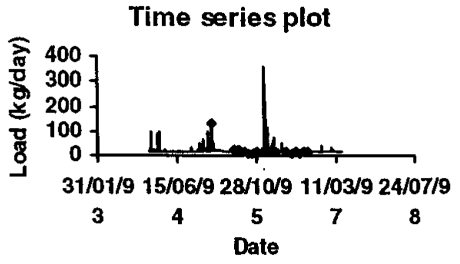




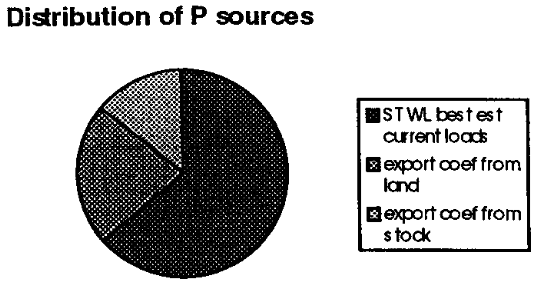


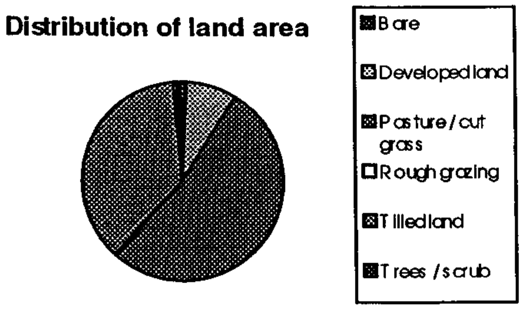
12180480 SWIFT (BRANSFORD BRIDGE) at:- 451900 282200

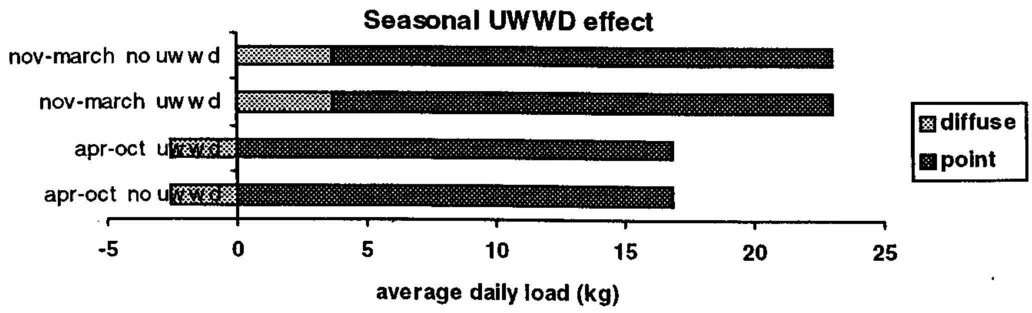




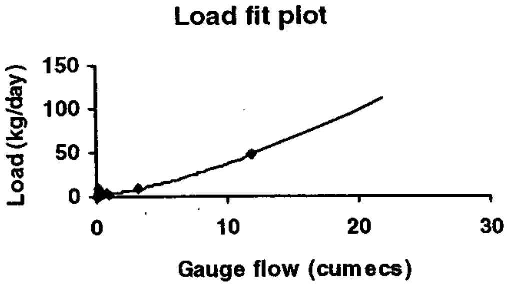
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	12861	19	19848
Total P entering system estimate	11428		
STWL best est current loads	7264		
Total diffuse estimate	4164		
export coef from land	2545		
export coef from stock	1620		
STWL qualifying works capped 2 mg/l	7264		
STWL qualifying works capped 1 mg/l	7264		
total measured load using y=ax^b+c	6692		1517

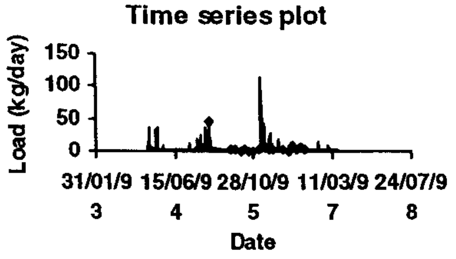




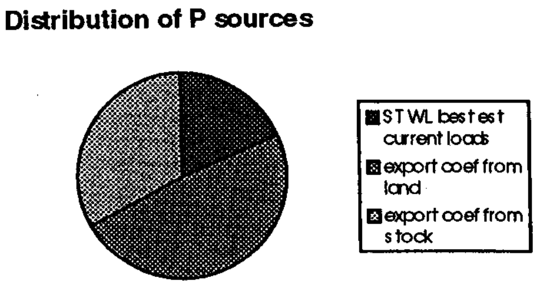


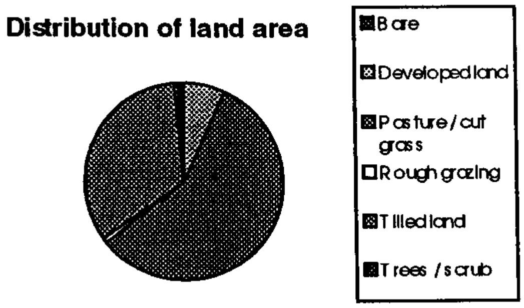
12182180 SWIFT (LUTTERWORTH) at:- 454800 284100

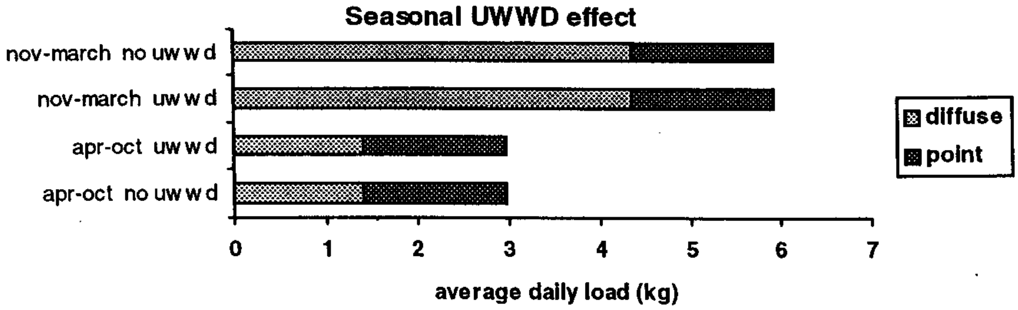




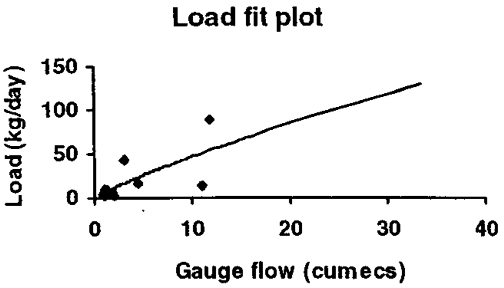
Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	2044	19	3736
Total P entering system estimate	3179		
STWL best est current loads	592		
Total diffuse estimate	2587		
export coef from land	1538		
export coef from stock	1050		
STWL qualifying works capped 2 mg/l	592		
STWL qualifying works capped 1 mg/l	592		
total measured load using y=ax^b+c	1540		726

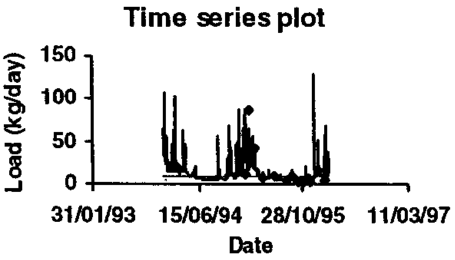




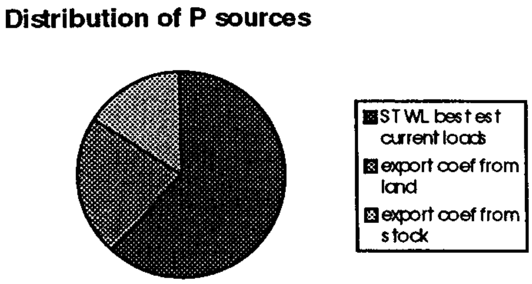


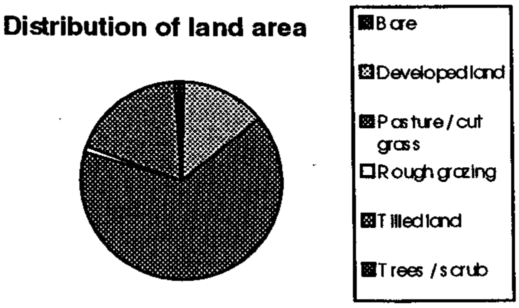
12385020 CLIFTON BROOK (RUGBY) CONF. AVON at:- 451500 276400

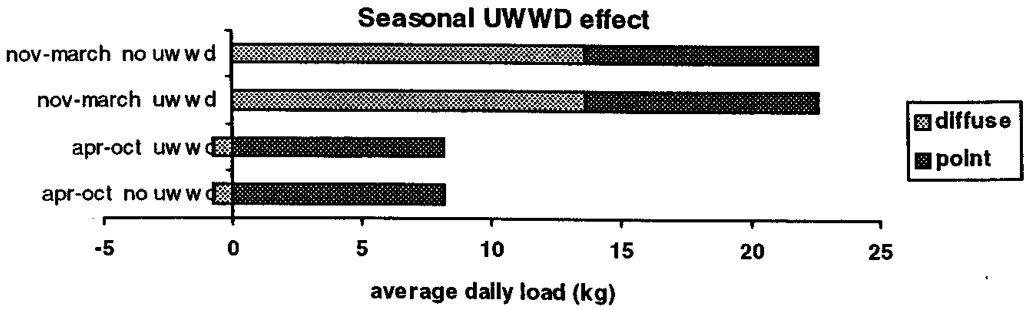




Estimate type	Value / Kg per annum	N samples	standard deviation
average instantaneous load	4422	19	7502
Total P entering system estimate	5446		
STWL best est current loads	3371		
Total diffuse estimate	2075		
export coef from land	1198		
export coef from stock	877		
STWL qualifying works capped 2 mg/l	3371		
STWL qualifying works capped 1 mg/l	3371		
total measured load using y=ax^b+c	5304		7630







APPENDIX 3

PRACTICAL METHODS OF REDUCING DIFFUSE RUNOFF OF PHOSPHORUS FROM AGRICULTURAL LAND

Descriptions of many of these methods have been obtained from Clouston (1996), Harper and Pacini (1995 a, b) and Mainstone *et al* (1996). A compilation of all the reported methods are given below. They are considered by a number of experts to be realistic approaches to reducing diffuse agricultural P pollution in the UK.

A Management Practices to Reduce the Input of Nutrients to the Field

A1 Reduction of fertiliser input whilst maintaining fertility

(McIntosh, 1994b; Tenor, 1994; Harper and Pacini, 1995b; Mainstone *et al* 1996, Higgs *et al* 1998)

For a large proportion of farms in the UK the import of phosphorus in the form of inorganic fertiliser, animal feed and imported manure greatly exceeds the export of P in the goods produced. As a result soil P concentrations are increasing. Historically this surplus was required to build up soil P reserves and improve fertility. However, in a few situations (a few percent of the total land area, T Harrod, Soil Survey. pers. comm.) soils are now saturated with P to the extent that a significant proportion of additional fertiliser applied to the field runs off with the next intense and prolonged storm event. The assumption is that the area of land which is P saturated will increase with time if present practices continue. The problem is exacerbated when farmers continue to make "maintenance dressings" to keep up soil fertility. An additional factor on livestock units is the disposal of animal waste to land without allowing for the nutrient content of the slurry before adding the recommended amount of inorganic fertiliser. All nutrient sources should be considered and the best management practice is to analyse the P content of the soil, assess the requirements of the crop and then add the minimum P fertiliser (organic or inorganic) to bring the field levels up to those required by the crop, taking into account the amount of P applied in livestock waste. Liming of acid soils can release some of the existing P reserve to reduce the requirement for additional P.

A2 Location of the fertiliser closer to the plant root

(Mainstone *et al*, 1996)

Use of very expensive machinery to localise fertiliser delivery close to the plant root.

A3 Timing of fertiliser applications – "incidental losses"

(Managing Maize; Mainstone *et al*, 1996)

This simply requires vigilance on the part of the farmer so that fertiliser, slurry or manure are not applied to the soil immediately before heavy rainfall is expected. Under the latter circumstances a significant proportion of the application can be lost since there is inadequate opportunity for plant uptake. Similarly, applications should be timed to supply nutrients at the time the crop needs them rather than application at a convenient time which is well before the crop need and increases the potential for nutrient loss from the soil.

A4 Nutrient management planning

(Clouston, 1996, Beegle *et al*, 1998)

When the above and other management practices are considered together they constitute a Nutrient Management Plan. This aims to ensure an accurate balance between the input of nutrients to a farm and the outputs. The farmer is aiming for the economically optimum level of P reduction, ie maximum crop return for minimum fertiliser input. However, from the publics' perspective there is also an underlying aim to prevent detrimental environmental impacts so that a holistic cost benefit analysis should also include the benefits of preventing nutrient loss through run-off or leaching. This latter aspect will probably not appear in the farmers cost benefit analysis. In order to develop a management plan, detailed technical assessments are required of the nutrient content of the soil, inorganic fertilisers and organic manures, animal crop nutrient demand and timing/application rates to be successful. A nutrient management plan will also consider the benefits of liming soil to release more of the residual P and will consider the need for continued maintenance dressing when organic manures are being used.

Nutrient management is reported as being a highly cost-effective management practice, since long term use of nutrient management plans has demonstrated economic benefits/higher profitability to farmers. It is also relatively inexpensive to implement compared to anything requiring capital investment. The efficiency is highly dependent on the amount of "over-fertilisation" prior to the implementation of a nutrient management plan and is estimated to be equivalent to the percentage reduction in fertiliser use recommended by the plan (Casman, 1990). As such, the efficiency is highly site specific. Dils *et al* (1998), in a study of the Loch Erne system in Northern Ireland, showed that savings from £2 to £22 per hectare were possible on farms collaborating in a nutrient management advice scheme. Tippett and Dodd (1995) using data from Camacho (1990, 1992) proposed a cost effectiveness of 10-45 $US/ kg P removed.

A5 Reduction in the dietary intake of P by livestock

(McIntosh, 1994b; Tenor, 1994; Valk et al., 1998)

Mass balance calculations at the farm scale show that much more P is brought onto a farm than goes out in the product. Although this is particularly important with regard to inorganic fertilisers it is also relevant to the feeding of livestock. It has been suggested that it should be possible to reduce the P content of livestock pellets by changing the formulation without changing the nutritional content significantly.

B Management Practices to Reduce the Run-off of Dissolved Nutrients from the Field

B1 Riparian buffer zones - also called buffer strips, riparian filters or filter strips.

(Understanding buffer strips; Managing Maize; Muscutt *et al*, 1993; Harper & Pacini 1995a; Clouston, 1996)

Strips of normally natural or naturalised vegetation 1-50 m wide situated alongside watercourses, particularly low-order streams. They slow down both above ground and through flow drainage water. Physical sedimentation occurs as overland flow is slowed; vertical infiltration of soluble nutrients is encouraged. Creating anaerobic zones encourages denitrification, whereas phosphorus removal is more effective in aerobic zones.

There are several potential problems with the use of buffer strips as a management option. They are not effective in areas containing sub surface land drainage systems (field drainage tiles) or where ground water recharge is a major water transport process. It has been suggested that as time progresses the soils in the buffer strip may become saturated with P and no longer retain it effectively but, at the moment, this appears to be a prediction based on the behaviour of some soils rather than an observed phenomenon. There may be problems of P removal if anaerobic zones are allowed to develop. Care must be taken to ensure that the system does not break down in high runoff events washing out all the stored sediment from earlier, lower flow events.

A minimum of 5 m width is recommended but the size depends on the vegetation, slope and soil type. De-nitrification is most effective in buffer strips containing stands of young stages of woodland succession because of the higher stem density. However, woodland strips are not very effective for P removal (presumably because of the anaerobic conditions) and grass strips are needed. The latter have a high retention, except in sandy soils. A combined N and P reduction system can be used where a grass strip is placed in between the woodland and the stream. Coppicing and grass cutting (or grazing) produce the most efficient buffer strips. The P reduction depends on the binding capacity of the soil, which can be estimated from the Fe: total P ratio. Ratios below 8 (g/g) indicate a poor retention capacity (Harper and Pacini, 1995a). However, buffer zones can achieve reductions up to 40 kg P/ha/year. Maximum potential reductions have been estimated at 60% for NO_3^- and 80% for P. Elsewhere efficiencies ranging from 30-100% for nutrients have been claimed.

As well as reducing P, N and suspended solids runoff, buffer strips restore semi-natural habitats with consequent beneficial effects on biodiversity. However, they reduce field size and can limit the use of large machinery. In some circumstances up to 30% of the field area can be required to remove P in the runoff. The main costs are the cost of purchasing the land and the loss of production each year. Operational costs will be incurred annually to maintain the strip by coppicing and grazing/cutting grass. However, grants can be obtained from the set-aside and other schemes.

B2 Wetland restoration

(Harper and Pacini, 1995a; Clouston, 1996)

This best management practice requires restoration of areas of land, which historically would have remained predominantly wet throughout the year. Such areas are recognised as facilitating both sedimentation and the 'fixing' of potential pollutants/contaminants through physical or biological processes. Horseshoe shaped wetlands at the stream outlet of sub-surface drains produce lower, more uniform nutrient outputs. An approximate ratio of wetland to catchment area of 0.1-1% is required. They can be created by introducing impermeable, adjustable sluices into natural/semi-natural banks and are apparently suitable for use in lowland sites, up to 4th order streams. Most design requirements are given for nitrogen reduction, ie anaerobic conditions. However, P retention is best achieved in aerobic conditions. The P retention and N removal can be separated in either time or space. The combination of a deep stratified basin favours denitrification followed by a shallow basin for active P sorption. However, if oxic/anoxic conditions can be controlled sufficiently well then oxic conditions can be provided in the summer and anaerobic ones in the winter when P will flush out down the system. This practice is not particularly helpful upstream of lakes or estuaries with long retention times. Design requirements are complex and require a significant amount of active management, including water level management. In general success rates for P retention appear to be low. Wetlands have a high habitat value but can form the site of biting fly populations and water quality is variable. They often have high operating costs.

B3 Artificial reed beds

(Harper and Pacini, 1995a)

Phosphorus removal by reed beds is mainly by physical sorption on the soil and roots. This is encouraged by the fact that reeds create an aerobic microclimate around their roots during photosynthesis. As a result anaerobic conditions can occur in the winter (non-growing season) releasing P to the water. This is not a problem unless there is a lake downstream or a long retention time estuary. Two systems in parallel are usually constructed so that one can be maintained while the other one is still in operation. A gravel base of approx. 60 cm thickness is required to encourage drainage. This is laid on an impermeable butyl liner and is covered by a sand or soil layer. An innoculum of plant risomes (eg phragmites) is planted in the system. The required area is reported by Harper and Pacini (1995) to be 2-6 m^2 (Harper and Pacini do not give consistent figures for areal loadings, 2-6 m^2 is the most consistent area.) loaded at about 170 l/day with a phosphorus load of 1.8 g day (ie a feed concentration of approximately 10 mg/l of P).

Reed beds have a low capital cost (unless a significant amount of earth moving is required) and generally low annual maintenance costs. The efficiency can fall off after a few years requiring re-packing of the filter bed. It may be possible to control this using anaerobic conditions in the winter to release sorbed P. Several water companies have used these systems for tertiary treatment.

B4 Oxidation pond

(Harper and Pacini, 1995a, b)

Like wetlands this approach can be used at the downstream end of land drainage collection channels. This is a simple pond with plug flow characteristics. Loadings are approximately 3-10 $m^3/m^2/y^{-1}$ to produce a retention time of 40-200 days. The pond must be sealed to prevent leakage and should be separated into short sections so that the wind cannot resuspend sediments. Order of magnitude reductions in P concentration can be achieved. Construction costs are low, apart from the cost of the land. Disadvantages include risk of insect nuisance and occasional high-suspended solids in the outflow due to algal blooms. Periodic removal of sediments is necessary. They are untried in the UK situation.

B5 Critical area planting

(Clouston, 1996; Mainstone, *et al*, 1996; MAFF, 1997)

This best management practice targets the best crop to those areas which are most sensitive to erosion and/or leaching. It can relate to standard arable crops, such as not growing maize on sloping land, but is more commonly used in relation to growing of grass either as long term ley or permanent pasture in areas adjacent to streams and rivers. Alternatively, bankside vegetation such as shrubs and trees can be provided to increase shade. It can also equate to excluding livestock from areas within fields where the soil disturbance is severe. The major benefits are the targeting of areas subject to high erodability. The effects of introducing livestock to sensitive locations should be considered carefully. It can be combined with permanent pasture, short/long term ley, livestock exclusion, feed/water trough location, livestock trails and field boundary/access points. This can also include integrated crop management, which matches the crop with the soil type and conditions. The areas most prone to erosion are converted to permanent grassland, woodland or set aside.

Tippett and Dodds (1995) reported costs in the US for conversion of arable land to grass ($50/ha/year) and trees (£35/ha/year). However these costs do not include the opportunity costs of the value of the lost crops and are probably not applicable to the UK situation where set aside and other grants could well change the costs dramatically. They also state that effectiveness is highly site dependent but is attributed to both stabilisation of the soil reducing particle losses and reductions in dissolved run-off resulting from the cessation of fertilisation. Effectiveness is suggested to range from 0 –90% reductions in P loss.

C. Management Practices to Reduce the Erosion of Particulate Nutrients from the Field

Reductions in slope gradient/length

C1. Strip cropping

(Managing Maize; Harper & Pacini, 1995b; Clouston, 1996)

This best management practice requires the growing of crops in bands or strips with a close growing crop such as grass, alternated with a strip of cropped land. The close growing crops reduce erosion by filtering sediment and also pollutants, which may be in suspension through run-off. Strip cropping can be combined with contour cropping/contour cultivation with the strips following the contour. The close growing crop should be of equal width to the arable crop. Tippett and Dodd (1995) suggested that their effectiveness was similar to vegetated field strips but reported no efficiency data for either field borders or strip-cropping. (See also C20, vegetated field strips.)

C2. Contour cropping

(Clouston, 1996; MAFF, 1997)

Contour cropping involves extending seed lines or sowing crops across the slope in line with contours. This reduces both erosion and run-off by removing natural drainage channels down the slope as with conventional cropping. The crops provide a perpendicular barrier to run-off due to stem density. Reducing run-off improves infiltration and reduces soil erosion. The technique is considered most cost effective where fields are wide along the contour and perpendicular to field boundaries. Economic savings are achieved through reduction in fuel consumption during cultivation. It can be combined with contour cultivations and strip cropping.

C3. Hedgerow Planting

(Harper & Pacini, 1995b; Clouston, 1996; MAFF, 1997)

This best management practice is attempting to achieve two objectives. The first objective is to intercept over land flow. A second objective is to reduce the concentration of animals or machinery operations in areas likely to suffer from erosion or run-off. It could, therefore, be combined with a field boundary/access point or livestock exclusion. This best management practice is incorporated within techniques provided through the Countryside Stewardship Scheme.

C4. Terracing

(Clouston, 1996)

Contrary to the normal UK usage, this best management practice does not reduce the length of a slope by creating a series of steps down the contour. In this context it means a series of regularly spaced embankments across a slope that form a channel on the up-slope side of the embankment to direct surface water or run-off, and also slow down run-off improving infiltration and sedimentation. “Terracing” is normally accompanied by contour planting and acts as a check on any contour row failure (Tippett and Dodd, 1995). Grassing the waterways (see C19 below) may increase the removal of pollutants. Terracing requires extensive earthworks and therefore significant potential changes to existing land-use practices. It may well result in significant loss of useable area and would, therefore, tend to be focused on areas where erosion is severe. Reductions of P in surface run-off of 66% have been reported (Langdale *et al*, 1985). However, if large storms cause overtopping of embankments the terraces can fail causing serious downslope erosion. Based on Langdale’s estimates of efficiency, Tippett and Dodd (1995) estimated a cost effectiveness of 50–130 $ US per kg P reduced but it is not clear how applicable to UK conditions this estimate is.

C5. Contour Cultivations

(Harper & Pacini, 1995b; Clouston, 1996; MAFF, 1997)

By focusing cultivations along contours any surface run-off and erosion will be directed across the slope rather than down existing seedbeds/tramlines. As a consequence the speed of run-off will be reduced improving infiltration and sedimentation. The success of this best management practice is dependent on very careful attention to field slope, shape and soil types. It can be combined with contour cropping. Operational cost savings can be made through reduced fuel usage.

Continuous ground cover

C6. Triple cropping

(Clouston, 1996)

This best management practice aims to maintain a crop cover throughout the year, particularly during likely heavy rainfall. This is achieved by growing three continuous crops through the year. Typical rotations seen in the USA include maize followed by a fodder crop of wheat/oats followed by soya bean. This continuous cover reduces potential erosion and run-off. The best management practice, however, would require significant changes to normal arable rotations within the UK. It could be more appropriate to intensive vegetable and arable growing areas.

C7. Green manure cropping

(Harper & Pacini, 1995b; Clouston, 1996)

This best management practice requires the growing of nitrate rich crops such as legumes, which are subsequently ploughed into the soil. The green manure crop provides cover during months when soil could be sensitive to erosion or run-off. The green manure crop provides the additional benefit of soil cover with provision of nitrogen for the succeeding crop, thereby reducing requirements for application of artificial nitrogen, and improves water retention capacity and soil structure. It can be combined with triple cropping, cover crops and conservation tillage. This best management practice is already included within recommended management techniques for Nitrate Sensitive Areas.

C8. Grassland rotation

(Harper and Pacini, 1995b; Clouston, 1996; MAFF, 1997).

Integrating grassland into crop rotations either on a permanent basis or short-term leys could strengthen soil structure and, depending on location, could provide filtration of pollutants and sediment. On arable farms short-term leys would be most appropriate to provide cover cropping on land sensitive to erosion. Permanent pasture would be used in very sensitive locations such as steep valley sites. If the permanent pasture is to be grazed, attention should be made to livestock best management practices. It could be combined with strip cropping, critical area planting, water diversions, grass watering, access tracks and livestock trails.

C9. Crop residue management

(Clouston, 1996)

This best management practice aims to reduce erosion by wind and water. This is achieved by leaving the crop residue on the soil surface. UK examples of crop residue would include chopped straw following harvesting of cereals. It does not include leaving crop stalks and roots in-situ, as this alone would not provide sufficient soil cover. This latter technique is covered by the conservation tillage best management practice. Managing crop residue forms a critical component of conservation tillage methods of cropping for the next sowing. It also provides additional sources of organic material, which provides similar benefits to green crops. Careful attention as to when residue crops can be incorporated is required, especially those with high nitrogen residues such as peas, and also the degree of cover.

C10. Cover crops

(Managing Maize; Clouston, 1996; MAFF, 1997)

This best management practice reduces erosion by maintaining soil cover during periods of high rainfall on soils subject to erosion and in particular on sloping land. The crop can also provide benefits when ploughed into the soil by causing temporary tie-up of plant nutrients especially nitrogen. A cover crop can be natural regeneration but is more typically a crop, which is sown together with a main crop. The cover is grown either forward of the main production cycle or provides ground cover following harvesting. It can be combined with a nutrient management plan, triple cropping, green manure cropping and strip cropping. It also includes techniques such as undersowing a cover crop or sowing autumn crops early to encourage ground cover over winter.

C11. Consideration of the time of planting

(Harper and Pacini, 1995b; MAFF, 1997)

The time of sowing can greatly influence the risk of erosion. For example, maize should be sown in the spring, whereas autumn crops should be sown early to allow the crop to establish before the winter.

Low disturbance tillage methods

C13. Conservation tillage

(Managing Maize; Harper and Pacini, 1995b; Clouston, 1996, MAFF, 1997)

In theory conservation tillage is defined as any tillage or planting system that leaves at least 30% of the soil surface covered with crop residue after planting and includes practices such as no-till and reduced tillage practices, including ridge-till, strip till, coulter, chisel and mulch till (Casman and Pacheco, 1989). However, in the UK the term is also used synonymously with non-inversion tillage, which aims to minimise soil disturbance by minimising soil preparation. Crops can be either directly sown into the uncultivated soil or following chisel ploughing or sown in bands of seedbed up to 4 inches in width. Minimising cultivation reduces erosion by maintaining a stable soil structure and reduces the potential for run-off along drainage channels created through normal cultivation such as ploughing. It can be combined with triple cropping, green manure cropping, crop residue management, contour cropping and cultivation. It generally requires specialist and expensive capital investment in machinery. There are different degrees of conservation tillage from no tillage through to minimum tillage or merely delaying tillage until immediately prior to seedbed preparation. The use of a chisel plough is predicted to reduce sediment load by 22% (Baumgart and McIntosh, 1994). Other approaches include the avoidance of fine seedbeds and not using rollers on seedbeds; creating tramlines only after emergence of the crop and, if this is not possible, removing wheelings by the use of a shallow tine behind the wheel.

Costs presented by Tippets and Dodd (1995) are specific to the US and do not include capital or other set-up costs. Efficiencies are very variable depending on the site and its previous agricultural history. Tippetts and Dodd quote reductions ranging from 0.3 to 1.0 kg P/hectare /year.

C14. Compaction management

This best management practice would, through cultivation techniques such as deep ploughing or provision of uncropped areas along the headland, reduce the effects of compaction. Compacted areas are more subject to the effects of run-off and possibly erosion through lower infiltration. It could be combined with provision of access tracks/livestock trails, field boundary/ access points and critical area planting.

C15. Soil spreading

(Clouston, 1996, MAFF, 1997)

This best management practice highlights to farmers the potential effects of spreading soil on areas prone to erosion or run-off. It is particularly relevant to the practice of clearing ditches and depositing on the riparian edge. It can therefore be combined with ditch management.

Maintenance of good soil structure

C16. Use bulky organic manures and/or incorporate straw residues

(MAFF, 1997)

Interception techniques

C17. Water/sediment basin/retention

(Clouston, 1996; MAFF, 1997)

This best management practice is located at the low point of a drainage slope or ditch or surface run-off area such as a farmyard. It aims to retain water flow encouraging infiltration and sedimentation. There are many different types of construction from engineered solutions to bunded areas following the natural landform. It can be combined with other best management practices which aim to direct run-off and erosion away from surface or groundwater such as water diversions, ditch management, grass waterways, grass hedges, roof and farmyard run-off intersection and porous pavements. It should, however, be regarded as a 'last resort' best management practice in that it cleans up from other poor management techniques. It, therefore, should only be employed when nothing else can be achieved.

C18. Water diversions

(Harper and Pacini, 1995a,b; MAFF, 1997)

This best management practice aims to reduce the speed of run-off and therefore erosion by intercepting water flow across the slope and diverting it to existing water structures or proposed structures such as retention/water basins. It is generally applied to soils subject to severe erosion due to soil type, slope or climate in order to reduce rill and gully formation. Tippett and Dodds (1995) could find no efficiency data. They suggested that reductions in dissolved nutrient loads/concentrations would be effectively zero and that any reductions in total P losses would be due to unquantified reductions in particulate P. Costs were reported as $81/ha/year but the transferability of this figure to UK conditions is unknown.

C19. Grass waterways

(Clouston, 1996).

This practice achieves two functions. Firstly it directs run-off to a sediment basin/wetland restoration area to allow ecological processes to remove pollutants. Secondly, unlike diversions or open ditches, it has been suggested that the vegetation within the channel may act as a filter in removing some of the sediment. It could, therefore, be combined with water diversions and possibly buffer strips.

However, Tippett and Dodds (1995) report data from Casman (1990) and Casman and Pacheco (1989) which show that under conditions of laminar flow, which are prevalent in grassed waterways, (dissolved) nutrient removal efficiencies approach zero and any sediment-bound nutrients deposited in the waterway tend to be resuspended and washed out later.

C20. Filter Strips/vegetated field strips/field borders

(Understanding Riverbank Erosion from a conservation perspective; Clouston, 1996)

Vegetated field strips (= filter strips) are strips of uncultivated vegetated land between a field and a stream, whereas field boundaries are the equivalent all round the edge of a field. These remove nutrients by settling out/trapping suspended solids in run-off and by plant uptake and soil adsorption/ binding of dissolved phosphorus, thereby reducing their potential effects on adjacent watercourses. The location of filter strips is however critical to ensure that during heavy run-off or erosion events a flushing effect through the filter area does not exacerbate potential pollution. There is no definitive width for these strips and are probably best combined with other best management practices such as permanent pasture, short/long term ley, buffer strips and best management practices reducing the potential volume of run-off from large areas. Tippett and Dodd reported no efficiency data for field borders or strip-cropping. However, numerous research studies showed variable results for field strips although there seems to be some agreement that they are more efficient at removing suspended solids in run-off than nutrients. Casman (1990) suggested that a figure of 30-90% efficiency was probably appropriate for filter strips 5-10 m wide. On the basis of this Tippett and Dodd (1995) proposed a cost effectiveness range from 40 – 100 $US/kg P reduced.

C21. Soil berms

(Clouston, 1996)

This best management practice aims to redirect run-off or erosion materials away from watercourses thereby reducing direct pollution. It can involve deep ploughing to a riparian edge to create a raised berm or more engineered operations such as strengthened soil structures around a retaining pond or sediment basin, or across a field slope to break it up into segments to prevent run-off. It should only be regarded as a 'last resort' as it does not reduce, by itself, pollution levels through management but prevents pollution entering watercourses. The pollutants still need to be managed.

C22. Riparian buffer

(Clouston, 1996)

This best management practice moves machinery operations and livestock access away from watercourses thereby reducing the potential direct pollution. Typical examples include prevention of fertilisers and sprays entering directly the watercourse whilst tractors turn on headlands, and preventing livestock polluting watercourses through urine/faeces and disturbing stream/river sediment. It can be combined with access tracks, livestock exclusion, filter strips and other techniques, which provide a barrier or filtration to run-off or erosion on the riparian edge.

C23. Critical area planting

(Clouston, 1996; Mainstone, *et al*, 1996; MAFF, 1997)

See B5 for general description but can also reduce particulate run-off.

D. General Farm Management

D1. Irrigation management

(Clouston, 1996; MAFF, 1997)

This best management practice follows a similar approach to pesticide and nutrient management. It requires matching demand and supply closely to avoid excess water and therefore potential for run-off and erosion. It requires detailed expertise about the moisture holding capacity of soils and water requirements of crops. It could be combined with pesticide and nutrient management plans.

D2. Ditch management

(Clouston, 1996; MAFF, 1997)

This best management practice aims to target the appropriate ditch management techniques to specific requirements. For example, the common practice of ditch clearance and deposition of soil adjacent to the riparian edge should only be undertaken where it is necessary to clear drainage outfalls. Where ditch management is required it should be undertaken in rotation to avoid excessive lengths of cleared vegetation and allow natural re-growth. Maintaining existing vegetation and thereby reducing water flow will improve sedimentation and bank stabilisation. It can be combined with soil spreading and other best management practices orientated towards management of water flow and livestock exclusion. Ditch management includes the maintenance of land drains.

D3. Hedgerow management

(Clouston, 1996)

This best management practice is aimed at returning to traditional hedgerow management practices such as hedge laying, with the objective of creating a more substantial barrier to erosion or run-off. Sedimentation and infiltration could therefore be improved. Current hedgerow management practices tend to create 'gappy' barriers requiring additional fencing. It could be combined with hedgerow planting and field boundary/access points. This best management practice is incorporated within techniques promoted through Environmentally Sensitive Areas.

D4. Access tracks

(Clouston, 1996; MAFF, 1997)

This best management practice can comprise of relocating an existing access route to areas less vulnerable to erosion or run-off, such as at the top of the slope rather than the bottom, and/or providing a hardened surface to reduce the erosive effects of livestock poaching or vehicle rutting. Access tracks can also act as conduits for moving run off from an eroded area to a water body. Careful design of tracks, e.g. location of ewes etc, and location can prevent direction to watercourses but to soakaways or other treatment areas. It could be combined with field boundary access points or livestock exclusion.

D5. Field boundary/access points

(Clouston, 1996)

Reorientation or re-establishment of field boundaries and access points to assist with other best management practices such as contour cultivations and cropping, access track locations and movement of livestock and machinery to and from the main farm buildings. The overall objective is to reduce the potential for over use of land and farming practices, which would lead to erosion and run-off.

D6. Vehicle movements

(Clouston, 1996; MAFF, 1997).

This best management practice aims to improve the movement of machinery from the main farm centre to individual fields. In particular, attention should be paid to moving the harvested crops from the land during wet conditions. Examples include ensuring tractors move over uncultivated land thereby reducing erosion and through gateways, which have been hardened to reduce the effects of rutting. It could therefore be combined with access routes, permanent pasture, long term leys and field boundary/access points.

D7. Farmyard run-off interception

(Clouston, 1996).

This best management practice aims to direct any potential run-off from the farmyard which may carry a high concentration of pollutants to either land, which is capable of infiltration or filtration of sediment, or to best management practices which can cater for the high concentrations such as water/sediment basin/retention ponds.

D8. Roof run-off interception

(Anon, 1995; Clouston, 1996)

This best management practice aims to intercept run-off from the roofs to farm buildings, prior to flow across highly polluted and concentrated areas within the farm centre, such as livestock collecting yards, silage feeding areas and chemical mixing and spraying wash-out areas. It could also prevent high volumes of water during heavy rainfall being directed to land subject to erosion and run-off. It can therefore be combined with water diversions, water/sediment basin/retention ponds and grass waterways.

E. Livestock

E1. Livestock exclusion

(Understanding riverbank erosion from a conservation point of view; Harper and Pacini, 1995b; MAFF, 1997)

This practice is aimed at removing livestock either temporarily or permanently from areas which could be sensitive to erosion or through intensive livestock movements leading to poaching and subsequent vegetation loss or pollution of watercourses through direct access. Examples could include fencing riparian edges, relocating feed/water troughs to hardened areas and provision of shelter away from steep slopes or wet areas. It can therefore be combined with many other livestock management best management practices. Fencing riverbanks also has the added benefit of removing direct manure inputs to the river.

E2. Feed/water trough location

(Clouston, 1996; MAFF, 1997)

By locating feed and water troughs to areas which are not sensitive to erosion, the effects of intensive livestock movements can be reduced. It may require provision of hardened areas within particularly sensitive fields. Mobile feeding facilities could be provided where it is not possible to provide permanent facilities. Careful location of feed/water troughs can also reduce the creation of stock trails, by drawing livestock away from watercourses and reducing erosion on steep land. It can be combined with livestock exclusion, access tracks and field boundary/access points.

E3. Livestock trails

(Clouston, 1996; MAFF, 1997)

This best management practice aims to force livestock movements within or between fields to areas not subject to erosion. It will also aim to reduce the potential for livestock trails to direct run-off along uncropped areas. It should aim to reduce erosion from unsurfaced trails and the potential losses of high concentrations of nutrients in these areas. It can also be combined with stream crossings to prevent animal trails eroding riparian edges when movements are between fields abutting watercourses. This best management practice could involve solely management of animal movements within existing fields or locate more permanent tracks away from sensitive areas and/or provide hardened surface.

E4. Grazing management

(Understanding Riverbank Erosion from a conservation perspective; Clouston, 1996; MAFF, 1997)

This best management practice requires detailed attention within the overall grazing management strategy to the intensity and duration of grazing appropriate to the soil conditions and proximity to adjacent watercourses. Objectives should be to maintain adequate grass cover together with appropriate recovery time. It can also be combined with livestock trails, field boundary/access points and stream crossings. It may require extensive changes to existing cropping rotations and livestock management.

E5. Waste management

(Harper & Pacini, 1995b; Clouston, 1996)

This best management practice draws together a wide range of techniques focused on the storage, treatment and disposal of farm effluent. Ideally, it should be combined with a nutrient management plan. The techniques are discussed widely in existing MAFF/EA guidelines. The main problem is that most farms do not have sufficient storage to keep manure until growing crops require it. Tippett and Dodd (1995) suggest that spreading at any rate up to and including 4x the US agronomic rate of application resulted in a 5% loss to surface waters of P spread on the land. The effectiveness of waste management, particularly the provision of storage will depend, to great extent, on the amount of land which a farm has available to spread manure, ie the number of hectares of spreadable land per livestock animal or bird kept. Tippett and Dodd (1995) also propose the use of anaerobic digesters to reduce P levels in manure, prior to spreading. However, it is uncertain how this could be achieved, since P is effectively conservative in these situations (unlike nitrogen which will be reduced by denitrification processes).

E6. Reduce strip grazing of fodder crops

(MAFF, 1997)

Crops such as kale are used to feed cattle in winter. The bare soil which is left and the high trampling which takes place encourage erosion.

E7. Location of outdoor pig units to minimise the risk of erosion

(MAFF, 1997)

F. Stream Bank Works

F1. Stream bank stabilisation

(Understanding Riverbank Erosion from a conservation perspective; Clouston, 1996).

By improving stream bank stabilisation, direct erosion from bank slippage should be reduced. Stabilisation could be achieved through a variety of techniques which vary considerably in cost from critical area planting, livestock exclusion and ditch management to engineering operations or using supporting materials. The latter include:

a. Well-managed trees can reduce riverbank erosion by 85-90% (Understanding Riverbank Erosion from a conservation perspective) but willows need to be copied/ pollarded every 5-0 years to maintain their efficiency.

b. Place stoning (large boulders) or pitching (dry stone walling at the toe of the bank) around or below the water level.

c. Fagotting: willow whips (bundles of small branches and twigs) laid near the waters edge and tied in place. Encourage silt deposition, especially when they shoot.

d. Re-seed bare ground on or adjacent to banks.

F2. Stream crossing

(Clouston, 1996)

This best management practice involves provision of a purpose built bridge/culvert across a watercourse for livestock and machinery. It could be combined with provision of watering facilities at the crossing, although as discussed later, direct access of livestock to watercourses should be limited. By focusing crossings to purpose built structures, erosion along stream banks would be reduced. It can be combined with livestock trails, access tracks and livestock exclusion.

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